LIMNOLOGICAL STUDIES ON LAKE TINISHU ABAYA, ETHIOPIA

BY

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A Dissertation Submitted to the Department of Zoological Sciences of Addis Ababa University in Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Biology (Fisheries and Aquatic Sciences)

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School of Graduate Studies
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Approved by the Examining Committee

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Chairman ______________________________Signature______ Date__________

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Declaration

I, the undersigned, declare that this disseration has been composed solely by myself and that it has not been submitted, in whole or in part, in any previous application for a degree or any other qualification. Except where duly stated by references or acknowledgments, the work presented in this disseration is entirely my own.

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ABSTRACT

Lake Tinishu Abaya, hereafter referred to as LTA, is a small-sized inland water body in the rift valley system of Ethiopia. LTA was stocked with Tilapia fish (*Oreochromis niloticus*) in 1997 to enhance fishery and provide animal protein source to the local community. This study designed to investigate the limnological features (physical, chemical, and biological) of the lake and determine the ecological productivity, water quality, and its potential service for users in the neighborhood and the nation at large. Moreover, the baseline information from the study can be used by policymakers to design strategies for sustainable use of LTA.

All the data for this study were collected and analyzed using standardized sampling techniques and methods. The water samples for the analyses of physicochemical parameters, phytoplankton, and zooplankton were collected monthly between January and December 2016 from two selected sampling sites (open water and shore area). Benthic macroinvertebrates were collected from March to August 2017 from five sampling sites. For analysis of fish gut content and related studies, *O. niloticus* fish samples were collected from March to May and July to September in 2017.

The results of the various physicochemical parameters suggested that LTA was well oxygenated, slightly warm, and alkaline. From the values of conductivity (as a proxy to salinity) the lake can be classified as a freshwater body compared some other rift valley lakes of Ethiopia and TDS values. TDS of LTA was always less than 3000 mgL^{-1} indicating freshwater system. The water of LTA was turbid, low water clarity, shallow euphotic area, and hypertrophic state with nutrient enrichment.

In LTA, 37 phytoplankton taxa belonging to six groups were found. Bacillariophyceae (40%) were the most diversified phytoplankton followed by Chlorophyceae (30%), Cyanobacteria (19%), Euglenophyceae (5%), Dinophyceae (3%), and Cryptophyceae (3%). In this study, 24 zooplankton taxa comprising of rotifers, cladocerans, and copepods were recorded. The abundance of zooplankton was dominated by copepods (54%) followed by rotifers (40%). The overall mean total standing biomass of zooplankton was 230.87 in open water and 164.2 $\mu g\ L^{-1}$ at shore area, and it was broadly dominated (94%) by the microcrustaceans (copepods & cladocerans).

The photosynthetic productivity of phytoplankton of LTA was determined in the open water, and the results showed that the rate of photosynthesis (A_{max}) ranged from 0.62 $g\ C\ m^{-3}\ h^{-1}$ to 2.02 $g\ C\ m^{-3}\ h^{-1}$. The hourly integral photosynthetic production ($\sum A$) and pattern of variation for the daily integral rates of photosynthesis ($\Sigma\Sigma A$) ranged from 0.247 $g\ C\ m^{-2}\ h^{-1}$ to 1.022 $g\ C\ m^{-2}\ h^{-1}$ and 5.43 $g\ C\ m^{-2}\ d^{-1}$ to 9.194 $g\ C\ m^{-2}\ d^{-1}$, respectively. Because of the number of phytoplankton and zooplankton taxa, their abundance, and

biomass, and considering marked values of photosynthetic productivity it was concluded that LTA is biologically productive aquatic resource to support fish and other aquatic organisms.

The ecological condition of LTA was assessed using benthic macroinvertebrates as well, and a total of 5735 benthos specimens comprising of 23 taxa were collected from all the study sites. There was a spatial effect on the distribution of benthic individuals (ANOVA; $p < 0.05$). It was high at Dacha riverside (n=2089) followed by Bobodo riverside (n=1145), Reference site (n= 963), Badober riverside (n= 859), and Main fish landing site (n=679). The Hemiptera family were the predominant macroinvertebrates that contributed the largest number (n=2546) of the total samples followed by Diptera (n=878), Coleoptera (n=835, 14.56%) and Gastropods (n=631). The majority (about 70%) of the benthic communities of LTA comprised pollution tolerant species as compared to pollution sensitive ones. This indicated the existence of organic pollution, and thus LTA is undergoing environmental stresses.

The diet and other related aspects of the stocked *O. niloticus* were also assessed. The relationship between the total length and standard length of *O. nilotucus* in LTA was linear and significant ($R^2 = 0.962$, TL=0.1456SL+1.8088). The relationship between total length and the total weight of *O. niloticus,* on the other hand, was curvilinear with a strong relationship ($R^2 = 0.9848$, $TW = 0.0194\ TL^{2.9876}$). The slope of the regression (b) was 2.9876, closer to the isometric growth value (b=3) of fish. The Fulton´s Condition Factor (FCF) of *O. niloticus* ranged from 0.96 to 3.51 (mean= 1.88). The well being of *O. niloticus* fish showed that the majority of the population in LTA were in good condition (FCF >1). Phytoplankton, detritus, zooplankton, and macrophytes were the most important food items while insects, nematodes, ostracods, and fish scales made up minor portions of the diet of *O. niloticus* in LTA. The importance of phytoplankton, macrophytes, and detritus increased with increasing fish-size and the significance of zooplankton, insects, and other animal origin food items declined with increasing fish size.

In a nutshell, LTA is a productive lake and an important aquatic resource for the surrounding area. But the lake and its watershed are facing many threats from intensive human activity. The major problems affecting LTA include excessive water abstraction, extensive shoreline modification, sedimentation, eutrophication, water pollution due to organic waste from agricultural area and overfishing. At the same time, efforts to ensure the health and normal functioning of this productive ecosystem are negligible. Thus, management solutions should be developed to avert the current and continuing degradation of LTA and its environs. Some of the actions that should be taken, but not limited to, are: demarcating a buffer zone, restoration of riparian habitats, regulating the excessive water abstraction, controlling mass fish catching, and empowering the local community to protect and conserve this aquatic resource.

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LIST OF ABBREVIATIONS

ANOVA	Analysis Of Variance
APHA	American Public Health Association
ASL	Above Sea Level
DT	Dominant Taxa
FAO	Food and Agricultural Organization
FCF	Fulton´s Condition Factor
HFBI	Hilsenhoff Family Level Biotic Index
L	Lake
LTA	Lake Tinishu Abaya
Meq	Molecule of equivalent
NFALRC	National Fisheries and other Aquatic Life Research Center
NTU	Nephelometric Turbidity Unit
OBBN	Ontario Benthos Biomonitoring Network Protocol Manual
RDA	Redundancy Analyses
RICH	Richness
SDI (H')	Shannon Diversity Index
SDOI	Schoener Diet Overlap Index
SNNPRS	Southern Nations, Nationalities, and Peoples' Region State
SODIS	Solar Disinfection
SPSS	Statistical Program for Social Studies
SRP	Soluble Reactive Phosphorous
TA	Total Alkalinity
TDS	Total Dissolved Solids
TP	Total Phosphorous
TSI	Trophic State Index
WFD	Water Framework Directive
WHO	World Health Organization
WQI	Water Quality Index

CHAPTER 1: GENERAL INTRODUCTION

1.1. Background and Justification

Limnological studies of tropical freshwater ecosystems are not extensive when compared to temperate ones (Hecky and Fee, 1981). But East African lakes have been studied fairly well in comparison with other regions of Africa. This is true of Ethiopian lakes even though rift valley lakes have been relatively better studied (Girum Tamire, 2014; Lemma Abera, 2016; Abenet Woldesembet, 2019).

Ethiopia is a country in the Horn of Africa with an estimated area of around 1.1 million km^2. The Wetlands cover 1.14 % of the total landmass of the country. It is often called the "Water Tower of Africa" (Tenalem Ayenew, 2009). Ethiopia is endowed with a large number of standing water bodies, whose sustainable use can contribute to the economy of the country. The lakes are critical to the survival of local communities as they are the actual and potential sources of food and income (Zinabu Gebremariam, 1994).

Scientific studies on water quality and productivity can contribute to the assessment of the suitability of water for particular uses such as domestic, irrigation, sanitation, fish production, and other socioeconomic values based on its physical, chemical, and biological characteristics (Akpofure, 2013). The quality and productivity of any water body are governed by its physicochemical factors and biological characteristics (Venkatesharaju et *al.*, 2010). Distribution and productivity levels of organisms are largely determined by these factors (Pawar *et al.*, 2006).

The productivity of any aquatic water body also depends on the plankton present in it (Pawar *et al.*, 2006). According to the same authors, plankton are very useful tools for the assessment of biotic potential and contributes to the overall estimation of the basic nature and general economic potential of the water body. The plankton communities that considered in the present study of LTA include phytoplankton and zooplankton.

The roles of phytoplankton in managing the bioenergetics of lakes and their function as bio-indicators have been known for a long time. Phytoplankton are used as bio-indicators since they reflect even the slightest changes taking place in their immediate environment by changing their species composition, biomass, community structure, chlorophyll pigment content, and productivity. As phytoplankton assemblages are at the base of the food web, changes in phytoplankton biomass and the pattern of primary production have implications for the whole community, including fish (Girma Tilahun and Ahlgren, 2010). For instance, phytoplankton production (PP) is regarded as a good predictor of fish yield in lakes and seasonal measurements of PP are a prerequisite for good such estimates (Hooker *et al.*, 2001).

The primary productivity of different water bodies has been widely investigated to assess the fish production capacity of a water body to formulate fishery management policies (Sarvesh *et al.*, 2015). The study of energy transfer in lakes and reservoirs is based on the measurement of primary productivity of phytoplankton and the environmental variables, which limit or control this productivity. Primary productivity of aquatic ecosystems depends basically upon the photosynthetic activity of autotrophic organisms, like phytoplankton particularly in the open area of the lake (Girma Tilahun and Ahlgren, 2010). Phytoplankton are the major primary producers in many aquatic systems and are important food for consumers (Choudhury and Ruma, 2014).

Primary productivity and biomass of phytoplankton are affected by an array of chemical, physical, and biological factors. The more frequently a lake is stirred by winds to the bottom, the faster the nutrients are recycled from the mud into the photosynthetic zone where they may accelerate the rate of productivity (Tadesse Fetahi *et al.*, 2014).

Zooplankton are planktonic animals/organisms that are an important component of aquatic ecosystems, as they participate in the natural purification of water (Seyoum Mengistou and Fernando, 1991a; Adane Sirage, 2006). The presence and dominance of zooplankton species play very significant roles in the functioning of freshwater ecosystems (Adamneh Dagne, 2010).

Some have observed that zooplankton constitute the main food for fish, and the adult fish not only consume them but also select them as detectable items. In freshwater environments, zooplankton plays relevant roles in energy transfer and nutrient transport and regeneration, owing to their position in the food web as the main direct consumers of phytoplankton (Eshete Dejen *et al.,* 2004). Zooplankton are also good bioindicators of the physical and chemical conditions of aquatic environments which cause changes in the qualitative and quantitative composition of zooplankton and influence their densities, expressed as number per area or volume units (Ayalew Wondie, 2006).

Biomass of zooplankton reflects the instantaneous quantity of live organic matter in an area and also provide a means of analyzing an ecosystem's productivity, despite taxonomic composition (Seyoum Mengistou and Fernando, 1991b). Studies that consider biomass values are important because they allow comparison of different environments, providing a common unit to evaluate zooplankton groups (Ghidini and dos Santos-Silva, 2011). The establishment of length-weight regressions are fundamental when determining the biomass of aquatic communities and also in most studies of food web interactions which contribute to our knowledge of how aquatic ecosystems function. For microcrustaceans (cladocerans and copepods), weight estimates from length-weight regressions have been the most frequently used technique employed to approximate the biomass of these organisms (Kenneth *et al.*, 2008).

Determining zooplankton size is an important tool for building the size-weight regressions, which are useful when only size data is available. Due to these reasons, estimation of zooplankton size and dry weight constitutes an important contribution for the study of trophic-web structure in aquatic ecosystems, considering its relationship with the trophic status of the water bodies (Pinto-Coelho *et al.*, 2005).

Accurate measurement of biomass is necessary to understand the structure and dynamics of biological communities (Marin *et al.*, 2001). Basic knowledge on the biomass of different zooplankton species provides the necessary data for the calculation of secondary production, and information about the competitive strategies responsible for their success in an environment (Adamneh Dagne *et al.*, 2008). Measurement of zooplankton biomass is also essential in studies of production ecology of zooplankton (Seyoum Mengistou and Fernando, 1991b). Generally, zooplankton directly affects the first industry and studying on zooplankton is, therefore, used to determine the ecological status of the water body (Letícia *et al.*, 2016).

Earlier, biological communities like plankton have been used for assessing water quality of rivers, lakes, and streams. Nowadays, benthic macroinvertebrates are used as the main biological indicators (Muralidharan *et al.*, 2010). This is because any modifications of water bodies due to pollutants entering aquatic systems have profound effects on the macroinvertebrate communities living in there and produce relatively quick undesirable changes in the abundance and species variations that reflect the degradation of the environment (Gabriels *et al.*, 2005).

Using benthic invertebrate communities as indicators of environmental degradation or restoration has become widespread and reliable for bio-assessment since the benthos broadly reflects environmental conditions. Because of their abundance and position as “middlemen” in the aquatic food chain, the benthos plays critical roles in the natural flow of energy and nutrients (Fernández-Díaz *et al.*, 2008). Most have narrow ecological requirements and are very useful as bio-indicators in determining the characteristics of aquatic environments (Benetti and Garrido, 2010). Some can tolerate higher loads of pollution than others i.e. they can respond to nutrient enrichment, oxygen availability, food quantity and quality, and changes in habitat structure (Lücke and Johnson, 2009).

Hence, by assessing indicator species, diversity, and functional groups of the benthic macroinvertebrate, it is possible to determine water quality and impact of pollutants on aquatic community and using biological indicators are more adequate to detect long-term changes in water quality, since aquatic organisms are adapted to specific environmental conditions (Aschalew Lakew and Moog, 2015).

Another very important biotic community besides the plankton and benthic macroinvertebrates, the role of fishes in the aquatic system must be dealt with. Fish are key elements in many natural food webs and an important source of food and recreation, which in some ways make them serve as environmental indicators (Shipton *et al.,* 2008**)**.

The studies of the food and feeding habits of freshwater fish species is a subject of continuous research because it constitutes the basis for developing successful fisheries management programme on fish capture and culture (Kamal *et al*., 2010). Food availability determined the well being of fishes and their reproductive potentialities in any aquatic ecological system (Elias Dadebo *et al*., 2014).

Weight and size of fish reflect food availability in the aquatic ecosystem (Olurin and Aderibigbe, 2006). According to the same authors, quantitative determination of the components of the diet, their nutritive values, and seasonal availability are the basic parts for the understanding of environmental impacts on the condition and growth of fish. Therefore, an understanding of fish diet and the growth of its influence can be essential for understanding the ecological role and the productive capacity of fish populations (Elias Dadebo *et al.*, 2014).

Although LTA is the only freshwater ecosystems for the surrounding communities, it has not been given the attention it deserves. This is most probably due to its remote location and having a small size compared to other many rift valley lakes of Ethiopia. Thus, there is a need to study the ecology and productivity of LTA. In addition, lack of baseline limnological knowledge of this freshwater system handicaps decision makers and local communities for proper management and conservations of these aquatic resources (Zinabu Gebremariam *et al.*, 2002; Zerihun Desta, 2003). This study investigated several limnological aspects of LTA, including its biota, its primary productivity, and possible physical, chemical, and biological factors affecting these. The study would provide a basis for management of the lake.

1.2 Statement of the problem

The limnology of some of the Ethiopian lakes is unexplored. LTA is one of such lakes, which has not been given due attention. The remote location and small size of the lake as compared to the other big rift valley lakes of Ethiopia may result in less attention for the lake. Although LTA has not received attention, it supports the life of the nearby communities directly and indirectly. It is potential for economic significance such as fish production; small-scale irrigation, recreation, and domestic use are among others. Studying the limnology of LTA would contribute to acquiring knowledge for the proper management and conservations of the aquatic resources. Thus this study is designed to investigate the overall ecological productivity of LTA in relation to the changes in the various abiotic and biotic factors.

1.4 Research objectives

1.4.1 General objective

The ultimate objective of the study was to investigate the limnological features (physical, chemical, and biological) of the lake and determine the ecological productivity, water quality, and its potential service for users in the neighborhood and the nation at large.

1.4.2 Specific objectives

- To determine the water quality and trophic state of LTA in relation to the physicochemical parameters;
- To study phytoplankton species composition and biovolume in relation to various physicochemical parameters;
- To study the photosynthetic productivity and biomass of phytoplankton;
- To investigate the ecology, abundance, and biomass of zooplankton in relation to some environmental variables;
- To explore the diversity of benthic macroinvertebrates communities;
- To observe the die*t and related aspects* o*f Oreochromis niloticus.*

1.5 Research questions

- What are the water quality and trophic status of LTA like presently?
- What are the major plankton and benthos taxa found in LTA?

- What is the abundance and biomass of the phytoplankton and zooplankton in LTA?
- What is the primary productivity of phytoplankton in LTA?
- What are the feeding habits and wellbeing of *O. niloticus* in LTA

1.6 Description of the study area

LTA is interchangeably called "Small Abaya" (Kassahun Asaninew *et al.*, 2011). The word "Tinishu" in the study lake is used purposefully. It is because there is another and a relatively big lake named "Lake Abaya" in the Southern Nation, Nationalities, and Peoples Regional State (SNNPRS) where LTA is found. Comparing the two lakes, LTA is 100 times smaller than the neighboring Lake Abaya. The term *"Tinishu"* in Amharic means *"the smaller"/ the little"*. To separate the two lakes, the communities called the smaller lake as "Tinishu Abaya" to confirm its smaller size.

LTA is a lake in the Southern Nations, Nationalities, and Peoples' Region State (SNNPRS) of Ethiopia. It is located 160 km far from South of Addis Ababa, which is the capital city of Ethiopia. The lake is found in Silttie Zone (SNNPRS). Silttie is bordered on the south by Alaba special Woreda, on the Southwest by Hadiya, on the North by Gurage, and on the East by the Oromia regional state of Ethiopia.

LTA is in a remote area of a small village (Kebele) known as, "*Gebrie-Ber"* which is at about 15 km in the eastern direction from the township of Kebet. The town Kebet used as a capital town for Silttie woreda where the name of the zone (Silttie zone) comes from. Worabie is the capital of Silttie zone. It is situated when one crosses the road from Addis Ababa to Arbaminch in the southwest direction.

LTA is found at 7^{0}29´03.65´´ N latitude and 38^{0}03`17.79´´ E longitude and at an altitude of 1835 m above sea level. The lake is comparatively small in size with a surface area of 1253 hectares (12.53 km^{2}) (Kassahun Asamine *et al.*, 2011). It is a shallow lake with a maximum and mean depth of 3.7 m and 2.9 m, respectively (survey of the present study). The lake has an oval shape (Fig. 1.1), bounded by the mountain at the eastern shore. The lake has two major feeder rivers - River Dacha (northern corner) and River Bobodo (southern corner). As a sort of spillway, River Badober (northern side) serves as an outlet for the lake especially during the rainy seasons (July to September). The lake water looks like a brown color even there is somehow green in some dry months (e.g. it was shallow green from January-March).

There are different species of aquatic habitats or macrophytes in and around LTA. The dominant macrophytes found in this lake are, *Cyperus dubius*, *Persicaria senegalensis*, *Potamogeton* sp., *Lugonia* sp., and many types of grass species (Appendix 4). The *C. dubius* and *P. Senegalensis* are predominately found in the western and southwestern shoreline of the lake. Macrophytes are sparse in the eastern direction of the lake, which is extensively affected by human and livestock activities. In all cases of the macrophytes, their abundance increases with the onset of the wet seasons (July to September). The *Potamogeton* sp. is apparently absent during the dry periods of the year (January-April).

Based on the request of the Silte Woreda Bureau of Agriculture, NFALRC (National Fisheries and other Aquatic Life Research Center, Sebeta, Ethiopia) in 1997 did some preliminary survey in LTA, and the study confirmed the absence of any fish of commercial importance except Barbus species and *Tilapia zilli* (Kassahun Asamine *et al.,* 2011). Soon after the survey, *Oreochromis niloticus* fries were stocked into the lake in 1997 to enhance fishery and availing cheap fish protein to the local community (Kassahun Asamine *et al.,* 2011). While, *Tilapia zilli* was reported as native to the lake (Kassahun Asamine *et al.,* 2011), no specimen of these species were captured during this study. However, *O. niloticus* and *Barbaus sp.* have been confirmed during this study period.

LTA is a source of livelihood and supports many socioeconomic activities of which fish production; irrigation, human consumption, recreation, bathing and washing, and animal watering are among others. The lake is also a vital environment for birds as it is north-south migration corridor of different bird species (e.g., Pelicans, Cormorants, Kingfishers, etc.) and other aquatic microorganisms. There are also many activities (Appendix 5) in LTA and its watershed such as cultivating crops, vegetables, and onions, and other related activities.

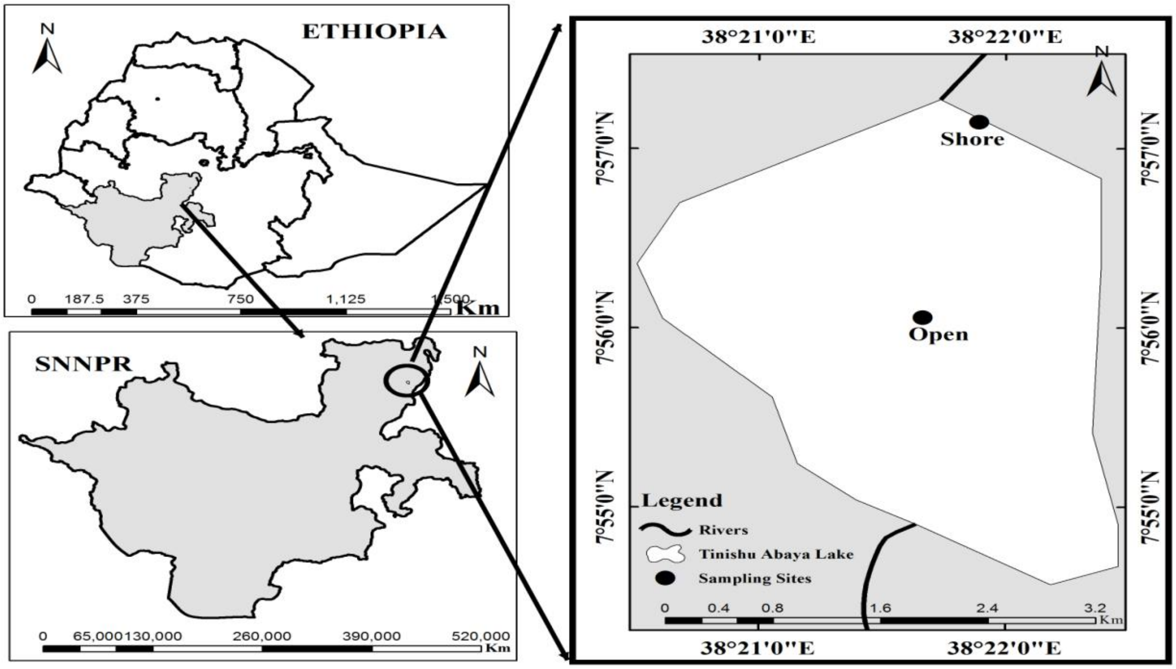


Fig. 1.1 Location of LTA and the sampling sites (black spots in the inset map)

1.7 Thesis outline

This study had a compilation of results that emanated from studies made on the limnological characteristics of a tropical small-sized and shallow tropical Rift Valley Lake namely LTA (Ethiopia). The study was organized into eight chapters. The first chapter is "general introduction" which introduces the review of the relevant literature. The last chapter consisted of the "general conclusion and recommendations" given based on the findings of the present study. From chapter two to chapter seven (six separate studies) the methodology and pertinent issues of the major limnological and fishery studies of the lake are included.

The first findings of this study (Chapter two) deal with "analysis of water quality and trophic state of Lake Tinishu Abaya in relation to physicochemical parameters". In this chapter, the temporal and vertical distributions of the physical and chemical parameters were measured and their possible causes and implications for the lake ecosystem conditions had been discussed. Within the chapter, the trophic state of the lake also calculated. Two articles have been published from the data of this study:

> Yirga Enawgaw and Brook Lemma (2018): Water Quality and Productivity Assessment of Lake Tinishu Abaya for Multiple Designated Water Uses, Ethiopia. *Journal of Environmental & Analytical Toxicology*. **8(3):** 1-12.

> Yirga Enawgaw and Brook Lemma (2018): Water Quality Index (WQI) in the Assessment of Lake Tinishu Abaya for the Suitability of Drinking Purpose. *International Journal of Modern Chemistry*. **10(2):** 256-267.

The second findings of this study (Chapter three), deals with the "species composition, fresh weight biomass and distribution patterns of phytoplankton in Lake Tinishu Abaya". This chapter discusses the temporal variations in species composition and abundance in terms of biovolume or growth performance (fresh-weight biomass) of phytoplankton in relation to the major physicochemical factors. A result of this chapter has been published in the following article:

> Yirga Enawgaw and Brook Lemma (2018): Phytoplankton Community Composition and Nutrient Conditions as an Indicator of Ecosystem Productivity in Lake Tinishu Abaya, Rift Valley, Ethiopia. *International Journal of Fisheries and Aquatic Studies*. **6(3):** 173-186.

The third findings of this study (Chapter four) deal with the "photosynthetic productivity and biomass of phytoplankton in Lake Tinishu Abaya". This chapter focused on the seasonality in the primary production and biomass in terms of chlorophyll-*a* content of phytoplankton to assess the potential ecological productivity of Lake Tinishu Abaya more emphasis given for fish production. Data of this chapter has been published in the following article:

> Yirga Enawgaw and Brook Lemma (2018): Seasonality in the Photosynthetic Productivity and Biomass of Phytoplankton in Lake Tinishu Abaya (Rift Valley, Ethiopia): The Basis for Fish Production. *International Journal of Fisheries and Aquaculture*. **10(8):** 95-108.

The fourth findings (Chapter five) investigated the " community structure and biomass of zooplankton in Lake Tinishu Abaya". This chapter primarily emphasized the community structure and standing biomass of zooplankton in relation to prevailing environmental conditions to show the potential water productivity of Lake Tinishu Abaya. From the results gathered a single article has been published:

> Yirga Enawgaw and Brook Lemma (2018): Zooplankton Communities as an Indicator of Ecosystem Productivity in Lake Tinishu Abaya, Rift Valley, Ethiopia. *International Journal of Fisheries and Aquaculture.* **10(5):** 53-70.

The fifth findings (Chapter six) investigated the " benthic macroinvertebrates and an indicator of water quality in Lake Tinishu Abaya". This chapter primarily emphasized on the ecological condition of Lake Tinishu Abaya in relation to the spatial variations of benthos communities from the littoral regions of the lake. The study also identified the pollution-sensitive and tolerant macroinvertebrates to assess the ecological stress or existence of organic pollution of LTA.

From the data, the following manuscript has been submitted for the publication and it is accepted:

> Yirga Enawgaw and Brook Lemma (2019): Ecological Conditions of Lake Tinishu Abaya in relation to Benthic Macroinvertebrate, Ethiopia. ***Oceanography & Fisheries***. Accepted.

The sixth findings of this study (Chapter seven) deal with "feeding habits and related aspects of Nile Tilapia (*Oreochromis niloticus* L.) in Lake Tinishu Abaya". In this topic, the food items found in the gut of O. *niloticus*, the well-being of the same species of fish in LTA and other related aspects of the fishery are presented. A single manuscript has been published in this chapter.

> Yirga Enawgaw and Brook Lemma (2018): Seasonality in the diet composition and ontogenetic dietary shifts of *Oreochromis niloticus* L. (Pisces: Cichlidae) in Lake Tinishu Abaya, Ethiopia. *International Journal of Fisheries and Aquatic Research*. **3(1):** 49-59.

CHAPTER 2: ANALYSIS OF WATER QUALITY AND TROPHIC STATE OF LAKE TINISHU ABAYA IN RELATION TO PHYSICO-CHEMICAL PARAMETERS

2.1 Introduction

Water quality deals with the physical, chemical, and biological characteristics in relation to all other hydrological properties (Aminu *et al.*, 2017). The quality of water in any ecosystem provides significant information about the available resources for supporting life in that ecosystem and it provides the basis for judging the suitability of water for its designated uses and to improve existing conditions (Shined *et al.*, 2011).

The fertility of water is related to its physicochemical properties and understanding of water chemistry serves for considering whether the water is rich or poor in biological production. The physical and chemical properties of water greatly influence the use of distribution and richness of biota (Zinabu Gebremariam, 1994). A property such as high dissolved oxygen in water is an essential pre-requisite for satisfactory aquatic life (Yezbie Kassa, 2016). The techniques of using physical and chemical properties to assess water bodies are essential. They also reveal the concentrations of known environmental contaminants which could render such water unfit for human consumption and other purposes (Venkatesharaju *et al.,* 2010).

The trophic state or stage of eutrophication, of a body of water, has been associated with the study and management of cultural eutrophication of lakes. The tropic state concept focuses on variables that are directly or indirectly related to primary productivity (i.e., algae and aquatic plants) (Dodds and Cole, 2007). According to Carlson (1970), the trophic state is defined as the total weight of living biological material (biomass) in a water body at a specific location and time. The trophic state is understood to be the biological response to forcing factors such as nutrient additions (Nauuman, 1917), but the effect of the nutrient can be modified by parameters such as seasons, grazing, mixing depth, etc.

Biological productivity, i.e., the trophic state of the inland water body can be determined by the direct and indirect method of measuring the level of nutrients, chlorophyll content and measuring water clarity (Carlson, 1977). Carlson's trophic state index (TSI) attempts to provide an unambiguous classification system of lakes based on their productivity levels. The index was intentionally designed to be numerical so it could be easily understood by the general public, with no loss of information or sensitivity to change.

For optimum development and management of aquatic resources for the beneficial uses, current information is needed which is provided by water quality programmers (Lloyd, 1992). In addition, to benefit from lakes it is a key to know the characteristics of the lakes under investigation (Shinde *et al.*, 2011). Therefore, it has become obligatory to analyze at least the important water parameters when ecological studies on aquatic ecosystems are carried out (Zinabu Gebremariam *et al*., 2002). The aim of this particular study was to determine the water quality and trophic state of LTA in relation to the changes in various physical and chemical parameters to observe the potential of the lake water for multiple designated water use.

2.2 Materials and Methods

2.2.1 Sampling protocol

For the analysis of various physical and chemical variables, routine water sample collections were carried out for a year monthly between January 2016 and December 2016 from two predefined sampling sites. The first was shore site (07^0 57.234' N & 038 22.037'E) where the high human impact was expected, as it is close to the edge of the lake and the second was from a relatively less human affected area (open-water site). The open-water site (07^0 56.658' N & 038^0 21.787' E) is found in the center (length and width of the lake is 6 and 2 km, respectively). This site is relatively protected from direct human impacts. The waste from domestic animals and household disposals, and other related agricultural byproducts cannot easily reach the center of the lake especially in the dry season where no flood carries the waste matter from the watershed. Hence, the site is relatively protected from human impact.

Water samples were collected in opaque 1L plastic bottles from the lake surface and it chilled in the icebox on site and transported to the Limnology Laboratory of Addis Ababa University for chemical analysis. The samples for physicochemical parameters were collected and, *in-situ* measurements were recorded in the early mornings between 8:00 and 11:00 am.

2.2.2 In-situ and laboratory measurements

In-situ measurements of temperature, dissolved oxygen, electrical conductivity, and pH were measured using a portable multimeter (Model HQ 40d Multi Hach Lange) at different depths within the euphotic area. TDS was calculated from the relationship of electrical conductivity for compatibility using the formula TDS=0.6 EC (Glenn, 2005). Turbidity of the lake water was measured using a portable digital turbidimeter (Model Oakton: T-100). Water transparency was measured using a standard Secchi Disc (28 cm in diameter with alternating black and white quarters). The euphotic depth (photosynthetic depth), the depth of 1% of the surface photosynthetic rate was determined using the relationship between euphotic zone (Zeu) and vertical light Extinction coefficient (K_d) as $Z_{eu} = 4.6/K_d$ (Kalff, 2002). Extinction coefficient (K_d) was found on a Secchi disk (Z_{SD}) relationship of Holmes (2000) formula as $K_d=1.44/Z_{SD}$.

In the Laboratory, total alkalinity was determined from 100ml of the unfiltered water sample taken from the surface by titration with 0.1N HCl with bromocresol green used as endpoint indicator (Wetzel and Likens, 2000). Chlorophyll content (Chl-*a*) of the water was determined spectrophotometrically (Talling and Driver, 1963). - 100 ml of water sample was filtered using Whatman GF/F glass microfiber filters (Diameter: 47 mm; Model PB-1825-047-BR) and the filters were deep-frozen overnight to facilitate extraction. The filters were then extracted in 90% acetone. Chl-*a* was determined spectrophotometrically at 665nm and 750nm after centrifugation. The major dissolved inorganic nutrients (soluble reactive phosphorus-SRP, total phosphorous-TP, dissolved silicate-SiO_2, nitrite-NO_2, nitrate-NO_3, and ammonium-NH_4) were determined using the standard method of APHA, 1995 (Table 2.1).

Table 2.1 Standard method for determining inorganic nutrients (after APHA, 1995)

Nutrients	Method
SRP	Ascorbic Acid
TP	Unfiltered water digested using potassium–peroxodisulphate, and autoclaved at 120 0C for 50 minutes then follow SRP procedure
SiO_2	Molybdosilicate
NO_2	The reaction between sulfanilamide and N naphthyl-(1)-ethylenediamine dihydrochloride
NO_3	Sodium salicylate
NH_4	Indo-phenol Blue

2.2.3 Trophic State Determination

The Trophic state of LTA was determined using Carlson (1977) trophic status index (TSI) determination method for an inland water body, which was calculated based on Secchi disk transparency (SD), chlorophyll-*a* content (CHLA), and concentration of total phosphorus (TP).

Carlson (1977) trophic state equation:

$$TSI\ SD = 60-14.41\ (\ln SD) \qquad \text{Equation 2.1}$$

$$TSI\ TP = 14.41\ \ln(TP) + 4.15 \qquad \text{Equation 2.2}$$

$$\text{TSI CHL}a = (9.8)(\ln \text{CHL}A) + 30.6 \quad \text{......} \text{Equation 2.3}$$

$$\text{CTSI (Average)} = (\text{TSI-SD} + \text{TSI-TP} + \text{TSI-CHLA})/3 \quad \text{......} \text{Equation 2.4}$$

Where: TSI stands for trophic state index, CTSI for Carlson's trophic state index, ln for natural logarithm, Z_{SD} for Secchi Depth (meter), TP for total phosphorous (μgL^{-1}), and CHL*a* for chlorophyll-*a* (μgL^{-1}). From this equation, Carlson's estimated the trophic state values ranging for Oligotrophic lakes (TSI, <40), Mesotrophic (TSI, 40-50), Eutrophic (TSI, 50-70), and Hypereutrophic (TSI, >70) states (Table 2.6).

2.2.4 Statistical analysis

To determine the relationship between the physicochemical parameters, the Pearson Correlation ‘'r” was used. Analysis of variance (ANOVA) was used to analyze the significant difference of environmental parameters among the study months or between seasons. Since there are only two study sites, t-test was used to check the presence or absence of spatial variations among the different variables. SPSS version 20 software was used for ANOVA and t-test analysis. The significance of the variables was set at 95% confidence interval (p=0.05).

2.3 RESULTS

2.3.1 Physicochemical features

Temperature and Dissolved Oxygen: In the study lake, as expected from a small-sized and shallow lake with frequent occurrence of complete mixing, there was little vertical difference in most of the physicochemical parameters (t-test; $p > 0.05$). However, there were notable temporal variations usually between the study months (ANOVA; $p<0.05$).

Marked temporal oscillation in surface temperature and dissolved oxygen was observed in the study months (Fig. 2.1). The value of surface water temperature ranged from 18.5 ^{0}C to 27^{0}C (mean: 23.08 ^{0}C±3.0 ^{0}C) at the open water site and 18.5^{0}C to 29.2^{0}C (mean: 23.23 ±3.32 ^{0}C) at the shore site. The surface dissolved oxygen (DO) ranged from 5.85 mgL^{-1} to 15.1 mgL^{-1} (mean: 9.04±3.07 mgL^{-1}) and 6.1 mgL^{-1} to 11.62 mgL^{-1} (mean: 8.53±1.77 mgL^{-1}) at the open water and shore sites, respectively (Table 2.2). The level of surface DO was relatively high during the dry period particularly from January to March and then it showed an oscillation. DO decline from January to May and then fairly evenly distributed from June to December (Fig. 2.1). Both temperature and dissolved oxygen had not shown significant spatial variations between the two sampling sites ($p > 0.05$) (Fig. 2.1). The two important parameters (temperature and DO) correlated negatively (r= -0. 24) with each other, but not strongly.

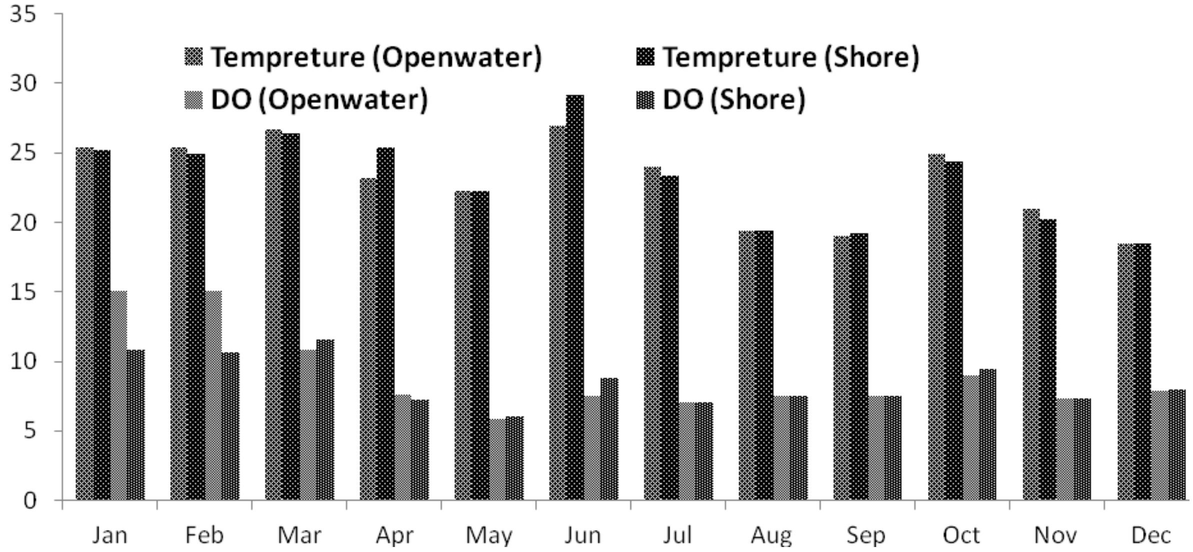


Fig. 2.1 Distributions of surface temperature (^{O}C) and dissolved oxygen (DO, mgL^{-1}) for LTA in the open water and shore areas (January -December 2016).

Table 2.2: Monthly physicochemical parameters (means and ranges) measured at the open-water and shore sites from January-December, 2016) for LTA. (SD is for standard deviation and See the details of the data in Table 2.3.

Parameters	Study site	Mean±SD	Range
Temperature (^{0}C)	Open-water	23.08±3.0	18.5-27
	Shore	23.23±3.32	18.5-29.2
TDS (mgL^{-1})	Openwater	273.3±53.62	117.7-653.9
	Shore	249±55.79	96-653.9
DO (mgL^{-1})	Openwater	9.04±3.07	5.58-15.1
	Shore	8.53±1.77	6.1-11.62
Chl-*a* (µgL^{-1})	Openwater	31.21±13.77	18.97-65.05
	Shore	26.30±13.30	12.13-48.77
Total alkalinity (meqL^{-1})	Open water	4.56±2.64	1.44-8.26
	Shore	3.89±1.99	0.96-6.94
Sechi Disk (cm)	Openwater	22.03±4.42	16-28.5
	Shore	14.22±1.40	12-16.5
K_d (m-^{1})	Openwater	6.78±3.58	5.05-9
	Shore	10.22±3.27	8.73-12
pH	Openwater	8.47± 0.39	8.11-9.27
	Shore	8.53 ± 0.35	8.15-9.22
Conductivity(µScm^{-1})	Openwater	420.53±352.34	181.1-1006
	Shore	384.53± 315.03	147.7-1006
Turbidity(NTU)	Openwater	111.5± 31.32	57-143
	Shore	135.42± 44.47	71-188

2.3.1.1 Depth profiles of temperature and dissolved oxygen

Temperature: The depth profiles of temperature and DO collected monthly at 0.25 m intervals are presented in Fig.2.2. The water temperature at the deepest depth (2.5-3m) at the open-water site ranged from at least 17.84 ^{0}C at 2.5 m in September to a maximum of 22.5 ^{0}C at 3 m in April. The temperature difference between surface and maximum depth (2.5-3m) at the open water site was 0 ^{0}C in August and 6.4 ^{0}C in February. The latter was representing the largest temperature difference between surface and the maximum depth of the column of the lake water at the open-water site.

In this lake, the temperature difference between 1 m intervals along the depth profile was above 1^0C in the measuring months from January to March indicating the existence of discontinuous warm polymictic nature of thermal stratification (Table 2.3). But the temperature gradient per 1m of depth was fairly even (< 1^0C) from November December and April to May) (Table 2.3). Depth profiles of temperature showed that similar temperature was recorded throughout the water column in August that was 19. 4^0C. A distinct thermal discontinuity has been observed where the thermocline extended up to 2 m during the period of January-March. It was shallower (up to 1 m) during the period of June, July, and December 2016. The water column below 2 at LTA represented the hypolimnion with a minimum temperature of 17.84^0C. Generally, LTA showed a short term and warm temperature stratifications in January, February, and March and mixes in the rainy periods (August-September) and early dry periods (April-May) (Table 2.3).

Dissolved oxygen (DO): From the study of depth profiles of dissolved oxygen of LTA, all depth profiles showed the maximum oxygen record in the upper layer of the water column and it declined with increasing depth. During the study period, the concentration of dissolved oxygen at the maximum depth (3 m) showed temporal variations from at least 4.84 mg O_2L^{-1} in August to a maximum of 7.32 mg O_2L^{-1} in April at the open water site. There was DO stratification in the first three sampling months of January, February, and March 2016. In these dry months, the upper DO was 15.1mgL^{-1} at the surface, which suddenly dropped to 6.0 mgL^{-1} at 2.5 m. But DO showed little variations in the water column in most of the sampling months (May to December 2016) (Table 2.3). Generally, the highest DO measurements were recorded between the surface and 0.5m depths in most of the sampling months, with the maximum recorded in January (15.1 mgL^{-1}) and February (15.09 mgL^{-1}) (Table 2.3).

Table 2.3 Depth profiles of some physical parameters recorded in the open water and shore areas in LTA during this study period (January-December, 2016).

Date	Depth (m)	DO (mgL^{-1})		Temperature (^{o}C)		pH		Conductivity (μScm^{-1})	
		Open	Shore	Open	Shore	Open	Shore	Open	Shore
January 2016	0	15.10	10.90	25.40	25.20	9.05	9.02	998.00	1006.00
	0.25	15.09	10.60	25.40	25.40	9.05	9.01	996.00	999.00
	0.5	11.9	7.50	24.30	23.60	9.26	8.99	940.00	992.00
	1	6.46	6.10	22.40	22.20	9.14	8.98	937.00	985.00
	1.5	6.09	5.30	22.00	21.60	9.07	8.86	937.00	943.00
	2	6.03		19.80		9.03		929.00	
	2.5	6.0		19.60		9.01		927.00	
February 2016	0	15.09	10.7	25.4	25.00	9.05	9.01	994.00	1002.00
	0.25	15.07	10.4	25.4	25.00	9.05	9.01	994.00	997.00
	0.5	11.8	7.3	24.1	24.00	9.24	8.99	938.00	991.00
	1	6.44	6.1	22.2	22.00	9.13	8.97	935.00	984.00
	1.5	6.08	5.3	22	22.00	9.07	8.84	936.00	941.00
	2	6.01		19.6		9.02		927.00	
	2.5	6.01		19.0		9.01		924.00	
March 2016	0	10.9	11.62	26.70	26.40	9.27	9.00	1006.00	615.00
	0.25	10.6	11.46	26.30	26.30	9.27	9.00	1003.00	590.00
	0.5	9.1	10.89	24.60	26.00	9.21	8.70	999.00	557.00
	1	8.87	9.05	24.40	25.20	9.00	8.60	990.00	518.00
	1.5	8.83	9.04	22.40	25.10	9.00	8.40	986.00	515.00
	2	8.75		21.70		8.94		979.00	
	2.5	8.01		20.40		8.57		963.00	
April 2016	0	7.66	7.25	23.20	25.40	8.81	8.46	424.00	392.00
	0.25	7.62	7.27	23.00	24.60	8.89	8.46	373.00	385.00
	0.5	7.68	7.31	22.70	24.60	9.13	8.54	387.00	386.00
	1	7.62	7.29	22.70	23.70	9.02	8.96	580.00	372.00
	1.5	7.63	7.29	22.70	23.30	9.03	8.89	397.00	373.00
	2	7.55	7.13	22.60	23.10	9.03	8.94	370.00	373.00
	2.5	7.52		22.60		9.09		390.00	
	3	7.32		22.50		9.00		370.00	
May 2016	0	5.85	6.10	22.30	22.30	8.30	8.55	224.00	231.00
	0.25	5.97	5.98	22.40	22.00	8.62	8.45	215.20	226.00
	0.5	5.77	5.98	22.40	22.00	8.62	8.43	232.00	225.00
	1	5.73	5.97	22.10	22.30	8.49	8.42	206.40	207.60
	1.5	5.71	5.95	21.00	22.20	8.52	8.36	210.40	206.40
	2	5.7	5.96	19.80	22.20	8.52	8.34	211.00	209.00
	2.5	5.7		19.60		8.40		203.40	
	3	5.7		19.10		8.22		203.40	
June 2016	0	7.54	8.81	27.00	29.20	8.11	8.49	232.00	239.00
	0.25	7.27	7.74	25.00	26.60	8.17	8.52	209.00	233.00
	0.5	7.06	7.14	23.50	25.00	8.19	8.50	209.30	213.40
	1	7.02	6.82	23.30	24.10	8.30	8.44	208.30	211.90
	1.5	6.88	6.78	23.10	23.30	8.51	8.42	208.10	211.40
	2	6.84	6.63	22.90	23.60	8.45	8.41	207.90	211.60
	2.5	6.82		22.80		8.48		207.40	
	3	6.63		22.30		8.4		205.50	

Date	Depth (m)	DO (mgL^{-1})		Temperature (^{o}C)		pH		Conductivity (μScm^{-1})	
		Open	Shore	Open	Shore	Open	Shore	Open	Shore
July 2016	0	7.07	7.09	24.00	23.40	8.26	8.25	199.70	201.20
	0.25	7.01	6.99	22.70	23.40	8.20	8.28	197.90	200.80
	0.5	6.91	6.99	22.10	23.30	8.10	8.23	194.00	199.30
	1	6.82	6.89	21.60	23.20	8.42	8.21	192.60	199.00
	1.5	6.75	6.61	21.40	22.70	8.37	8.11	192.30	193.60
	2	6.73	6.21	21.20	22.20	8.31	8.14	191.90	188.60
	2.5	6.73		21.20		8.25		191.50	
	3	6.36		21.20		8.18		191.20	
August 2016	0	7.51	7.52	19.40	19.40	7.69	7.64	183.30	149.90
	0.25	7.47	7.45	19.40	19.60	7.69	7.63	183.30	184.70
	0.5	7.48	7.42	19.40	19.60	7.69	7.63	183.20	184.70
	1	7.44	7.4	19.40	19.60	7.69	7.63	183.30	184.60
	1.5	7.45	7.38	19.40	19.60	7.69	7.63	183.30	184.50
	2	7.26	6.93	19.40	19.50	7.69	7.63	183.20	186.90
	2.5	6.7		19.40		7.68		183.20	
	3	4.84		19.40		7.68		177.46	
September 2016	0	7.51	7.51	19.10	19.20	8.25	8.33	181.10	147.70
	0.25	7.45	7.45	19.10	19.40	8.25	8.33	181.10	182.50
	0.5	7.42	7.42	18.70	19.40	8.25	8.23	181.20	182.50
	1	7.44	7.4	18.70	19.60	8.24	8.29	183.30	184.60
	1.5	6.88	6.78	18.10	18.30	8.15	8.20	178.10	189.20
	2	6.84	6.63	18.00	18.20	8.15	8.21	169.90	183.60
	2.5	6.70		17.84		8.15		163.20	
	3	6.70		17.84		8.15		163.20	
October 2016	0	9.05	9.51	25.00	24.40	8.49	8.81	213.30	217.50
	0.25	8.80	9.42	23.50	24.10	8.47	8.78	214.50	217.10
	0.5	7.26	7.03	22.10	23.60	8.35	8.74	206.20	207.40
	1	7.13	6.72	21.90	21.10	8.73	8.61	205.50	201.90
	1.5	7.11	6.42	21.80	20.80	8.71	8.66	205.20	200.70
	2	7.08		21.80		8.69		203.50	
	2.5	6.87		21.50		8.73		203.00	
	3	6.43		21.10		8.68		202.30	
November 2016	0	7.32	7.33	21.00	20.30	8.48	9.22	222.00	221.00
	0.25	7.36	7.27	20.90	20.60	8.75	9.16	222.00	221.00
	0.5	7.37	7.28	20.80	20.70	8.66	9.06	222.00	221.00
	1	7.28	7.25	20.90	20.70	8.77	8.93	222.00	221.00
	1.5	7.2	6.46	20.80	20.30	8.71	8.93	222.00	219.70
	2	6.99		20.50		8.7		220.00	
	2.5	6.93		19.80		8.54		217.90	
December 2016	0	7.92	7.96	18.50	18.50	8.74	8.46	223.00	222.00
	0.25	7.90	7.95	18.40	18.40	8.66	8.42	223.00	222.00
	0.5	7.91	7.95	18.40	18.40	8.68	8.84	223.00	221.00
	1	7.90	7.88	18.40	18.20	8.66	8.80	223.00	221.00
	1.5	7.90	7.78	18.40	18.10	8.66	8.71	223.00	220.00
	2	7.91		18.40		8.66		223.00	
	2.5	7.90		18.40		8.58		223.00	

pH and Alkalinity: The water of LTA was alkaline throughout the year. The pH ranged from 8.11-9.27 (mean: 8.47 ^{0}C± 0.39) at the open water site and 8.15-9.22 (mean: 8.53 ^{0}C± 0.35) at the shore site. Total alkalinity varied from 1.44 to 8.26 meqL^{-1} (mean: 4.58±2.64 meqL^{-1}) and 0.96 to 6.94 meq/L (mean: 3.89±1.99 meqL^{-1}) at the open water and shore sites, respectively (Table 2.2). The depth profiles of pH in LTA showed there was no meaningful difference down the water column. The maximum pH difference between the lower depth (2.5 m) and the surface (0 m) was only 0.7 in March. However, it was mostly higher in the upper area compared to the lower depth. pH ranged from at least 7.68 at 3 m in August to a maximum of 9.27 at the surface in March at the open water site and it ranged from 7.63 at 3 m in August to 9.02 at the surface in January in the shore area. In the study area, pH was above 7.0 at any measuring depth throughout the study months which tells the lake was alkaline.

PH showed seasonal variation with a significant difference in the sampling months (ANOVA; P<0.01). The high values of pH were seen during the dry period (January-May and October to December) (Fig. 2.2). The total alkalinity of the study lake also showed marked temporal fluctuations (ANOVA; P<0.05). It was high from January to April (dry period) and low from July to September (rainy period). pH and alkalinity correlated significantly and positively (r= 1). pH correlated positively and significantly with Secchi disk (r=0.583) but it correlated negatively with turbidity (r= -0.468).

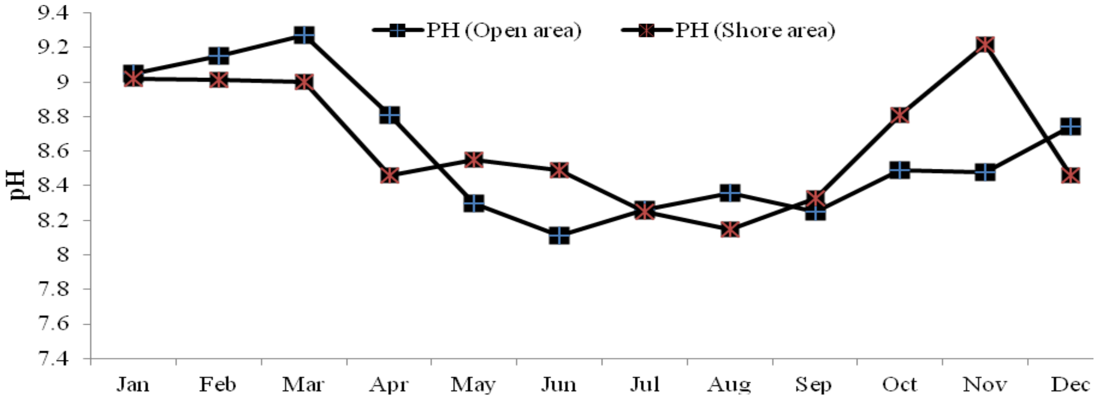


Fig. 2.2 Spatiotemporal variations of pH for LTA at the open water and shore areas (January - December 2016).

Electrical Conductivity (EC) and Total Dissolved Solids (TDS): In this study, fairly high readings of EC were reported. The electrical conductivity of surface water for the study lake varied from 181.1 to 1006 μScm^{-1} (mean: 420.53±352.34 μScm^{-1}) and 147.7 μScm^{-1} to 1006 μScm^{-1} (mean: 384.53±315.03 μScm^{-1}) at the open water and shore sites, respectively. There were no meaningful differences along the water column in the measurement of EC. In LTA, the depth profiles of electrical conductivity, generally, was evenly distributed down a column from May to December (Table 2.3). But surface electrical conductivity showed marked variations between the study months (ANOVA; $p < 0.05$). The higher values were recorded during the dry period (January to March) and it was sharply decreasing during the main rainy season (June-September). The peak value of conductivity was reported in March and January.

In this study lake, the total dissolved solids (TDS), the portion that passes through a filter, was, calculated from the EC readings of the water. The results of TDS of LTA varied from the low values of 117.7 mgL^{-1} to the high value of 653.9 mgL^{-1} (mean: 273.3 mgL^{-1}) at the open water site and 96 mgL^{-1} to 653.9 mgL^{-1} (mean: 249.9 mgL^{-1}) at the shore site (Table 2.2). High TDS measured from January to March (dry period) and sharply decreased from April to September. Low TDS values reported from June to September (rainy months). Conductivity was negatively correlated with turbidity ($r= -0.443$).

Turbidity and Transparency: LTA was very turbid throughout the year. Turbidity ranged from 57 to 143 NTU (mean: 111.5±31.32 NTU) and 71 to 188 NTU (mean: 135.42±44.47 NTU) at the open water and shore sites, respectively (Table 2.2). The water transparency of LTA varied from the lowest value of 16 cm to the highest value of 28.5 cm (mean: 22.03±4.42 cm) at the open-water site and 12cm to 16.5 cm (mean: 14.22±1.40) at the shore site (Table 2.2).

A marked temporal variation of turbidity was observed among the study months (ANOVA; $P < 0.05$). High values of turbidity recorded during the main rainy season (June and July) and low turbidity reported from January to April (dry period) (Fig. 2.3). Turbidity correlated positively and strongly with, NO_2 ($r= 0.577$) and TP ($r= 0.523$) while it correlated negatively with DO ($r= -0.510$), chlorophyll-*a* ($r= -0.515$), and conductivity ($r= -0.443$). Secchi disk showed significant differences between the two sites (t-test; $P< 0.01$) and different months (ANOVA; $P <05$). Low values of transparency were recorded during the rainy season (June -September) (Fig. 2.4) at both sites.

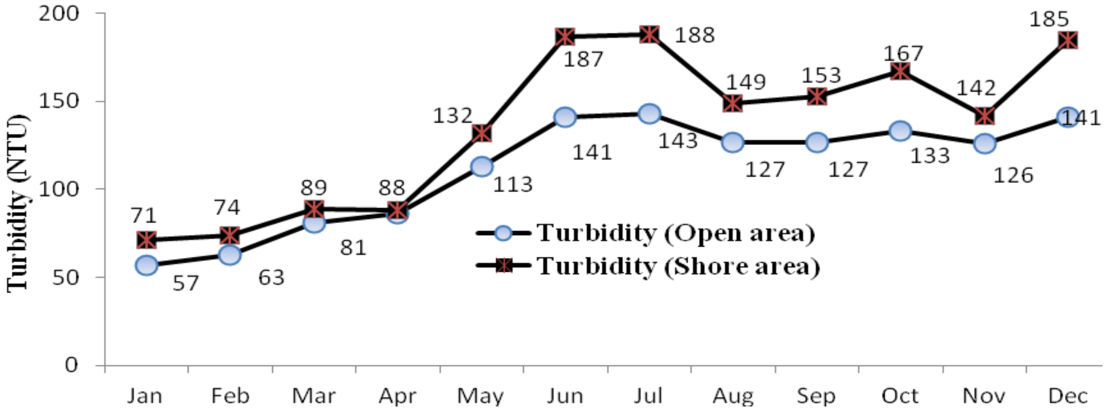


Fig. 2. 3 Temporal variations of the water turbidity (NTU) of LTA at the open water and shore areas (January -December 2016).

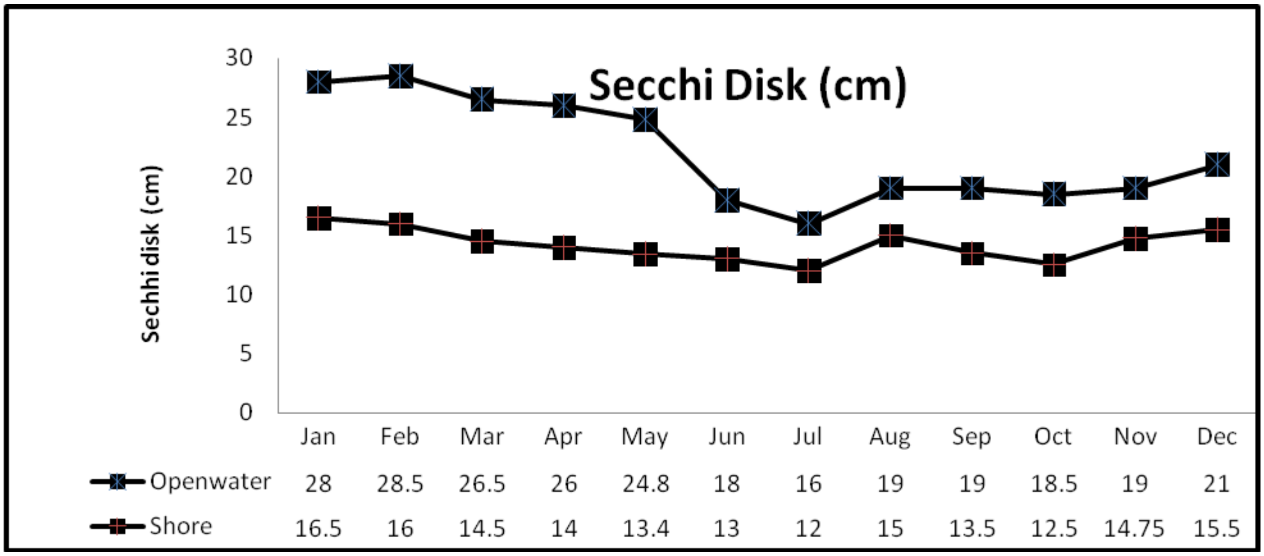


Fig. 2. 4 Spatiotemporal disstribution patterns in the water transparency (Secchi disk, cm) of LTA at the open-water and shore area (January -December 2016).

Euphotic area: In LTA, a shallow euphotic area was observed year round. The euphotic depth (Zeu), the depth at which 1% of the surface photosynthetic active radiation (PAR) was recorded ranged from 0.51m in July to 0.91m in February with a mean value of 0.7 m. The euphotic depth was deeper from January to May (dry period) and was relatively shallower from June to December (rainy season) (Fig. 2.5).

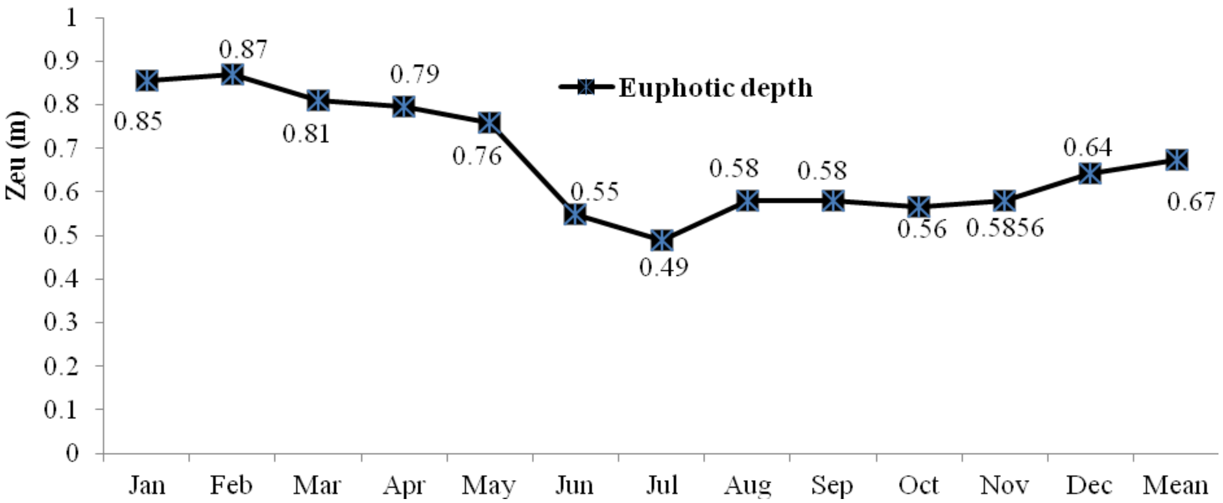


Fig. 2. 5 The Euphotic Depth (Zeu, M) of LTA at the open water area during the study period (January - December 2016)

2.3.2 Inorganic Nutrients

The major inorganic nutrients analyzed in the present study were nitrogen (nitrite-NO_2, nitrate-NO_3-N, and ammonium-NH_4-N), phosphorus (soluble reactive phosphorus-SRP and total phosphorus-TP), and dissolved silicate (SiO_2). The concentrations of each of the measured nutrients are found in Table (Table 2.4).

The lake was relatively rich with inorganic nutrients. The mean values were 503.8±427.8 μgL^{-1} for NO_2, 625.8±442.6 for NO_3, 192±55.8 for NH_4, 36.1±8.6 for SRP, 172.1±95.3 for TP and 62.8±53.9 for SiO_2 in samples from the open water and 196.9±11.6 for NO_2, 47.4± 13.9 for NO_3, 51.3±20.3 for NH_4, 42.3±11.6 for SRP, 184.4±92 for TP, and 62.5±51 for SiO_2 in samples from the shore.

The nitrite and ammonium showed a clear seasonality among the study months (ANOVA; $p<0.05$). Their high concentrations observed during the rainy time (June to September) and showed a reduction in most of the dry period (November to February) (Table 2.4). The reverse was true for the concentration of nitrate with its peak value in November (Table 2.4).

In LTA, phosphorus as soluble reactive phosphorus (SRP) and total phosphorus (TP) were investigated. The existence of the SRP and TP showed significant variations among study months (ANOVA; $p<0.05$). The peak values of SRP were seen from January to March (dry period). A clear oscillation was observed in the concentrations of TP in LTA. It increased from

May to July and reduced in August, then after it increased again from September to October. The lower value of TP was seen in January and February (dry period).

In LTA reasonably low dissolved silica was reported throughout the study period (Table 2.4). There were no meaningful differences in the concentration of dissolved silica between the study sites (t-test; $P<0.01$) (Table 2.4). However, there was a clear and remarkable fluctuation in the concentration of dissolved silica between months (ANOVA; $P< 0.05$). It was low in most of the dry months (January-May) and comparatively high from July-September (rainy month) and October-December (the beginning of the dry season). The peak SiO_2 values reported in November at both sampling sites. The relatively high concentration of SiO_2 (154.8 μgL^{-1} at the open-water site and 146.49 μgL^{-1} at shore site) also recorded in August (main rainy month) at both study sites (Table 2.4).

Table 2.4: Concentrations of Inorganic nutrients (μgL^{-1}) in LTA in the open water and shore areas during the study period (January-December, 2016).

Date	**NO_2-N(μg/L)**		**NO_3-N(μg/L)**		**NH_4-N(μg/L)**		**SiO_2(μg/L)**		**SRP(μg/L)**		**TP(μg/L)**	
	open water	**Shore**	**Open water**	**shore**	**open water**	**shore**	**open water**	**shore**	**open water**	**Shore**	**open water**	**Shore**
Jan	94.30	87.12	165.90	163.95	35.54	42.72	22.27	24.45	48.13	44.69	56.73	75.65
Feb	68.11	80.96	163.00	162.65	71.00	95.85	21.00	20.99	45.00	41.25	84.00	108.67
Mar	174.35	454.51	184.78	193.90	49.36	43.28	28.27	27.92	48.13	44.69	275.17	275.17
Apr	214.37	494.53	200.41	230.35	52.12	44.38	30.00	27.23	29.73	28.52	75.65	103.17
May	241.05	601.26	351.44	341.02	54.33	52.11	32.77	34.16	31.10	27.83	104.89	125.53
Jun	273.58	366.97	166.55	154.84	59.26	83.62	25.49	31.39	25.77	60.17	266.57	273.45
Jul	807.20	914.00	141.80	152.20	66.50	66.50	45.30	55.30	24.10	30.90	268.30	295.50
Aug	727.2	660.47	175.70	191.29	38.30	37.75	154.80	146.49	32.70	30.93	91.10	96.29
Sept	1234.10	1434.30	171.80	186.08	36.10	33.33	68.50	67.10	37.80	41.25	285.50	294.09
Oct	1314.18	1447.6	165.25	162.65	30.01	34.99	68.83	67.79	42.97	60.17	283.77	290.65
Nov	487.04	513.72	240.77	238.16	28.91	34.43	185.67	180.89	28.35	55.01	106.61	110.05
Dec	410.33	460.36	178.27	185.76	47.84	47.15	70.74	66.01	39.53	41.68	167.24	164.66
Mean	**503.8±427.8**	**625.8±442.6**	**192 ± 55.8**	**196.9±11.6**	**47.4± 13.9**	**51.3±20.3**	**62.8±53.9**	**62.5±51**	**36.1±8.6**	**42.3±11.6**	**172.1±95.3**	**184.4±92**

2.3.3 Trophic state

The trophic state of LTA was determined during the study (Table 2.5). Trophic state index (TSI) in terms of Secchi disk (TSI_{SD}), total phosphorus (TSI_{TP}) and chlorophyll-*a* (TSI_{CHLA}) ranged from 78.3-86.4 (mean: 82.04±8.1), 59.25-81.66 (mean: 72.34±22.41) and 59.56-71.56 (mean: 62.5±15) in samples from open water, respectively. The values were relatively higher in samples from shore area and it ranged 85.96-90.55 (mean: 88.17±4.56) for TSI_{SD}, 66.49-86.12 (mean: 77.56±19.63) for TSI_{TP}, and 55.06-67.91 (mean: 61.52±14.91) for TSI_{CHLA}. Carlson's Trophic State Index (CTSI-the mean of the three STI's) varied between 67.72 and 77.92 (mean: 72.31±10.2**)** at the open-water site and 71.23 and 78.9 (mean: 75.75±7.76) at the shore site (Table 2.5).

The results of TSI values in the present investigations strongly suggested that LTA can be classified as a hypertrophic system based on the mean values of TSI_{SD} and TSI_{TP} and eutrophic state based on the mean value of $TSI_{CHL}a$. Based on the average value of the three TSI's (CTSI), LTA is classified as a hypereutrophic state of the inland water body. In this lake, the trophic state index in terms of chlorophyll-*a* was lower than the trophic state indices of Secchi disk and total phosphorus indicating the hypertrophic state of the lake was not the result of algal turbidity but rather due to nutrient as a result of high sediment from the upper watershed of the lake.

There were seasonal fluctuations in the TSI of LTA. TSI_{TP} was high during the rainy time (June to September) and relatively low during most of the dry period (January to May). TSI_{SD} was high from May (post a dry month) to July (rainy period) and it low from August to September (rainy period). TSI_{SD} were above 70 (hypertrophic state) throughout the study period. There was a clear oscillation in the TSI_{CHLa}. It was progressively decreasing from January to August and then increased from September to December in 2016. The peak CTSI value was observed in March, September, June, and July. Except for December (TSI_{CHLa}=71.56), TSI_{CHLa} was found in between 50-70 (eutrophic state) all the other study months. CTSI was increased from January to March and sharply decreased in May and then it showed an increasing pattern from May to June with a pointed declined in August (Fig. 2.6). The pattern of CTSI showed a decreasing order starting from September to November and again an increasing towards December (Fig.2.6).

Table 2. 5 Trophic State Index (TSI) for LTA in terms of Secchi disk (SD), total phosphorous (TP) and chlorophyll a (CHLA) and the average of each TSI's (CTSI-Carlson's trophic state index) at open-water and shore sites during the study (January -December 2016).

TSI parameters	**Study site**	**Mean±SD**	**Range**
TSI_{SD}	Openwater	82.04±8.1	78.3-86.4
	Shore	88.17±4.56	85.96-90.55
TSI_{TP}	Openwater	72.34±22.41	59.25-81.66
	Shore	77.56±19.63	66.49-86.12
TSI_{CHLa}	Open water	62.5±15	59.56-71.56
	Shore	61.52±14.91	55.06 - 67.91
CTSI	Open water	72.31±10.2	67.72-77.92
	Shore	75. 75±7.67	71.23-78.9

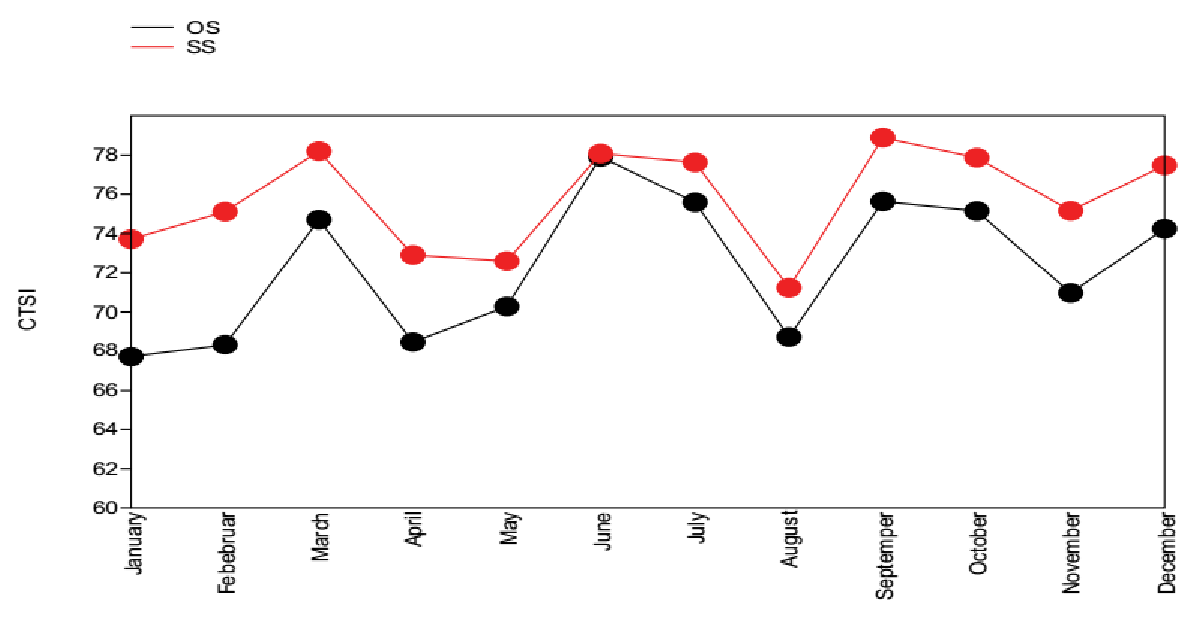


Fig. 2. 6 Distribution of Carlson trophic state index (CTSI) in LTA at open water site (OS) and shore site (SS) during the study period (January-December, 2016).

2.4 DISCUSSION

2.4.1 Physicochemical characteristics

Temperature and dissolved oxygen: In LTA, there is a relatively high surface temperature indicating the lake is fairly warm. The fairly high temperature in the study lake could be due to a rise in the air temperature due to climatic factors, excessive silt in the water that captures heat, humidity, solar radiation, and other related issues. In the study, a considerable surface temperature fluctuation was observed. This is possibly due to changes in weather conditions of the area. However, the low water temperature was recorded towards the end of the rainy period (August and September) likely due to the cold winds in the evenings and early morning after midnight, which is fairly regular in the general climatic conditions of Ethiopia. High surface temperature is recorded in most of the dry months owing to the high solar radiation, low water level, and clear atmosphere.

The surface temperature of LTA recorded during this study is comparable with other rift valley lakes of Ethiopia. It is closer to the freshwater lakes like Hawassa, with 20.5-28.4^{0}C (Demke Kifle, 1985), Ziway with 18.5-27.5^{0}C (Girma Tilahun, 1988), Langano with 18-27) (Kassahun Wodajo, 1982), and Chamo with 23-30^{0}C (Eyasu Shumbulo, 2004) and the saline Lake Abijata with 18-27^{0}C (Kassahun Wodajo, 1982). These temperature differences in the study area and the other lakes of Ethiopia are not unusual for tropical water systems and it may be most likely due to the weather condition of the area.

Temperature below 14°C and above 39.5°C is harmful to fish particularly to tropical species (Hossain *et al.,* 2012). The temperature of 25-30^{0}C is a normal temperature range of natural tropical water for the survival of aquatic life (Eyo *et al.,* 2008). In this regard, the upper temperature of LTA (27 ^{0}C at open-water site & 29.2 ^{0}C at shore site) falls within the normal temperature range of natural tropical waters and thus the lake water is considered suitable for aquatic lives.

Presence of dissolved oxygen in water may be due to direct diffusion from the air and photosynthetic activity of autotrophs (Jun *et al.,* 2014). In this study, the concentration of oxygen is relatively high suggesting that the lake is well oxygenated and it is suitable for the survival of most of the aquatic life as prescribed by WHO (2004, 2012) (Table 2.6).

In the study lake, temporal variations in dissolved oxygen have been observed. It has been low during the rainy period compared to the dry months. The presences of low DO during the main rain period (July-August) apparently due to the dissociation effect of temperature as the peak temperature was seen in July. Bachmann *et al.*, (2000) considered that whereas wind is a major oxygenator in large lakes, dissolved oxygen in smaller lakes is largely determined by the photosynthetic action of plankton, and because of the shallowness of the lake that allows frequent top-down mixing. A similar fact has been observed in LTA with dissolved oxygen well distributed in its water column. Beadle (1981) argued that the major determinant of circulation in tropical lakes is wind rather than the seasonal fluctuations of illumination and atmospheric temperature, which is the case with LTA.

Depth profile of temperature: the vertical temperature variation top to bottom in LTA is less than 1^0C in most of the sampling months (April-December) when measurements were taken. The decline in temperature by about 1 ^{0}C or more with increasing depth has been up to 2m depth in the months of January, February, and March. In the 0-1m depth layer, temperature gradients varied from 2.3^0C in March to 3^0C in January and in the 1-2m depth layer, temperature gradients varied from 2.6^0C in February to 2.9^0C in January and there is no temperature stratification observed below 2 m in all sampling months.

LTA shows a short term and warm temperature stratifies in the mid-dry periods (January-March) and mixes in the post-rainy periods (August-September) and early dry periods (April-May). As expected from the shallow and small-sized lake, persistent thermal stratification in LTA has not been observed throughout the study period. In addition, a true metalimnion was rarely formed. In shallow lakes (less than about 10 to 12 feet deep), a common feature of LTA, wind forces is usually strong enough to mix the water from top to bottom and thwart summer stratification (Thoma *et al.,* 1996). Lakes with a lot of water flowing through them (i.e., a short water residence time) also do not develop persistent thermal stratification. While a temperature gradient from the warmer surface to cooler bottom waters may exist in such lakes, a true metalimnion rarely is formed (Skowron, 2015). LTA is too shallow (3.7 m maximum depth) which mixes frequently and it is a polymictic lake. This may persuade the absence of deep-seated thermal stratification in most of the sampling months.

Stratification in very shallow lakes is generally rare since they have warm water mixing throughout their water column due to wind energy input (Santos *et al.,* 2015). However, cold water stratification can occur even in the shallowest lakes under the right climatic conditions (Thomas *et al.,* 1996; Boehrer and Schultze, 2008; Brook lemma, 2009; Santos *et al.,* 2015). The same plausible reason has been given for short term temperature differences between the surface and down the water column in the mid dry period (January-March 2016) in LTA. Polymictic lakes occur in regions of low seasonal temperature variations, subject to rapidly alternating winds and often with large daily (diurnal) temperature variations (Arnold *et al.,* 1992). These lakes have frequent periods of circulation and mixing and may be subdivided into cold polymictic, which circulate at temperatures close to 4 °C, and warm polymictic which circulate at higher temperatures (Zofia *et al.,* 2010).

As expected from the shallow and small-sized lake, persistent and deep-seated thermal stratification in LTA has not been observed during the study period. This is typical to shallow tropical lakes, where the narrow temperature differences between surface and deeper waters are lost due to cool breathes of air in the night, allowing frequent or sometimes daily complete mixing (Brook Lemma, 2009). Two types of thermal stratifications, superficial and deep-seated, were observed in East African lakes at altitudes of 1500-2000 m (Baxter *et al*., 1965). Superficial stratification is generated daily by solar heating and destroyed by nocturnal cooling and mixing, which affects the 0-3 m stratum in which steep gradients of temperature can develop during calm weather. According to Baxter *et al*. (1965) in lakes with maximum depth over 15-30m may show pronounced and deep seasonal stratification with complete or nearly complete mixing. But LTA is very shallow reasonably the absence of continues and persistent stratification in the lake.

Thermal stratification with thermoclines extending between 2 and 7 m in a Crater Lake Hora Kilole (Ethiopia) was observed in 1990-1991 (Brook Lemma, 1994); which was associated with the increased depth attained after diversion of River Mojo and the cone-shaped water basin, which prevented wind-induced mixing. In LTA similar thermal stratification was not observed even if the lake depth increased during the rainy season because of the high volume of runoff entering the lake trough feeder Rivers Dacha and Bobodo. However, this did not prevent the mixing as the lake is exposed to wind action due to its slumped shores and the small difference between top to bottom temperature differences lost in the night, making the lake vulnerable to frequent mixing.

Depth profiles of dissolved oxygen: Oxygen-maximum were observed between 0-0.5m depths in most of the sampling months (March, May-December) and at shallow depth near the surface (0.5m depth) in some other sampling months (January, February, and April). On the other hand, all depth profiles showed oxygen maximum in the upper layer of the water column and it declined with increasing depth.

In this lake, dissolved oxygen sharply and progressively decreased from the surface up to 2.5 m in the mid-dry months (January-March) (Fig. 2.2). For example, it was 15.1 mgL^{-1} at the surface and suddenly become 6.46 mgL^{-1} at 1m in January. The same trend is observed for the sampling months of February and March (Table 2.3). This is most likely linked with the temperature stability of the epilimnion. This is also possibly due to the mixing of the surface water with atmospheric air and the photosynthetic effect of the phytoplankton in the upper layers of the water column. Phytoplankton population declines with depth, exposure to air is lost, and whatever dissolved oxygen available is taken up by bacterial action and respiration of other living things (Eyasu Shumbulo, 2004).

The observed oxygen-maximum at or near the surface associated with a superficial thermal stratification, which usually implies a steep temperature gradient in the uppermost 0-2m stratum during warm and calm weather, and this gradient is a barrier to turbulent mixing, resulting in *in-situ* accumulation of oxygen produced by photosynthesis (Baxter *et al*., 1965). From the study of oxygen depth profiles of Ethiopian shallow lakes Kuriftu (Zelalem Desalgn, 2015), and the Ethiopian rift valley lakes such as Chamo, Hawassa, and Ziway (Girma Tilahun, 2006) indicated that they showed oxygen maxima in the uppermost 0-5 m layer, as recorded in LTA.

Usually, the values of DO were low in the part of the water column between the surface and the depth of oxygen maxima (Table 2.3), a situation observed in L. Kilole (Brook Lemma, 1994). The decline in oxygen concentration near the water surface in some tropical lakes is accentuated by the influence of rising temperature on the saturation concentration (Baxter *et al*., 1965) and photoinhibition (Goldman *et al*, 1963; Melack, 1979a). It could also result from the effect of high light intensity on the photosynthetic generation of molecular oxygen as most depth profiles of photosynthesis determined in the present study exhibited depressed photosynthetic rates at the surface.

Abrupt drops in DO (Oxyclines) corresponded to the depth of thermocline following thermal stratification. Similarly, DO showed discontinuities between 0.5-1.5 m and between 3-4.5 m with complete deoxygenating below 5 meter in the saline L. Abijata (Elizabeth Kebede *et al*., 1994) and also in crater L. Kilole (Brook Lemma, 1994), L. Bishoftu (Tadesse Ogato, 2007) and L. Koriftu (Zelalem Desalgn, 2007). In stratified Ethiopian lakes, Pawlo and Bishoftu, oxygen depletion was associated with the barrier of thermal gradients (Wood *et al*., 1984). This situation was not observed in LTA may be due to the shallowness of the lake.

pH and alkalinity: pH is an important parameter which determines the suitability of water for various purposes (Chaturvedi and Bassin, 2010). In this study, the pH values are always >7. This reasonably indicated that the water of LTA is alkaline and not acidic. There was not a significant pH difference in the depth profiles of the lake (Table 2.3). The pH of most natural water bodies ranges from 6.5 to 8.5 (WHO, 2006). BSI (2003) recommendation of pH is 6.5-8.5 for optimal survival of most aquatic life. The minimum values of pH (i.e., pH= 8.11 at the open-water site and 8.15 at shore site) are within the maximum permissible limits of (WHO, 2006) for recreation, fish production and for the survival of other aquatic life (6.5-8.5 or 9.0).

The increase in pH values in aquatic systems such as LTA is associated with the daily cycle of photosynthetic activity, where by day the peak hours of solar radiation enhances photosynthesis followed by a rise in pH, leading to a more alkaline situation than earlier in the morning. The decrease pH values during the period of the main rainy season owing to dilution caused by the rainwater and may result from the production of CO_2 from biological oxidation process (Tadesse Ogato, 2007) and may have contributed to the reduction of pH.

In LTA, there is fairly high total alkalinity; with the recorded highest value was 8.24 meqL^{-1} at open-water and at a value of 6.81 meqL^{-1} at the shore site. The expected total alkalinity in nature is 0.4 to 40 meqL^{-1} (Lind, 1979). As the highest total alkalinity of LTA is found within the range of the expected maximum boundary for the productive lake (Lind 1979), it is considered fairly productive.

Electrical conductivity (EC) and total dissolved solids (TDS): In the present study, high EC has been recorded during the dry period compared to the rainy months. The decline of conductivity in the rainy season may be due to the dilution of water because of increasing water level via the feeder rivers (Dacha and Boboda). EC was to some extent higher at the surfaces of the lake compared to the lower depth; this is probably due to higher surface water temperature that might have lent itself to higher chemical activity (Brook Lemma, 2009).

EC value of <1000 μScm^{-1} as recommended by FAO (2006) suggests that the water of LTA is suitable for crop and livestock agriculture. The mean conductivity of LTA (420.53µS/cm) is within the maximum permissible limit FAO guideline for livestock and poultry. However, the mean value of conductivity obtained in LTA is above the recommended value of 300 μScm^{-1} by WHO for drinking.

TDS is a term used to describe the inorganic salts and small amounts of organic matter present in solution in water. TDS in water supplies originate from natural sources, sewage, urban and agricultural run-offs, and industrial wastewaters (WHO, 1996). In this study, TDS was estimated from EC.

Water with total ionic concentration < 3000 mgL^{-1} is fresh. The highest value of TDS of LTA is 659.9 mgL^{-1}. This makes it easy to see LTA is a freshwater system. According to FAO, the acceptable range of TDS for livestock drinking is 100-1500 mgL^{-1} and TDS levels above 1200 mgL^{-1} are unacceptable to most consumers (WHO, 2003). LTA can be recommended for livestock drinking and most other consumers. Elevated TDS hurts the palatability of water. As WHO (2003) stated, the palatability of drinking water has been rated by panels of tasters in relation to its TDS level in Table 2.7 below.

However, water with low concentrations of TDS may be unacceptable because of its flat and insipid taste. In addition, a high level of TDS in drinking water (greater than 1200 mgL-1) is also objectionable to consumers (WHO, 2005, 2008). Based on this, the water of LTA is categorized as fair in its palatability of taste for drinking. But, for drinking, the maximum concentration level set by EPA and WHO is 500 mgL^{-1}. The highest value of TDS for LTA (659.9 mgL^{-1}) is above the recommended value for drinking.

Table 2. 6 The recommended standard of some of the physicochemical parameters for drinking purpose as per the respective recommending agencies and the values of LTA (January-December, 2016).

Physicochemical Parameters	**LTA (present study)**		**Standard values for drinking**	**Recommending agency**
	Open-water site	**Shore site**		
pH	8.11-9.27	8.15-9.22	6.5-8.5	BIS
DO (mgL^{-1})	9.04±3.07	9.53±1.77	5	WHO
Conductivity (µScm^{-1})	420.53±352.34	384.53± 315.03	300	WHO
Nitrate-NO$_3^-$ (mgL^{-1})	0.192 ± 0.05	0.197±0.012	45	BIS
Phosphate (mgL^{-1})	0.172±0.095	0.184±0.092	0.4	WHO
Turbidity (NTU)	111.5±31.32	135.42±44.47	5	WHO
Total alkalinity (mgL^{-1})	228±132	194.5±99.5	120	WHO
Temperature (^{0}C)	23.08±3.0	23.23±3.32	40	BIS

Table 2. 7: Water quality rating by WHO (2003)

TDS rank	**TDS rating**	**TDS values**
1	Excellent	< 300 mgL^{-1}
2	Good	300 and 600 mgL^{-1}
3	Fair	600 and 900 mgL^{-1}
4	Poor	900 and 1200 mgL^{-1}
5	Unacceptable	> 1200 mgL^{-1}

Water clarity, turbidity, and euphotic area: LTA is turbid with low water clarity and shallow euphotic area. Turbidity was at its peak during the main rainy seasons (June to September). The maximum turbidity recorded in July (rainy period) (188 NTU) in the open water was nearly 2.5 times greater than the minimum value measured in January (dry period) (57 NTU).

The higher turbidity of LTA could result from the high amount of silt brought by runoffs from degraded watershed via the feeder Rivers Bobodo and Dacha and other temporal floods, domestic wastes from animal watering, bathing, and other related activities like resuspended sediments from the bottom of the lake during stratification.

Turbidity in drinking water is caused by particulate matter that may be present from source water due to inadequate filtration or from re-suspension of sediment in the distribution system (Theofanis *et al.*, 2009). According to the European Union Drinking water directives (EU DWD, 2019); drinking water turbidity value must not exceed 1.0 NTU in the water treatment works.

The appearance of water with a turbidity of less than 5 NTU is usually acceptable to consumers, although this may vary with local circumstances. The turbidity of LTA water at the open-water site (71-188 NTU) and shore site (57-143) is indicated that it is much higher than the recommended value for drinking purpose. Thus the LTA water is unfit for human consumption.

However, people in the nearby lake used the lake water for drinking purpose with no form of treatment. This may cause a health risk, particularly for waterborne diseases. Excessive turbidity, or cloudiness, in potable water is aesthetically unappealing, and may also represent a health concern. Turbidity can provide food and shelter for pathogens. If not removed, turbidity can promote re-growth of pathogens in water distribution systems, leading to waterborne disease outbreaks, which have caused significant cases of gastroenteritis throughout the world (Theofanis *et al.*, 2009). A similar circumstance has been given for LTA; however, further waterborne disease prevalence study should be done to give a tangible reason for this prediction.

LTA had low water clarity. The significant lower water transparency is most likely due to wind-induced re-suspension of bottom sediments and causes the large turbulence and sediment instability which reduces the depth of light penetration. During the period of sampling, a lot of human activities including swimming, washing clothes, watering animals, dumping of wastes, and different agricultural activities were observed at the shoreline of the lake. These activities are likely to contribute and cause re-suspension of particles, and hence low water clarity of the study lake.

Generally, shallow lakes, like the case of LTA, exhibits low water clarity (Euphotic depth < 1 m) are regarded as a productive ecosystem (Wetzel, 1990; Eyasu Shumbulo, 2004). The water clarity of LTA is low and its euphotic depth is less than 1m year round. This is highly indicating thta the lake water is considered as a reasonably productive aquatic resource.

2.4.2 Inorganic Nutrients

The major inorganic nutrients (NO_3-N + NO_2-N + NH_4-N, TP and SRP) measured in this study lake were relatively high (Table 2.8). Higher concentrations of NO_3-N + NO_2-N + NH_4-N (all are nitrogen families) in LTA are reported compared to other nearby rift valley lakes of Ethiopia such as, L. Ziway (Girma Tilahun, 1988; 2006), L. Hawassa (Elizabeth Kebede and Amha Belay, 1994; Girma Tilahun, 2006), and L. Chamo (Girma Tilahun, 2006). This significant concentration of nitrogen nutrients in LTA seems to indicate that the lake water is supportive for the survival of most of the aquatic life. The high concentrations of inorganic nutrients in LTA may be associated with the presence of higher external loading of nitrogen which most probably originated from domestic sewage dumped and agriculture.

Basically, all the nutrients including the nitrogen families were relatively high during the rainy season. This is because of the possible influx of nitrogen-rich flood water into the lake water from the farm fields in the watershed and the sewage released from settlements. Domestic wastes, animal droppings, and other organic matters that reach the lake water through runoff during the rainy period, especially via River Dacha (Appendix 5), increase the level of nitrogen compounds in the lake. It is a known fact that nutrients of nitrogen families can become important in waters receiving agricultural runoff, wastewater through runoff, failure on-site septic systems, runoff from animal manure storage areas, and industrial discharges that contain corrosion inhibitors (USEPA, 1991).

Nitrate is one of the most important nutrients in any ecosystem (WHO and FAO, 2004). High nitrate content (>1 mgL^{-1}) is not conducive to aquatic life (Samuel *et al.*, 2008). Ayers and Westcott (1985) stated that the required maximum concentration of NO_3^- for livestock and irrigation were 100 and 30 mgL^{-1}, respectively and the concentration below 5 mgL^{-1} will not affect flora. The concentration of nitrate in LTA is within the permissible limits for flora, livestock, and irrigation. In most lakes, the concentration of NO_3-N is usually from 4-1500 μgL^{-1}. But in polluted lakes, the level extends to over 5000 μgL^{-1} (Brönmark and Hansson, 2002) and. in unpolluted waters, the level of nitrate-nitrogen is less than 0.1 mgL^{-1}. In the present study, the maximum values of nitrate obtained at the open water site were 0.353 mgL^{-1} and at the shore, the site was 0.341 mgL^{-1} which are all below the WHO recommended standards of 45 mgL^{-1} for drinking water.

Ammonium concentrations are lower than nitrate-nitrogen concentrations in most productive lakes (Wetzel, 2001). The concentration of NH_4 in well-oxygenated water is also usually low relative to other forms of inorganic nitrogen due to its ability to be readily oxidized and its rapid and preferential uptake by phytoplankton (Kalff, 2002). A similar phenomenon was observed in LTA in which ammonium nitrogen is lower than other forms of nitrogen nutrients because the lake is well oxygenated. Prochazcova *et al.*, (1970) in his study on fresh waters also observed phytoplankton preference for NH_4-N over NO_3-N even when the concentration of NO_3-N exceeds that of NH_4-N by sixfold. Conway (1977) has also demonstrated the inhibitory effect of NH_4 on the uptake of nitrate than ammonium by phytoplankton.

Table 2. 8 Mean and range value of major inorganic nutrients (µgL^{-1}) of LTA at open water and shore sites during the study period (January-December, 2016).

Nutrients (µgL^{-1})	**Site**	**Mean±SD**	**Range**
NO_2	Open-water	503.82±427.77	68.11-1314.18
	Shore	626.32±442.56	80.98-1447.6
NO_3	Open-water	192.14± 55.75	141.8-351.44
	Shore	42.26±11.57	152-341.02
NH_4	Open-water	47.44± 13.92	28.91-71
	Shore	51.34 ±20.27	33.33-95.85
SRP	Open-water	36.11±8.63	24.1-48.13
	Shore	42.26±11.57	27.83-60.17
TP	Open-water	172.13± 95.3	56.73-285.5
	Shore	184.41± 92.02	75.65-294.5
SiO_2	Open-water	62.80±53.89	21-185.67
	Shore	62.48± 50.96	20.99-180.89

Lagus *et al.*, (2007) studied impacts of nutrient enrichment and sediment on phytoplankton community structure in the northern Baltic Sea. Their results showed obvious changes in phytoplankton composition in response to adding both NO_3-N and PO_4-P. When nitrogen was the limiting factor, the dominant group was cyanobacteria, followed by Bacillariophyta. The dominant phytoplankton group of LTA is cyanobacteria followed by Bacillariophyta (See Chapter 3). Thus in LTA, Nitrogen is most probably the limiting nutrient compared to the phosphorous one. However, nutrients, in general, are in sufficient amounts in LTA, if not in excess.

In this study, a comparatively high level of phosphorous nutrients is documented (Table 2.8). This is presumably due to the different agricultural activities in the watershed area of the lake. Regeneration of phosphorus by zooplankton can be high enough to raise the ambient concentration to a level capable of supporting algal growth (Morris, 1980). Aquatic organisms play a great role either in depletion or enrichment of phosphate. Livingstone and Melack (1984) determined the excretory rate of zooplankton and found a rate of 100 mg m^{-2} d^{-1} for phosphate from some lakes of Sub-Sharan Africa. As described above for the N-nutrients, an appreciably high proportions of the phosphate compounds (SRP, organic phosphate and all included in TP) flow into the lake through the feeder rivers, runoff that washes down household and possible industrial wastes (most probably from Beer and Bottled water wastes from Gurage highland) into the lake.

Daily phosphate release by cladocerans zooplankton was 35-60% of their total body phosphorus and for rotifers values of 3-4 times higher (Lehman and Branstrator 1994). The fairly high level of phosphate (TP) in LTA is also probably a result of zooplankton excretion since their relative abundance is fairly high (See Chapter 5 for zooplankton abundance). The release of phosphate from anaerobic sediments and its subsequent transport to the epilimnion during mixing may also contribute to the high level of phosphate in the study lake (Mortimer, 1971).

As expected from the occurrence of complete mixing in lakes, the concentration of both SRP and TP in the present study has not shown marked spatial variations (Table 2.8). The slightly higher concentrations of phosphate at the shore site is probably associated with the greater exposure of the site to wind blowing and sediment runoff from the watershed close to the shore site and settlements. Most of the time phosphate concentrations in LTA are low in the months of high phytoplankton biomass (see also Girma Tilahun, 1988).

The maximum allowable concentration of phosphate for irrigation water is 2 mgL^{-1} (Ayers and Westcot, 1985). Most natural lakes (not affected by man) have phosphorus concentrations of between 1 and 100 μgL^{-1} (Brönmark and Hansson, 2002). According to the same authors, the observed values of the concentration of total phosphorus in LTA indicate that the lake is impaired by anthropogenic factors. The phosphorus concentration not greater than 300 μgL^{-1} showed the lake was not disturbed by anthropogenic factors (USEPA, 2005).

Total phosphate is mostly higher in hypertrophic lakes as compared to those of oligotrophic lakes in which the ratio of TN:TP is low, indicating nitrogen limitation, while TP is low for ultra/oligotrophic lakes and thus TN:TP is high, resulting in phosphorous limitation (Graciano *et al.*, 2006). The trophic state of LTA is hypertrophic. The hypertrophic state of LTA is apparently because of high phosphorus concentration other than algal boom. Thus it is probably safe to conclude that nitrogen is a more limiting factor than phosphorous in the study area.

The concentration of dissolved silica in LTA is, generally, low (Table 2.8). Over the last two decades, the Ethiopian Rift Valley lakes showed a decline in silicate concentrations (Zinabu Gebremariam *et al.*, 2002; Girma Tilahun, 2006; Zelalem Desalegn, 2007). The depletion of silica can be related to its removal from solution in diatom-dominated lakes like Shalla (Ethiopia) (Elizabeth Kebede and Willén, 1998) or its slower rate of regeneration resulting from the accumulation of organic matter as shown for alkaline lakes in Africa (Hecky and Kilham, 1973).

In L. Chamo, Amaha Belay and Wood (1984) found a relation between decreasing silica concentration and larger diatom growth and concluded severe depletion of silica associated with increasing diatom growth and lower dissociation rates of silicic acid from diatom frustules. It was also described from results recorded at Lakes Victoria (Talling and Talling, 1965) and Chad (Lemoalle, 1978) that silica depleted during the abundance of main abstractors, diatoms. The low value of silica in LTA might result from the above situations as diatom populations in this study lake are considered high (See Chapter 3).

LTA is small in size and polymictic nature and complete mixing for a prolonged period. Thus, nutrients including silica have probably been not locked up in the sediments by the phenomenon of ‘reverse weathering’, a major process in closed basins of Ethiopian lakes (Wood *et al*., 1984).

Even the greater input of the nutrients with runoff from the steep slopes watershed would contribute to silica inputs at shore site; which eventually dissipates into the pelagic zones of the lake.

2.4.3 Trophic State

The trophic state of LTA is determined using Calson's (1970) trophic the determination of inland water bodies. Accordingly, LTA can classify as classified as a hypereutrophic state. High concentration of phosphorus and low water clarity is the main factors for the occurrence of the high trophic stage of LTA. The hypertrophic state of LTA indicates there is a signal of eutrophication of the lake. Even if anoxic condition because of eutrophication due to algal bloom was not observed, the probability of its occurrence is high since high anthropogenic influences found in a nearshore site of the lake results in the load of nutrients into the lake.

TSI in terms of TSI_{PT} and TSI_{CHLa} of LTA is higher than the Ethiopian highland lake, L. Hayq which was categorized as eutrophic ($TSI_{CHL\ a}$= 55.7 and TSI_{TP}=63) (Tadesse Fetahi, 2010). The result of the average value of TSI for LTA is fairly comparable with L. Hawassa (TSI= 72.6; Adimasu Woldesenbet, 2015). Shallow lakes do not exceed three meters in depth and exhibit polymictic nature with diurnal stratification and nocturnal mixing usually have a high trophic state (Scheffer and Jeppesen, 2007; Grosman, 2008). This situation is observed in LTA.

The slightly high TSI at the shore site was probably associated with the greater exposure of the site that received runoffs and frequent human impacts. This record also corroborates with the relatively low water transparency throughout the study period in both sampling sites makes it the lake has high TSI_{SD} value.

The lowest phytoplankton biomass (See Chapter 4) has been observed during the rainy season (June- September) which resulted in a relatively low value of STI_{CHLa} in the rainy time. The heavy precipitation in the rainy season may have led to high volumes of surface runoff that brought particulate materials and nutrients into the lake with a consequent accumulation of suspended matter in the water with the reduction in light penetration and hence limiting photosynthesis. This may ultimately cause the occurrence of high TSI in LTA as a result of high water turbidity and nutrient concentration.

CHAPTER 3: SPECIES COMPOSITION, FRESH WEIGHT BIOMASS AND DISTRIBUTION PATTERNS OF PHYTOPLANKTON IN LAKE TINISHU ABAYA

3.1 Introduction

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). Roach (2004) and Falkowski and Raven (2007) argue that half of the world's oxygen is produced via phytoplankton photosynthesis and hence the global primary production.

Phytoplankton are important since they comprise the major portion of primary producers in the aquatic environment (Barupal and Gehlot, 2015). Temporal variability in the structure and function of the phytoplankton community is of fundamental importance to the metabolism of an aquatic system (Calijuri *et al*., 2002). Aquatic environments are subject to high temporal variability, with the frequent reorganization of the relative abundance and species composition of phytoplankton because of the interactions between physical, chemical, and biological variables (Duarte *et. al.*, 2006; Tadesse Fetahi *et al.*, 2014).

The study of changes in phytoplankton diversity, species composition, and spatiotemporal variability in water bodies is fundamental to the understanding of water quality (Girma Tilahun, 2006) and fisheries management (Falkowski and Raven, 2007). Knowledge of the spatial and temporal variations and community structure of phytoplankton is also fundamental to the understanding of ecosystem dynamics (Tadesse Fetahi *et al*., 2014). Therefore, this study investigated the temporal dynamics in the species composition and abundance of phytoplankton communities in relation to various physicochemical factors to observe the potential of the productivity of LTA, Ethiopia.

3.2 Material and Methods

3.2.1 Phytoplankton sampling and identification

Phytoplankton samplings for their quantitative analysis were carried out for a year monthly between January and December 2016 from two predefined sampling sites (in open-water and shore area) using bottle water sampler (Kemmerer: Model: A-S310 MONDSEE: Moosbachweg 10: 3 L capacity) within the euphotic area.

For qualitative analysis and identification of plankton species, nets (15 µm mesh size) samplers were also used. Composite samples were prepared by mixing equal proportions of water samples collected from the euphotic depths (0.25 m interval). From the composite water sample, 1L was taken and fixed it with Lugol's iodine solution. In the laboratory, the samples were homogenized and added to a 1-liter capacity graduated cylinders and kept in the dark for 24 hours to let the stained phytoplankton settle at the bottom overnight. After 24 hours, the upper 900 ml of the sample was siphoned out top-down and the remaining 100 ml was homogenized and poured into labeled clean bottles. For the identification of the phytoplankton so collected, samples were cleaned with cold H_2O_2 cleaning technique (Taylor *et al*., 2007).

From the homogenized sample, 1 mL sample was pipette into Sedgwick Rafter cell (1 mm^2) to identify the possible lowest taxonomic groups of phytoplankton using identification keys as of Whitford and Schumacher (1973), Gasse (1986), Komarek and Kling (1991), Cronber and Komarek (2004), Komárek and Anagnostidis (2000), and crosschecked to other relevant sources available in the web.

3.2.2 Phytoplankton counting and biovolume estimation

Individuals of each taxon of the phytoplankton were counted using a counting chamber (1000 cells with l ml in dimension). In the counting chamber, cells/individuals/colonies were counted randomly commonly 30 fields (a minimum of 400 individuals) under an inverted microscope (Motic/AE31) with a fixed camera at a magnification of 4-100X depending on the sizes of the individual phytoplankton.

The cell or individual number per ml was calculated according to Wetzel and Likens (2000). The volume of individual taxa was estimated by applying equivalent geometric shapes to cell forms by direct measurement of the cell dimensions (Hillebrand *et al*., 1999; Sun and Liu, 2003). After counting the number of individuals and volume of each individual, the biovolume (mm^3L^{-1}) of the dominant phytoplankton were calculated by multiplying the volume of individual phytoplankton and their number or abundance per liter.

3.2.3 Statistical Analysis

The relationships between the dominant phytoplankton taxa and environmental variables were analyzed using constrained Redundancy Analysis (RDA, CANOCO for Windows 4) using past software (Leps and Smilauer, 2003). Pearson correlation "r" was also used to check the correlation between the various physicochemical variables and the abundance of phytoplankton. Shannon Diversity Index (SDI) was applied to show the diversity of phytoplankton. Two-way ANOVA statistics were used to analyze the temporal and spatial variations of phytoplankton abundance. SPSS software package version 20 was used for ANOVA analysis. Statistical significance was set at P = 0.05.

3.3 RESULTS

3.3.1 Species composition of phytoplankton

During this study, 37 phytoplankton taxa belonging to 6 divisions including Bacillariophyceae (Diatoms), Chlorophyceae (Green algae), Cyanobacteria (Blue-green algae), Euglenophyceae (Euglenoids), Cryptophyceae, and Dianophyceae were recorded in LTA (Table 3.1). The species composition of phytoplankton showed Bacillariophyceae was the most diverse (15 taxa, 40.5%) group of phytoplankton followed by Chlorophyceae (11 taxa, 29.7%) and Cyanobacteria (7 taxa, 18.9%). Euglenophyceae (2 taxa, 5.4%), Dinophyceae (1 taxon, 2.7%), and Cryptophyceae (1 taxon, 2.7%) were represented by low species richness throughout the study period (Fig. 3.1). The Shannon diversity index (SDI) of each division of phytoplankton appreciably varied between the two sites. At the open water site, Bacillariophyceae (2.098) had the highest diversity, while at the shore site Chlorophyceae (2.161) took over the other taxa. This had shown that the two groups of phytoplankton (Bacillariophyceae and Chlorophyceae) were more diverse than other phytoplankton in the study lake.

Table 3.1: Phytoplankton taxa recorded in LTA (January-December, 2016)

Phytoplankton communities			
Bacillariophyceae	**Chlorophyceae**	**Cyanobacteria**	**Euglenophyceae**
Asterionella sp.	*Ankistrodesmus sp.*	*Anabaena sp.*	*Euglena sp.*
Aulacoseira sp.	*Botryococcus sp.*	*Cylindrospermopsis africana*	*Phacus sp.*
Cyclotella sp.	*Closterium sp.*	*Merismopedia sp.*	**Cryptophyceae**
Cymbella sp.	*Coelastrum sp.*	*Microcystis aeruginosa*	*Cryptomonas sp.*
Epithemia sp.	*Cosmarium sp.*	*Oscillatoria sp.*	**Dianophyceae**
Fragilaria sp.	*Oocystis sp.*	*Pseudoanabaena sp.*	*Peridinium sp.*
Gyrosigma sp.	*Pediastrum duplex*	*Spirulina sp.*	
Melosira sp.	*P. simplex*		
Navicula sp.	*Scenedesmus armatus*		
Nitzschia sp.	*Volvox sp.*		
Pinnularia sp.	*Zygnema sp.*		
Stauroneis sp.			
Surirella sp.			
Synedra ulna			
Thalassiosira sp.			

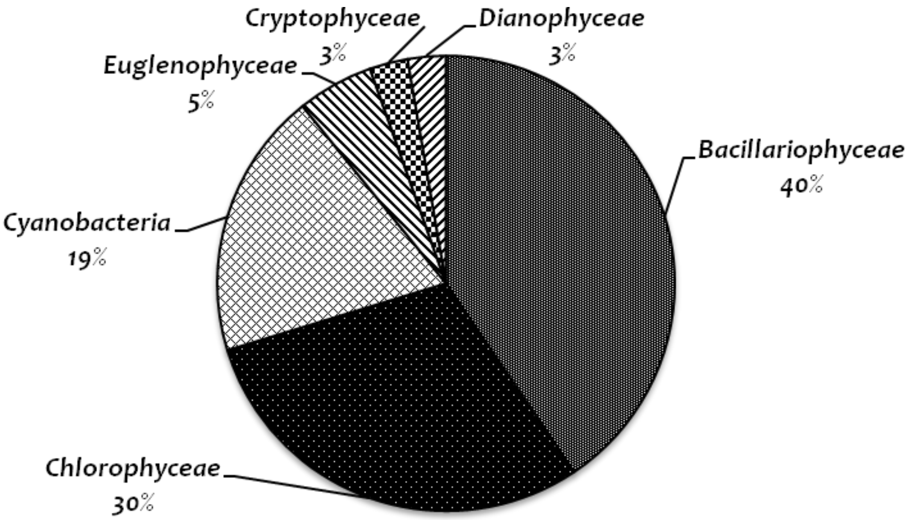


Fig. 3.1: The percentage composition in taxa richness of phytoplankton in LTA (January-December 2016).

3.3.2 Biovolume (abundance) of the major phytoplankton

In this study, the abundance of the dominant phytoplankton taxa was anticipated in terms of its fresh weight biomass considered as biovolume (mm^3L^{-1}). The total biovolume of phytoplankton in LTA ranged from the lowest value of 7.96 mm^3L^{-1} to the highest value of 315.29 mm^3L^{-1} (average= 117.19 mm^3L^{-1}) at the open-water site and 6.47 mm^3L^{-1} to 239.12 mm^3L^{-1} (average= 87.96 mm^3L^{-1}) at the shore site (Table 3.2). There was a temporal effect on the total biovolume of phytoplankton (ANOVA; $p < 0.05$). The biovolume record was higher during the rainy months than the dry months, which was concurrent with the high nutrient enrichment from runoff. It was lower during the dry season along with relatively low water clarity. Total biovolume of phytoplankton progressively decreased from January to April (dry period) and increased from May to October (inclusive of the wet season) with a sharp increase in August (the peak rainy month) at the open-water site and in December at the shore site (Fig. 3.2).

Microcystis aeruginosa was the most conspicuous and commonly occurring phytoplankton. But regarding the freshweight biomass, this species comprises only 10% of the total biovolume of phytoplankton. In the study area, about 75% of the total biovolume of phytoplankton constituted by the genera *Pediastrum, Anabaena, Cylindrospermosis, Nitzchia,* and *Perdinium.* The remaining 25% of the total biovolume of phytoplankton was shared by other taxa (Table 3.3).

The largest biovolume of phytoplankton constituted the cyanobacteria. This group of phytoplankton comprising about 54% and 34% of the total biovolume of phytoplankton at open-water and shore sites, respectively. Chlorophyceae and Bacillariophyceae comprised relatively high biovolume next to the Cyanophyceae. The former constituted 31% at the shore site and 12.8% at the open-water site. The latter comprising 22.3% and 22.9% at the open-water and shore sites, respectively. Euglenophyceae and Dianophyceae had comparatively low (10%) biovolume.

The calculated biovolume of cyanobacteria ranged from 63.43 to 201.07 mm^3L^{-1} (mean: 79.43±62.53 mm^3L^{-1}) and 0.5 to 99.44 mm^3L^{-1} (mean: 31.16±45.5 mm^3L^{-1}) at the open-water and shore sites, respectively (Table 3.2). Significant variations in the biovolume of cyanobacteria were observed among the study months ($p < 0.05$). The maximum biovolume of cyanobacteria was recorded in August at the open-water and in October and September at the shore site. Generally, a high abundance of cyanobacteria in LTA observed in the sampling months of August and September (rain months) and October, November, and December (dry months).

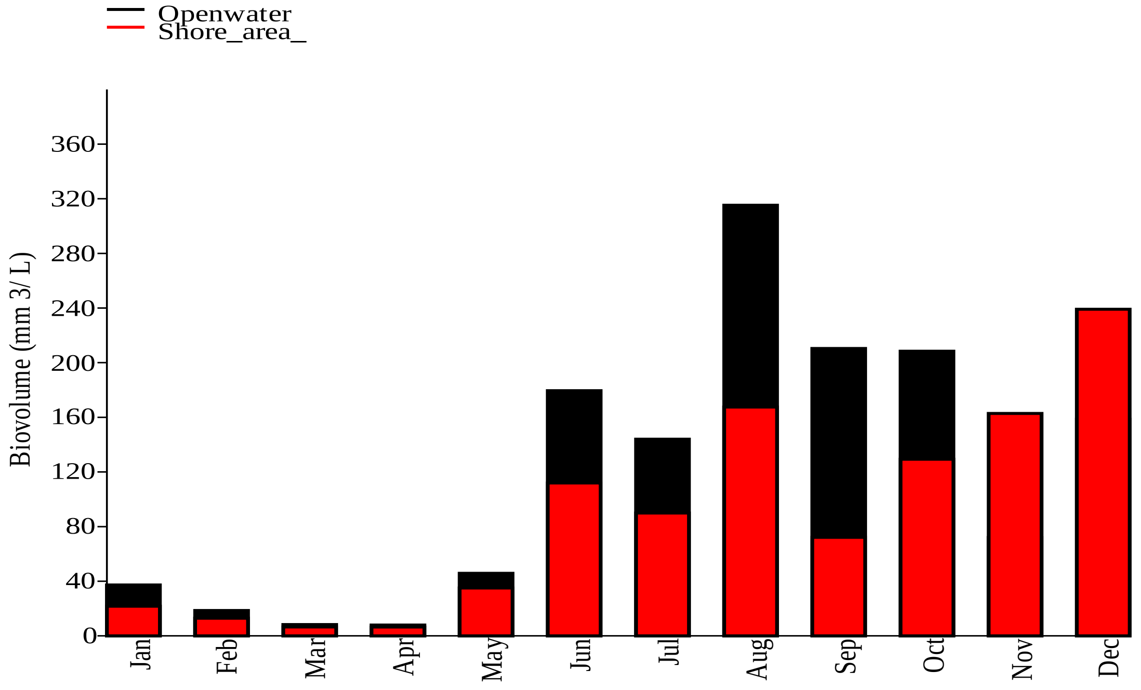


Fig. 3.2: Temporal patterns of the total biovolume (mm^3L^{-1}) of phytoplankton in LTA during the study period (January-December, 2016) in the open water and shore area.

Table 3.2: The biovolume (mm^3L^{-1}) of the major phytoplankton group for LTA (January-December, 2016) at open-water and shore sites.

Major phytoplankton	Site	Minimum	Maximum	Mean	Std. Dev
Chlorophyceae	Open-water	0.3015	89.3153	15.05906	24.20505
	shore	0.1873	170.1202	27.4825	56.30104
Cyanophyceae	Open-water	63.42692	201.0735	79.42982	62.35
	shore	0.503	99.4459	31.16089	34.49752
Bacillariophyceae	Open-water	0.83211	84.00768	26.12501	32.52048
	shore	0.58405	57.71488	20.19285	22.34142

Anabaena circinalis was co-dominant with *M. aeruginosa.* At the open water site, the maximum percentage (25%) of the total biovolume of phytoplankton was contributed by *Anabaena circinalis. Cylindrospermopsis africana, Pseudoanabaena*, and *Oscillatoria* were also occurring regularly. *Cylindrospermopsis* comparatively had a high percentage contribution (18%) at the open water site. *Pseudoanabaena* and *Oscillatoria* contributed insignificant amounts (< 2%) of the total biovolume of phytoplankton.

Seasonality of major taxa of the cyanobacteria was evident. The biovolume of *M. aeruginosa* was relatively high in September, October, and November (Fig. 3.3). Relatively, high biovolume of *Anabaena* was seen at mid of rainy season (Agust) and the beginning of the dry period (October). It was progressively declining from February to May (dry season) with a quick drop in April and May. The higher biovolume of *C. africana* was computed in September, (end of the rainy month) and October and November (beginning of dry months). It increased with time, from July to September (rainy period) and November to December (dry season). Though *Oscillatoria sp* conspicuously occurred, the species comprised insignificant amount (< 2%) for the total phytoplankton abundance.

Among the Cyanobacteria species found in the present study lake, *Spirulina sp.*, and *Merismopedia sp* occurred hardly. *Spirulina* reported only in January, February, and April. *Spirulina* was observed at the open-water sites in January and April and at the shore site in

February. *Merismopedia* did not occur from July to October at both sites. The two species contributed a negligible amount (< 0.5%) of the total biovolume of phytoplankton.

The biovolume of Chlorophyceae (green algae) ranged from 0.30 to 89.32 mm^3L^{-1} (mean: 15.1±24.2 mm^3L^{-1}) and 0.19 to 170.12 mm^3L^{-1} (mean: 27.48±56.3 mm^3L^{-1}) at the open-water and shore sites, respectively (Table 3.2). There were significant seasonal variations in the biovolume of Chlorophyceae ($p <0.05$). It was high in the dry period (January, February, March, December) and progressively decreased from January to April until it pointed low in May. The maximum biovolume of Chlorophyceae was seen in December (dry period) in the open-water site.

Pediastrum simplex, *Coelastrum*, *Cosmarium*, and *Botryococcus* sp. were the green algae which occurred regularly. However, the biovolume of green-algae was broadly dominated (about 80%) by a single species *Pediastrum simplex.* This species also consisted of a significant amount of biovolume (29.6%) for the total phytoplankton at the shore site. *Cosmarium sp.* contributed relatively high (12%) value of the total biovolume of Chlorophyceae next to *Pediastrum simplex.*

High biovolume of *Pediastrum* was seen mostly during the dry season (January to February and November to December). In June and July (rainy period) relatively high *Pediastrum* biovolume was also seen. The biovolume of *Coelastrum sp.* was high in July at both sites, whereas *Cosmarium* and *Botryococcus* species were at their peak in August (Fig. 3.3).

The contribution of Bacillariophyta (diatoms) to the total biovolume of phytoplankton was comparatively high (22%) next to the blue-green and the green algae. The value ranged from 0.83 to 84 mm^3L^{-1} (mean: 26.13±32.52 mm^3L^{-1}) and 0.58 to 57.72 mm^3L^{-1} (mean: 20.19±22.34 mm^3L^{-1}) at open-water and shore sites, respectively. The biovolume of diatoms in this lake showed significant temporal variation ($p <0.01$). Its high values were reported from June to August (rainy season) and low biovolume was reported from January to April (dry period).

Nitzschia, *Cymbella*, *Synedra*, *Frgilaria*, *Navicula*, and *Melosira* were the dominant taxa of diatoms that occurred commonly. Of whom, *Nitzschia sp.* was extensively dominated (85%) the biovolume of diatoms. *Nitzschia sp.* also contributed 19% of the total biovolume of phytoplankton, which is considered relatively high. *Synedra ulna* and *Cyclotella sp.* were co-dominant with *Nitzschia sp.* They together contributed a significant amount, about 10%, of the biovolume of the diatoms. The contribution of *Cymbella*, *Frgilaria*, *Navicula*, and *Melosira* for the total phytoplankton biovolume were almost negligible (<1.5%) (Table 3.3). The relatively low biovolume of *Synedra ulna* was seen from February to April (dry season) and August to September (wet season) while high biovolume was observed in June (wet) and May and December (dry period). *Cylotella sp.* was more conspicuous in December, and it was relatively high in July and August (wet period) and October and December (dry period).

The computed biovolume of Euglenophyceae showed the maximum values of 86.75 mm^{3}L^{-1} and 47.26 mm^{3}L^{-1} at open-water and shore sites, respectively. There was a temporal effect ($p < 0.01$) on distributing the biovolume of Euglenophyceae. It was high during the rainy season (June-September) and low in most of the dry season (January-April). From the Dianophyceace, *Perdinum* was the only species to be recorded throughout the study period and it has commonly occurred at shore site. This species contributed somewhat high percentage (10%) to the total biovolume of phytoplankton.

Table 3. 3 Percentage contribution (%) of the major phytoplankton taxa for the total biovolume of phytoplankton in LTA. **Note**: The superscript "a" represents the species which relatively contributed meaningfull values.

	Study site	
Major phytoplankton species	**Open-water**	**Shore**
Anabena circinalis	25.46^{a}	15.03^{a}
Cosmarium sp	1.58	1.02
Cyclotella sp	0.78	1.41
Cylindrospermopsis	17.59^{a}	8.03^{a}
Euglena sp	0.53	0.47
Melosira sp	0.16	0.17
Microcystis aeruginosa	8.68^{a}	9.99^{a}
Navicula sp	0.34	0.32
Nitzschia sp	19.00^{a}	18.46^{a}
Oscillatoria	1.20	1.81
Pediastrum duplex	0.29	0.33
Pediastrum simplex	10.63^{a}	29.59^{a}
Peridinium sp	10.20^{a}	9.88^{a}
Pseudoanabaena sp	1.19	0.56
Synedra ulna	1.76	2.30

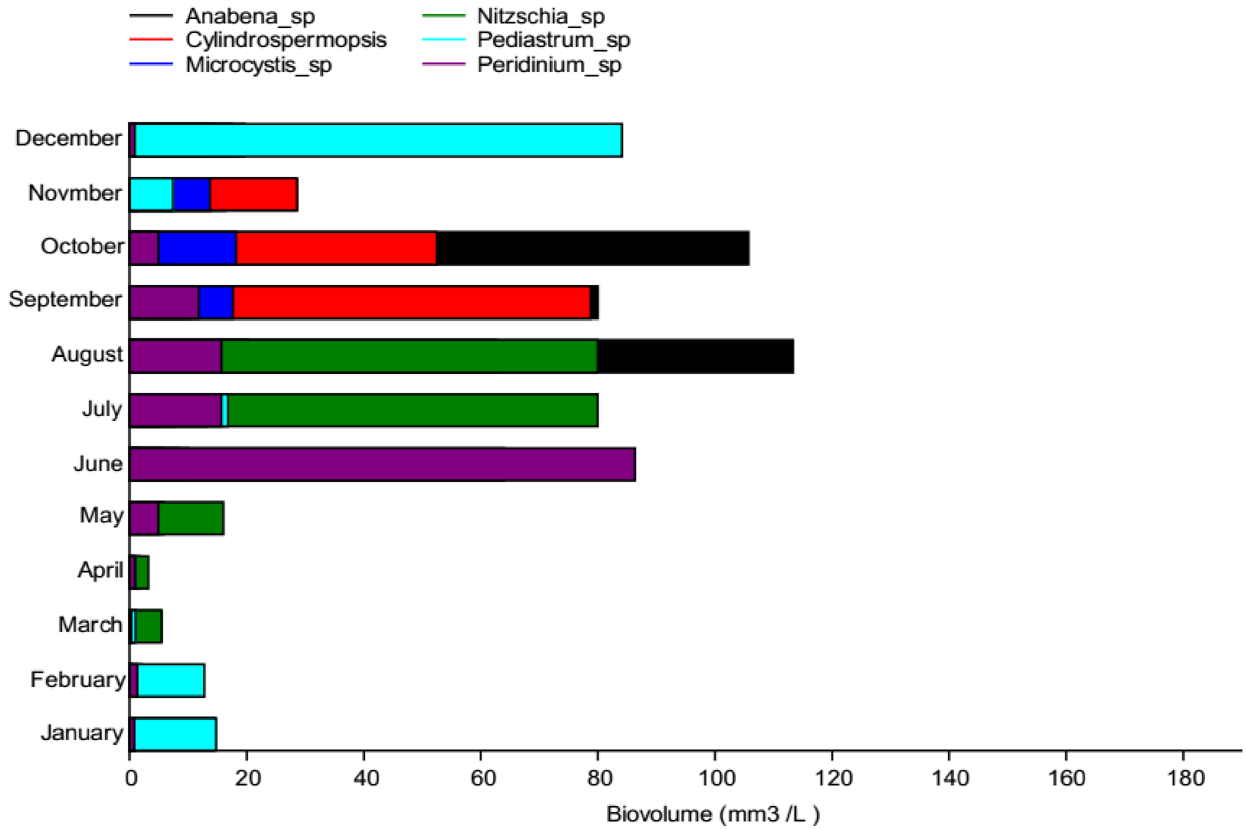


Fig. 3.3 Temporal patterns of the biovolume of major phytoplankton taxa in LTA in the open water (January-December, 2016).

3.3.4 Relationship between biovolume of dominant phytoplankton taxa and some physicochemical parameters

The correlation of environmental parameters and phytoplankton abundance was analyzed using Constrained Redundancy Analyses (CRA) graph (Fig. 3.4). The first and second axes together explained 79.20% of the cumulative percentage variance of species-environment relations (Table 3.4). The first axis explains 65.30% and the second axis explains 13.90% of species-environment relations (Table 3.6). The *M. aeruginosa* and *Coelastrum sp.* correlated positively and strongly with turbidity, NO_2, and TP while these dominant species correlated negatively and strongly with temperature, total alkalinity, pH, DO, Secchi disk, and conductivity. *Nitzschia sp.* correlated positively and strongly with turbidity, and most of the inorganic nutrients (NO2, NO_3, TP, NH_4). *Pediastrum sp.* correlated positively and strongly with SRP.

Table 3.4 Summary of the statistics of the DA diagram

Axes	**1**	**2**	**3**	**4**
Eigenvalues:	0.597	0.127	0.087	0.044
Species-environment correlations:	0.976	0.961	0.970	0.892
Cumulative percentage variance				
of species data :	59.7	72.4	81.1	85.5
of species-environment relation:	65.3	79.2	88.7	93.6
Sum of all eigenvalues	1.000			
Sum of all canonical eigenvalues	0.914			

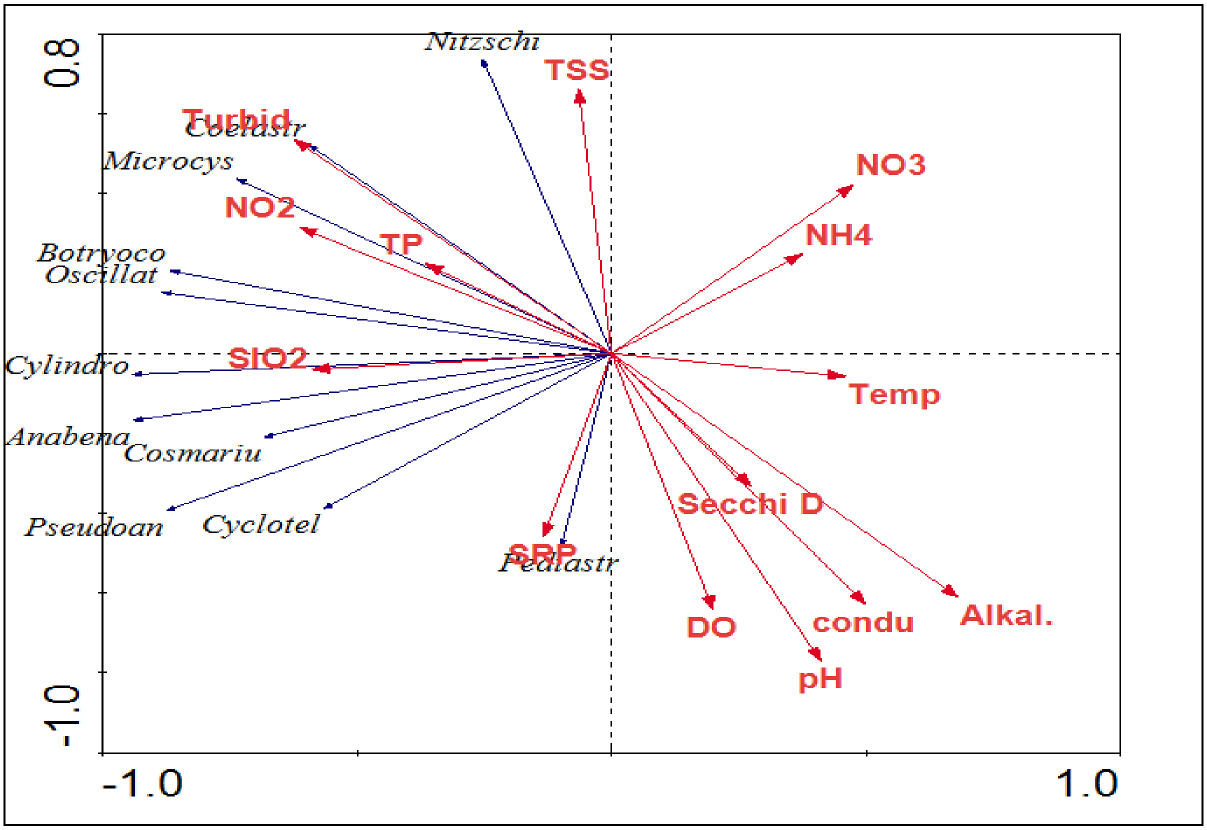


Fig. 3.4: Bi-plot of the Constrained Redundancy Analyses (CRA, CANOCO for Windows 4) for dominant phytoplankton species (blue arrows) and environmental variables(red arrows): Pediastr-*Pediastrum simplex*, Cyclotel-*Cyclotella* sp., Pseudoan-*Pseudoanabena* sp., Cosmariu-*Cosmarium* sp., Anabena-*Anabaena* sp., Cylindro *Cylindrospermopsis africana*, Oscillat-*Oscillatoria* sp., Botrycoco- *Botryococcus* sp., Microcys-*Microcystis aeruginosa*, Coelastr-*Coelastrum* sp, Nitzschi-*Nitzschia* sp.: Turbid-turbidity, Condu-conductivity, Secchi D-Secchi disk, Temp-temperature, Alkal-Alkalinity.

3.4 DISCUSSION

3.4.1 Phytoplankton communities

The phytoplankton taxa recorded in LTA (37 taxa) in this study were higher than those of the Ethiopian smaller crater Lakes Babogaya (32 species: Yeshiemebet Major, 2006), Bishoftu (22 species; Tadesse Ogato, 2007) and Kuriftu (25 species; Zelalem Desalgn, 2007). The number of phytoplankton taxa recorded in LTA relatively closer to the Ethiopian highland Lake Hayq (40 species: Tadesse Fetahi, 2010). However, the species composition of LTA is low compared to those of the nearby Ethiopian freshwater Rift Valley Lakes, L. Chamo (44 species; Elizabeth Kebede and Willen, 1998, 53 species; Girma Tilahun, 2006), L. Ziway (67 species; Elizabeth Kebede and Willen, 1998, 58 species; Girma Tilahun, 2006), and L. Hawassa (70 species; Elizabeth Kebede and Willen, 1998, 80 species; Girma Tilahun, 2006) (Table 3.5).

The difference in the number of phytoplankton taxa between LTA and other lakes of Ethiopia may be due to the difference in water chemistry, sampling time, the level of inorganic nutrients, turbidity of the lake water, and occurrence of frequently top-down complete mixing. It may also be associated with the interplay of physical (e.g. lake size, shoreline modification) and biological factors (e.g. vegetation covers and presence/absence of predators/grazers) and a greater variability of taxonomic literature on tropical algal flora (Girma Tilahun, 2006).

Table 3.5 Species richness of LTA and other Ethiopian lakes

Lake	# of species	Sources
LTA	**37**	**Present study**
Kuriftu	25	Tadesse Ogata (2007)
Bishoftu	22	Zelalem Desalgn (2007)
Babogaya	32	Yeshimebet Major (2006)
Hayq	40	Tadesse Fetahi (2010)
Chamo	44	Elizabeth Kebede and Willen (1998)
	53	Girma Tilahun (2006)
Ziway	67	Elizabeth Kebede and Willen (1998)
	58	Girma Tilahun (2006)
Hawassa	70	Elizabeth Kebede and Willen (1998)
	80	Girma Tilahun (2006)

Blue-green algae and green algae were species-rich taxonomic groups in most tropical and temperate lakes (Agusti *et al*., 1990). Green algae followed by blue-green algae have contributed the most to the total of species richness in Rift valley Lakes Ziway, Hawassa, and Chamo, Ethiopia (Girma Tilahun and Ahlgren, 2010). The diatoms followed by the green algae and blue-green algae contributed the highest number of taxa of phytoplankton in LTA as reported during the present study.

Recent research in Ethiopian lakes shows increasing dominance of blue-green algae because of phytoplankton groups suffering disproportionably higher losses, mainly by mortality (mostly grazing by macrozooplankton and herbivorous fishes) (Agusti *et al*. 1990). Cyanobacterial dominance is a common phenomenon in lakes and Reservoirs in Ethiopia (Elizabeth Kebede and Willen, 1998; Adane Sirage, 2006; Ayalew Wondie, 2006; Girma Tilahun, 2006; Yeshimebet Major, 2006; Tadesse Ogato, 2007; Zelalem Desalgn, 2007). The same situation has been observed in the present study because of the highest abundance of phytoplankton in terms of their growth or biovolume is contributed by the blue-green algae.

Rivers Dacha and Bobodo are the feeders for LTA. These rivers collect considerable quantities of nutrients from the catchment, in particular during the rainy seasons ending up in the lake and ultimately resulting in the growth of phytoplankton. As the dilution of the lake increases with high inflow and runoffs in the wet seasons, water level and nutrient concentration reach their seasonal peaks, creating a favorable environment for the proliferation many species of phytoplankton, particularly, the blue-green algae (Misgina Belachew, 2010; Redeit Abate *et al.*, 2017).

In LTA, phytoplankton abundance increases throughout the rainy time, reaching it's maxed at the start of the dry period. Various physicochemical parameters including major inorganic nutrients of water analyzed depicted a positive correlation with diversity and distribution of phytoplankton and support most of the phytoplankton development and growth. During the early dry season (January-April), most taxa of phytoplankton, basically the blue-green algae, and diatoms (the two dominant groups of phytoplankton), showed sharp decline following the decrease in water temperature and nutrient depletion. The correlation between the observed declines of phytoplankton with diminishing nutrient concentrations appears to be predominantly causal.

Temporal patterns in phytoplankton with some environmental aspects observed in Lake Malawi indicated that the highest abundance occurred in June-July (rainy) (related to cooling and vertical entrainment) and the lowest in January-March (dry), which was related to high surface temperature and pronounced stratification (Iles and 1960; Harding 1963). In LTA, the relatively high surface temperature recorded in June and July (See Chapter 2) may lead to the occurrence of high biovolume in the two rainy months. The water of LTA is very turbid throughout the year. The influence of turbidity on the abundance of phytoplankton may be compensated by increasing nutrient inputs through rivers inflows and the consequent lake water dilution.

The decline in phytoplankton abundance was observed towards the dry period (January to April) corresponding with the higher chlorophyll-*a* content and increased the abundance of rotifer (See Chapter 5). Rotifers are an important source of mortality for phytoplankton and in bringing about seasonal changes in the abundance of phytoplankton (Adamneh Dagne, 2010). In January, February and December of 2016 algal biomass attained its peak (Fig. 3.2) but cell count of individual species per ml decreased.

During the early dry season (January-March), biovolume of phytoplankton (especially, blue-green algae and diatoms) has shown a sharp decline following the decrease in water temperature and nutrient depletion. A combination of low temperature and increased grazing pressure from the higher density of zooplankton especially rotifers (See Chapter 5) may have accounted for the generally reduced abundance of phytoplankton during the early dry season. The correlation between the observed decline of phytoplankton with diminishing nutrient concentrations and decreased temperature during the early dry season that the relationship appears to be predominantly causal in Ethiopian small crater Lakes, Bishftu (Tadesse Ogato, 2007) and Koriftu (Zelalem Desalgn, 2007) and one of the large Ethiopian rift valley lake, L. Chamo (Eyasu Shumbulo, 2004).

Microcystis aeruginosa is the most conspicuous population in LTA. The dominance of *M. aeruginosa* and most other phytoplankton are high during the rainy months (June-September) and the beginning of the dry season (October). This might be associated with nutrient enrichment and turbidity of the lake water. The dominance of *M. aeruginosa* during mixing and rainy period than a period of thermal stability was observed in the nearby rift valley lakes Hawassa, Chamo, and Ziway (Girma Tilahun, 2006), the tropical high land Lake, L. Hayq (Tadesse Fetahi, 2010) and the largest Ethiopian Lake, L. Tana (Ayalew Wondie, 2006), and other small-sized Ethiopia crater lakes, L. Kuriftu (Tadsse Ogato, 2007) and L. Bishoftu (Zelalem Desalegn, 2007).

The persistence of fairly high levels of nutrients, particular phosphate in a generally turbid and mixing water column favors the dominance of blue-greens owing to their structural and physiological adaptation for buoyancy regulation (Hammer *et al*., 1983). The same ground has given for LTA for the dominance of blue-green algae over the other phytoplankton. Shapiro (1990) suggested that high temperature helps the dominance of cyanobacteria. Robarts and Zohary (1987) also highlighted high levels of temperature favor the optimal growth of *Microcystis* in lakes and reservoirs of the temperate and tropical regions. In general tropical lakes show cyanobacterial dominance during drought and falling water level (Harris and Baxter, 1996). This probably results in the remarkable abundance of cyanobacteria in the period in October (beginning of the dr time that starts water falling of the lake).

Shapiro (1984) in his studies indicated that low CO_2 concentration can readily lead to the cyanobacterial dominance of *M. aeruginosa*. Badger and Price (2003) also indicated *M. aeruginosa* dominance due to its well developed and most effective CO_2 concentrating mechanisms. The factors for the dominance of blue-greens in the Ethiopian crater Lakes Bishoftu (Tadesse Ogato, 2007) and Kuriftu (Zelalem Desalgn, 2007) appear to be low CO_2, superficial mixing, and nutrient availability. This situation may persuade the predominance *M. aeruginosa* in particular and cyanobacteria in LTA.

Other investigations also stressed cyanobacterial dominance associated with different factors such as shallow mixing (Reynolds, 1987), low light (Smith, 1986, 2006), high temperature (Shapiro, 1990), low CO_2 (Shapiro, 1984), high total phosphate (Watson *et al*., 1997), luxury consumption of phosphorus (Peterson *et al*., 1993), ability to minimize grazing (Hanley, 1987), and buoyancy regulation (Reynolds, 1987).

The Bacillariophyceae (diatoms) are the most diverse group of phytoplankton in LTA. They also contributed a significant amount of the abundance of phytoplankton next to the blue-green algae. The high concentration of nitrogen and phosphorous nutrients resulting from low of other competing algal groups and biomass most probably favored the dominance of diatoms next to blue-green algae. Wet period dominance of diatoms in LTA is associated with superficial mixing resulting from nocturnal cooling despite the absence of superficial stratification. Talling (1986) confirmed diatoms dominate water columns in the presence of strong mixing. The same plausible reason has forwarded for the remarkable abundance of diatoms in LTA.

Diatoms commonly dominated during periods of overturning in wind exposed (turbulence) lakes with a sufficient supply of available dissolved silica (Diaz and Pedrozoa, 1993). The correlation between the biovolume of diatoms and silica in this study lake is positive but not strong (r=0.176). The nutrients required by algal cells for growth and multiplication may not always be found in relative proportions required by phytoplankton rather phosphorous and nitrogen in all groups and silicon in diatoms are regarded as growth-limiting nutrients (Hecky and Kilham, 1988).

Nitzschia is commonly the dominant genus in abundance (biovolume) among the diatom group. *Cyclotella* and *Synedra* are co-dominants following *Nitzschia sp.* Studies on the seasonality of phytoplankton in Lake Tana showed that diatoms (*Nitzschia* and *Melosira*) are important elements in the seasonal pattern (Ayalew Wondie, 2006). In LTA, *Nitzschia* and *Cyclotella* have shown similar patterns. Unlike Lake Malawi (Hecky and Kling, 1981) where there was no well-established co-dominance of diatoms and blue-green algae have seen, in LTA, although not so strong, there is a positive relationship (r=0.301) between the diatom and blue-green algae, indicating the two dominant algae co-exist. However, the predominance of diatoms generally occurred in periods of strong vertical mixing (June-August) and that of blue-green algae under relatively less mixing water column (October). In addition, the lake is smaller in size and very shallow which results absent well-marked thermal stratification enabled it to have its own pattern of seasonality compared with other large lakes.

In studies of tropical phytoplankton dynamics, Livingstone and Melack (1984) have described large inter-annual changes in the soda lakes involving multiple shifts of dominant algal species against a background of changing salinity, zooplankton density, and nutrient dynamics. Several Soda lakes, such as Lake Arenguadie (Wood, 1986) and Rift Valley Lakes in Ethiopia and Lake Simbi in Kenya (Melack, 1979b) exhibited year long-dominance by single blue-green algae species, *Spirulina platensis.* In LTA, *Microcystis aeruginosa* has predominantly occurred year round.

Coelastrum sp., Cosmarium sp. and *Botryococcus sp.* have also co-existed with *M. aeruginosa. Botryococcus* genus was also reported as an important taxon in the phytoplankton assemblages of the Ethiopian rift valley Lakes Chamo (Girma Tilahun, 2006) and Kenyan lakes Naivasha and Oloidien (Kalf and Watson, 1986). The predominance of *Botryococcus sp.* may be attributed to their ability to change their position by buoyancy alterations and better use lower depths of greater nutrient abundance and upper strata of light abundance, and their ability to avoid photoinhibition at Lake Surface by their protective carotenoid pigmentation (Wetzel, 2001). These large Chlorococcal colonies cannot be effectively grazed by zooplankton (Girma Tilahun, 2006). *Euglenoids* and *Phacus sp.* contributed the least abundance of phytoplankton relative to other taxonomic groups. John (1986) associated high abundance and bloom of Euglenophyceae with organic enrichment from farm animals or plants decaying in overgrown vegetation. The same explanation seems plausible for the abundance of *Euglenoid* in LTA as an input of organic materials originate from domestic animals from the nearby communities and related agricultural byproducts washed through runoff in the near area and the Gurage highlands.

3.4.2 Phytoplankton diversity as an Indicator of water conditions

Diversity is a parameter of community structure related to the number of species and its abundance. One of the common diversity indices used widely in environmental monitoring (Washington, 1984) is the Shannon-Wiener Diversity Index (H′). According to the study by Salusso and Moraña (2000), there are three classes of pollution status based on H'. In their scale, water bodies with H′ over 3 has no contaminants, H′ values 1-3 contain a moderate level of contaminants, and H′ <1 indicates high pollution level. Based on this classification, LTA is moderately polluted seeing H' index values of 1.741 to 2.161.

CHAPTER 4: PHOTOSYNTHETIC PRODUCTIVITY AND BIOMASS OF PHYTOPLANKTON IN LAKE TINISHU ABAYA

4.1 Introduction

In order to meet the growing food requirements of the world population, great effort is needed in the development and utilization of the biological wealth of the aquatic environment (Wilfred, 2016). The study of energy transfer in lakes and reservoirs is based on the measurement of primary productivity of phytoplankton and the environmental variables, which limit or control this productivity (Rediet Abate *et al.,* 2014). Primary productivity of aquatic ecosystems is basically dependent upon the photosynthetic activity of autotrophic organisms (Wetzel and Likens, 1979). Phytoplanktons are the major primary producers in many aquatic systems and are an important food for consumers (Sarvesh *et al.,* 2015).

As phytoplankton assemblages are at the base of the food web, changes in phytoplankton biomass and the pattern of primary production have implications for the whole community, including fish (Ahlgren *et al.,* 2005). For instance, phytoplankton production (PP) is regarded as a good predictor of fish yield in lakes and seasonal measurements of PP are a prerequisite for good such estimates (Hooker *et al.,* 2001). The primary productivity of different water bodies has been widely investigated to assess the fish production of a water body to formulate fishery management policies (Girma Tilahun and Ahlgren, 2010).

Primary production by phytoplankton constitutes the principal source of energy to higher trophic levels in open waters. Their small size and short life cycle enable them to respond quickly to environmental changes and their abundance, primary production, and species composition have been used to indicate the trophic status and changes (interannual or seasonal) in water bodies around the world (Darchambeau *et al.,* 2014).

It is now well known that primary productivity of warm tropical lakes is two to three times higher than that in temperate ones (Talling and Lemoalle 1998; Darchambeau et al. 2014) because the maximum rate of photosynthesis per unit of biomass (photosynthetic capacity) is a function of temperature. Higher incident irradiance is the other factor that promotes elevated primary production in tropical lakes (Lewis, 1987).

There is considerable seasonal variation in phytoplankton productivity in African lakes (Ndebele-Murisa *et al.,* 2010), but understanding these variations in relation to nutrient concentrations and community structure has been poorly reported in comparison to temperate lakes (Lewis, 2000). This has changed recently, however, with a number of studies on the temporal dynamics, community structure and primary productivity of phytoplankton in tropical African lakes (Ndebele-Murisa *et al.*, 2010; Tadesse Ogato, 2010, Zelalem Desalgn, 2010; Darchambeau *et al.,* 2014, Rediet Abate e*t al.*, 2014; Tadesse Fetahi *et al.,* 2014) appearing in the scientific literature. This study basically designed to investigate and discusses the temporal variations of photosynthetic productivity and biomass of phytoplankton in relation to various physical and chemical variable and nutrient concentrations to observe the potential productivity of LTA.

4.2 Materials and Methods

4.2.1 Sampling protocol

One mid-lake sampling site (open water) was selected for regular sampling site for the estimation of primary production. The open-water site is relatively less turbid (good for light penetration) than shore sites; thus the open-water site (mid-lake) is considered a representative for studying the photosynthetic activities of the present study lake. Collections of water samples for the incubation purpose were made at this site with monthly intervals between January 2016 and December 2016. Composite samples collected from the depth (0.25 m interval) were prepared by mixing equal proportions of water samples collected from the euphotic depths using a Kemmerer (Model: A-S310 MONDSEE: Moosbachweg 10) water sampler.

4.2.2 Measurement of phytoplankton biomass

Biomass of phytoplankton for the study lake was estimated as chlorophyll-*a* content (Chl-*a*). For Chl-*a*, 100 ml of water samples were filtered onto duplicate Whatman GF/F glass microfiber filters (0.7 mµ pore size; 47 mm Diameter, Model PB-1825-047-BR) and the filters were deep-frozen overnight to facilitate extraction. Then the filters were homogenized and extracted in 90% acetone. Chl-*a* was determined spectrophotometrically after centrifugation at 665 and 750 nm (Talling and Driver, 1963) without phaeopigments corrections.

4.2.3 Measurement of photosynthetic productivity of phytoplankton

Photosynthetic rates of phytoplankton were estimated at the open-water site by the light-and dark-bottle technique using composite water samples with dissolved oxygen determined by the Winkler method (Wetzel and Likens, 2000). Both light-and-dark Winkler bottles were filled with water taken from the composite (250 ml) and incubated for 3 hours during the day time (11:00 AM to 02:00 PM) at different depths (within the euphotic depth). Dark bottles were kept in light-proof dark-bags and the top of the bottles was wrapped with aluminum foil. Another pair of bottles with composite water sample (250ml) was fixed with Winkler reagents immediately after incubation of the light-and-dark bottles which used as an initial in the calculation of PP.

After incubation, the bottles were retrieved and immediately fixed with Winkler's reagent and transferred to Limnology Laboratory of Addis Ababa University in the icebox. Then it was immediately acidified and well-mixed. The concentration of oxygen (O_2 mgL^{-1}) in each bottle (initial, dark, and light bottles) was determined by titration with standard sodium thiosulfate solution (0.0125 N), using starch as an indicator (Lind, 1979). The measurement of oxygen (O_2 mgL^{-1}) then converted into grams of the carbon contents (gC) using the formula stated by Wetzel and Likens (2000).

Gross and net photosynthetic rates (GP and NP) were calculated using the formula in Wetzel and Likens (2000). The hourly rate of gross photosynthesis per unit area ($\sum A$) was estimated from depth profiles of gross photosynthesis by the Gird Enumeration Analysis (Olson, 1960). The daily integral rates of photosynthesis ($\sum\sum A$ mg $O_2 m^{-2} d^{-1}$) of the lake were determined from the hourly-integrated rates by multiplying with the factor used by Talling and Talling (1965) for East African Lakes which is 0.9. The products then multiplied by the number of hours of sunshine often considered for tropical lakes (that is, 10) (after Tadesse Fetahi, 2010). The estimated various photosynthetic parameters were found in Table 4.1.

Table 4. 1 The photosynthetic parameters estimated in LTA.

B = Phytoplankton biomass as chlorophyll-*a* (mgChl *a* m^{-3});

ΣB=Phytoplankton biomass in the euphotic zone (mgChl-*a*m^{-2}) (the product of phytoplankton biomass (B, chlorophyll *a* (mg Chl-*a* m^{-3}) and euphotic depth;

A= Gross photosynthetic rate (mg $O_2 m^{-3} h^{-1}$);

A_{max} = Light-saturated volumetric rate of gross photosynthesis (mg$O_2 m^{-3} h^{-1}$);

ΣA= The hourly rate of photosynthesis per unit area (mg O_2 $m^{-2} h^{-1}$) estimated from the gross photosynthesis depth profile (Talling *et al.*, 1973);

$\Sigma\Sigma A$=Daily rate of photosynthesis per unit area estimated by multiplying ΣA by the commonly used factors 10 and 0.9 (Talling 1965);

P (A/B) = Specific rate of gross photosynthesis per unit biomass (mg O_2 mg Chl$a^{-1} h^{-1}$);

P_{max} (A_{max}/B = A_{max}/Chl *a*) = Light-saturated specific rate of gross photosynthesis per unit biomass (mg O_2 mg Chl a^{-1} h^{-1}) (Wetzel and Likens, 2000)

4.4 RESULTS

4.4.1 Biomass of phytoplankton

The results of changes in phytoplankton biomass measured as a chlorophyll-*a* concentration of LTA is shown in Fig. 4.1. The concentration of chlorophyll-*a* (chl-*a*; µgL^{-1}) and the phytoplankton biomass within the euphotic zone (ΣB; mg Chl-*a* m^{-2}) ranged from 18.97 to 65.05 µgL^{-1} (average: 31.21 µgL^{-1}) and 9.33 to 41.74 mg Chl-*a* m^{-2} (average: 21.89 mg Chl *a* m^{-2}), respectively (Table 4.2). Biomass of phytoplankton has shown marked differences among the study months (ANOVA; $p<0.05$). It showed a declined pattern from January to May (dry period) with a sharp decrease in July and August (rainy period) and then an increasing pattern from October to December (dry period) with a peak recorded in December (dry month) (Fig.4.1).

The biomass of phytoplankton in the study area was correlated with some of the physicochemical variables. It correlated positively and strongly with pH (r=0.543), conductivity (r=0.485), Secchi disk (r=0.467) and alkalinity (r=0.576). However, it correlated negatively and strongly with NO_2 (r= -0.550), TP (r=-0.411) and TSS (r=- 0.658).

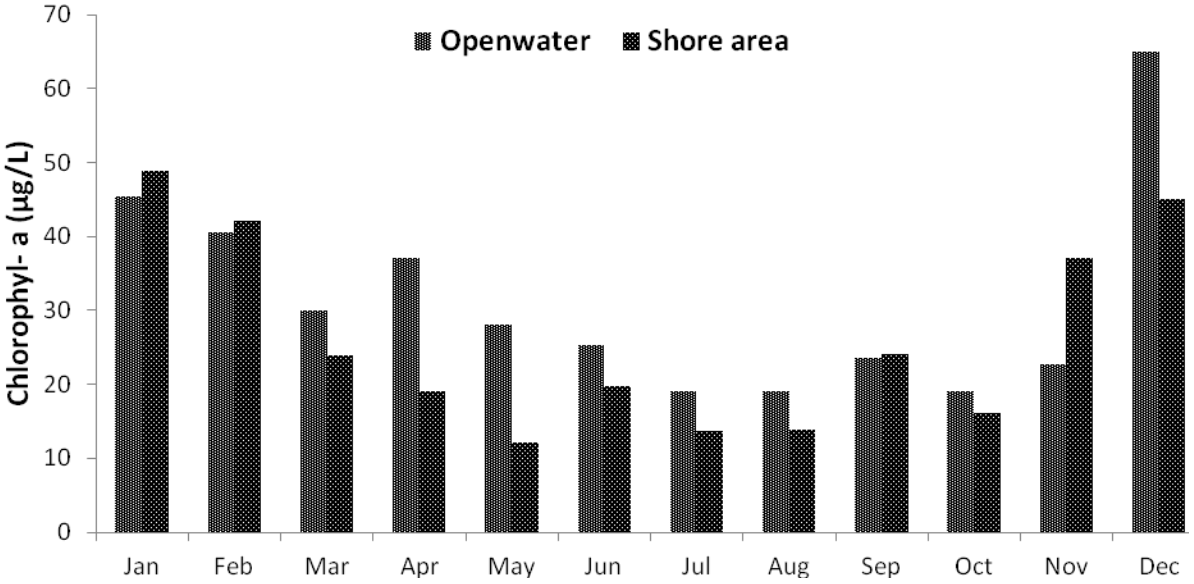


Fig. 4. 1 Spatiotemporal variations in the biomass (chlorophyll-*a* contents) of LTA at open water and shore sites during the study period (January-December, 2016).

4.4.2 Photosynthetic production

4.4.2.1 Depth-profiles of gross photosynthesis

Depth profile of photosynthetic rate (GP) as a gram of carbon evolved per unit volume per given time ($gCm^{-3}h^{-1}$) was estimated for each sampling date (Fig. 4.2). In the estimation of photosynthetic production along with the vertical profiles using composite samples, the same amount of algal biomass was presumably exposed to the same levels of algal nutrients. The depth profiles considered three main regions including the near-surface region of light-inhibition (<0.25m), a sub-surface region of light-saturation (0.25-0.5m), and a lower region of light-limitation (>0.5m).

The results of the depth-profiles of gross photosynthesis exhibited a subsurface maximum rate of gross photosynthesis at 0.25 and 0.5 m in all sampling months. In six measuring months (March, April, May, June, November and December) maximum rate of gross photosynthesis were recorded at a sub-surface region of light-saturation at a depth of 0.25 m and in the other six months (January, February, July, August, September, and October) the maximum rate of gross photosynthesis were estimated at a sub-surface region of light–saturation at a depth of 0.5 m (Fig. 4.2).

In this study, the maximum gross photosynthetic rate (1613.5 $mgO_2\ m^{-3}\ h^{-1}$) (2.02 $gC\ m^{-3}\ h^{-1}$) observed in July coinciding with photosynthetic biomass of 19.09 Chl-*a* mgm^{-3} and minimum value (495 $mgO_2m^{-3}h^{-1}$) (0.62 $g\ C\ m^{-3}\ h^{-1}$) reported in March associated with biomass of 29.9 Chl-*a* mgm^{-3}. Net photosynthetic rates showed a depth distribution pattern to that of gross photosynthesis with maximum volumetric rates at a depth of 0.25 and ranged from a low value of the absolute value of 285 mg O2 $m^{-3}\ h^{-1}$ (4.34 $gCm^{-3}h^{-1}$) in August to a high value of 1285 mg $O_2m^{-3}h^{-1}$ (7.28 $gCm^{-3}h^{-1}$) in December. The extent of percentage reduction in gross photosynthesis from a maximum light-saturated rate of photosynthesis (A_{max}) due to photoinhibition was estimated by calculating the difference between maximum gross photosynthetic rate and the gross photosynthetic rate at the surface and expressing it as a percentage of the latter. The reduction in photosynthetic rates due to photoinhibition varied from 12.5% in March to 62.07% in November with its average values of 34.45%.

4.4.2.2 Photosynthetic characteristics

During the present study, besides depth profile photosynthetic productivity, different characteristics related to phytoplankton productivity were measured (Table 4.2). The average maximum light-saturated rate of gross photosynthesis (A_{max}) was 1.269 $gCm^{-3}h^{-1}$. It lowest and the highest values of were 0.62 $gCm^{-3}h^{-1}$ in March and 2.02 g $Cm^{-3}h^{-1}$ in July, respectively. The lowest and highest values of gross photosynthetic rates were recorded correspondent with the biomass values of 29.9 and 19.09 mg Chl-*a* m^{-3}, respectively. The highest value of light-saturated gross photosynthesis coincides with the lowest of the value of chlorophyll *a*. A_{max} and chlorophyll *a* correlated negatively but it was weak (r= -0.162). A_{max} was high during the rainy period when the concentration of inorganic nutrients becomes high and comparatively low phytoplankton biomass. A_{max} correlated positively and significantly with other photosynthetic parameters such as P_{max} (r=0.804), ∑A (0.742), and ∑∑A (0.744).

Biomass-specific rate of gross photosynthesis at light saturation (P_{max}) has been used to estimate the photosynthetic capacity (assimilation number) of phytoplankton. P_{max} ranged from 0.02 g C (mg Chl *a*)$^{-1}$ h^{-1}) to 0.106 g C (mg Chl–*a*)$^{-1}$h^{-1}) with the annual mean value of 0.048 g C (mg Chl *a*)1 h^{-1}). P_{max} showed considerable seasonal variations. It was high during a rainy period concurrently with increasing water turbidity and nutrients enrichment. P_{max} correlated positively with water turbidity (r=0.638), A_{max} (r=0.804), and ∑∑A (r=1).

4.4.2.3 Production rates per unit area

The area enclosed by each depth profile is a measure of the integral rate of photosynthesis per unit area of lake surface (ΣA, mg O_2 m^{-2} h^{-1}) (g C m^{-2} h^{-1}), values were obtained by the Grid Enumeration Analysis (Olson, 1960) (Table 4.2). The hourly integral rate of gross photosynthesis per unit area (∑A) ranged from 0.60 g C m^{-2} h^{-1} in April to 1.022 g C m^{-2} h^{-1} in July with an average value of 0.642 g C $m^{-2}h^{-1}$. Daily production rates per unit area (ΣΣA, g C $m^{-2}d^{-1}$) which are used as descriptive characteristics of a water body were also estimated from the hourly integral rates and it ranged from a lower value of 5.43 g C $m^{-2}d^{-1}$ in April to a higher value of 9.19 g C m^{-2} d^{-1} in July with an average value of 5.78 g C $m^{-2}d^{-1}$.

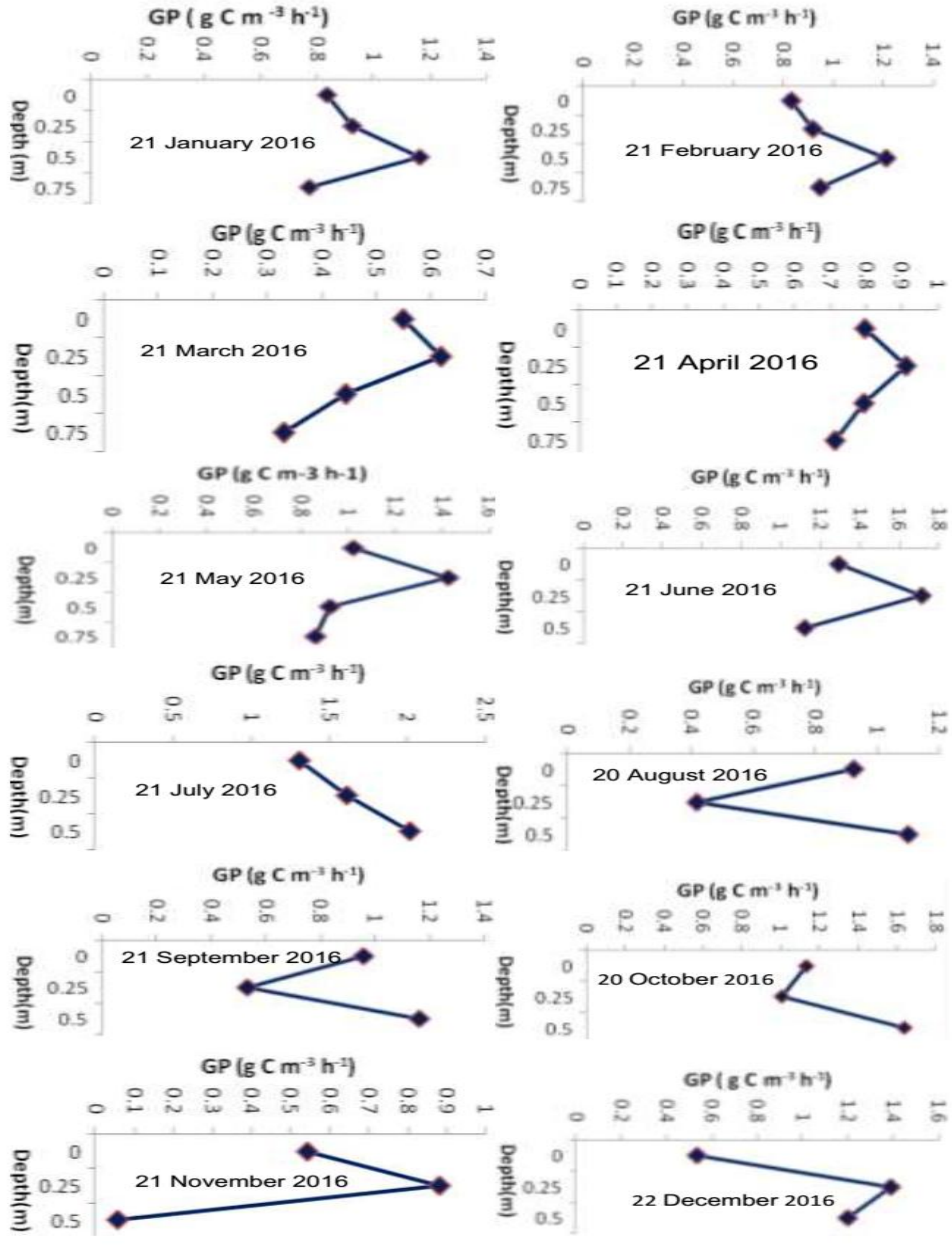


Figure 3. Depth profiles of gross photosynthesis per unit volume (GP, g C m^{-3} h^{-1}) at open station during the study period (January to December, 2016).

Fig. 4. 2 The depth profiles of the growth photosynthesis (GP, g C m^{-3-h}) in LTA (January-December, 2016).

The areal and daily rates of photosynthesis varied with a similar pattern of aerial gross photosynthesis which means they were high during the rainy period coinciding with high ambient inorganic nutrients and relatively low phytoplankton biomass. ∑A and ∑∑A correlated with A_{max} having 'r' values of 0.742 and 0.744, respectively. The seasonal peaks of hourly integrals coincided with seasonal peaks in A_{max}, which were observed when phytoplankton biomass was relatively low. The production rates per unit area seem influenced by the light-saturated rate of gross photosynthesis as the maximum depth of the photosynthetic zone was the same for all depth-profiles. The positive and strong correlation (r= 0.742) between A_{max} and ∑A seems to corroborate the conclusion that gross photosynthesis per unit area is influenced by the light-saturated rate of photosynthesis.

Table 4.2 Photosynthetic parameters measured in LTA (January -December 2016). Phytoplankton biomass (Chl-*a* per unit water volume (B, mgChl-*a*m^{-3}), phytoplankton biomass within the euphotic zone/depth (ΣB, mg Chl-*a* m^{-2}), maximum light-saturated rate of photosynthesis (A_{max}, mgO$_2$m^{-3}h^{-1}) (gCm^{-3}h^{-1}), Specific rate of photosynthesis at maximum light saturation ($P_{max,}$ mg O_2 (mgChl-*a*)$^{-1}$h^{-1}) (gC (mgChl-*a*)$^{-1}$h^{-1}), Hourly integral rates of photosynthesis (ΣA, mgO$_2$m^{-2}h^{-1}) (gCm^{-2}h^{-1}), Daily integral rates of photosynthesis (ΣΣA, mgO$_2$m^{-2}d^{-1}) (gCm^{-2}d^{-1}), and Percentage reduction (% Reduction) of gross photosynthesis.

	Photosynthetic parameters						
Date	**B**	**∑B**	**A_{max}**	**P_{max}**	**∑A**	**∑∑A**	**% Redu.**
Jan	45.33	38.78	930 (1.16)	20.5 (0.026)	555 (0.69)	4995 (6.23)	38.81
Feb	40.52	35.29	970 (1.21)	23.94 (0.03)	586.88 (0.74)	5281.88 (6.6)	45.86
Mar	29.9	24.21	495 (0.62)	16.56 (0.02)	291.56 (0.36)	2624.06 (3.28)	12.5
Apr	37.01	29.40	730 (0.91)	19.72 (0.025)	46.54 (0.60)	482.91 (5.43)	14.3
May	28.01	21.23	1140 (1.43)	40.7 (0.05)	635.9 (0.795)	5723.16 (7.15)	39.02
Jun.	25.34	13.94	1371 (1.71)	54.1 (0.068)	551.25 (0.964)	4961.25 (8.678)	32.27
Jul	19.09	9.33	1613.5 (2.02)	84.52 (0.106)	659.17 (1.022)	5932.5 (9.194)	23
Aug	19.12	11.10	880 (1.1)	46.02 (0.057)	325.83 (0.58)	2932.5 (5.22)	18.92
Sept	23.56	13.68	930 (1.16)	39.47 (0.049)	353.33 (0.56)	3180 (5.04)	21.57
Oct	18.97	10.72	1310.48 (1.64)	69.08 (0.086)	504.03 (0.63)	4536.29 (5.67)	44.44
Nov	22.67	13.16	705 (0.881)	31.1 (0.039)	197.5 (0.247)	1777.5 (2.22)	62.07
Dec	65.05	41.74	1112.9 (1.39)	17.11(0.021)	416.66 (0.521)	3750 (4.688)	61.59
Mean	**31.21**	**21.89**	**931.65 (1.269)**	**38.57(0.048)**	**394.77(0.642)**	**3558.27(5.78)**	**34.53**

4.5 DISCUSSION

4.5.1 Biomass of phytoplankton

The biomass of phytoplankton in LTA measured as concentration of chlorophyll-*a* contents, is high, given the data from other shallow rift valley Lakes of Ethiopia (Table 4.3), such as Lake Hawassa (10-25 mg chl-*a* m^{-3}; Tadesse Fetahi, 2005; 13-26 mg chl-*a* m^{-3}; Girma Tilahun, 2006) and Lake Koka (22 mg chl-*a* m^{-3}; Melaku Mesfin *et al.,* 1988; 16 mg chl-*a* m^{-3}; Elizabeth Kebede and Willén, 1998). However, it is much lower than the Ethiopian small-sized and saline Crater Lake Chitu (2600 mg chl-*a* m^{-3}; Wood and Talling, 1988) and Bishoftu (60 mg chl-*a* m^{-3}; Zinabu Gebremariam, 1994; 30 mg chl-*a* m^{-3}; Tadesse Ogato, 2007). The biomass for LTA is relatively closed to those of the largest freshwater lake in the central rift valley of Ethiopia, L. Ziway (27.8-45.2 mg chl-*a* m^{-3}, 26.3-44.8 mg chl-*a* m^{-3}; Getachew Benberu, 2005).

Table 4. 3: Measurements of phytoplankton biomass on LTA and other Ethiopian Lakes as reported by various authors: (OS is for open water site and SS for shore site)

Lake	Date	Site	B (mg chl-*a* m^{-3})	Data Source
LTA	**2016**	**OS**	**19.02-65.05 (mean: 31.21)**	**Present study**
		SS	**12.13-48.77 (mean: 26.3)**	
Koka	Feb 1995		22	Melaku Mesfin *et al.*, (1988)
	Mar-May, 1991		16	Elizabeth and Willen (1998)
Hawassa	2004		10-25	Tadesse Fetahi (2004)
	2006		13-26	Girma Tilahun (2006)
Ziway (measured at different sites)	2003-2004		30.1-57.9	Getachew Benbiru (2005)
			27.8-45.2	
			26.3-44.8	
	Feb,1987-88		150-222	Girma Tilahun (1998)
	Mar' 91		154	Elizabeth and Willén (1998)
Chitu	Aug'1966		2600	Wood and Talling (1988)
Bishftu	Apr'1992		60	Zinabu Gebre-Mariam (1994)
	July 2007		30	Tadesse Ogato (2007)

Considering the differences in biomass values observed between the two sampling sites of the present study, the t-test conducted at p =0.05 gave a value of 0.42, indicating that the spatial variation of phytoplankton biomass was not significant. This appears to be related to the rarely marked spatial variations of levels of nutrients which presumably resulted from the frequent horizontal mixing of the water body. Despite statistically insignificant (t-test: $p>0.05$), the biomass was higher in the open water compared to the shore area. This seems associated with the generally good water transparency (less turbidity) at the open-water site. The macrophytes found at the mouth of the Feeder Rivers (Bobodo and Dacha) may act as a nutrient filter moderating the spatial input of nutrients into the lake. Similar trends were reported for Ethiopian Lakes Ziway (Getachew Beneberu, 2005) and Babogaya (Yeshiemebet Major, 2006) in which the nearshore site had lower biomass than the open-water site.

The temporal variations of biomass in the study area point out that the low phytoplankton biomass has been commonly shown during the rainy season (June to September). This seasonal minimum biomass in the rainy seasons coincided with a period of heavy precipitation that resulted in land runoff which brought particulate materials into the lake with a consequent reduction in light penetration. Low phytoplankton biomass in lakes during periods of heavy rainfall was also reported for other Ethiopian lakes such as L. Tana (Ayalew Wondie, 2006), L. Bishoftu (Tadesse Ogato, 2007), and L. Kuriftu (Zelalem Desalgn, 2007) and other greater African lake, L. Victoria (Lung *et al.,* 2001).

The peak in phytoplankton biomass in December (early dry period) for the open-water site may coincide with an increase in NO_3-N levels while that of the shore site occurred in January (mid-dry period) concomitant with high water transparency which allowed improved light penetration. The decline of phytoplankton biomass at the end of the dry season (May) probably associated with a fairly low level of most of the inorganic nutrients and the peak abundance of rotifers (See Chapter 5).

Ganzalez (2002) noted that rotifers can have a considerable effect on phytoplankton as they resist the toxicity of blue-greens although they mainly ingest particulate material. He also noted that temporal dynamics of phytoplankton biomass are controlled by loss processes including grazing by zooplankton and fish. The Cyclopoid copepod with a special peak of *Thermocyclops decipins* followed by smaller rotifers dominated the zooplankton community in LTA (See Chapter 5).

This may have also controlled the seasonality of the biomass of phytoplankton in the study lake. Phytoplankton biomass is controlled by a combination of “bottom-up” factors such as temperature, light, and nutrients (Talling, 1986) and “top-down” factors such as zooplankton predation (Brook Lemma *et al.,* 2001; Brook Lemma, 2004, 2009).

4.5.2 Photosynthetic productivity

4.5.2.1 Depth profiles of photosynthetic rates

The depth profiles of photosynthetic activity of LTA exhibited depressed rates of photosynthesis at the near-surface of the water columns during the sampling months. Profiles with reduced surface photosynthetic activity or rate during measurement of primary productivity in water bodies have been documented for many African lakes including those in Ethiopia (Demeke Kifle and Amha Belay, 1990; Eyasu Shumbulo, 2004; Tadesse Ogato, 2007; Tadesse Fetahi, 2010; Zelalem Desalgn, 2015): in Kenya (Melack, 1981; Vareschi, 1982) and in Tanzania (Melack and Kilham, 1974b).

Lower photosynthetic rates of phytoplankton at a lake's surface are linked to photoinhibition, which is believed to occur when light exceeds physiological saturation and results over photons that do not become dissipated by photosynthetic carbon fixation (Falkowski and Raven, 1997). The extent of surface depression of photosynthetic rates is not a function of only the intensity of incident irradiance. Experimental studies have shown that the extent of photoinhibition varies with the photo-acclimatization state (Talling and Lemoalle, 1998) and species-specific differences in photo-acclimatization strategies (Behrenfield *et al*., 1998).

The decrease in photosynthetic rates is associated with photo-oxidative disruption of pigment systems, inactivation of photosynthetic enzymes, and increased photorespiration (Osmond, 1981). Exposure to inhibiting irradiance for prolonged periods has been shown to increase the effect of photo-inhibition (Dubinsky and Berman, 1981). In the present study, the light irradiance (photosynthetic active radiation-PAR) was not measured. However, the water transparency (Secchi disk) generally is too low. The water turbidity due to sediment-laden runoff from the watershed is prominent. From this, it can be concluded a high possibility for photo-inhibition in the study lake causes the lower photosynthetic rates of phytoplankton at a lake's surface than a sub-surface region of the light-saturation area. The correlation between maximum gross photosynthesis and water turbidity and Secchi disk in the present study is also negative.

The depth profiles of photosynthesis have been shown steep curves in most months at the shallow depths (Fig. 4.2). Similar depth profiles of photosynthesis with a similar pattern of photosynthetic curves were observed for in some lakes of Ethiopia such as L. Kilole, L. Ziway, L. Hawassa, L. Chamo, L. Kuriftu, and L. Bishoftu. In Lake George and many soda lakes of East Africa with abundant phytoplankton (Melack and Kilham, 1974a), the euphotic zone is less than 1m deep (the common feature of LTA) and photosynthetic depth profiles are correspondingly condensed. Such kind of dense phytoplankton assemblage with a condensed photosynthetic zone can have implications for intense CO_2-demand by photosynthesis per unit water volume, and such demand leads to considerable CO_2-depletion (Talling and Lemoalle, 1998).

Estimation of percentage reduction of photosynthesis at the surface from the volumetric rate of gross photosynthesis at light-saturation (A_{max}) below water surface showed low value, 12.5% in March, may be coincident with cloud cover. Similarly, Melack (1979a) and Tadesse Ogato (2007) in Lake Bishoftu, Zelalem Desalegn (2015) in Lake Kuriftu, Eyasu Shumbulo (2004) in Lake Chamo, and Girma Tilahun (2006) in Lake Hawassa and Ziway, observed depth profiles of photosynthesis lacking a region with reduced rates near the surface during a cloudy day. In the study area, high percent reductions were seen in November (62.07%), December (61.59%), February (45.58%) and October (44.44%) corresponding with the high water clarity.

On all occasions, below the depths of A_{max}, volumetric rates of gross photosynthesis declined progressively. In rainy months the curve terminated at shallower depths following low water clarity which affected light penetration (Fig. 4.2).

The observed photosynthetic rates of LTA are high; a feature of tropical lakes (Melack, 1979a). From the comparative studies of photosynthetic rates of temperate and tropical lakes, Lemoalle (1981) concluded that generally higher tropical rates stem mainly from higher values of photosynthetic capacity associated with higher temperature. Others suggested that nutrient regeneration is more rapid and primary production is high in tropical than in temperate lakes due to high temperatures at all depths (Hecky and Kling, 1987).

3.4.4.2 Photosynthetic characteristics

The present maximum rate of photosynthesis (A_{max}) of LTA is broadly comparable to those reported for other Ethiopian Lakes; e.g. L. Chamo (716-1789; Eyasu Shumbulo, 2004) and L. Bishoftu (410-1630; Tadesse Ogato, 2007). It was higher than those reported for the other nearby rift valley lake, L. Hawassa (212-425; Demeke Kifle and Amha Belay, 1990) and the small-sized crater Lake Babogaya (106-407; Yeshiemebet Major, 2006). But, the upper rates of photosynthesis in the study lake is much lower than values reported for other small-sized Ethiopian crater lakes such as L. Kilole (4000-10000; Talling *et al.*, 1973) and L. Arenguade (10000-30000; Talling *et al.*, 1973) (Table 4.4). The difference in the values of the maximum rates of photosynthesis between the present study lake and the other nearby Ethiopian lakes may be due to the level of nutrients, water turbidity, and polimictic nature of LTA.

The wide range of the saturation parameter per unit water volume is a function of primarily variable biomass concentration (B, as chlorophyll-content) and photosynthetic capacity (the light-saturated specific rate per unit biomass, P_{max}, (Talling and Lemoalle, 1998). The correlation between A_{max} and phytoplankton biomass for LTA in the present study is negative (r= -0.162), although much higher correlations are not uncommon (Silva *et al.,* 2002). Girma Tilahun (1988) in Lake Ziway (r= 0.36) and Esyasu Shumbulo (2004) in Lake Chamo (r = 0.3), however, found a positive but weak correlation between A_{max} and biomass. Lack of correspondence between biomass and A_{max} was also reported for phytoplankton of several reservoirs in Sri Lanka (Silva *et al.,* 2002) and Lakes Aranguade (Tailing *et al.*, 1973) and Hawassa (Demeke Kifle and Amha Belay, 1990) in Ethiopia.

According to Talling and Talling (1965) and Hammer (1981), high maximum rates associated with low algal biomass result from high specific activity (P_{max}). The strong correlation between A_{max} and P_{max} (r= 0.804) found in this study area may explain the association of high light-saturated rates with low algal biomass observed for LTA. A strong and positive correlation in Lakes Ziway (r =0.54) and Chamo (0.71) was also found between A_{max} and P_{max}.

Generally, high values of A_{max} are shown during the rainy time concurrently with high ambient inorganic nutrients (NO_2, NH_4, SRP, and TP) and high water turbidity. The seasonality of the maximum light-saturated rate of photosynthesis has shown low values during the time of dry period coincided with high DO, total alkalinity, electrical conductivity, and water transparency, pH, and phytoplankton biomass.

In LTA, a temporal variation in A_{max} seems influenced more by the concentration of nutrients than algal biomass. Smith (1986) has also shown a strong and positive correlation between A_{max} and inorganic nutrients (N and P). From his studies on tropical African lakes, Melack (1979a) concluded that euphotic depth was inversely correlated with maximum photosynthetic rates, a similar circumstance was observed in the present study area.

In the study area, the specific rate of photosynthesis at maximum light saturation ($P_{max,}$) has been compared with other nearby Ethiopian lakes (Table 4.4). The upper values of P_{max} for LTA is considerably higher than those reported for Lake Ziway (9.6-22.5; Amha Belay and Wood, 1984; Girma Tilahun, 1988), L. Hawassa (4-19; Demeke Kifle and Amha Belay, 1990), L. Abijata (14.8; Amha Belay and Wood, 1984), L. Aranguade (11-18; Talling *et al.*, 1973), L. Chamo (10-34; Eyasu Shumbulo, 2004), Lake Kilole (16-33.7; Talling *et al.*,1973), all in Ethiopia and Lakes Simbi and Sonachi, Kenya (15-17 and 8-14; Melack, 1981). The range of P_{max} in LTA is nearly comparable with the small-sized Ethiopian crater Lake, L. Bishoftu (29-91; Tadesse Ogato, 2007). In lakes of temperate regions, values rarely exceed 20 mg O_2 (mgChl-*a*h^{-1} (Bindloss, 1974).

Various authors suggested explanations for the unusually high values of photosynthetic capacity obtained from tropical waters (Lemoalle, 1981). These variations in P_{max} may be attributed to the lake size, the existence of complete top-down mixing, and the availability of high concentration of algal and inorganic nutrients.

The highest photosynthetic capacity (84.52 O_2 (mg Chl-a^{-1} h^{-1}) in LTA is associated with the lowest phytoplankton biomass (19.09 mg Chl-*a* m^{-3}). The relation between photosynthetic capacity and phytoplankton biomass is inverse (r= -0.683). An inverse correlation between photosynthetic capacity and phytoplankton biomass is often encountered worldwide and the trend is represented in the tropics as in L. George, Uganda (Ganf, 1972), L. Macllwaine, Rhodesia (Robarts, 1979), L. Ziway (Demeke Kifle and Amha Belay, 1990), Lake Babogaya (Yeshiemebet Major, 2006), L. Hawassa (Demeke Kifle and Amha Belay, 1990), L. Bishoftu

(Tadesse Ogato, 2007), L. Kuriftu (Zelalem Desalegn, 2007), L. Kilolie (Talling *et al*., 1973), and L. Chamo (Eyasu Shumbulo, 2004) in Ethiopia.

It has been demonstrated that temperature (Eppley, 1972), light (Falkowski, 1981), nutrient regimes (Falkowski and Stone, 1983) and cell size (Malone, 1971) directly affect photosynthetic capacity. As temperature and light are uniformly high in the tropics, algal type including cell size, nutrients, and CO_2 supply may be considered of greater importance in determining the magnitude of photosynthetic capacity of phytoplankton (Talling *et al.,* 1973; Amha Belay and Wood, 1984; Girma Tilahun, 1988; Siliva *et al.,* 2002; Rediet Abate *et al.,* 2017).

Various explanations have been given to the relatively high values of the photosynthetic capacity obtained from tropical water bodies. Comparison of mean photosynthetic production in the trophogenic zone of tropical and temperate lakes led to the conclusion that higher tropical rates originate from high photosynthetic capacity, which is probably the result of the usually higher tropical temperature (Lemoalle, 1981). This view is supported by the results of the studies made on Lakes Macilwaine, Rhodesia (Robarts, 1979) and Chad (Lemoalle, 1983), which showed a drop in photosynthetic capacity during the markedly cooler season. In LTA, temperature and photosynthetic capacity are positive. As it has been shown by Tailing *et al.,* (1973) for Lake Aranguade, Ethiopia, Melack (1979b) for Lake Simbi, Kenya and Lemoalle (1983) for Lake Chad in Chad, tropical soda lakes can show a combination of high phytoplankton standing crop and above-average biomass-specific rates, partly due to the large reserve of CO_2 for localized photosynthetic activity in condensed photosynthetic zones (Talling *et al.*, 1973).

4.4.4.3 The areal and daily rates of photosynthesis

The area enclosed by each depth profile is a measure of the integral rate of photosynthesis per unit area ($\sum A$, $mgO_2m^{-2}h^{-1}$) ($gCm^{-2}\ h^{-1}$) of lake surface values obtained by the Grid Enumeration Analysis (Olson, 1960). The upper boundary of the hourly integral rate of photosynthesis per unit area ($\sum A$) of LTA is lower than Lake Bishoftu (3880; Tadesse Ogato, 2007), Lake Ziway (1600; Demeke Kifle and Amha Belay, 1990) and Lake Babogaya (1800; Yeshiemebet Major, 2006) (Table 4.4).

Table 4.4: The various photosynthetic parameters ($mgO_{2)}$ of LTA and other Ethiopian lakes

Lake	A_{max}	P_{max}	ΣA	ΣΣA	Source
LTA	**495-1631.5**	**16.56-84.52**	**46.5-635.9**	**482.91-5723.16**	**Present study**
Hawassa	212-425	4-19		3300-7800	Demeke Kifle and Amha Belay (1990)
Bishoftu	410-1630	29-91	890-3880	4880-8030	Tadesse Ogato (2007)
Kilole	4000-10000	16-33.7		1400-2400	Talling *et al*., (1973)
Arenguade	10000-30000	11-18			Talling *et al*., (1973)
Ziway	1640-4670	9.6-22.5	300-1600		Demeke Kifle and Amha Belay (1990)
Babogaya	106-407		400-1800	1010-5590	Yeshiemebet Major (2006)
Chamo	716-1789	10-34		3800-18580	Eyasu Shumbulo (2004)

The seasonal peaks of hourly integrals coincided with seasonal peaks in $A_{max.}$ The positive and strong correlation (r = 0.742) between A_{max} and ∑A seems to corroborate the conclusion that gross photosynthesis per unit area is influenced by the light-saturated rate of photosynthesis. Although the correlation between phytoplankton biomass and hourly integral photosynthesis is negative and poor (r= -0.162) gross photosynthesis per unit area (∑A) seems to depend on the biomass-specific rate as Talling *et al.* (1973) have shown for the Ethiopian lakes, Aranguade and Kilole. Negative and poor correlation between phytoplankton biomass and hourly integral photosynthesis were also found in Lake Chamo (r = -0.2; Eyasu Shumbulo, 2004).

The daily production rates per unit area (ΣΣA, $mgO_2m^{-2}d^{-1}$) ($gCm^{-2}d^{-1}$) is used as the descriptive characteristics of a water body (Talling and Lemoalle, 1998). To make the data comparable to those of many African lakes, the hourly rates were converted to daily rates by a factor used by Talling (1965) for other East African lakes (i.e. 0.9) assuming the specific activity of phytoplankton does not show strong diurnal variation.

The upper values of daily integral photosynthetic rates estimated in LTA is more or less similar to those of the reported daily integral production values of the shallow productive Ethiopian lakes such as L. Hawassa (7.8 $gO_2m^{-2}d^{-1}$; Demeke Kifle and Amha Belay, 1990), L. Babogaya (5.59 gO_2 $m^{-2}d^{-1}$; Yeshiemebet Major, 2006), L. Bishoftu (8.0 $gO_2m^{-2}d^{-1}$; Tadesse Ogato, 2007) and other tropical African lake such as L. Baringo, Kenya (3.8 $gO_2m^{-2}d^{-1}$; Patterson and Wilson, 1995).

The upper daily integral photosynthetic rates in LTA is much lower than the nearby rift valley lakes, L. Chamo (18.58 gO_2 m^{-2} d^{-1}; Eyasu Shumbulo, 2004**)**, L. Ziway (17.6 $gO_2m^{-2}d^{-1}$; Girma Tilahun (1988), and African tropical Lake Mizrahi, Rwanda (9.5 gO_2 $m^{-2}d^{-1}$; Mukankomeje *et al.,* 1993). But the maximum daily integral production rates of LTA are higher than those reported for Ethiopian Crater Lake, L. Kilolie (1.4-2.4 gO_2 m-2 d^{-1}; Talling *et al*., 1973).

In lakes not enriched by human activities, gross photosynthetic rates of 30 gO_2 (9.36 gC) $m^{-2}d^{-1}$ or greater are seldom encountered (Melack and Kilham, 1974a, b; Tailing and Lemoalle, 1998). Melack (1979a) reported similar values for the phytoplankton of Lake Simbi (Kenya). High values of daily production rates (43 and 57 $gO_2m^{-2}d^{-1}$) were reported for Lake Aranguade in Ethiopia (Talling *et al.,* 1973). Other examples for exceptionally high daily integrals include those determined from oxygen evolution and 14C uptake by bottled phytoplankton in Amaravathy Reservoir, India (56.9 gO_2 or 17.75 $gCm^{-2}d^{-1}$; Sreenivasan, 1965) and Red Rock Tarn, Australia (56.1 gO2 m^{-2} or 17.5 gC $m^{-2}d^{-1}$; Hammer, 1981).

Studies on phytoplankton primary productivity suggested that the mean primary productivity of streams and lakes of the world is about 0.6 g C m^{-2} d^{-1} (Melack and Kilham, 1974b; Oduor and Schagerl, 2006, 2007b), making these the most productive aquatic environments anywhere in the world. For example, high daily primary productivity have been reported in some African lakes such as L. Aranguade of Ethiopia (47–57 g O_2 m^{-2} day^{-1}) (Talling *et al.,* 1973), L. Bogoria (4.0-22.2 g O_2 m^{-2} day^{-1}) (Oduor and Schagerl, 2007), L. Elmentaita (2.0 to 23.2 g O_2 m^{-2} day^{-1}) (Oduor and Schagerl, 2007), L. Naivasha (12 g O_2 m^{-2} day^{-1}) (Millbrink, 1977), L. Sonachi (4.5 to 22.3 g O_2 m^{-2} day^{-1}) (Melack, 1988), L. Nakuru (0.0-30.0 g O_2 m^{-2} day^{-1}) (Oduor and Schagerl, 2007).

In LTA, the daily photosynthetic productivity is almost 10 fold (mean value= 5.78 $gCm^{-2}d^{-1}$) compared to other most productive aquatic water bodies which are 0.6 g C m^{-2} d^{-1} as stated by Melack and Kilham, 1974b; Oduor and Schagerl, 2006, 2007b. Thus, the lake is apparently considered as productive lake and a large consumer population includes zooplankton, fish and birds, among others, are supported by this phytoplankton productivity directly and indirectly.

CHAPTER 5: COMMUNITY STRUCTURES AND BIOMASS OF ZOOPLANKTON IN LAKE TINISHU ABAYA

5.1 Introduction

Zooplankton community comprises a mixture of species belonging to many taxonomic groups. 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 are also good bioindicator of the physical and chemical conditions of aquatic environments which cause changes in the qualitative and quantitative composition of zooplankton and influence their densities (Karabin, 1985).

The dominant zooplankton in most tropical freshwater ecosystems are rotifers and microcrustaceans made up of cladocerans and juvenile copepods (Wetzel, 1983; Seyoum Mengistou and Fernando, 1991b; Brook Lemma, 2001, 2004; Ayalew Wondie, 2006; Adamneh Dagne, 2010). This is because; tropical lakes are generally believed to be characterized by small seasonal variation in water temperature (Burgis, 1978). High predation is of key importance for the dominance of small-bodied zooplankton in warm shallow lakes (Iglesias *et al.,* 2011). The study of Brook Lemma *et al.,* (2001) on the fish predation pressure on and interactions between Cladocerans: observations using enclosures in three temperate Lakes of Germany (i.e. Lakes Feldberger Haussee, Groge Fuchskuhle, and Dagowsee) and one Tropical Lake (i.e, L. Bishoftu, Ethiopia) also concluded that fish indirectly guarantees the proliferation of small-bodied zooplankton protected from fish predation by their small sizes.

Zooplankton abundance, expressed as number per area or volume units does not necessarily provide accurate information about community biomass because zooplankton consists of a great variety of groups or animal species of a large size range (Ayalew Wondie, 2006). The biomass of the zooplankton species is thus an important and necessary parameter to calculate the secondary production of this community (Melão and Rocha, 2004).

Biomass reflects the instantaneous quantity of live organic matter in an area and also provides a means of analyzing an ecosystem's productivity, despite taxonomic composition. Studies that consider biomass values are important because they allow comparison of different environments, providing a common unit to evaluate zooplankton groups (Esteves and Sendacz, 1988). The establishment of length-weight regressions are fundamental when determining the biomass of aquatic communities and also in most studies of food web interactions and secondary productivity (Ghidini and dos Santos-Silva, 2011), which contribute to our knowledge of how aquatic ecosystems function.

Accurate measurement of biomass is necessary to understand the structure and dynamics of biological communities (Seyoum Mengistou and Fernando, 1991b). Basic knowledge on the biomass (length-weight) of different zooplankton species provides the necessary data for the calculation of secondary production, and information about the competitive strategies responsible for their success in an environment. Measurement of zooplankton biomass is essential in studies of production ecology of zooplankton.

Studying zooplankton in the present work was, therefore, intended to quantify biomass, abundance, and community structure of the zooplankton species to bear out the ecological productivity of LTA.

5.2 Materials and Methods

5.2.1 Zooplankton sample collection and identification

Zooplankton sample collection was carried out from two sampling sites (open water and shore sites) monthly from January to December 2016, using 30 µm net sampler with vertical halls (1.5-2m below the surface). The sample was immediately preserved with 4% formalin and transported to Limnology Laboratory of Addis Ababa University for further studies. In the laboratory, from the homogenized sample, 10 ml of sub-sample was taken from the preserved volume of zooplankton sample for the identification. Zooplankton were identified with the possible lowest taxonomic classification under a stereoscope microscope with the fixed camera using different identification references such as Koste (1978), Defaye (1988) and Fernando (2002), and other supplementary sources from the web.

5.2.3 Estimation of relative abundance

The zooplankton taxa were counted using a counting grid (14 in number) as separate groups from a 10 ml sub-sample. In the counting grid, individuals were counted randomly usually up to at least 400 individuals under a stereoscope microscope (magnification 40x) with a fixed camera, and the final estimation of zooplankton relative abundance (individual L^{-1}) of lake water, was estimated for each site and months using the formula of Edmondson and Winberg (1971).

5.2.4 Estimation of biomass

The biomass (dry weight) of the two groups of microcrustaceans (i.e., copepods and cladocerans) zooplankton was determined using length-weight relationships (Dumont *et al.,* 1975). The dry weight of each copepod and cladocerans was measured on a sensitive balance (a PerkinElmer AD6) after drying at 60^{O}C in an oven for 30 ± 6 h. (Tadesse Fetahi, 2010). The weIght of zooplankton was measured in batch then it divided for the number of individuals to have got the weight of single zooplankton. The body length (total body length, excluding caudal setae for copepods and cladocerans) (Dumont *et al.,* 1975) of zooplankton was measured using measuring bored which is fixed with the microscope.

Length-weight regression equations were derived for the different species of copepods (230-adults, 148-copepodites, and 166-naupli) and cladocerans (340 adults). Once the length-weight regression equation for each species of microcrustaceans was derived, the equation applied throughout the study period.

The biomass of the small-bodied rotifers was calculated from length measurements and biovolume approximations. The biovolume of each species of rotifers was computed from linear dimensions applied to simple geometric formulae appropriate to body shape (Ruttner-Kolisko, 1977). A wet-weight/dry-weight conversion factor of 0.1 (Doohan, 1973) was used for all genera except *Asplanchna*, for which a factor of 0.039 (Dumont *et al*., 1975).

5.2.5 Statistical analysis

The relationships between the dominant zooplankton taxa and significant environmental variables were analyzed using Constrained Redundancy Analyses (CRA, CANOCO for Windows 4) using PAST software (Leps and Smilauer 2003). Pearson correlation 'r' was also used to check the correlation between various physicochemical parameters and relative abundance and biomass of the zooplankton communities. Shannon diversity index was applied to show the diversity of zooplankton communities. Analysis of variance (ANOVA) was used to analyze the spatiotemporal effect of zooplankton abundance and biomass. SPSS software package version 20 was used for ANOVA statistical analysis. The statistical significance was set at p=0.05.

5.3 RESULTS

5.3.1 Diversity of zooplankton

In this study, 24 taxa of zooplankton comprising of the large-sized microcrustaceans (copepods and cladocerans) and the small-bodied Rotifera were recorded in LTA during the study time. The smaller rotifers had the highest number of species (11 species) followed by the larger cladocerans (5 species), and copepods (2 species) (Table 5.1). The Shannon Diversity Index was higher for Rotifera (2.45) than cladocerans (2.395) and copepods (2.384) indicated the small-bodied rotifers were more diverse than the larger microcrustaceans in the study area.

Table 5.1 Zooplankton taxa recorded in LTA (January-December, 2016).

Copepods	Cladocerans	Rotifers
Mesocyclps aequatorialis	*Bosmina longirostris*	*Anuraeopsis fissa*
Thermocyclops decipiens	*Ceriodaphnia cornuta*	*Ascomorpha ovalis*
	Daphnia barbata	*Asplanchna sieboldi*
	Diaphanosoma excisum	*Brachionus angularis*
	Moina micrura	*Brachionus caudatus*
		Brachionus quadridentatus
		Brachionus calyciflorus
		Brachionus falcatus
		Euchlanis spp
		Filinia opoliensis
		Filinia pejleri
		Filinia terminalis
		Hexarthra intermedia
		Keratella tropica
		Polyarthra vulgaris
		Pompholyx complanata
		Trichocerca sp.

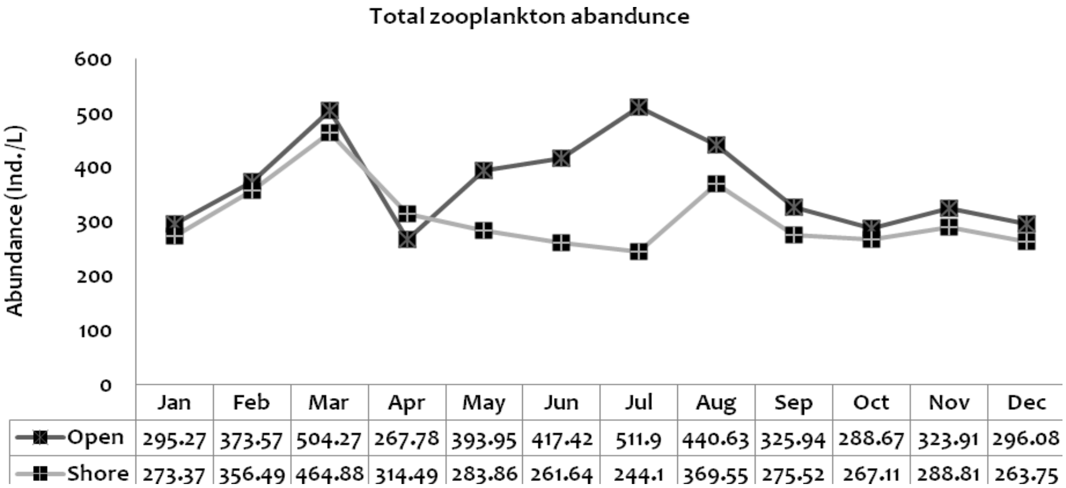
5.3.2 Relative abundance of zooplankton

The total abundance of zooplankton in LTA ranged from 267.8-511.9 (mean: 369.95±84.3) and 244.1-464.88 (mean: 305.3±63.08) individual L^{-1} at the open water and shore sites, respectively (Table 5.2). The abundance of zooplankton was dominated by copepods (54%) followed by Rotifers (40%). Relatively large-sized cladocerans accounted lower percentage (6%) for the total zooplankton abundance compared to the smaller rotifers and copepods.

Table 5.2: The relative abundance (individual L^{-1}) of zooplankton of LTA at the open-water and shore site during the study period (January-February, 2016) (NOTE: SD is for standard deviation.

Zooplankton	**Study site**	**Minimum**	**Maximum**	**Mean±SD**
Copepods	Open-water	80.51	356.03	203.98±94.41
	Shore	71.87	289.70	151.02±53.85
Cladocera	Open-water	9.25	35.34	23.65±9.93
	Shore	10.84	48.88	24.53±10.42
Rotifers	Open-water	82.40	207.40	142.32±38.91
	Shore	75.80	213.00	129.74±42.82
Total	Open-water	267.78	511.90	369.95±84.30
	Shore	244.10	464.88	305.3±63.08

Although relatively higher relative abundance was estimated at the open water than the shore area, there were no significant differences between the two sites (ANOVA; $p > 0.05$). However, there was a seasonal variation among the sampling months (ANOVA; $p < 0.05$). The peak total abundance was reported in July (main rainy period) at the open-water and March (dry time) at the shore site (Fig. 5.1). Generally, the total abundance of zooplankton was relatively high during the rainy season (June-September) concurrently with high water turbidity and high nutrient enrichment. The relative abundance of zooplankton in LTA was comparatively low during the dry season (January-May) coincides with increasing water clarity, low ambient inorganic nutrients, and relatively high phytoplankton biomass.



	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Open	295.27	373.57	504.27	267.78	393.95	417.42	511.9	440.63	325.94	288.67	323.91	296.08
Shore	273.37	356.49	464.88	314.49	283.86	261.64	244.1	369.55	275.52	267.11	288.81	263.75

Fig.5.1: Temporal distributions of the total relative abundance (ind. L^{-1}) of zooplankton in LTA in the open water and shore area (January-February, 2016).

Significant temporal variations in the abundance of copepods have been observed among the study months (ANOVA; $p<0.01$). High abundance (60-65%) of copepods occurred during the rainy months (June-August) and low abundance (15-30%) of copepods mostly occurred during the dry period, particularly in January, November, and December. The numerical density of copepods ranged from 80.51-356.03 individual L^{-} (average: 203.98 individual L^{-1}) and 71.87-289.70 individual L^{-} (mean: 151.02 individual L^{-}) at the open-water and shore sites, respectively (Table 5.2).

Thermocyclops decipins and *Mesocyclops aequatorialis* were the two copepods found in LTA. The adult *T. decipiens* was widely dominated (25-45%) over the abundance of all other zooplankton in most of the sampling months. The relative abundance of *T. decipiens* was comparatively high (>50%) in all the rainy season (June-September) and low (<15%) during the dry season (January-April and November-December (Table 5.3).

The total numerical density of *T. decipins* was ranged from 8 to 155 individual L^{-1} and 12 to 93 individual L^{-1} at the open-water and shore sites, respectively (Table 5.3). *M. aequatorialis*, on the other, contributed a relatively low percentage (5%) for the total zooplankton abundance and the copepods. The total numerical density of *M. aequatorialis* ranged from 8 to 24 individual L^{-1} at the open-water and 2 to 27 individual L^{-1} at the shore sites (Table 5.3). The relatively high numerical abundance of *M. aequatorialis* was observed in the dry season (January-April). The low abundance of *M. aequatorialis* was shown from the pre-rainy season (May) towards the rainy season (June-September) (Table 5.3).

The pre-adult stages of cyclopoid copepods (copepodites and nauplii) contributed a significant amount (35%) to the total zooplankton abundance. The numerical density of copepodites was ranged from 2 to 98 individual L^{-1} at the open-water site and 10-59 individual L^{-1} at the shore site. The density of Nauplii ranged from 5-147 individual L^{-1} (total for the year was 955.93 individual L^{-1}) and from 11 to 130 individual L^{-1} (total for the year was to be 895.92 individual L^{-1}) at the open-water and shore sites, respectively (Table 5.3).

Table 5.3: The abundance (individual L^{-1}) of the adult *T. decipins* and *M. aequatorialis* and the pre-adult stages (Copepodites and Nauplii) of the copepods in LTA at open-water and shore sites during the study (January-February, 2016).

Date	***M. aequatorialis***		***T. decipins***		**Copepodites**		**Nauplii**		**Total**	
	Open-water	**Shore**	**Open-water**	**Shore**	**Open-water**	**Shore**	**Open-water**	**Shore**	**Open-water**	**Shore**
Jan	19.55	6.87	16.12	16.12	39.90	44.12	4.94	52.85	80.51	119.96
Feb	24.12	10.60	7.83	37.65	49.54	36.99	105.03	95.78	186.52	180.99
Mar	21.14	6.94	122.86	92.81	65.06	59.82	146.97	130.10	356.03	289.70
Apr	18.16	1.98	30.05	45.91	30.05	25.10	76.95	70.02	155.21	143.01
May	7.60	2.64	43.93	37.98	41.94	30.05	129.13	100.10	222.60	170.75
Jun	10.57	3.57	154.96	33.95	50.99	36.07	101.99	80.06	318.51	153.65
Jul.	13.87	4.62	154.96	14.00	2.21	10.04	133.43	120.20	304.47	148.88
Aug	8.98	27.00	152.72	56.01	98.29	41.48	26.42	11.10	286.41	135.54
Sept	9.25	0.92	30.12	20.08	62.89	48.09	100.93	95.12	203.19	164.21
Oct	16.12	8.45	42.54	30.52	31.44	42.54	45.32	61.03	135.42	142.54
Nov	16.38	13.20	20.34	16.12	24.04	16.91	44.92	44.92	105.68	91.17
Dec	12.09	4.16	8.12	12.09	33.09	21.01	39.90	34.61	93.20	71.87
Total	**177.83**	**90.89**	**784.55**	**413.24**	**529.44**	**412.12**	**955.93**	**895.92**	**2447.80**	**1812.30**

Unlike the small-sized copepod microcrustaceans, the large-sized cladocerans consisted of low relative abundance (about 6%) of the total zooplankton. The peak abundance of cladocerans was reported in January (15%) and December (25%). The numerical density of cladocerans ranged from 9.25-35.34 individual L^{-1} (average: 23.65 individual L^{-1}) and 10.84 -48.88 individual L^{-1} (average: 24.53 individual L^{-1}) at the open-water and shore sites, respectively (Table 5.2). The relative abundance of cladocerans did not show significant variation in the study months ($p > 0.05$). However, its value was high during the rainy season (June-September) concurrently increasing with water turbidity that provided them with refuge from fish visual predation.

Moina micrura and *Daphnia barbata* were commonly occurring and widely dominated (75%) the abundance of cladoceran zooplankton. The remaining 25% of the cladoceran abundance comprised *Diaphnosoma excisum, Cerodaphnia cornuta,* and *Bosmina longirostris. M. micrura* and *D. barbata* accounted for about 40% and 38% of the cladoceran relative abundance, respectively. *Diaphnosoma excisum* was the third most abundant of Cladoceran species next to *M. micrura* and *D. barbata.* It contributed about 23% at the open-water site and 17% at the shore site. The two cladocerans species (i.e, *C. cornuta* and *B. longirostris)* contributed insignificant amounts (<5%) for the relative abundance of cladocerans and the total zooplankton abundance.

The relative abundance of *M. micrura* ranged from less than a single individual in April to 21 individual L^{-1} in March at the open water site and it ranged from at least 1 individual L^{-1} in July to a maximum of 21 individual L^{-1} at the shore site (Table 5.4). There were marked seasonal variations in the abundance of *M. micrura* ($p < 0.05$). It showed three patterns. It was high during the beginning of the study period (January to March) and became reduced from April to October and then it showed an increasing pattern from November to December (dry season). The peak (21 individual L^{-1}) abundance of *M. micrura* was estimated in August and March at the open-water and shore sites, respectively.

The distribution of the relative abundance of the larger *D. barbata* significantly varied between the study months and the study sites ($p < 0.05$). It was relatively high in the dry period (March-May) and the middle of the rainy time (July) at the open-water site. At the shore area, the abundance of *D. barbata* was high during all the rainy period (June- August). *D. barbata* was frequently occurring and became dominant among the cladocerans in abundance in all the rainy months (July-September) (Table 5.4).

In most of the sampling months, low abundance (< 5 individual L^{-1}) of *D. excisum* was recorded. The numerical abundance of *D. excisum* ranged from total absence to 14 individual L^{-1} at the open-water site and 23 individual L^{-1} at the shore site. It was high at the beginning of the sampling dates such as January and February and the end of the study period (December). The peak abundance (14.1 individual L^{-1} at the open-water site and 22.7 individual L^{-1} at the shore site) of *D. excisum* was seen in December at both sites. Low abundance of *D. excisum* was reported in all the rainy seasons (June-September) (Table 5.4).

Low (<5%) abundance of *C. cornuta* and *B. longirostris* were seen throughout the study period. Usually, on average, less than a single individual per litter was counted. The estimated total abundance of *B. longirostris* in the study area was relatively low and it was to be 20.53 individual L^{-1} at the open water site and 10.37 individual L^{-1} at the shore site. Total abundance of *C. cornuta* was 8 individual L^{-1} and 4.3 individual L^{-1} at the open-water and shore sites, respectively. The relative higher abundance of *C. cornuta* and *B. longirostris* was observed during dry season particularly from January to May. They were absent in the rainy months (June to September). *C. cornuta* was also absent in November and October (dry period) (Table 5.4).

Table 5. 4 The relative abundance (individual L^{-1}) of the cladoceran species at the open-water site and shore site in LTA during the study period (January-December 2016).

Date	***C. cornuta***		***B. longirostris***		***D. barbata***		***D. excisum***		***M. micrura***		**Total**	
	Open water	**Shore**	**Open water**	**Shore**	**Open water**	**Shore**	**Open water**	**Shore**	**Open water**	**Shore**	**Open water**	**Shore**
Jan	2.91	0.99	3.96	1.06	0.99	0.99	11.1	7.27	14.0	14.0	32.96	24.31
Feb	0.99	0.33	2.64	0.99	0.99	0	13.87	7.27	11.56	6.61	30.05	15.2
Mar	0.99	0.99	2.97	2.97	10.57	13.21	0.66	0.66	20.61	20.15	35.34	37.98
Apr	1.32	0.33	3.63	1.65	13.21	13.21	1.32	1.32	0.99	13.87	20.47	30.38
May	1.32	0.66	4.95	2.64	18.16	6.52	5.61	1.32	2.61	13.87	32.65	25.01
Jun	0	0.66	1.32	0	8.32	3.18	5.02	1.32	1.85	14.53	16.51	19.69
Jul	0	0	0	0	18.16	17.17	5.02	1.32	2.25	0.93	25.43	19.42
Aug	0	0	0	0	8.32	17.83	1.59	1.59	20.15	1.59	30.52	21.01
Sep	0	0.33	0	0	5.81	12.15	1.59	1.59	1.85	10.04	9.25	24.11
Oct	0	0	0	0	4.62	9.25	2.78	1.85	1.85	6.47	9.25	17.57
Nov	0.26	0	1.06	1.06	0.79	1.85	1.85	1.85	6.87	6.08	10.83	10.84
Dec	0.26	0	0	0	4.16	16.91	14.07	22.72	12.09	9.25	30.58	48.88
Total	8.05	4.29	20.53	10.37	94.1	112.27	64.48	50.08	96.68	117.39	283.84	294.4

In LTA, small-bodied rotifers were the second most important group of zooplankton in terms of the relative abundance next to the copepods. They contributed a considerable amount (about 40%) of the total abundance of zooplankton. There was a notable significant variation in the abundance of rotifers among the study months ($p < 0.05$). High relative abundance (50-55%) of rotifers was observed during the dry months (January, November, and December) with a peak value of 70% in November. The relatively low percentage (< 30%) composition of rotifers was reported towards the rainy season (June, July, and September). The minimum percentage of contribution (about 25%) of rotifers to the total percentage of zooplankton was seen in June

The total numerical density of rotifers ranged from 82.4 to 207.4 individual L^{-1} (mean: 142.32 ± 38.91 individual L^{-1}) at the open-water site and 75.80-213 individual L^{-1} (mean: 129.74 ± 42.82 individual L^{-1}) at the shore site. Three rotifer species including *Brachionus sp.*, *Filinia sp.* and *Hexarthra sp.* contributed the largest (about 90%) relative abundance of the group. These species dominated the abundance of Rotifera more than half of the sampling periods. The three rotifer species together comprised a meaningful amount (about 25%) of the overall zooplankton abundance.

Compared to the other species of rotifers, *B. angularis* was the most dominant with 55 to 70% relative abundance from April to August. But, at the beginning of the study period (January to March), *B. caudatus* and *K. tropica* dominated (>45%) the abundance of Rotifera. Most of the Rotifera species (*K. tropica, B. falcatus, F. pejleri, A. ovalis, P. complanata, P. vulgaris, Trichocera* sp., *Euchlanis* sp., *A. sieboldi)* were high during the dry seasons. But *B. calyciflrous* dominated the abundance of rotifers in most of the rainy months (June -August) (Fig. 5.2).

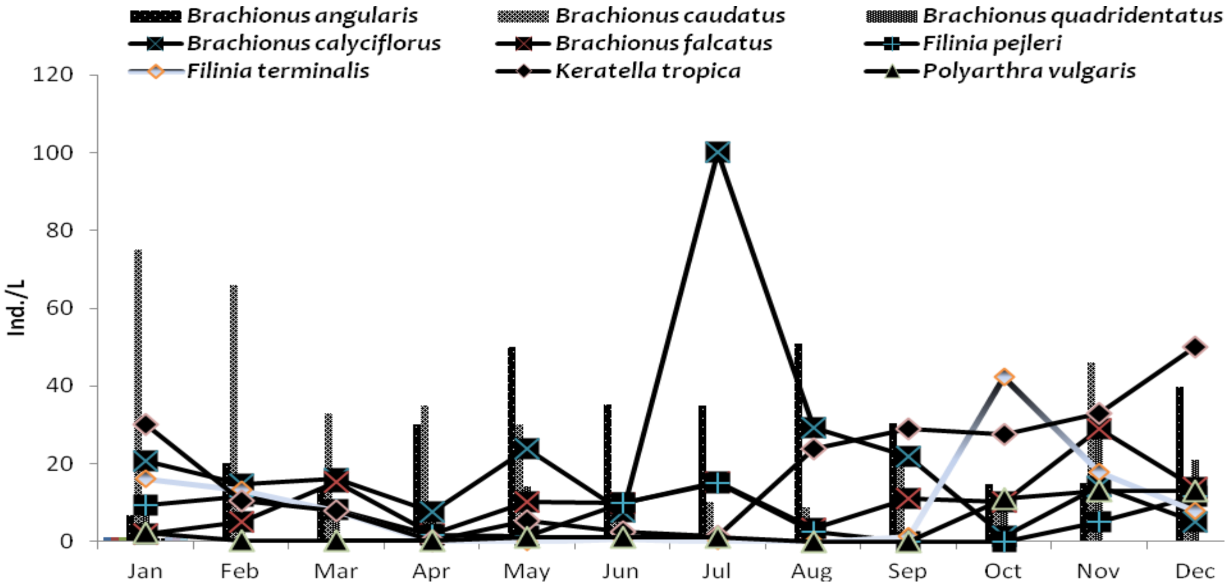


Fig. 5. 2 Temporal patterns in the abundance of the dominant rotifer species (*B. angularis, B. calyciflorus, B. caudata, B. falcatus, B. quadridentataus, F. pejleri, F. terminalis, K. tropica, P. vularis*) in LTA at the open water site (January- December 2016).

5.3.3 Biomass of zooplankton

The overall mean total zooplankton biomass of LTA was 230.87 µgL^{-1} and 164.2 µgL^{-1} in the open water and shore area, respectively (Table 5.5). The mean total biomass of copepods, cladocerans, and rotifers were 174.4, 46.76, and 13.67 µgL^{-1} in the open water and 101.8, 51.35 and 10.9 µgL^{-1} at the shore sites, respectively. The standing biomass of zooplankton was broadly dominated (94%) by the two large-sized microcrustaceans (copepods and cladoceran). The rotifers contributed lower percentages (6%) of the total zooplankton biomass as compared to the copepods and cladocerans. The largest (about 75%) biomass of zooplankton in the study area was contributed by *T. decipiens, M. aequatoriais, D. barbata*, and all the pre-adult stage of copepods (copepodite, and nauplii) (Table 5.5).

Table 5. 5 Mean standing biomass (DW, µgL^{-1}) of microcrustaceans (with pre-adult stages) and total rotifers with their percentage contributions within and between groups for LTA during the study period (January-December, 2016) at open-water and shore sites.

Species / stages	**Mean Biomass**		**%**		**%**		**%**	
	Study Site		**Within-group**		**Total crustacean**		**Total zooplankton**	
	Open	**Shore**	**Open**	**Shore**	**Open**	**Shore**	**Open**	**Shore**
M. aequatorialis	45.11	22.16	26.5	21.8	20.8	14.5	19.5	13.5
T. decipiens	72.11	34.98	42.3	34.4	33.2	22.8	31.2	21.3
Cyclopoid copepodite	32.43	25.24	19.0	24.8	14.9	16.5	14	15.4
Cycopoid nauplii	20.79	19.42	12.2	19	9.4	12.7	9	11.8
C. cornuta	0.31	0.14	0.7	0.2	0.1	0.1	0.1	0.1
B. longirostris	0.84	0.40	1.8	0.8	0.4	0.3	0.4	0.2
D. barbata	34.92	41.30	74.7	80.4	16.1	26.9	15.1	25.2
D. excisum	7.60	4.73	16.3	9.2	3.5	3.0	3.3	2.9
M. micrura	3.09	4.78	6.5	9.3	1.4	3.1	1.3	2.9
Total Copepods	**170.44**	**101.80**			78.5	66.4	**73.8**	**62.1**
Total Cladocerans	**46.76**	**51.35**			21.5	33.5	**20.2**	**31.3**
Total Crustaceans	**217.20**	**153.15**					**94.0**	**63.4**
Total Rotifers	**13.67**	**10.90**					**6.0**	**6.6**
Total zooplankton	**230.87**	**164.2**						

The pattern in the standing biomass of zooplankton did not show marked variations between the two sites ($p > 0.05$). But, there was a seasonal effect between the study months ($p < 0.05$). It was high during the main rainy season (June and July) concurrently with a high number of individual zooplankton, increasing nutrients enrichment, and water turbidity. Although biomass was mostly high during the rainy months, a relatively high mean biomass of zooplankton was also reported in March (dry period) (Fig. 5.3).

There was significant variation in the mean biomass of copepods between study sites ($p < 0.05$) and months ($p < 0.01$). The biomass of copepods was higher at the open-water site than shore area. This happened during the rainy period (June to August) and in the post-rainy month

(December) (Fig. 5.3). The two adult copepods, *T. decipiens,* and *M. aequatoriais* contributed the highest (35-51%) biomass of zooplankton in most of the study period.

The standing biomass of *T. decipins* was ranging from 8.04 to 215.39 µgL^{-1} (average=72.11 µgL^{-1}) at the open-water area and 9.35 to 83.53 µgL^{-1} (34.98 µgL^{-1}) at the shore site. The biomass of *M. aequatoriais* was ranging from 23.4 to 72.3 µgL^{-1} (average= 45.11 µgL^{-1}) and 2.33 to 81.93 µgL^{-1} (average=22.26 µgL^{-1}) at the open water and shore sites, respectively. High biomass *T. decipins* was commonly observed during the main rainy season (June to September). But the standing biomass of *M. aequatoriais* was relatively high in most of the dry months (January to April) (Table 5.6)

The larval stages of cyclopoid copepods (copepodites and nauplii) together comprised relatively high (about 23-27%) percentage of the total zooplankton standing biomass. The former contribution for its biomass was higher (14-15%) than the later (9-12%). High biomass of the two larval stages was observed during the rainy season (June-September). But, high relative biomass of naupli was also observed in the two dry months (March and May). The average biomass of copepodites was ranging from 1.86-70.8 µgL^{-1} (average=32.48 µgL-1**)** and 8.43-47.1 µgL^{-1} (average= 25. 24 µg/L) at the open-water and shore sites, respectively. The biomass of nauplii ranged from 1.35 to 37 µgL^{-1} (average: 20.79 µgL^{-1}) at the open water and 2.55 to 31.2 µg/L (average: 19.4 µgL^{-1}) at a shore site (Table 5.6).

Table 5. 6 Biomass (µgL^{-1}) of the adult (*M. aequatorialis* and *T. decipins*) and the larval stages (Copepodite and Naupli) of copepods in LTA (January-December, 2016) at the open-water site and shore site.

	M. aequatorialis		***T. decipins***		**Copepodite**		**Naupli**		**Mean**	
Date	**Open**	**Shore**	**Open**	**shore**	**Open**	**Shore**	**Open**	**shore**	**Open**	**Shore**
Jan.	57.92	20.40	17.57	17.57	29.93	33.09	1.33	14.3	26.69	21.33
Feb.	71.64	31.39	8.53	41.04	37.16	27.74	28.36	25.9	36.42	31.51
Mar.	72.30	23.73	110.57	83.53	46.84	43.07	35.28	31.2	66.25	45.39
Apr.	63.56	6.93	36.66	56.01	23.44	19.58	19.24	17.5	35.73	25.01
May	26.37	9.16	53.16	45.96	33.13	23.74	32.29	25	36.24	25.97
Jun.	37.95	12.82	215.39	47.19	43.85	31.02	31.62	24.8	82.2	28.96
Jul.	51.87	17.28	213.84	19.32	1.86	8.43	37.36	33.7	76.23	19.67
Aug.	27.30	81.93	134.40	49.29	70.77	29.87	6.08	2.55	59.64	40.91
Sept.	23.40	2.33	23.49	15.66	44.02	33.67	30.28	28.5	30.3	20.05
Oct.	40.30	21.13	31.90	22.89	20.75	28.08	9.52	12.8	25.62	21.23

	M. aequatorialis		*T. decipins*		Copepodite		Naupli		Mean	
Date	**Open**	**Shore**	**Open**	**shore**	**Open**	**Shore**	**Open**	**shore**	**Open**	**Shore**
Nov.	32.76	26.44	11.78	9.35	13.22	9.30	8.09	8.09	16.46	13.30
Dec.	35.91	12.36	8.04	11.97	24.16	15.34	9.98	8.65	19.52	12.08
Mean	**45.11**	**22.16**	**72.11**	**34.98**	**32.43**	**25.24**	**20.79**	**19.4**	**42.61**	**25.45**

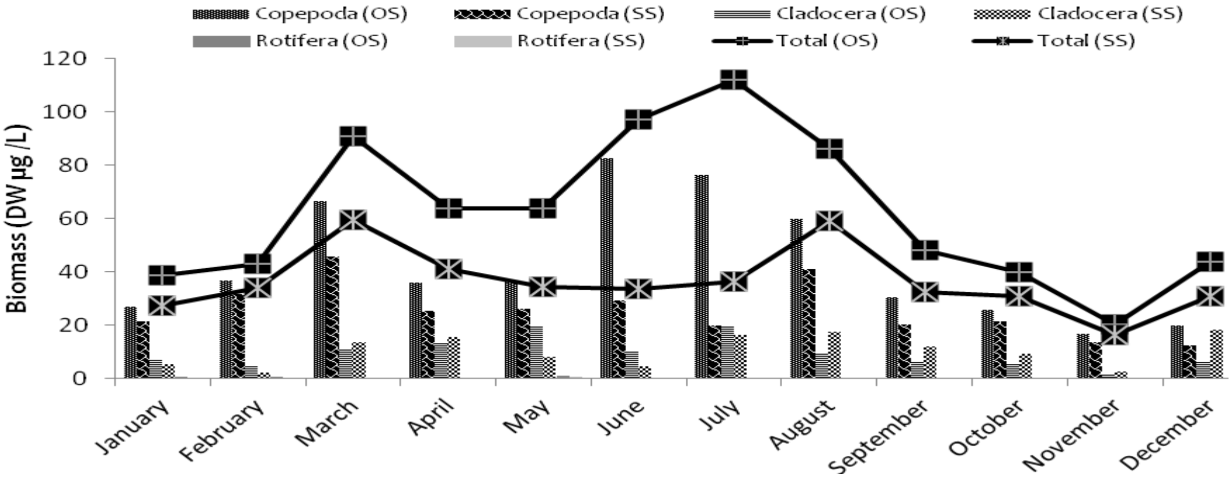


Fig. 5. 3 Temporal patterns of Biomass (µgL^{-1}) of Zooplankton groups (Copepods, cladocerans, Rotifers) and their total mean biomass in LTA at the open water site (OS) and shore site (SS) (January-February, 2016).

The large-sized cladocerans consisted of relatively high (20-30%) biomass next to the copepods. There was marked seasonal variations in the biomass of cladocerans among the study months ($p < 0.05$). It was high toward the main rainy season (June, July, and August) concurrently with high water turbidity and increasing nutrient enrichment. *D. barbata* had large body-size (754 µm mean length) and body dry weight (4.4 µg mean dry weight) relative to all the other species of zooplankton (Table 5.6). This species consisted of the highest (about 80%) biomass of the cladocerans. This species was also comprised the largest (25%) biomass of the total zooplankton at the shore site.

Diaphnosoma excisum contributed relatively high (16%) biomass for cladoceran following *D. barbata*. The biomass of *D. barbata* and *D. excisum* was relatively high during the rainy time (June-September). The reverse was true for *M. micrura, C. cornuta,* and *B. longirostris* whose biomass was high during the dry period (Fig. 5.4). Though the maximum abundance of cladocerans was contributed by *M. micrura*, this species consisted of low (only 9%) biomass of the cladocerans. *C. cornuta* and *B. longirostris* were also contributed to an insignificant amount (<5%) for the biomass of cladoceran and the overall zooplankton standing biomass.

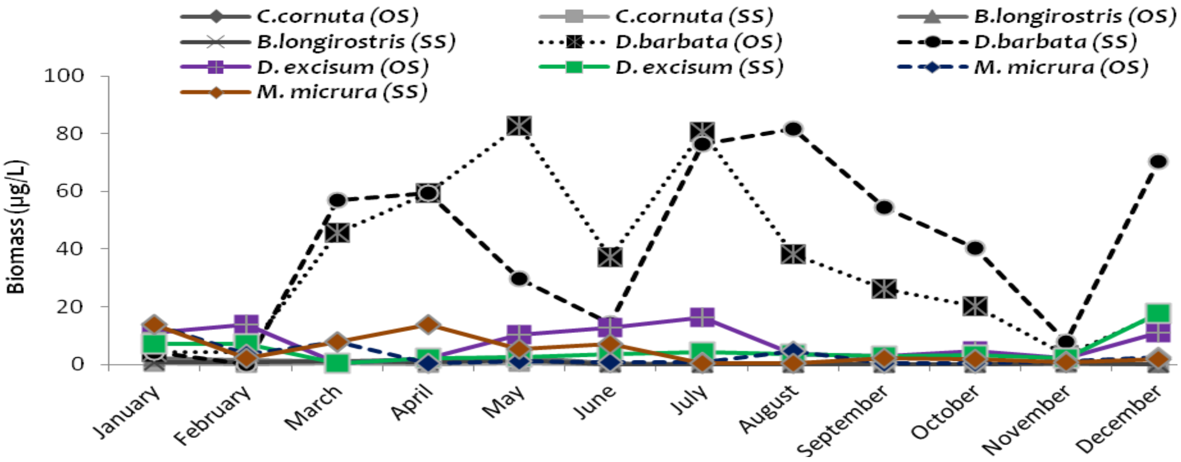


Fig. 5. 4: Temporal patterns of the Biomass (µgL^{-1}) of cladoceran species *(B. longirostris, C. cornuta, D. barbata, D. excisum,* and *M. micrura*) in LTA at the open-water site (OS) and shore site (SS) (January-February, 2016)

The biomass of the small-bodied rotifers was approximately 15 times lower than the larger microcrustaceans (copepods and cladocerans). They only constituted about 6% of the total biomass of zooplankton. The overall total standing biomass of rotifers ranged from 5.54 to 18.69 µgL^{-1} and 5.17 to 20.7 µgL^{-1} at the open-water and shore sites, respectively. There was significant seasonal variation in the pattern of rotifers biomass among the study months (ANOVA; $p < 0.05$). It was high during the dry season (January, February, May, and December) and low in all the rainy period (June-September).

The maximum percentage (45%) contribution for the total rotifer biomass was made by *B. angularis* followed by *B. calyciflorus* (17%), *B. caudatus* (12%), and *B. quadridentatus* (10%). The individual dry weight of *A. sieboldi* was much higher (4.1 µgL^{-1}) as compared to the other all species of rotifers. However, its percentage contribution to the biomass of rotifers was relatively low (10%) in relation to its body size. There was a seasonal effect in distributing the different species of rotifers biomass. At the beginning of the study period (January) (dry month), *Filinia* sp. contributed the highest (35%) percentage whereas, in May (dry month), August (main rainy month) and October (beginning of dry period) *A. sieboldi* contributed the maximum (41%) percentage of the biomass of Rotifera.

Table 5.7: Length-weight regressions (mean values), lnW = lna + blnL (µm), W-dry weight µg, L-length µm, a is an intercept, b is the slope of the regression line, ln-natural logarithm, n-sample size for the microcrustacean zooplankton of LTA.

Species/stages	Lna	B	Length: mean (range)	Individual DW	n	R^2
M. aequatoriais	1.9848	6.0499	889 (862-912)	2.97	76	0.9656
T. decipiens	1.4473	3.0529	614 (539-693)	0.99	154	0.9778
Cyclopoid copepodite	1.7688	2.4503	464 (425-560)	0.73	148	0.9227
Cycopoid nauplii	1.1451	3.9375	174 (169-180)	0.25	166	0.9536
C. cornuta	1.1481	1.6707	364 (277-435)	0.29	72	0.9493
B. longirostris	1.6701	2.0329	302 (257-339)	0.27	58	0.7723
D. barbata	1.7100	0.7984	754 (701-793)	4.4	66	0.9128
D. excisum	1.9289	2.8769	599 (411-767)	1.65	60	0.9662
M. micrura	0.8744	2.4654	463 (438-526)	0.3	84	0.9753

Table 5.8: Percentage contributions of the rotifers species for the total rotifer biomass (µgL^{-1}) in LTA at the open-water area and shore site during the study period (January-December, 2016).

SN	Rotifers	Percentage (%)	
		Open-water	Shore
1	*Brachionus angularis*	22.98	44.64
2	*Brachionus calyciflorus*	18.95	16.25
3	*Brachionus caudatus*	12.19	11.88
4	*Brachionus quadridentatus*	12.28	6.96
5	*Asplanchna sieboldi*	10.09	6.88
6	*Filinia pejleri*	6.84	1.34
7	*Euchlanis spp*	4.56	2.23
8	*Brachionus falcatus*	3.77	2.23
9	*Polyarthra vulgaris*	3.77	3.30
10	*Pompholyx complanata*	2.54	2.77
11	*Hexarthra intermedia*	0.96	0.80
12	*Trichocerca sp*	0.09	0.06
13	*Filinia terminalis*	0.70	0.36
14	*Filinia opoliensis*	0.07	0.01
15	*Ascomorpha ovalis*	0.04	0.06
16	*Keratella tropica*	0.04	0.09
17	*Anuraeopsis fissa*	0.02	0.03

5.3.4 Physico-chemical variables and zooplankton abundance

5.3.4.1 Redundancy Analysis (RDA)

The correlation of some of the environmental regulations and distribution of dominant zooplankton species was analyzed using a constrained Redundancy Analyses (RDA, CANOCO) graph (Fig. 5.9). Table 5.9 below indicated that the first and the second axes of RDA graph explain 55.5% of the cumulative percentage variance of species-environment relation of zooplankton for the study area. *T. decipiens* and *B. angularis* correlated positively with NO_2 and NH_4. *B. calyciflorus* correlated positively with TSS and turbidity. *F. opoliensis*, *B. caudatus*, and *M. micrura* correlated positively with conductivity and DO. *D. barbata* correlated negatively but strongly with DO and conductivity. *B. quadridentatus*, *D. excisum,* and *M. aequatorialis* correlated negatively and strongly with TSS, turbidity, temperature and nitrogen nutrient (NH_4 and NO_2) (Fig. 5.5).

Table 5.9 Summary of the statistics of RDA diagram

Axes	**1**	**2**	**3**	**4**
Eigenvalues:	0.266	0.107	0.096	0.074
Species-environment correlations:	0.920	0.952	0.838	0.865
Cumulative percentage variance				
of species data :	26.6	37.2	46.9	54.3
of species-environment relation:	39.6	55.5	69.8	80.9
Sum of all eigenvalues	1.000			
Sum of all canonical eigenvalues	0.671			

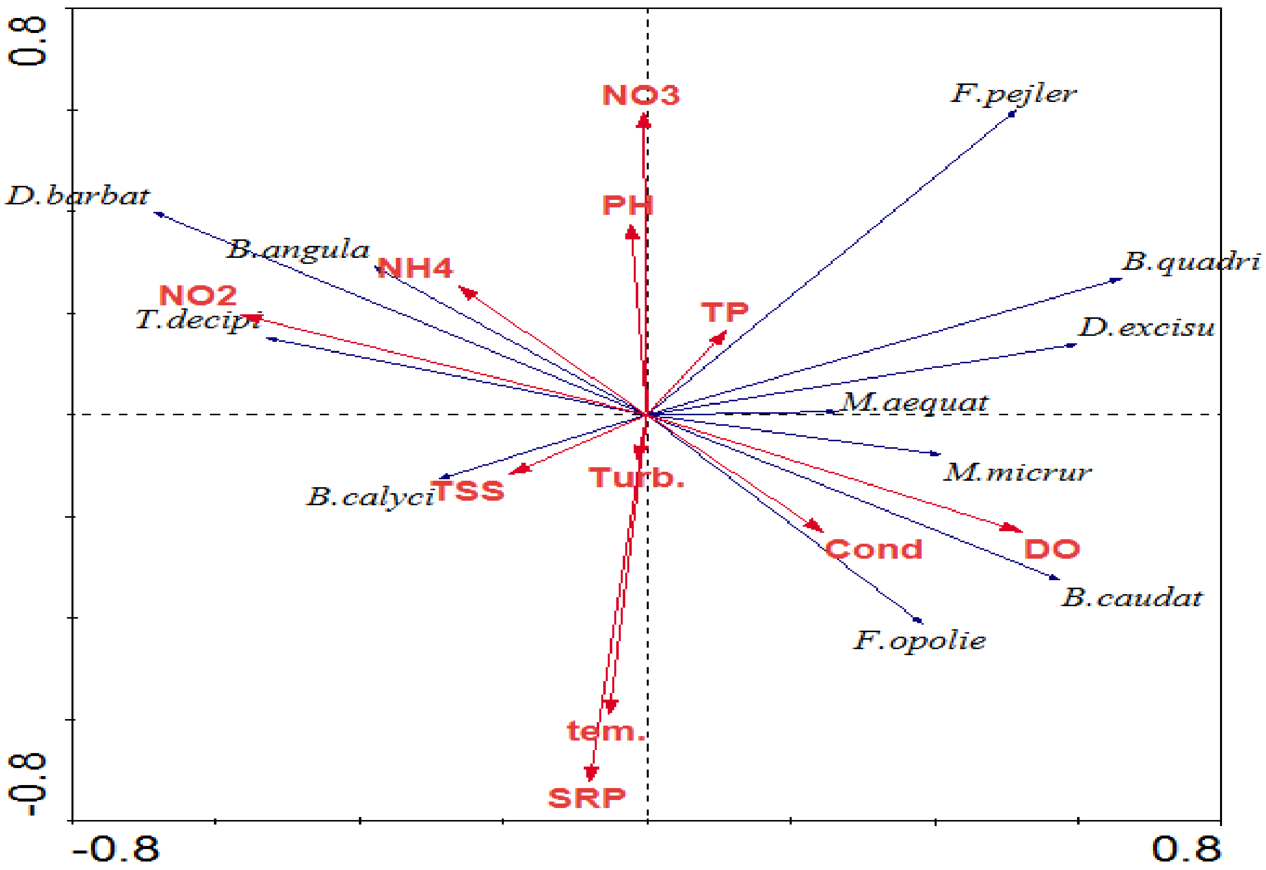


Fig. 5. 5: Bi-plot of the Redundancy Analyses (RDA) dominant zooplankton species (blue arrows) and environmental variables (red arrows): *M. aequat-M. aequatorialis, T. decipi-T. decipins; M. micrur- M. micrua; D.excisu- D. excisum; D. barbat-Daphnia barbata; B. angula-Brachionus angularis; B. calyci- Brachionus calyciflorus; B. caudat-Brachionus caudatus; B quadri-Brachionus quadridentatus; F pejler-Filinia pejleri and F. opolie-Filinia opoliensis.*

5.4 DISCUSSION

5.4.1 Community structure of zooplankton

Zooplankton taxa found in LTA are fairly high compared to some other Ethiopian productive lakes (Table 5.10). Zooplankton diversity in the study lake is more than twofold of Lake Hayq (11 species; Tadesse Fetahi, 2010) and the saline rift valley Lake Abijata (17 species; Kassahun Wodajo and Amha Belay, 1984). The low species diversity in Lakes Abijata and Shalla compared to LTA of the present study may be due to low nutrient enrichment and phytoplankton biomass of the lakes. The diversity of phytoplankton in LTA, however, is much lower than the nearby rift valley Lakes; L. Hawassa (53 species; Seyoum Mengistou and Fernando, 1991a) and L. Ziway (69 species; Adameneh Dagne, 2010). The differences in the taxa richness in the study lake and other lakes of Ethiopia owing to the level of nutrients, development in the growth and biomass of phytoplankton, the turbidity of the lake water, physical structure of the lake (e.g., size and shallowness), the mixing nature of the lake water, and some other related factors.

Table 5.10: Zooplankton species composition of LTA (January-December, 2016) and other lakes of Ethiopia

Lake	Copepod	Cladoceran	Rotifers	Total	Source
LTA	**2**	**5**	**17**	**37**	**Present study**
Ziway	3	7	49	69	Adameneh Dagne (2010
Abijata	4		13	17	Kassahun Wodajo and Amha Belay (1984)
Hayq	2	3	6	11	Tadesse Fetahi (2010)
Hawassa	8	5	40	53	Seyoum Mengistou and Fernando (1991)

The diversity of zooplankton in LTA reflects typical tropical aspects, with rotifers being by far the dominant taxa in species richness (Seyoum Mengistou and Fernando, 1991b; Adamneh Dagne *et al.*, 2008; Tadesse Fetahi, 2010). Rotifers diversity dominating over the crustaceans in the study lake agrees with reports from other Ethiopian Lakes (Seyoum Mengistou and Fernando, 1991a; Adamneh Dagne, 2010; Tadesse Fetahi *et al.*, 2011). In his study on zooplankton communities from East African lakes, Green (1993) found that rotifers have been more specific in the plankton than copepods and cladocerans. Green and Seyoum Mengistou (1991) also found that the mean momentary rotifers species number from Ethiopian water samples was higher than the mean for the rest of the world.

Size relationships between plankton and fish have been documented in the literature. For instance, lakes with resident Salmonoid fish have crustacean zooplankton always smaller than 0.75 mm and substantial rotifer populations, while lakes without fish contained much larger crustacean species and fewer rotifers (Hrbáček, 1962). The study of Brook Lemma *et al.*, (2001) on the fish predation pressure on and interactions between cladocerans: observations using enclosures in three temperate Lakes of Germany (i.e. Lakes Feldberger Haussee, Groge Fuchskuhle, and Dagowsee) and one Tropical Lake (i.e, L. Bishoftu, Ethiopia) also concluded that fish indirectly guarantees the proliferation of small-bodied zooplankton protected from fish predation by their small sizes. In LTA, *Oreochromis niloticus* fish were introduced into the lake from the nearby Lake Ziway in 1997 (Kassahun Asaminew *et al.*, 2011). This may ravenously prey upon the larger crustaceans, resulting in a top-down effect which leads to a high diversity of the small-sized rotifers. As the smaller zooplankters have a little grazing impact upon blue-green algae water quality if aquatic systems deteriorate (Bouvy *et al.*, 2000).

In this study area, the relative abundance of zooplankton has been dominated by the cyclopoid copepods. This is under several other authors, who reported the dominance abundance of copepods in tropical and sub-tropical lakes (Burgis, 1974; Lewis, 1979; Seyoum Mengistou and Fernando, 1991a; Amarasinghe *et al.*, 1997; Irvine and Waya, 1999; Brook Lemma, 2004, Eshete Dejen *et al.*, 2004; Ayalew Wondie, 2006; Adamneh Dagne, 2010; Tadesse Fetahi *et al.*, 2011). The dominance of the abundance of copepods and rotifers over the cladocerans in LTA could have resulted from their competitive advantage.

Feeding experiments did show that Rotifera and smaller zooplankton (e.g., copepods) were dominant in the presence of blue-green filamentous algae (Fulton and Paer, 1988). The low percentage of the larger cladocerans abundance compared to copepods and rotifers in LTA could have been due to high predation pressure by juvenile tilapia. It has been stated in several studies that the formation of large colonies or filamentous limits their exploitation by zooplankton through a physical constraint on ingestion, nutritional inadequacy, and toxicity of such large or filamentous algae (Gilbert, 1990; DeMott and Moxter, 1991; Haney *et al.*, 1994).

There is a considerable temporal variation in the distribution pattern of zooplankton occurrence and their relative abundance. This marked temporal variation was probably due to some proximal causes such as changes in resource availability, weather, and predation pressure. Since zooplankton is an important diet of planktivorous fish (Fernando and Ponyi, 1981; Brook Lemma, 2001), the effect of predation on zooplankton temporal variation cannot be ruled out in LTA. The breeding season of Tilapia, *O. niloticus* in LTA occurs between December and May peaking in January, February, and March, where a minor peak was also observed in June, July, and August (Kassahun Assamenew *et al.,* 2011). Meanwhile, fish data collection, the numbers of juveniles were high (See chapter 7). This significant number of juvenile tilapia (*O. niloticus*) could have a great role in the seasonal structuring of the zooplankton communities in LTA.

The relative abundance of copepods and rotifers in the study lake correlated negatively with chlorophyll-*a* (food) indicated there is a considerable grazing pressure on phytoplankton. This is a well-established fact for other Ethiopian lakes (Seyoum Mengistou and Fernando, 1991a; Eshete Dejen *et al.,* 2004; Ayalew Wondie, 2006; Adamneh Dagne *et al.,* 2008; Tadesse Fetahi, 2010). Compared to the other factors the abundance of copepods is correlated with water clarity (r= -0.504) and pH (= 0.545) which signified that the two environmental factors had a profound effect on the seasonality of the copepod abundance. Several studies revealed that temporal variations of tropical zooplankton were associated with turbidity, water level, temperature, and stratification (Irvine and Waya, 1999; Eshete Dejen *et al.,* 2004; Melão and Rocha, 2004; Isumbisho *et al.,* 2006).

Thermocyclops decipiens had notably dominated the total zooplankton abundance throughout the sampling period. The dominance of *T. decipiens* abundance over the other species of zooplankton could result from its omnivorous feeding habits (Burgis, 1969, 1971) and its fast movement to evade the predation of fishes (Brook Lemma, 2001, 2004, and 2009). This species was reported to feed on *Microcystis* in Lake George Uganda (Burgis, 1969). It may have gained an advantage from the dominance of blue-green algae and the fact that it is less attractive for a predator due to its small body size. The dominance of *Microcystis* over the other phytoplankton (See Chapter 3) most probably results in the presence of a high abundance of *T. decipiens* in LTA. The RDA analysis graph (Fig. 5.1) also indicated that the abundance of *T. decipins* is strongly correlated with most of the inorganic nutrients (NO_2, NO_3, NH_4), and other various physicochemical variables like pH and turbidity.

The relative abundance of the large-sized *M. aequatorialis* (862-912 µm in length) is lower compared to the small-sized *T. decipiens* (539-693 µm in length). A possible reason for the absence of a high abundance of *M. equatorialis* could be a high number of juveniles (<10 cm in total length) *O. niloticus* (See Chapter 7), which may pose a significant predation pressure on the larger adult copepods, *M. aequatorialis,* resulting in abundance lower than *T. decipens*. Larger copepods like *M. aequatorialis* take refuge among macrophytes to avoid fish predation. However, the macrophyte coverage for LTA is not much extensive. This results *M. aequatorialis* in difficulty to hide them from predation which enhances its low abundance.

In shallow lakes where the water transparency is generally low (one of a known feature of LTA), diel vertical migration may hardly be observed (Brook Lemma 2001, 2004 and 2009). In such shallow lakes, large-bodied zooplankton migrates horizontally from the open water to the littoral into the macrophytes seeking refuge against predators (Burks *et al.,* 2002). In LTA, large-sized zooplankton (e.g. *M. aequatorialis* and *D. barbata*) are dominantly found at the shore where macrophytes are concentrated.

Several studies indicated *Chaoborus* (the insect Diptera) as a major component of the limnetic food web in tropical lakes (e.g. Lake Chad; Saint-Jean, 1983; Lake Malawi; Irvine, 1997) or in shallow reservoirs (Aka *et al.,* 2000; Pagano *et al.,* 2003). *Chaoborus* can eliminate 2-90% of the population of its prey per day or 20-29 prey items per individual per day (Pagano *et al.*, 2003). An experimental study on *Chaoborus* predation on zooplankton also indicated that large zooplankton like adult *Mesocyclop* species is affected. A consistently high selection of *Chaoborus* for *M. aequatorialis* was reported from Lake Malawi (Irvine, 1997) and an increase in vulnerability of *Mesocyclops* with an increase in size from Lake Valencia (Lewis, 1979) and Lake Ziway (Adamneh Dagne, 2010). The same reason has been given for the presence of low abundance *M. aequatorialis compared to T. decipiens* in LTA since the presence of Chaoborus is evident in the lake.

The majority of the cladocerans relative abundance comprises *M. micrura, D. barbata,* and *D. excisum.* Except for *D. barbata* where its relative abundance is high in the rainy time, all the remaining abundance of the cladoceran species is comparatively high during the dry season. A high abundance of most cladocerans in most of the dry season may be due to the occurrence of a high level of grazable phytoplankton biomass since high biomass of phytoplankton has been reported during the dry season (See Chapter 4).

The RDA graph also indicated that *M. micrura* and *D. excisum* correlated and strongly with temperature. This shows the temperature is likely to be a controlling factor for the two dominant species of cladocerans besides food availability. The high abundance of *D. brbata* reported during rainy time owing to high ambient inorganic nutrients that indirectly stimulated the growth of grazable phytoplankton for daphnians to thrive on and high water turbidity that provided them refuge from fish predation. The occurrence of *D. barbata* correlated strongly with phosphorous nutrients than others (RDA graph) (Fig. 5.5).

The smaller rotifers comprise a significant amount of the overall abundance of zooplankton following the copepods. The relative abundance of rotifers is dominated by *Brachionus* species (*B. angularis, B. caudatus, B. calciferous,* and *B. quadridentatus*) more than half of the sampling periods. There was a remarkable seasonal variation in the abundance of rotifers. They were high during the dry season probably due to high phytoplankton biomass and low abundance of rotifers has been recorded during the rainy season coinciding with cladoceran dominance that outcompeted them in phytoplankton grazing.

5.4.2 Biomass of zooplankton

The standing biomass of zooplankton in LTA is comparable with other nearby rift valley lakes of Ethiopia such as L. Ziway with 90.87 mgDWm^{-3} for total zooplankton biomass, 78.3 mg DW m^{-3} for copepods, 9.4 mgDWm^{-3}, for cladocera, and 3.1 mgDWm^{-3} for Rotifera (Adamneh Dagne, 2010), L. Hawassa with 44.9 mg DW m^{-3} for total zooplankton biomass, 36.93 mgDWm^{-3} for copepods biomass (Seyoum Mengistou and Fernando, 1991a) and L. Hayq with 236.4 mg DW m^{-3} for total zooplankton biomass, 161.46 mg DW m^{-3} for copepods biomass, and 75 mg DW m^{-3} for cladocerans biomass (Tadesse Fetahi *et al.*, 2011).

The total mean standing biomass of the larger microcrustaceans (copepods and cladocerans) and smaller rotifers in the study lake is fairly high compared to other Ethiopian lakes that suggest the lake is considerably productive and supports most of the aquatic lives including fish. The occurrence of the reasonably high biomass of zooplankton in the present study could have been due to sufficient inorganic nutrients washed down from the watershed through runoff, generating comparatively high algal biomass (food) with well-oxygenated water due to frequent mixing.

Large body-sized *D. barbata* may also rally round in the presence of high zooplankton biomass in the study lake. An increase in zooplankton biomass has often been associated with a rise in the trophic status of the environment (Esteves and Sendacz, 1988). Eutrophic environments support higher biomass of zooplankton than more oligotrophic ones (Esteves and Sendacz, 1988). The trophic state of LTA is hypereutrophic (high nutrient enrichment; See Chapter 2) may result in the high biomass of zooplankton.

In LTA, the two large-sized microcrustaceans including copepods and cladocerans contributed a remarkable amount in the highest biomass of zooplankton. Having a considerably high relative abundance of the copepods particularly *T. decipiens*, and having large body size for the crustaceans (i.e. *D. barbata*) may be the major factor for the existence of high biomass of the microcrustaceans than the smaller rotifers.

CHAPTER 6: BENTHIC MACROINVERTEBRATES AS AN INDICATOR OF WATER QUALITY FOR LAKE TINISHU ABAYA

6.1 Introduction

Benthic macroinvertebrates are one of the key components of lake ecosystems (Venkatesharaju *et al.,* 2010). They form an important link between primary producers, detrital deposits and higher trophic levels in aquatic food webs (Stoffels *et al,* 2005). Benthic invertebrates may indicate eutrophication but in addition several other modes of lake degradation (Hooda *et al.,* 2000). In consequence, the more holistic assessment based on benthic invertebrates is expected to result in different classifications than that based on planktonic communities, especially for lakes subjected to multiple impacts (Hooda *et al.,* 2000).

Using benthic invertebrate communities as indicators of environmental degradation or restoration has become widespread and reliable for bioassessment since the benthos broadly reflects environmental conditions (Rosenberg and Resh, 1993: (Heino *et al.*, 2003; Soldner *et al.,* 2004). Among the biological communities considered bio-indicators of water quality, the most commonly used are the benthic macroinvertebrates (Bonada *et al.*, 2006a,b), because several characteristics make them easy to study and show clear responses when faced with adverse environmental conditions (Moreno *et al*., 2009).

According to the European Water Framework Directives (WFD, 2000), assessment by benthic invertebrates could be useful by these reasons: (i) Diversity and ecological functions of a lake ecosystem are not solely based in the pelagic water body. Benthic invertebrates include several trophic guilds and consumer levels and they are closely interrelated to ecosystem processes. Thus they may well reflect ecosystem health. (ii) Pressures are different in the pelagic and littoral zones of lakes. Littoral benthic invertebrates thus indicate different pressures (e.g. land use or non-point inputs in the catchment, hydro-morphological changes) than indicated by plankton. (iii) Benthic macroinvertebrates integrate combined and varying pressures better than plankton, as they exhibit life cycles ranging between several months and several years. (iv) Benthic macroinvertebrates can be found in most lakes, and during most of the year, and this is a substantial practical advantage over other biological communities, e.g. phytoplankton (highly variable in time, present only during vegetation period) or macrophytes (not present in some lake types and degradation levels, developed only in summer, slow reaction to pressures due to hysteresis effect.

The littoral zone of lakes plays a crucial and dynamic role in regulating the flows of nutrients and materials from the watershed. In this lake zone, benthic invertebrates take an intermediary position between primary producers and microbial decomposers on one side and vertebrate predators (mainly fish) on the other. Hence, the energy flow to the pelagic depends also on a well-functioning benthic energy channel from the littoral zone (WFD, 2000; Chatzinikolaou and Lazaridou, 2007; Pérez-Quintero, 2007; Benetti and Garrido, 2010' Hooda *et al.*, 2000).

Notably, changes in any ecosystem process are potentially reflected by related changes in the structure (abundance and species composition) of the benthic invertebrate assemblages. Benthic invertebrates mostly respond to eutrophication pressure, as phytoplankton, neglects that many lakes are subject to other significant human impacts. This means that the ecological integrity of lakes cannot be assessed by solely surveying phytoplankton. Hence, an assessment system based on benthic invertebrates could effectively indicate multiple pressures. Therefore, this study was aimed to investigate the diversity of benthic macroinvertebrate communities in the littoral zones of LTA to assess the ecological condition of the lake, which has been strongly modified by human activities.

6.2 Materials and Methods

6.2.1 Description of sampling sites

General features of the lake were observed along with five sampling stations. The first two were along the two feeder rivers. One of them was in the northern corner of the lake along Dacha River and the other in the southern corner along Bobodo River and they were named as Dacha riverside site and Bobodo riverside site, respectively. River Badober is relatively small used as an outlet for the lake when water volume increases during the rainy period. The third sampling site was found along river Badober and named a Badober riverside site. The other sampling site was located at the main fish landing and animal watering site along the shore of the lake and hence it was named as a Main fish landing site. At this site, many shoreline modifications have been made (e.g., small irrigation, a demonstration site for seedling, water fetching, domestic waste disposal, and other related wastes). The last sampling site is Reference site. This site is in a relatively pristine environment, as it is protected by the natural rim, and not suitable for agricultural activities, which protected it from impairment by human activities. The physical features of the sampling sites are summarized in Table 6.1.

Table 6.1 Description of sampling sites for benthic macroinvertebrates along LTA

Site	GPS location	Description
Dacha riverside (site 1)	N 07 ^{0}57' 220" E 038$^{0.}$20' 938" 1817 m a.s.l	Dacha River coming from the Gurage Mountains and enters LTA, is a high probability with organic matter, fertilizers, domestic wastes and possibly pesticides. Along this river, there is evidence of high erosion and siltation. Sometimes along the shore, there is considerable coverage of littoral macrophytes. The selected site here is generally considered as highly impaired/disturbed.
Bobodo riverside (site 2)	N 07^{0}55' 396" E 038^{0} 22' 237" 1819 m a.s.l	Boboda River is found on the southern side of the lake. There is a high amount of sediment entering the lake from the southern corner *via* River Boboda. Along this site, seasonal crop production (e.g. maize and horticultural crops) is taking place and thus there is a high probability of organic pollution and inflow of other excess agricultural inputs. Sometimes along the shore of the river, there is relatively good coverage of littoral macrophytes. However, the site is still considered highly impaired/disturbed.
Badober riverside (site 3)	N 07^{0}55'397"	Badober River is in the northwest direction of the lake and serves as an outlet of LTA of overfill. The shore of this river is usually used as a grazing field

Site	GPS location	Description
	E 038^{0}22'241" 1821 m a.s.l	for livestock. Except for some grass that covers the site, there are no aquatic macrophytes along the shore of the river, and hence the site is relatively considered as low/minimally disturbed.
Main fish landing site (site 4)	N 07^0 57'234" E 038^0 22' 637" 1823 m a.s.l	This site is found at the main entrance road to LTA. This site is an indication of high shoreline modification. Different human activities are carried out. Predominantly bathing, livestock watering, car washing, swimming, water fetching, fish landing, and recreation are taken place. At the upper side of the site, extensive irrigation takes place. Fertilizers and pesticides are regularly applied here to grow horticultural crops by abstracting water from the lake. The surrounding area has a steep slope towards the lake exposed for wind and runoff erosion. The site is without macrophytes and it is stony. The site is highly disturbed.
Reference (site 5)	N 07^0 58'234" E 038^0 22' 628" 1821m a.s.l	This sampling site is minimally affected by humans as compared to the other sites. It is protected by a natural cone-shaped rim. Since the location is mountainous, domestic cattle and humans do not easily access it and thus no agricultural activates are carried out and relatively organic and domestic wastes do not enter it. This site is also sheltered by afforestation (e.g. *Eucalyptus* trees) with fewer shoreline modifications. Although the macrophyte coverage is low, the shoreline is covered by grass creating a sort of buffering system for the river. The site is minimally disturbed and relatively pristine.

6.2.2 Sampling protocol

Benthic macroinvertebrate samples were collected from all five predefined sampling sites (Table 6.1) in six sampling periods during the dry (March to May 2017) and rainy seasons (June to August 2017). The benthic macroinvertebrates were collected from the littoral (sampling depth: 0.25-0.5 m) using standardized kick sampling with a hand D-net (28cm x30cm in diameter) with a horizontal transect (up to 50 m) from the shore towards the lake or river was taken, when the depth was less than 0.5m and a diagonal transect was taken when the depth was greater than 0.5 m. To maintain the consistency of sampling effort, a sample was obtained within 10 minutes at each site with 3 replicas. The collection of benthic macroinvertebrates were carried out based on the Ontario Benthos Biomonitoring Network Protocol Manual (OBBNPM, 2007).

The collected animals were washed and separated from sediment using 500 µm sieves and they were stored in plastic bags, fixed with 10% formaldehyde, and transported to the Limnology Laboratory of Addis Ababa University for further analysis. In the laboratory, the macroinvertebrates were washed, sorted, and identified to the lowest possible taxonomic resolution (usually family level) under a stereomicroscope using keys by Edmondson (1959), Jessup, *et al*., (1999), Gooderham and Tysrlin (2002), and Bouchard (2004). The identified macroinvertebrates were counted and the diversity, the number of individuals and distribution patterns were determined for specific metrics and correlated with various physic-chemical variables. The physicochemical parameters responsible for the diversity and abundance of the benthic macroinvertebrates were also measured using a standardized method (See Chapter 2).

6.2.3 Benthic macroinvertebrate diversity metrics

Benthic macroinvertebrate metrics measure different components of the community structure and have different ranges of sensitivity to stress. Therefore, it is recommendable to use several metrics because an integrated approach assures a valid assessment (Klemm *et al*., 1990). In the present study, the biotic Hilsenhoff Family-level Biotic Index (HFBI), Total number of taxa (Family level Richness), Percentage Dominant taxa (% DT), Percentage Chironomidae (%CHIR), Shannon Diversity Index (SDI), and Percent Ephemeroptera (% E), indexes were applied.

6.2.3.1 Hilsenhoff Family-level Biotic Index (H-FBI)

HFBI is calculated by multiplying the number of individuals of each family by an assigned tolerance value, assuming these products and dividing them by the total number of individuals (Hilsenhoff, 1988). Each family is given a score between 0-10 depending on its tolerance value for organic pollution (Bouchard, 2004). Those taxa that are least tolerant of pollution (such as mayflies and stoneflies) are given the lowest scores of H-FBI (Mason, 1996). High HFBI values indicate organic pollution, while low value indicative of clear-water conditions. The Hilsenhoff Family-level Biotic Index (H-FBI) was calculated as equation 6.1, below.

$$\mathbf{H-FBI=\Sigma(Xi\times ti)/n} \quad \text{...Equation 6.1}$$

Where X_i is the number of individuals within the taxon, *ti* is the tolerance value of a taxon, and *n* is the total number of organisms in the sample.

According to Hilsenhoff (1988), HFBI values of 0-3.75 indicate excellent water quality, while 3.76-4.25, 4.26-5, 5.01-5.75,5.76- 6.5, 6.51-7.25, and 7.26-10 indicate very good, good, fair, fairly poor, poor and very poor water quality, respectively.

6.2.3.2 Total Number of Taxa (TR)/Family Level Richness (RICH)

Taxa richness is used as an indicator of habitat quality. This metric is the measure of communities diversity, the number of different families found in samples of each site (Mason, 1996). Reductions in community diversity have been positively associated with various forms of environmental pollution, including nutrient loading, toxic substances, and sedimentation. Generally, taxa-richness increases with increasing water quality, habitat diversity, and habitat suitability (Barbour *et al*., 1996).

6.2.3.3 Percentage of Dominant Taxa (% DT)

Percent DT is the percent contribution of the most numerous family, and measures community balance, or evenness of distributing individual families of the sample. A community dominated by relatively few species would indicate environmental stress (Klemm *et al*., 1990). A high percent contribution by a single taxon indicates community imbalance (Plafkin *et al*., 1989; Bode *et al*., 1996; Barbour *et al*., 1999). The index uses abundance of the numerically dominant family relative to the rest of the population as an indication of community balance, i.e., dominant taxa greater than 35% indicates poor water quality, between 25% to 35% indicates fair water quality, and less than 25% indicates good water quality (Plafkin *et al*., 1989).

6.2.3.4 Percentage of Chironomidae (% CHIR)

This is a diversity measure for which the percentage of chironomids from the sample is calculated. The percentage of Chironomidae increases with a decrease in water quality because chironomids are very tolerant of nutrient enrichment or pollution condition of other types (Bode *et al.*, 1996).

6.2.3.5 Shannon Diversity Index (SDI)

The SDI combines taxa richness and community balance (evenness) to characterize species diversity in a community (Rosenberg and Resh, 1993). The combination of abundance and richness in SDI indicates the state of the macroinvertebrates communities (Rosenberg and Resh, 1993). A community with only a few taxa or with a few abundant taxa has low SDI. But a community exhibits high diversity (high SDI) if many taxa with equal or nearly equal numbers are present. A high SDI indicates good benthic habitat and non-impacted water quality. Benthic macroinvertebrate dynamics of LTA were also examined using the Shannon index (Washington, 1984) (equation 6.2) as:

$$\mathbf{H' = -\sum(ni/N)\ ln\ (ni/N)} \text{..} \text{Equation 6.2}$$

Where H' is the Shannon-Weaver Diversity Index, ni is the abundance of species i, and N is the total number of individuals in the community.

6.3 Statistical analysis

The relationships between the occurrence of benthic macroinvertebrates and significant physicochemical drivers were analyzed using Constrained Redundancy Analyses (RDA, CANOCO for Windows 4) using past software (Leps and Smilauer, 2003). The spatiotemporal variation of the environmental regulations and benthic communities was analyzed using two-way analysis of variance (ANOVA). SPSS software package version 20 was used for ANOVA analysis. Statistical significance was set at P = 0.05.

6.4 RESULTS

6.4.1 Physico-chemical factors

The results of the physicochemical factors responsible for the diversity and distribution of benthic macroinvertebrates at each site are shown in Table 6.2. There was spatial effect among the sampling sites on distributing most of the physicochemical parameters (ANOVA; $p<0.05$). It can be observed from Table 6.2, below, that high temperature of 27.42 ^{0}C was recorded, while dissolved oxygen was low at 5.67 mgL^{-1} at Dacha riverside site. But pH of 8.68 and electrical conductivity of 597.4 µScm^{-1} was higher at the Reference site than the other sites. Turbidity was high at the two feeder riverside sites (Dacha riverside, 179.8 NTU and Bobodo riverside, 172.15 NTU) and the Main fish landing site (138.67 NTU). Turbidity of water was relatively low at the Reference site (93.835 NTU) and Badober riverside sites (121.79 NTU).

All the major inorganic nutrients considered in the present study were high at the Main fish landing site, Dacha river site, and Bobodo riverside sites. These sites were highly influenced by human activities. The different ambient inorganic nutrients were low at the Reference site coincides with the relative less anthropogenic factors. At Dacha riverside site, nitrogen nutrients in particular NO_3 were comparatively higher than the phosphorous nutrients. However, in other sampling sites, the phosphorous families (SRP and TP) were higher than the nitrogen families (NO_2 and NO_3). Thus, most probably nitrogen was the limiting nutrient in the study lake, and in the sites.

There was a significant effect on distributing the physicochemical factors between the two sampling seasons (ANOVA; $P<0.05$). High temperature observed during the rainy months. The reverse was true for dissolved oxygen in most cases. But the difference was statistically insignificant (ANOVA; $p >0.05$). Nearly, an equivalent amount of pH was observed in the two sampling seasons at all the sampling sites. Electrical conductivity and water turbidity were significantly varied between the sampling seasons (ANOVA; $p<0.05$). The former was high at the dry season while the latter was high during the wet season.

Table 6.2 Physicochemical variables at different benthic macroinvertebrate sampling sites: Temperature (^{o}C), DO (mgL^{-1}), Turbidity (NTU), Conductivity (µScm^{-1}), Nutrients (µgL^{-1}).

		Physicochemical variables: mean values								
Sampling sites	**Season**	**Temp**	**DO**	**PH**	**Cond**	**Turb**	**NO_2**	**NO_3**	**SRP**	**TP**
Dacha riverside	Dry	25.53	5.99	7.37	236.33	161.00	0.72	1.12	0.05	0.43
	Wet	29.30	5.34	7.30	210.00	196.33	0.94	1.16	0.61	1.49
	Mean	27.42	5.67	7.34	223.17	179.80	0.80	1.15	0.33	0.96
Bobodo riverside	Dry	24.7	6.73	8.12	274.00	136.30	0.45	0.29	0.49	1.21
	Wet	24.00	7.47	8.30	196.70	208.00	0.77	0.42	0.49	1.22
	Mean	24.35	7.10	8.21	235.35	172.15	0.61	0.355	0.49	1.22
Badober riverside	Dry	21.70	7.30	8.26	172.67	122.83	0.43	0.93	0.07	0.55
	Wet	22.85	7.19	8.22	184.50	120.75	0.43	0.88	0.07	0.53
	Mean	22.28	7.245	8.24	178.59	121.79	0.43	0.91	0.07	0.54
Main fish landing site	Dry	24.30	8.24	8.64	400.33	70.33	0.83	1.30	0.62	1.09
	Wet	23.20	7.39	8.34	206.17	207.00	1.21	1.26	0.77	1.66
	Mean	23.75	7.82	8.49	303.25	138.67	1.02	1.28	0.70	1.38
Reference site	Dry	24.80	9.43	9.02	995.67	71.67	0.04	0.03	0.01	0.03
	Wet	28.06	7.18	8.34	199.13	116.00	0.18	0.22	0.04	0.07
	Mean	26.43	8.31	8.68	597.40	93.835	0.11	0.125	0.03	0.05

6.4.2 Diversity of benthic macroinvertebrates

In this study, 5735 benthic macroinvertebrate specimens comprising of 23 taxa were collected from all the study sites. The number of benthic individuals significantly varied among the sampling sites (ANOVA; $p<0.05$). It was high at Dacha riverside (n = 2089) followed by Bobodo riverside (n = 1145), Reference sites (n= 963), Badober riverside (n= 859), and Main fish landing site (n = 679) (Table 6.3). There were significant differences in the number of counted benthic individuals between the two seasons (ANOVA; $P<0.05$). A higher number of benthic individuals were collected in the wet season (n=3583, 62.5%) than in the dry season (n= 2152, 37.5%).

The Hemiptera Families were the predominant taxa that contributed the largest number (n= 2546, 44.39%) of the total samples followed by the Diptera (n= 878, 15.31%), Coleoptera (n= 835, 14.56%), and the Gastropoda (n= 631, 11%). But the Ephemeroptera (n= 171, 2.98%), Hirudinae (n= 335, 5.84%), and Oligochates (n= 116, 2.02%) were represented by relatively low number of specimens (Table 6.3).

Table 6.3: Benthic macroinvertebrates taxa and its number of individuals collected from all study sites during this study in LTA, and pollution tolerance value for each taxon (Bouchard, 2004)

	Study site						
Taxa	Dacha riverside	Bobodo riverside	Badober riverside	Main fish landing site	Reference site	Total	Tolerance value
Ephemeroptera	**6**	**12**	**33**	**3**	**117**	**171**	
Baetidae	2	1	21	0	89	113	**4**
Caenidae	4	11	12	3	28	58	**7**
Odonata	**108**	**67**	**14**	**10**	**24**	**223**	
Libellulidae	68	35	9	8	11	131	**9**
Ashnidae	0	1	0	0	11	12	**3**
Coenagrionidae	40	31	5	2	2	80	**9**
Diptera	**277**	**284**	**136**	**142**	**39**	**878**	
Chironomidae	258	281	108	121	33	801	**8**
Ceratopogonidae	2	1	20	2	5	30	**6**
Psychodidae	8	0	1	11	0	20	**10**
Tabanidae	9	2	7	8	1	27	**6**
Hemiptera	**1287**	**462**	**237**	**206**	**354**	**2546**	
Belestomatidae	8	6	13	145	88	260	**10**
Corixidae	512	328	114	0	0	954	**9**
Notonectidae	691	97	83	45	251	1167	**9**
Naucoridae	0	0	2	0	3	*5*	**5**
Gerridae	1	0	1	7	0	*9*	**6**
Nepidae	56	5	10	7	1	*79*	**8**
Mesovellidae	16	26	14	0	2	*58*	**8**
Coleoptera	**146**	**128**	**181**	**106**	**274**	**835**	
Dytiscidae	8	17	56	1	47	129	**5**
Hydrophildae	138	110	80	105	170	603	**5**
Gyrinidae	0	0	24	0	38	62	**4**
Elimidae	0	1	21	0	19	41	**5**
Gastropoda	**152**	**82**	**201**	**61**	**135**	**631**	**7**
Hirudinae	**78**	**86**	**47**	**110**	**14**	**335**	**10**
Oligochates	**35**	**24**	**10**	**41**	**6**	**116**	**8**
Total number of individuals	**2089**	**1145**	**859**	**679**	**963**	**5735**	
Number of taxa	**19**	**19**	**22**	**16**	**20**		

The largest (about 51%) abundance of benthic macroinvertebrates comprised by Notonectidae, Corixida, and Chironomidae taxa. These are all pollution tolerant taxa). The highest number of benthos individuals counted in taxa of Notonectidae (n = 1167), then followed by Corixida (n = 954), and the red Chironomidae (n = 801). The Baetidae (n = 113), Libellulidae (n = 131), Dytiscidae (n = 129), and Belestomatidae (n = 260) were also comparatively represented by high number of individuals. All the remaining taxa were low (<100 individuals). The Elimidae, Ceratopogonidae (n = 41), Tabanidae (n = 30), and Psychodidae (n = 20) taxa were represented by low number of animals. Ashnidae, Gerridae, and Naucoridae were the minor components (<0.5%) of the benthic communities and collected only 12, 9, and 5 individual, respectively (Table 6.3).

In LTA, the majority (70%) of the benthic composition comprised of the pollution tolerant taxa (such as red Chironomidae) as compared to that of sensitive ones. Among the EPT group (Ephemeroptera, Plecoptera, and Trichoptera) that are considering pollution sensitive taxa, only Ephemeroptera families (i.e., Baetidae and Caenidae) was found in LTA. These pollution sensitive taxa, Ephemeroptera, varied significantly among the sampling sites and between the study seasons (ANOVA; $p< 0.05$). This group prevailed extensively at the Reference site, indicating the relatively pristine nature of the area. Comparing the two sampling seasons, the number of Ephemeroptera specimen was high during the dry period coincides with low water turbidity and relatively high readings of electrical conductivity, pH, and dissolved oxygen.

6.4.3 Environmental factors and benthic communities

The correlation of the environmental parameters and benthic macroinvertebrates occurrence and abundance was analyzed using Constrained Redundancy Analyses (RDA graph) (Fig. 6.1) (Table 6.4). In the RDA graph, the first and second axes together explained 94.5% of the cumulative percentage variance of species-environment relation. The first axis and second axis explained 76.9 % and 16.5% of the species-environment relation, respectively (Table 6.4).

In RDA graph in Fig. 6.1 below, most of the inorganic nutrients were concentrated at the Bobodo riverside site. The site is an area of high sedimentation and siltation through runoff via Bobodo River from the southern corner. Both phosphorous nutrients (SRP and TP) and nitrogen nutrients

(NO_2 and NO_3), and water turbidity correlated positively and significantly with axis 1. But DO pH, and electrical conductivity were found to correlate positively and significantly with axis 2. Water temperature at each sampling site correlated positively and strongly with axis 1. As shown in Fig. 6.1, temperature and dissolved oxygen were correlated in the opposite direction.

Table 6. 4 Summary of the statistics of the DA diagram

Axes	**1**	**2**	**3**	**4**
Eigenvalues:	0.769	0.176	0.043	0.012
Species-environment correlations:	1	1	1	1
Cumulative percentage variance				
of species data:	76.9	94.5	98.8	100
of species-environment relation:	76.9	94.5	98.8	100

pH and DO were an important factor for the abundance of benthic taxa at the Reference site. The temperature was the prominent regulators for the abundance of benthic communities at Dacha riverside site relative to the other parameters; inorganic nutrients were also important factors at this site (Fig. 6.1). But pH and DO were major regulators for the occurrence and abundance of benthic macroinvertebrates at Main fish landing and Bobodo riverside sites (Fig. 6.1).

The most sensitive taxa having low tolerance value (Table 6.3) reported in LTA were Ashnidae (tolerance value = 3), Baetidae (tolerance value = 4), and Gyrinidae (tolerance value = 4). Naucoridae, Dytiscidae, Hydrophildae, and Elimidae benthic taxa are a moderately pollution tolerant taxa (tolerance value = 5) that were recorded in LTA. All these pollution sensitive groups account about 30% of the total of benthic taxa in LTA.

Sensitive taxa (Ashnidae, Baetidae, and Gyrinidae) and moderately tolerant taxa (Naucoridae, Dytiscidae, Hydrophildae, and Elimidae) correlated positively and strongly with electrical conductivity, dissolved oxygen, and pH. However, they correlated negatively with most of the inorganic nutrients (Fig. 6.1). All these pollution sensitive taxa concentrated at the Reference site as compared to other sites concurrently with minimal human-induced factors.

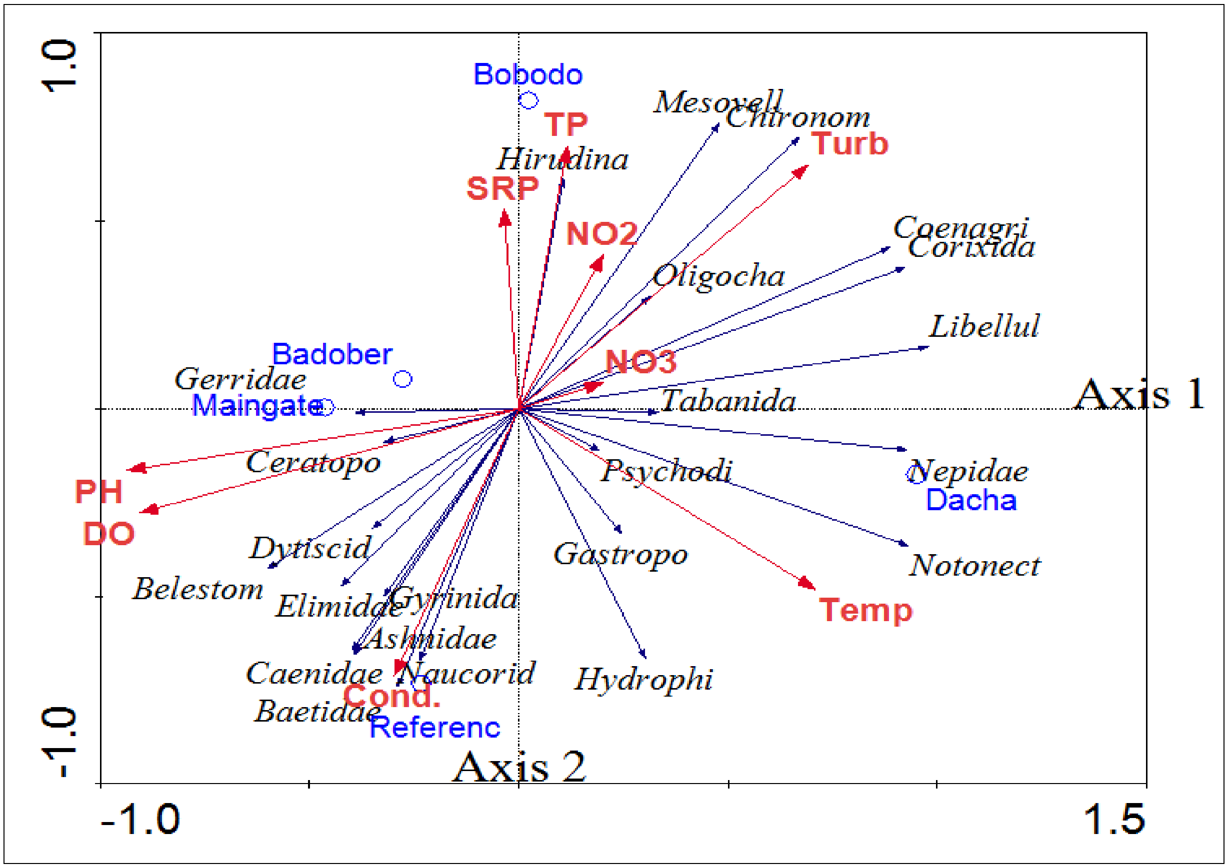


Fig. 6.1: Tri-plot of the constrained Redundancy Analyses (RDA, CANOCO for Windows 4) for benthic macroinvertebrates taxa (blue arrows) and environmental variables (red arrows) at each sampling sites (Blueprint). Mesovell-Mesovellidae, Hirudina-Hirudinae, Chironom-Chironomidae, Oligocha-Oligochates, Coenagri-Coenagrionidae, Corixida-Corixidae, Libellul-Libellulidae, Tabanida-Tabanidae, Gyrinida-Gyrinidae, Ceratopo-Ceratopogonidae, Dytiscid-Dytiscidae, Belestom-Belestomatidae, Naucoridae-Naucoridae, Notonect-Notonectidae, Gastropo-Gastropoda, Hydrophi-Hydrophildae, Psychodi-Psychodidae,

Hirudinae and Oligochaetes are pollution tolerant taxa having a tolerance value of 10 and 7, respectively (Table 6.3). Hirudinea correlated positively with SRP and TP, while the Oligochaetes correlated positively with NO_2 and NO_3. These taxa were high at Bobodo riverside site coincided with having high water turbidity. The other high pollution tolerant taxa found at this sampling site were Coenagrionidae, Corixidae, and Libellulidae. All these species had similar pollution tolerance value, which is 9 (Table 6.3). The occurrence of all the three tolerant taxa (Coenagrionidae, Corixidae, and Libellulidae) correlated positively and strongly with water turbidity and NO_3.

The red-blooded Chironomidae and Mesovellidae are high pollution tolerant taxa (tolerant value = 8) (Table 6.3). The two pollution tolerant taxa correlated positively and significantly with water turbidity (Fig. 6.1) and thus water turbidity seemed to be the possible reason for the remarkable occurrence and abundance of the Chironomidae and Mesovellidae. pH and DO seemed to influence the ecology of Belestomatidae and Ceratopogonidae taxa. Both are pollution tolerant taxa but the former is of higher pollution tolerance (10) than the later, which has a pollution tolerance value of 6 only (Table 6.3).

Gerridae (tolerance value =6) correlated negatively with most of the inorganic nutrients and this taxon was found high at the Main fish landing site (Fig. 6.1). In LTA, Notonectidae (Backswimmers) was widely dominated by the abundance of benthos specimens. This predominant taxon correlated positively and strongly with temperature, water turbidity, and nitrogen nutrient (NO_3). Notonectidae was preferable at Dacha riverside site concurrently with high temperature and water turbidity at that site.

6.4.3. Benthic macroinvertebrate diversity metrics

6.4.3.1 Taxa Richness (TR)

During this study, different macroinvertebrate diversity metrics were used (Table 6.5). There was no spatial effect on distributing the total number of taxa among the sampling sites (ANOVA; p >0.05). The total number of taxa at the Badober riverside site was high (22), followed by the intermediate (20 taxa) at the Reference site, and low (19 taxa) at Dacha riverside and Bobodo riverside sites (Table 6.5). Although statistically insignificant (ANOVA; p >0.05), the value of percent taxa richness (%TR) varies among sampling sites. The value of this metric ranged from a low value of 69.57% at the Main fish landing site to a high value of 95.65% at Badober riverside site (Table 6.5).

Table 6.5 Benthic metrics at the sampling sites in LTA

Sampling sites	Benthic macroinvertebrate diversity metrics					
	Taxa richness (% TR	Dominant taxa (% DT)	Ephenoptera (% E)	Chironomidae (% CHIR)	% HFBI	SDI
Dacha riverside	82.61	61.60	0.29	12.35	8.40	1.97
Bobodo riverside	82.61	40.30	1.05	24.54	8.42	2.09
Badober riverside	95.65	27.60	3.84	12.57	7.30	2.46
Main fish landing site	69.57	30.30	0.44	17.87	8.20	2.12
Reference site	86.96	36.80	12.15	3.43	6.90	2.23

6.4.3.2 Shannon Diversity Index (SDI) and Percentage Dominant taxa (% DT)

The Shannon Diversity Index varied among the sampling sites (ANOVA; $P<0.05$). The value of SDI was high (2.462) at Badober riverside with low (27.6%) %DT and low (1.97) at Dacha riverside site corresponding to high (61.6%) %DT (Fig. 6.2). The Reference site had higher SDI (SDI= 2.23) than Bobodo riverside site (SDI= 2.09) and Main fish landing site (SDI= 2.12). Usually, the SDI and %DT were inversely correlated (Fig. 6.1). Distributing %DT was appreciably differences among sampling sites. It was high at Dacha riverside (61.6%) and low (27.6%) at Badober riverside (27.6%) (Table 6.3).

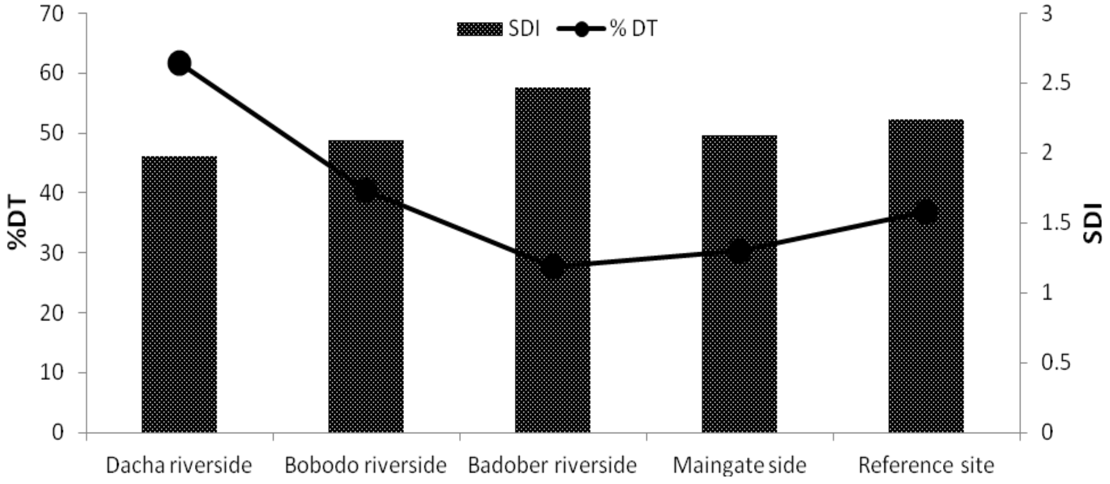


Fig. 6.2: Spatial distribution of the Shannon Diversity Index (SDI) and Percentage Dominant Taxa (% DT) at sampling sites in LTA.

6.4.3.3 Percentage Chironomidae (%CHIR)

The large abundance of Chironomidae is an indication of organic pollution and nutrient enrichment. In this study, the percentage of Chironomidae (%CHIR) considerably varied among the sampling sites. The lowest (3.45%) and highest (25.54%) values of these metrics were found at the Reference site and Bobodo riverside sites, respectively. Generally, high percent Chironomidae were recorded in the most of the sampling sites corresponding with 17.8%, 12.35%, and 12.57% at Main fish landing, Dacha riverside, and Badober riverside sites, respectively, indicating the sites were environmental stressed.

6.4.3.4 Percentage Ephemeroptera (% E)

Organisms in the Ephemeroptera, Plecoptera, and Trichoptera (EPT) groups are pollution sensitive and qualified as indicators of good water quality. In this study, Plecoptera and Trichoptera were not found. It was only Ephemeroptera families reported here (Table 6.3). However, Ephemeroptera in the study lake was, generally, low (<3%).

Distributing % E varied notably among sampling sites and between seasons. The percentage of Ephemeroptera was the highest (12.15%) at the Reference site. As compared to the Reference site, the percentage of Ephemeroptera was low in the other sites, having values of 0.29%, 0.44%, 1.05%, 3.84%, at Dacha riverside, Main fish landing site, Bobodo riverside, Badober riverside sites, respectively (Table 6.5).

6.4.3.5 Hilsenhoff Family-level Biotic Index

The Modified Family Biotic Index (FBI) was developed to detect organic pollution in these study sites. In LTA, most of the sampling sites showed high FBI values; suggesting comparatively low water quality. The value of HFBI ranged from 6.90 to 8.40. The value of this index was high at the Bobodo and Dacha riverside sites. The lowest value of this index was recorded at the Reference site (Table 6.5), indicating that this site is relatively good in water quality.

6.5 DISCUSSION

6.5.1 Physicochemical parameters

The various physicochemical parameters correlated with the abundance of benthic invertebrates in LTA along the study sites are put in Table 6.2. It is observed from the results that, most of the physicochemical parameters were noticeably high at all sampling sites. Considerably high levels of DO, pH and other physical parameters seemed to support the survival of most of the benthic invertebrate communities in LTA and its associated rivers. The distribution of most of the chemical and physical parameters varied significantly among the sampling sites. The water turbidity and all inorganic nutrients are high at Dacha riverside, Bobodo riverside, and Main fish landing site. But dissolved oxygen, pH, and electrical conductivity are relatively high Reference site and Badober riverside.

Dissolved oxygen is an important parameter that determines the spatial and temporal distribution of aquatic organisms as this is essential for their respiration (Samal *et al.*, 2009). In this study sites, dissolved oxygen was low at most of the sampling sites. This is because of the presence of relatively high temperature all the study sites and the numerous dissolved chemicals coming from the watershed. The RDA graph (Fig. 6.1) shows temperature and dissolved oxygen correlated negatively and strongly. More dissolved oxygen is present in water with a lower temperature compared to water with a higher temperature (Peeters *et al.*, 2002). The reason for this relationship between dissolved oxygen and temperature is that the solubility of the gas in a liquid is an equilibrium phenomenon (Mazumdar *et al.*, 2007). Bachmann *et al.*, (2000) considered that in smaller lake complete oxygen depletion was not observed apparently because of significant water movement through the lake. A similar fact was observed in all sampling sites of the present study because oxygen was not completely depleted at any site and season. The absence of complete depletion of oxygen in all the sampling sites may lead to the existence of various types of benthic macroinvertebrates in the study area.

The water temperature has been recognized as a major factor in the distribution, abundance and richness of aquatic organisms in general (Burgmer *et al.*, 2007) and benthic communities in particular (Fengqing *et al.*, 2012). Temperature also controls the dynamics of sustainability and survivorship of various aquatic organisms that live in lakes though different life forms have a recommended temperature variety (Samal *et al.*, 2009). Temperature affects the physiological processes of organisms, and hence temperature dynamics may change life cycle patterns and trophic interactions (Klanderud and Totland, 2005). The higher the water temperature, the greater the bio-geo-chemical activity influenced by the environmental intrinsic and extrinsic parameters (Samal *et al.,* 2004). In this study, a considerable high temperature has recorded at all sampling sites. This marked high temperature might have induced higher metabolic processes in DO rich environment and hence the high proliferation of or occurrence of high benthic organisms in all the study sites.

Other important physical parameters considered to regulate the abundance and diversity of benthic macroinvertebrates in the present study are pH and electrical conductivity. The variation in the pH could have an effect on biodiversity and richness (Hopkins *et al.,* 1989). Acidification has severely affected freshwater resources (Wigington *et al.,* 1996). Experiments conducted at a range of spatial scales, from laboratory experiments to whole-stream manipulations, demonstrate adverse effects of acidic pH on benthic invertebrate communities and differences in sensitivity among aquatic insects (Hopkins *et al.,* 1989). In the present investigation, the values of pH show that the water of all the sampling sites are alkaline and acidification has not been observed since pH is always greater than 7. The absence of acidic conditions in the study area may elucidate diverse types of macroinvertebrates in large numbers (Table 6.3 and Fig. 6.1).

There were no remarkable differences in pH among the sampling site. But it was high at the Reference site corresponding to low human perturbation and low at Dacha riverside site where high human perturbations occurred. A relatively higher mean value of pH has been observed during the dry period compared to the wet period. The decrease in pH values during the periods of rainy season owes its occurrence to the dilution caused by the rainwater and may result from the production of CO_2 from increased biological oxidation and photosynthetic processes of aquatic plants that may have ultimately contributed to the reduction of pH (Melack, 1996).

Electrical conductivity (EC) is much of salinity in water. It has long been considered an important factor in determining aquatic macroinvertebrate fauna in lentic and lotic systems (Mathi and Dorris, 1968; Hart *et al.,* 1991). In the present study, relatively high electrical conductivity has been recorded at all the sites. The distributions of these ionic concentrations have gone from high values at the minimally affected site (Reference site) to low values at high affected sites (Dacha riverside and Bobodo riverside). This might have supported one way or the other the survival of most of the benthic macroinvertebrate communities in the indicated sites along the LTA.

High turbidity was recorded at all sampling sites of benthic macroinvertebrates. One of the plausible reasons for high turbidity is the entrance of high sediment-laden from the agricultural area through runoff via the Feeder Rivers. The two Feeder Rivers Dacha and Bobodo were actively utilized during the sampling periods, mainly in the rainy months of July and August. These rivers carry complex types of sediments and garbage from the Guragae highlands where large settlements and towns such as Worabie to end up in LTA.

The best-known adverse consequence of agriculture on watersheds is the large sediment exported into adjacent rivers and streams (Suren and Jowett, 2001). Sedimentation in aquatic ecosystems is generally accompanied by deterioration of water quality, reduced light penetration, and the filling of interstitial spaces in benthic substrates (Cre´taz and Barten, 2007). During the present study, high agricultural activities are underway in the watershed of LTA. The lake rarely has any buffer zone as such. The agricultural activities that exposed the topsoil have promoted the transport of sediments that directly end up in LTA. It is understandable that all these factors affect the diversity and abundance of benthic communities in the study lake.

Streambed degradation by sedimentation can ultimately result in the extinction of benthic macroinvertebrates because these organisms predominantly dwell on stream substrates and require specific geomorphological structures for completion of oviposition and other life cycle stages (Quinn and Stroud, 2002; Miserendino and Pizzolon, 2004). Thus, the overall success of diverse benthic macroinvertebrate communities depends strongly on the integrity of their physical environments (Rempel and Church, 2009).

As Jun *et al.,* (2011) summarized in their paper on the effects of land use on benthic macroinvertebrate communities in the comparison of two mountain streams in Korea, different studies have documented multiple adverse effects of sedimentation on benthic macroinvertebrates, such as decline in species diversity and density (Quinn *et al.,* 1992), reduction in the number of suitable habitats (Bojsen and Jacobsen, 2003; Brierley and Fryirs, 2005), declines in food resources and food quality (Broekhuizen *et al.,* 2001), reduction in feeding and respiration (Stewart *et al.,* 2001), and increases in drifting rates of the organisms (Suren, and Jowett, 2001).

The major inorganic nutrients that occurred in appreciable proportions in the present study sites are NO_2 and NO_3 and SRP and TP. In all the sampling sites, all these nutrients are the preeminent regulating factors for the diversity and abundance of benthic communities. All the nutrients measured during the study were relatively high at all the sampling sites that support the existing invertebrate taxa. The anthropogenic activities are believed to be the main sources of high nitrogen and phosphorous loads into the lake resulting in high eutrophication along the shores of this site, which affected the survival of benthic macroinvertebrates at this site.

There were noticeably clear variations in the changes on benthic abundance along nitrogen and phosphorus gradient among sampling sites. Some invertebrate taxa were less pronounced on nutrient-poor sites (Reference site and Badober riverside site). For instance, Ephemeroptera showed significant declines with nutrient enrichment. Redundancy Analyses (RDA) graph (Fig. 6.1) also revealed a clear distinction in taxonomic composition and abundance between Reference sites (not nutrient enrich site) and other nutrient enriched sites.

6.5.2 Diversity of benthic macroinvertebrates

Diversity values for real communities are often found to fall between 1.0 and 6.0 (Stirling and Wilsey, 2001). Staub *et al.,* (1970), had also set diversity index less than 1 for poor diversity, 1-2 moderate diversity, 2<3 for average diversity and >3 for high diversity. Based on the value of Shannon index, Dacha riverside site is moderately diversified; whereas all the remaining sampling sites are diversified on average. Because of the SDI considered among study sites, the diversity of benthic macroinvertebrates in LTA is bounded from moderate to average diversity of benthic macroinvertebrate communities. This average diversity of the benthos in the study area might be due to the existence of good coverage of aquatic macrophytes at the littoral area of most of the sampling sites, presence of high level of ambient inorganic nutrients, relatively high water temperature, the alkaline nature of the lake, and possibly the weathering condition of the surrounding area.

Shannon Diversity Index (H') decreased as the perturbation increased (Salusso and Moraña, 2000). According to the study by Salusso and Morana (2000), there are three classes of pollution status based on H'. In their scale, water bodies with H′ value of over 3 have no contaminant, H′ values ranging from 1-3 contain moderate contaminants and H′ <1 indicates high pollution level. In LTA, the H' of benthic macroinvertebrate ranged from 1.97 at Dacha riverside site to 2.462 at Bobodo riverside site; indicating that all the sampling sites are moderately polluted. This leads to the inference that LTA is moderately polluted in according to the values of SDI. Different shoreline modifications (crop cultivations, irrigation farms, disposal of wastes, sedimentation, and siltation) on the watershed area of the lake have been observed during the meantime of this study. These resulted in the high rate of organic pollutions.

In this study, the number of benthic macroinvertebrate taxa or richness is comparatively similar among the sites and the changes have no meaningful differences among them but the composition of each taxon with their tolerance value among sites tells a lot about the impacts. Plafkin *et al.,* (1989) stated that sites with greater than 26 taxa can be non-impacted, 19-26 as slightly impacted, 11-18 as moderately impacted and 0-10 as severely impacted. Based on these criteria, all the sampling sites considered in the present study area (Table 6.5), fall in the slightly impacted range. Considering the total benthic macroinvertebrates taxa (19-23) that have been observed from all sampling sites, the ecological condition of LTA is slightly affected because of different human perturbations such as irrigation activities at the nearshore area, bathing, and washing, release waste, and other related possible suggestions.

The total number of benthic individuals was considerably different among the study sites. These remarkable differences in the abundance of benthic communities among the study sites are probably because of differences in the physic-chemical relations (RDA graph (Fig. 6.1) and because of the differences in the coverage of aquatic macrophytes. The largest number of invertebrate communities was recorded at Dacha and Bobodo riverside sites concurrently with a high level of all types of inorganic nutrients, the occurrence of high temperature, and increasing water turbidity. During this study, relatively good coverage of aquatic macrophytes has been observed at Dacha and Bobodo riverside sites. This ultimately resulted in the high number of specimens found at the two indicated study sites.

Compared to the two sampling seasons, high numbers of specimens have been recorded during the wet season than in the dry season. The difference in water temperature between the two sampling seasons resulted in the seasonal variations in the number of benthic individuals. The seasonal difference in the number of benthic individuals may be also a significant difference in habitat coverage between the two seasons. During the time of data collection, the littoral macrophytes were condensed during the rainy time compared to the dry period. *Cyperus dubius*, *Persicaria senegalensis,* and *Potamogeton* sp. were dominant in the littoral region of the lake during the rainy season (June-August). However, all the larger macrophytes were apparently disappeared in most of the dry season (January-April), with the water regressing back and the water level dropping that left the macrophytes exposed to dry.

In LTA, Hemiptera (true bugs) family particularly, the Notonectidae and Corixida are predominant taxa that represented the largest number of benthic specimens. Hemipterans are relatively tolerant to many forms of pollution (Bouchard, 2004). Surface dwellers, have little physical connection with water and therefore are less dependent on the water quality (Gooderham and Tysrlin, 2002). This feature of Hemiptera might have resulted in their wide dominance in the study area.

The Notonectidae, or commonly known as Backswimmers, are extensively dominated the number of benthic individuals. Backswimmers can fly easily and can, therefore, colonize new wetlands quickly. They are very active predators feeding on a mixture of invertebrates from the open-water and the water surface. Their ability to colonizing new environments and feeding on open-water and the surface with predating behaviors might have facilitated a high number of individuals in the study lake. Notonectidae is correlated strongly and positively with water temperature (RDA graph in Fig. 6.1). Fairly high temperature is likely the cause for the higher abundance of the Notonectidae than other benthic individuals in LTA.

Diptera families (particularly the red Chironomidae or non-biting midges) are other important benthic invertebrates comprising a large number of specimen next to the Hemiptera in the study sites. The extreme variability of dipterans extends to their tolerance to pollution (Bouchard, 2004). Some animals among the dipterans actually prefer organically polluted sites and thrive in sewage and carrion (Gooderham and Tysrlin, 2002). The same plausible reason was given for the dominance of Diptera in general and the high abundance of Chironomidae in particular in the present study sites. Diverse and well-studied families such as the Chironomidae can provide much information about their surrounding environment. Different species indicate different environmental conditions such as water chemistry. Sometimes these changes also indicate environmental impacts. In extreme cases of pollution, the deformities of the more tolerant species can be studied to assess environmental impacts.

In LTA, pollution tolerant taxa dominated the total benthic invertebrates at the high affected sampling site. These families belonged to Notonectidae, Chironomidae, and Belestomatidae with tolerance ratings ranging from 8 to 10 (See Bouchard, 2004) (Table 6.3). The Notonectidae, Chironomidae, and Belestomatidae are the dominant benthic taxa at Dacha rivers side, Bobodo riverside, and Main fish landing sites. All these sites are anthropogenically highly modified with shoreline modification.

The high percentage of a single or some common taxa at the affected sites, particularly at Dacha and Bobodo riverside sites might be due to the instability of the area by the inflow of rivers Dacha and Bobodo runoff. However, Bode *et al.*, (1996), point out that some unstressed habitats are dominated by only a few taxa due to habitat, flow, and seasonal effects. The better stability in Badober and Reference sites promoted the low percentage dominance; moreover promoted the relatively high number of sensitive groups than the tolerant ones, indicating reasonably better water quality of the site.

According to Barbour *et al.*, (1996), percent dominant taxa (%DT) above 45 indicates an impaired, 40-45 possibly impaired and less than 40 unimpaired conditions of the site. Based on this classification, %TD at Dacha riverside site is greater than 40 indicating the site is anthropogenically impaired, Bobodo riverside site is within 40-45 indicating the possibly impaired site. %DT at Badober riverside and Reference sites is less than 40, indicating that the two sites are unimpaired conditions (Table 6.3).

The presence or absence of certain taxa is related to water quality rather than other ecological factors (Mason, 1996). According to Hilsenhoff (1988), the presence of pollution sensitive groups rather than pollution tolerant groups can indicate better water quality because pollution tolerant groups can inhabit both habitats according to their niche preference. During the present study, the benthic community at and around LTA is broadly dominated by pollution tolerant taxa as compared to pollution sensitive taxa. This clearly indicates the poor water quality of the lake in relation to benthic macroinvertebrate assemblages.

Pollution-sensitive groups varied among sampling sites and seasons. It is commonly found at the Reference site and rainy period. The pollution sensitive groups are low in abundance during the rainy period. At this time, high organic pollution from the watershed entered to the littoral region of the lake through runoff. This resulted in the high occurrence of organic pollution to which these pollution sensitive groups have a low tolerance (Bouchard, 2004), and hence a low number of specimens have been reported during the wet season than in the dry one.

The percentage composition of high pollution tolerant communities such as Gastropods, Odonata, Diptera, Hirudinae (Leeches), and Oligochaetes (the aquatic earthworm) are high at the Dacha riverside, Bobodo riverside and Main fish landing sites where high human interference exists. A significant number or percentage of Chironomidae in the study lake shows the existence of organic pollution that resulted in less water quality indicating the lake was under long term environmental stress.

Hilsenhoff Family Biotic Index (HFBI) is used to detect organic pollution. All study sites considered in this study, showed relatively high FBI values; suggesting low water quality. Relatively lower FBI was calculated at Reference site; suggesting comparatively high water quality and high FBI was estimated at Badober riverside site and Dacha riverside site; suggesting comparatively low water quality. As Hilsenhoff (1988) noted, HFBI values of 0-3.75 indicate excellent water quality. HFBI values of 3.76-4.25, 4.26-5, 5.01-5.75,5.76- 6.5, 6.51-7.25, and 7.26-10 indicate very good, good, fair, fairly poor, poor and very poor water quality, respectively. In this study, the HFBI value at Reference site represented poor water quality while the values at all the other sites represented very poor water quality in relation to HFBI.

Generally, the study of benthic macroinvertebrates along the different sites of LTA is an important aspect for the judgment of the water condition of the lake. The results benthic assemblage shows there is the existence of organic. This indicates LTA has undergone environmental stress which obviously calls for participatory and interdisciplinary actions of watershed management and wise use of aquatic resources with compliance to commonly set national agenda of sustainable use of natural resources.

CHAPTER 7: FEEDING HABITS AND RELATED ASPECTS OF NILE TILAPIA (*OREOCHROMIS NILOTICUS* L.) IN LAKE TINISHU ABAYA

7.1 Introduction

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). 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.*, 2014).

Quantitative determination of the components of the diet, their nutritive value, and seasonal availability are the basic parts for an understanding of environmental impacts on the condition and growth of fish (Kamal *et al.*, 2010). Therefore, an understanding of fish diet and its influence on growth can be essential for the understanding of the ecological role and the productive capacity of fish populations (Alemayhu Negassa and Prabu, 2008). Studying feeding can be also considered as an important step to make growing fish more successful (Shipton *et al.*, 2008).

Based on the request of the Siltie Woreda Bureau of Agriculture, National Fisheries and other Aquatic Life Research Center (NFALRC), Sebeta, Ethiopia, *Oreochromis niloticus* fry were stocked into LTA in 1997, to develop fishery and availing inexpensive fish protein to the local community (Kassahun Asaminew *et al.*, 2011). Following the stocking, Kassahun Asaminew *et al.*, (2011) conducted a study on the adaptability, growth, and reproductive success of *O. niloticus* stocked in the lake. However, there was no information on the feeding ecology of *O. niloticus* in the lake and thus, studying the feeding habit can be a guide for the management of the stock. The aim of this work was, therefore, to explore the feeding habits, seasonal variation in diet composition and ontogenetic dietary shift of *O. niloticus*, in LTA, to observe the suitability of the lake ecosystem for the introduced fish.

7.2 Materials and Methods

7.2.1 Fish sampling and measurements

Seasonal fish samples from March to May (dry season) and July to September (wet season) in 2017 were collected using gill nets of 6 cm, 8 cm, 10 cm, and 12 cm stretched mesh sizes. To obtain fingerlings, beach seine (4 cm mesh size) was used in the shallower part of the lake. In addition, fish were purchased from the commercial landings of the fishermen (catch with 10 cm & 12 cm mesh size net). Total length (TL) and total weight (TW) of all fish specimens were measured using a measuring board and a sensitive balance (mark: WEI-IB 413C, Shangahi) to the nearest value of 0.1 cm and 0.1g, respectively. Length-Length and Lenght-Weight of the collected fish samples were estimated according to Pauly (1984). The well being of fishes was studied using Fulton´s Condition Factor (FCF) (%) based on the estimation method of Pauly (1984).

An equation used in the estimation of length-weight relationships, condition factor, and sex-ratio of *O. niloticus* was:

Length-Weight relationship of fish was estimated according to Pauly (1984) as the following equation (Eq. 7.1):

$$\mathbf{TW = aTL^b} \quad \text{..Equation.7.1}$$

TW is the total weight of fish (g), TL is the total length of fish (cm), *a* is the regression constant, b is TLTW coefficient

To find the slope of the regression (b) the equation was transformed into the logarithmic function as:-

$$\mathbf{logTW = loga + blogTL} \quad \text{……..........................…… Equation.7.2}$$

Fulton´s condition factor (FCF) (%) was calculated using the formula of Pauly (1984) in equation 7.3 below

$$\mathbf{FCF(k) = \frac{100TW}{TL^3}} \quad \text{…...Equation.7.3}$$

Where TW is the total weight of fish (g), TL is the total length of fish (cm), *a* is the regression constant, b is TLTW coefficient

7.2.2 Gut content analysis

In the field, the gut content of each fish was kept in labeled sampling bottles containing 5% formalin solution. The gut contents were then examined using a dissection microscope (Model: Leica, DME) and also a compound microscope (Leica DME) with a magnification of 100x. The relative importance of the different food items found in the gut contents was determined using the frequency of occurrence and volumetric methods (Bowen, 1980; Hyslop, 1980).

Frequency occurrence: The number of examined guts in which one or more of a given category of food items was found was expressed as a percentage of the total number of non-empty guts (Windel and Bowen, 1978). This method gives information on the proportion of a particular population of fish that fed on that particular food item. The proportion of the population that feeds on a particular food item was estimated and the frequency of occurrence was calculated (Hyslop, 1980; Bowen, 1983) as equation 7.4 below:

$$\mathbf{F}\boldsymbol{i} = \mathbf{100 * n}\boldsymbol{i}\mathbf{/n} \quad \text{eq.7.4}$$

where

F*i*: Frequency of occurrence of the *i* food item in the sample

n*i*: Number of stomachs in which the *i* item is found

n: Total number of stomachs with food in the sample

Volumetric analysis: The food items that were found in the guts were categorized into different taxonomic groups and the volume of each group was measured (Bowen, 1983). Then the volume of a given category of food items was expressed as a percentage of all the categories of food items present in the samples. The importance of different food items for different size classes was determined by dividing the fish into four size classes (I, < 10 cm TL, II, 10-19.9 cm TL, III, 20-29.9 cm TL and IV, > 30 cm TL) and determining percentage mean volume of food in each size class.

7.2.3 Statistical analysis

Chi-square test ($\chi 2$ test) was used to compare the frequency of occurrences of the different food categories during the different seasons. The non-parametric Mann Whitney’s U-test was used to compare the volume of the different food categories consumed during the seasons. ANOVA was used to see the significant variation in the condition factor between sexes. Dietary overlap between different length-classes was calculated using Schoener Diet Overlap Index (SDOI) (Schoener, 1970) using the following formula:

$\alpha = 1 - 0.5(\sum_{i=1} n\,|pxi - pyi|)$...Equation.7.6

Where α is percentage overlap, SDOI, between length classes *x* and *y*, *pix* and *psi* are proportions of food category (type) *i* used by length classes *x* and *y,* and *n* is the total number of food categories. Overlap in the index is generally considered biologically significant when the value α exceeds 0.60 (Mathur, 1977).

7.4 RESULTS

7.4.1 Length-weight relationship, condition factor and sex ratio

428 fish samples from 2.5 to 30.9 cm in TL and 1.1 to 475 g in TW were used to determine length weight and length-length relationships. Total length (cm) and standard length (cm) relationship of *O. nilotucus* in LTA was linear and significant ($R^2 = 0.962$, $TL = 0.1456SL + 1.8088$). Whereas, the relationship between total length and the total weight of *O. niloticus* was curvilinear with a strong relationship ($R^2 = 0.9848$, $TW = 0.0194\ TL^{2.9876}$ (Fig. 7.1). The slope of the regression (b) was to be 2.9876 closer to the isometric growth value of b=3 of fish. The intercept (a) is 0.0194 g/cm and it signifies the average condition factor index of Tilapia in LTA.

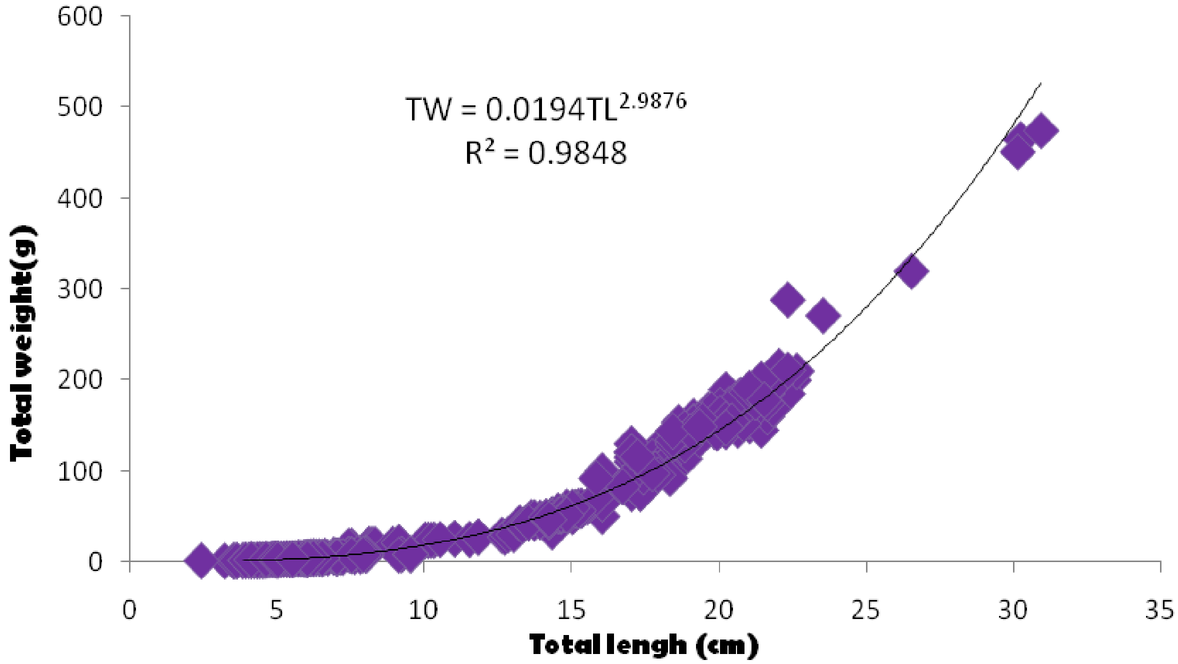


Fig. 7.1 Total length (cm) and total weight (g) relationship of *O. niloticus* in LTA (n=428)

There was significant variation in the number of fishes caught in the sampling season ($\chi^2 = 6.14$; $p < 0.05$). A high number of individual fishes was collected during the rainy season. There was no significant difference between sexes in the total catch ($\chi^2 = 3.65$, $p > 0.05$). But the number of males was slightly higher than females. The sex ratio (Female: Male) of the total fish catch was 1:1.4 which deviated a little from the theoretical or expected 1:1 ratio. The study discovered that the mean Fulton´s Condition Factor (FCF) of *O. niloticus* in LTA was 1.88 and it ranged from 0.96 to 3.51. There was significant variation in FCF between the two sexes (ANOVA, $P < 0.05$). High value was observed for males (mean FCF=2.21) than females (mean FCF=1.93).

7.4.2 Diet composition

Among the 428 gut contents used, 55 (12.85%) were empty and 373 (87.15%) were observed with one or more food items (Table 7.1). Phytoplankton, detritus, and zooplankton were usually the dominant food items in the fish gut. Macrophytes were intermediate; while insects, nematodes, fish scales, and ostracods were low. Phytoplankton constituted the largest component of the diet that occurred in 351 (94.11%) followed by detritus in 277 (74.26%) and zooplankton in 267 (71.58%) fish specimen. Macrophytes have moderately contributed to the frequency of occurrence of the food items and they occurred in 251 (53.9%) of the gut contents (Table 7.1). Volumetrically, the maximum total food items were constituted by phytoplankton (39.49%), followed by macrophytes (25.81%), detritus (14.39%) and zooplankton (12. 96%) (Table 7.1).

Phytoplankton, detritus, zooplankton, and macrophytes were the most important food items of *O. niloticus* in LTA. Other than the four major food items, insects, nematodes, ostracods, and fish scales made up a minor portion of the diet of *O. niloticus*. The former contributed 20.9% and 4.74% of the total gut content and the total volume of food items, respectively. The diet compositions of nematodes, ostracods, and fish scale found in the gut of *O. niloticus* accounted for 5.1%, 13.7%, and 2.9%, respectively. Volumetrically, they accounted for only 0.12 %, 2.32%, and 0.17 % of the total food items, respectively (Table 7.1).

Among the phytoplankton, blue-green algae (*Microcystis, Cylindrospermopsis, Anabaena,* and *Oscillatoria*), green-algae (*Pediastrum, Cosmarium, Ankistrodesmus, Botryococcus,* and *Coelastrum*), and diatoms (*Fragilaria, Pinnularia, Navicula, Nitzschia, Synedra, Cyclotella*) comprised most of the algae consumed by *O. niloticus.* The frequency of occurrence of blue-green algae, green algae, and diatoms was 85%, 79.4%, and 74.3%, respectively. Blue-green algae constituted the maximum value by volume (18.21%) followed by green algae (12.35%) and diatoms (5.31%) (Table 7.1).

The small-bodied rotifer (*Branchionus, Keratella*, and *Filinia* species) were the most important groups of animal origin in the diet of *O. niloticus*. The frequency of occurrence of rotifers was 67.3%. However, volumetrically accounted for relatively low volume (6.25%) of the total food items. The two large-bodied microcrustaceans (copepods and cladocerans) contributed a moderate amount of the food content of the fish gut next to rotifers. Copepods (*Mesocyclpos aequatorialis* and *Thermocyclops decipins*) and Cladocerans (*Diaphnosoma excisum, Daphnia barbata*, and *Moina micrura*) frequently occurred in most of the gut sample. The frequency of occurrence of copepods and cladocerans was 48.5% and 31.9%, respectively. Volumetrically, copepods comprised 4.12% and cladocerans accounted for 2.58% of the total volume of the food items (Table 6.1).

Table 7.1: The frequency of occurrence and volumetric contribution of food items consumed by *O. niloticus* in LTA **(**n=373). Note 'a' represents the food items significantly contributed to the diet of the fish.

	Frequency of occurrence		**Volumetric contribution**	
Food items	**Occurrence**	**Percent**	**Volume (ml)**	**Percent**
Phytoplankton	**351**	**94.1**[a]	**387**	**39.49** [a]
Blue green algae	317	85	178.5	18.21
Green algae	296	79.4	121	12.35
Diatom	277	74.3	52	5.31
Euglenoids	22	5.9	5.5	0.56
Others	183	49.06	30	3.06
Zooplankton	**267**	**71.6** [a]	**127**	**12.95** [a]
Copepods	181	48.5	40.4	4.12
Cladoceran	119	31.9	25.3	2.58
Rotifers	251	67.3	61.3	6.25
Macrophytes	**201**	**53.9** [a]	**253**	**25.81** [a]
Detritus	**277**	**74.3** [a]	**141**	**14.39** [a]
Insects	**78**	**20.9**	**46.5**	**4.75**
Diptera	57	15.3	16.2	1.65
Ephemeroptera	40	10.7	9.1	0.93
Coleoptera	13	3.5	7.5	0.77
Hemiptera	47	12.6	8.1	0.83
Plecoptera	8	2.2	5.6	0.57
Nematodes	**19**	**5.1**	**1.2**	**0.12**
Ostracods	**51**	**13.7**	**22.7**	**2.32**
Fish scales	**11**	**2.9**	**1.7**	**0.17**

7.4.4 Seasonality in the diet of O. niloticus

There was notable variation in the diet and the proportions consumed by *O. niloticus* in LTA during the dry (March-May) and wet (June-August) seasons. The frequency of occurrence of macrophytes significantly varied during the dry (n= 134) and wet (n= 239) seasons ($\chi2$-test, $P<0.05$). Similarly, the volumetric contributions of macrophytes, phytoplankton, and detritus significantly differed during the two seasons (U-test; $P <0.01$). During dry period phytoplankton extensively dominated the fish gut. They occurred in 98.5% of the gut and constituted 56.5% of the total volume of food items. But, their contribution declined to 32.9% of the total volume during the wet season.

Unlike phytoplankton, the contribution of macrophytes was lower during the dry season than the wet season. During the dry season, macrophytes were found only in 20.2% of the gut contents and constituted 13.8% of the total volume of food items. But, during wet months macrophytes occurred in 72.8% of the gut contents and constituted 30.62% of the total volume of food items (Table 6.2). Detritus was the second most important food item in the diet of *O. niloticus* in the wet season. It occurred in 56.7% and 84.1% of the gut content during the dry and wet seasons, respectively. Volumetrically, detritus comprised 11.6% in the dry season and 15.5% in the wet season from the total food bulk.

Like that of the macrophytes and detritus, the contribution of zooplankton was higher in the wet season than the dry season. They occurred in 52.2% and 82.4% of the gut during dry and wet seasons, respectively. Volumetrically, zooplankton constituted 2.9% in the dry season and 17% in the wet season (Table 7.2). Comparing the two seasons, the contribution of insects was relatively low during the dry months, occurring in 8.2% of the gut contents examined. However, it accounted for a relatively high percentage (17%) of the total volume of the food bulk. During the wet months, the contribution of insects was comparatively high occurring in 28% of the gut examined. Volumetrically, the contribution was insignificant (0.8%) in the wet season. The contribution of nematodes and ostracods to the diet of *O. niloticus* was relatively higher during the wet season than the dry season. The reverse was true for fish scale composition. Ostracods contributed 6.7% and 17.6% of the gut content of the fish diet in the dry and wet months, respectively. The contributions of nematodes and fish scales were generally negligible.

Table 7.2: Relative percentage contributions (%) of different food items in the diet of *O. niloticus* during dry (n= 134) and wet (n= 239) season from LTA. Note that the sum of the major categories of food items in bold add up to 100% in volumetric analysis. Note: 'a' represents the food items which significantly contributed to the diet of the fish in the two seasons.

	Frequency of occurrence (%)		**Volumetric contribution (%)**	
Food items	**Dry season**	**Wet season**	**Dry season**	**Wet season**
Phytoplankton	**98.51**^{a}	**91.63**^{a}	**56.5**^{a}	**32.97**^{a}
Blue green algae	97.76	77.82	28.0	14.43
Green algae	96.27	69.87	9.01	12.01
Diatom	69.40	76.99	4.43	4.68
Euglenoids	5.97	5.86	0.69	0.51
Others	78.36	32.64	14.37	1.34
Zooplankton	**52.24**^{a}	**82.43**^{a}	**22.22**	**16.95**^{a}
Copepods	29.853	58.99	3.34	4.43
Cladoceran	8.21	45.19	0.65	10.57
Rotifers	82.84	58.58	18.23	1.95
Macrophytes	**20.15**	**72.80**^{a}	**13.80**	**30.62**^{a}
Detritus	**56.72**^{a}	**84.10**^{a}	**11.62**	**15.52**
Insects	**8.21**	**28.03**	**16.89**^{a}	**0.77**
Diptera	30.6	6.69	4.10	0.77
Ephemeroptera	24.63	2.93	2.72	0.7
Coeoptera	9.70	0	2.72	0
Hemiptera	33.58	0.84	2.03	0.36
Plecoptera	5.97	0	2.03	0
Nematods	**2.24**	**6.69**	**0.11**	**0.13**
Ostracods	**6.72**	**17.57**	**0.88**	**2.89**
Fish scales	**4.48**	**2.09**	**0.22**	**0.16**

7.4.5 Ontogenetic Dietary Shift

Dietary overlap between different length-classes of *O. niloticus* for LTA was calculated using Schoener Diet Overlap Index (SDOI, α). The importance of different food items from different size classes was determined by dividing the fish into four size classes (I, < 10 cm TL, II, 10-19.9 cm TL, III, 20-29.9 cm TL and IV, > 30 cm TL) (Fig. 7.2). Percent means volume contribution of different food items with size class of *O. niloticus,* indicated that there were high variations in the diet of individuals in the size class I and III (α= 0.556), I and IV (α= 0.491), and II and IV (α= 0.559). Overlap of food items was also seen as the size class was closer to each other and it was shown in the size class I and II (α =0. 692), II and III (α =0.783), and III and IV (α =0.661).

Fig. 7.2, below, indicated that the ontogenetic diet shift was evident. In the first size class (< 10 cm TL), the dominant food items were zooplankton (45.7%) followed by phytoplankton (21.3%) and detritus (11.2%). In that size class, the contribution of insects, macrophytes, nematodes, ostracods, and fish scales was relatively low. They constituted 7.7 %, 4.5 %, 5.2%, 3.3%, and 1.1% of the total volume of the food items, respectively (Fig. 7.2). Of the zooplankton community, rotifers contributeted the highest percentage volume (40.7%) of the food bulk at that size class. Phytoplankton and detritus were essential food sources of plant origin in the diet of *O. niloticus* in the study lake in that size class of *O. niloticus*. They contributed a fairly high proportion of the total food items. Generally, in this size class, 78.2% of the total volume of food consumed by the *O. niloticus* was zooplankton (particularly rotifers), phytoplankton, and detritus. The remaining food items constituted 22% of the total volume of food items.

In size class 10.0-19.9 cm TL of fishes, foods of plant origin was most important than animal origin. Phytoplankton, macrophytes, and part of the detritus comprised 33.7%, 11.5%, and 22.6% of the total volume of the food items consumed, respectively. But the contributions of animal origin foods declined. However, zooplankton accounted for relatively high (23%) for the total volume of the diet. Insects comprised low (5.5%) volume of the food items. The less important food items namely, nematodes (1.2%), ostracods (2%), and fish scales (0.5%) collectively constituted less than 5% of the volume of food consumed within the size class.

Foods of plant origin accounted for the highest proportion of the food items in the 20-29.9 cm TL size class of fishes. The dominant food items found in the gut of the size class 20-29.9 TL were phytoplankton (47.6%), macrophytes (18.3%), and part of the detritus (15.5%). These three food items collectively constituted 81.4% of the total food bulk within the size class. In this size class, the contribution of foods of animal origin was low. Zooplankton and insects origin of food comprising about 10.5% and 6% of the total volume of food items, respectively. The contributions of nematodes (0.65%), ostracods (0.5%) and fish scales (1%) were negligible and collectively constituted only 2% of the total volume of food consumed in that size class (Fig. 7.2). In the largest size class (>30 cm TL), the order of the importance of dominant food items was toward Macrophytes (40.5%) >detritus (25.9%) > phytoplankton (20.4%), in that order. The three dominant food items collectively contributed about 87% of the total food items. Foods of animal origins were relatively less important in this size class. The contribution of zooplankton, insects, nematodes, ostracods, and fish scales were 4.8%, 5.2 %, 0.4%, 1.5%, and 1.3% of the total volume of food items consumed, respectively.

Generally, the dietary overlap between different length-classes of *O. niloticus* showed that the importance of phytoplankton, macrophytes, and detritus increased with size; whereas the importance of zooplankton, insects, and other animal origin food declined with fish size. Phytoplankton, zooplankton, detritus, and macrophytes were the dominant food items in all size classes, whereas the contribution of insects, nematodes, ostracods, and fish scales was low.

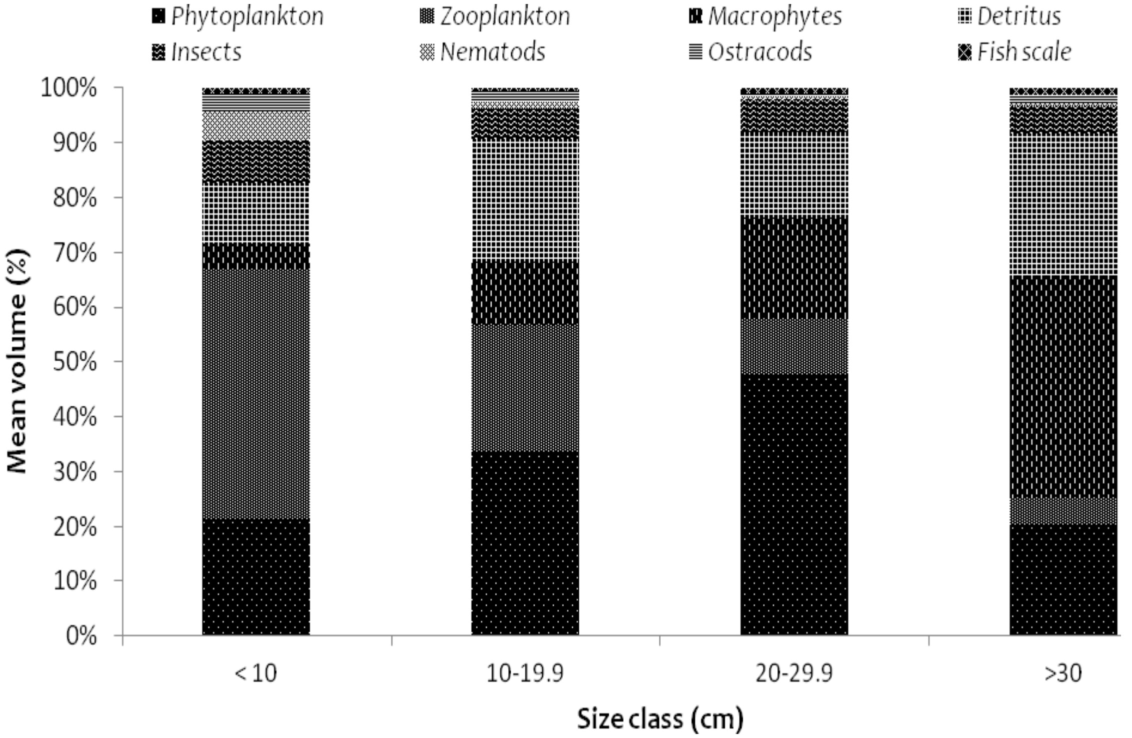


Fig. 7.2: The relative proportion (%) of different prey items in the diet of *O. niloticus* at different size classes from LTA.

7.5 DISCUSSION

7.5.1 Length-weight relationships and condition factor

The relationship between TL and SL of *O. niloticus* in LTA is linear indicating either of the length measurement could be converted to the other easily because of their strong relationship. The estimated length weight coefficient (b) value of *O. niloticus* in the study area (b=2.9876) is within the range of b values (2-4) in fishes in general (Bagenal and Tesch, 1978) and in tropical fishes (b=2.5-3.5) in particular (Gayannilo and Pauly, 1997). The slope of the regression line (b) indicated isometric growth pattern of the fish; where its shape and specific gravity do not change as the fish grows in size (Elias Dadebo *et al.,* 2014). It may be due to the growth that occurs at the same rate for all parts of the organisms (Flipos Engdaw *et al.,* 2013).

Several authors have reported isometric and allometric growth pattern of *O. niloticus* from different water bodies. The isometric growth pattern of *O. niloticus* observed in the present study (b=2.9867) agrees with the earlier findings of Kassahun Asaminew *et al.,* (2011) in the same lake where the b value was reported to be 2.99. The isometric growth pattern of *O. niloticus* was reported in other nearby rift valley lakes of Ethiopia such as in L. Ziway (b=3.11; Zenebe Tadesse, 1998), L. Chamo (b=2.98; Yirgaw Teferi and Demeke Admassu, 2002), L. Hawassa (b=3.01; Demeke Admassu, 1998), and L. Koka (b= 3.0541; Flipos Engdaw *et al.,* 2013). The coefficient for *O. niloticus* in this study is relatively higher than the coefficient for the same species in Ethiopian lakes; L. Abaya (b=2.6; Berhanu Kebede 2016), L. Hawasaa (b=2.91; Demeke Admassu, 1990), and L. Tana (b=2.74; Zenebe Tadesse, 1997).

Such differences in the length-weight relationship coefficient are not unexpected and may result from differences in habitats (Froese, 2006; Flipos Engdaw *et al.,* 2013). According to the same authors, the variation in the value of '*b*' happens due to season, habitat, gonad maturity, sex, diet, gut fullness, health, preservation techniques, and annual differences in environmental conditions. The isometric growth of *O. niloticus* in the present study lake may also be due to the high productivity of the lake. The growth of phytoplankton (See Chapter 3) and their photosynthetic productivity (See Chapter 4), and abundance and biomass of zooplankton (See Chapter 5), which is the most important food in the fish diet, are reasonably high. This may cause the proper growth of fish in the present study lake.

The study of condition assumes that heavier fish of a length is in a better condition (Jones *et al.*, 1999). According to the same authors, the indices have been used by fishery biologists as indicators of the general “well-being” or “fitness” of the population under. The FCF of *O. niloticus* in LTA (mean: 1.88, ranged: 0.96-3.51) is higher than the other nearby rift valley lakes, L. Abaya where FCF is 1.43 (Berhanu Kebede, 2016) and the Ethiopian highland Lake, L. Hayq where FCF value is 1.81 (Workie Worie and Abebe Getahun, 2014). But, FCF in LTA is lower than that of the same species in some other Ethiopian lakes. FCF of *O. niloticus* is 2.29 in Lake Tana (Zenebe Tadesse, 1997), and 2.23 in the nearby Lake Chamo (Yirgaw Teferi and Demeke Admassu, 2002).

Therefore, the fish in this study lake appears to be in a better condition when compared to those in Lakes Abaya and Hayq; but lower than those in the two large Ethiopian Lakes Tana and Chamo. The data in this study further shows that more than half of the fish samples examined have condition factors above 1.5 and 43.5% of it is above the mean (1.88) indicating the majority of the population in LTA are in good condition. The condition factor higher than one (>1) suggests good fish health condition and indicates an isometric growth, which is desirable in a fish farm (Ayode, 2011). Thus *O. niloticus* in LTA is in a healthy condition according to FCF values. This could be attributed to various factors such as quality and quantity of food, the rate of feeding, reproductive potential, high abundance of phytoplankton and zooplankton, and complete mixing of the lake water which provoke the dynamics of nutrient enrichment and the consequent productivity of the ecosystem.

Comparisons of length-weight equations fitted in the present study with that of Demeke Admassu (1990), Zenebe Tadesse (1997) and Workie Worie and Abebe Getahun (2014) could provide more additional evidence. For example, at 20 and 30 cm total length, *O. niloticus* have weights of 156 and 474 grams in L. Tana, 124 and 383 grams in L. Hawassa and 138 and 461 grams in L. Hayq, respectively. In LTA, it has obtained 145 and 451.5 grams at 20 and 30.1 cm, respectively. This comparison shows that the species would be heavier in LTA than of a similar sized fish in L. Hawassa but lighter than those in L. Tana.

Hence, it can be concluded that *O. niloticus* population has a rapid growth rate in LTA than in L. Hawassa but slower than that in L. Tana and comparative with L. Hayq. This difference may be attributed to the difference in the availability and quality of food in the different lakes. LTA may have better productivity with quality food than L. Hawassa. However, further study is needed to verify the plausible reason for this hypothesis.

The condition factor of both sexes in LTA is appreciably different. High value was observed for males (mean FCF=2.21) than females (mean FCF=1.93) indicating male fishes are in a better condition than females in the study area. However, when individual male and female fish of equal length are compared but where the female is gravid (full of eggs), relatively high FCF is mostly seen for female than male. Similar results were reported in another nearby rift valley Lakes, L. Hawassa (Demeke Admassu, 1994) and L. Abaya (Berhanu Kebede 2016), and in the same lake studied by Kassahun Asaminew *et al.,* (2011) where male *O. niloticus* is in a better condition than females.

In this study, the overall sex-ratio (Male: Female) for *O. niloticus* was 1.4:1 This agrees with the results obtained for the same species in L. Tana (Zenebe Tadesse, 1997), L. Victoria (Njiru *et al*, 2006), and the same lake (Kassahun Asaminew, 2011). Sex ratios of the same species where females population dominate over males were reported in L. Hawassa (Demeke Admassu, 1994), in Abu-Zabal Lake, Egypt (Shalloof and Salama, 2008), in L. Coatetelco, Mexico (Gómez-Márquez *et al*, 2003).

Although concrete evidence couldn't be drawn for biased sex ratio from the present study, it may be caused by sexual segregation during spawning, behavioral differences between the sexes, gear type, and fishing site (Demeke Admassu, 1994). Sex ratio may be affected by several factors such as sex-based difference in maturity, mortality, migratory and other behaviors (Matsuyama *et al.,* 1988).

Besides sex basis, during the measurement of the length of the fish, the size of males are larger than females in most samples. Male Nile tilapia typically grows larger than females in small water bodies and riverine habitats (Lowe-McConnell, 1958). Demeke Admassu (1998) also found that after 2 years of age, female *O. niloticus* grow more slowly than males in L. Hawassa and this may be related to the small size of the lake. This owing to be true for LTA since the size of the lake is too small (seven times smaller than L. Hawassa).

de Graaf *et al*., (1999) also reported that mean growth rates for males were greater than for females. The onset of sexual maturity could be another factor responsible for differences in sizes. Since females invest more energy in reproduction than males, they grow slower than males (Demeke Admassu, 1994). Larger males may also be related to high water temperature; as higher temperatures induce sex reversal and higher proportions of males (Peterson *et al.*, 2004). LTA is warm with a fairly high temperature (See Chapter 2). This warm temperature of the lake may favor the growth of males than females.

7.5.2 Feeding habits

In the diet of *O. niloticus* in LTA both plant and animal origin food items are found. The food from gut analyses consisted of eight groups of food items: phytoplankton, zooplankton, macrophytes, detritus, insects, nematodes, ostracods, and fish scales. The food items found in the guts of *O. niloticus* are similar to what has been reported by several authors in different rift valley lakes of Ethiopia. Adult *O. niloticus* reported to feed on variety of food items including phytoplankton, macrophytes, benthic invertebrates, insects, and detritus (Getachew Tefera, 1987a, b; Getachew Tefera and Fernando, 1989; Yirgaw Teferi *et al*., 2000; Oso *et al*., 2006; Alemayehu Negassa and Prabu, 2008). Juveniles are generally omnivorous feeding on zooplankton, insect larvae (Todurancea *et al*., 1988) and phytoplankton (Flipos Engdaw *et al.*, 2013).

Phytoplankton, detritus, zooplankton, and were dominantly found in the diet of *O. niloticus*, found. On the other hand, the occurrence of insects, nematodes, fish scales, and ostracods in the food bulk of *O. niloticus* was low. The present work confirmed that *O. niloticus* in LTA consumed large quantities of phytoplankton. Phytoplankton were the most important food items in most lakes in Ethiopia for which data are available (Getachew Teferra, 1993; Zenebe Tadesse, 1988, 1998; Yirgaw Teferi *et al*., 2000; Elias Dadebo *et al.*, 2014; Flipos Engdaw *et al.*, 2013). Besides phytoplankton, other foods of plant origin (macrophytes and detritus) were also consumed in large quantities. Zenebe Tadesse (1988, 1998) in L. Ziway and Yirgaw Teferi *et al*., (2000) in L. Chamo reported the importance of macrophytes and detritus in the diet of *O.*

niloticus. Bowen (1980) also reported large quantities of detritus in the diet of O. *niloticus* in L. Valencia (Venezuela). Several authors (e.g., Zenebe Tadesse, 1988; Getabu, 1993; Shipton *et al*., 2008; Oso *et al*., 2006; Kamal *et al*., 2010) also provided similar interpretations about the importance of detritus and macrophyte in different parts of Africa.

In the present study, the proportions of different food items varied considerably between the dry and wet seasons. The proportion of macrophytes, zooplankton, and detritus were higher in the wet season than in the dry season. The reverse was true for the proportion of phytoplankton where the high proportion was observed during the dry season. This is similar to the findings of other investigators in the Ethiopian rift valley lakes (Zenebe Tadesse, 1988, 1998; Yirgaw Teferi *et al*., 2000). A pronounced seasonal succession of phytoplankton has been found in some Ethiopian rift valley lakes, Hawassa, Ziway, and Chamo (Zenebe Tadesse, 1988, 1998; Elizabeth Kebede and Amha Belay, 1994; Yirgaw Teferi, *et al*., 2000) and Abaya (Elias Dadebo *et al.*, 2014) and in the Ethiopian largest lake, L. Tana (Zenebe Tadesse, 1997). In most of the Ethiopian rift valley lakes for which data are available, phytoplankton predominantly constituted the diet of *O. niloticus* during the dry season whereas macrophytes were important in the wet season.

The composition differences and the relative contribution of food items may partly be explained by the difference in microhabitat occupied by the fish before their capture. During the wet months, fish move to shallow parts of the lake for reproduction and stay there for longer period feeding on macrophytes. In addition, high flooding from the catchment during the wet months may cause fluctuations in water level and increasing turbidity of the lake. This mostly decreases the penetration of light in the lake and affecting the growth and abundance of phytoplankton in the water (Brook Lemma, 1994). During the dry seasons, when the water recedes away from the vegetation, the fish move further into the open-water and feed mainly on suspended phytoplankton. During the wet seasons, fish spend most of the time feeding in the littoral zones, where macrophytes are abundant (Zenebe Tadesse, 1998). It is highly probable that changes in the habitat choice of the fish due to water level fluctuations may bring variations in the composition of the fish diet in LTA. This happened in L. Tana as shown in the work of Zenebe Tadesse (1998).

The ontogenetic dietary shift of *O. niloticus* in LTA highlights significant variations of food items as the fish grows older. Ontogenetic diet shift was apparent during the present study with the importance of phytoplankton, macrophytes, and detritus increased with size whereas zooplankton, insects, and other animal origin food declined with fish size. Foods of animal origin, mainly zooplankton, are the most important food items of the fish below 10 cm of TL.

Zooplankton in particular rotifers contributed the largest proportion (46%, by volume) of the food items in the smallest size class (<10cm TL). In the Ethiopian rift valley L. Koka, *O. niloticus* <10 cm TL mainly fed on zooplankton and insects and this food type declined sharply as the size of the fish increased above 10 cm TL (Flipos Engdaw *et al.*, 2013). The ontogenetic dietary shift of *O. niloticus* in L. Victoria also showed that zooplankton were most important food items for fish less than 5 cm TL and little importance for larger fish with sizes over 10 cm TL (Njiru *et al*., 2006). The findings of the present study corroborate well with Tudorancea *et al*., (1988), Abdel-Tewwab and El-Marakby (2000), Yirgaw Tefri *et al*., (2000) and Alemayehu Negassa and Prabu (2008) where the contribution of zooplankton in the smaller size of *O. niolticus* is significant.

The possible reason for juveniles feeding on zooplankton may be due to the sizes of their mouth gape, the smallness by volume of their guts that may not support big macrophyte and detritus. The volumes of the gut are not large enough to make filter feeding energetically (Brook Lemma *et al.*, 2001; Elias Dadebo *et al.*, 2014). It is also possible the lack of physiological preparedness to process large and hard foods (Brook Lemma *et al.*, 2001). The reason for taking less zooplankton during adult life may be the fish change its mode of feeding by gulping the water within its area. Conversely, with larger size classes (10-19.9 and 20-29.9 cm TL), phytoplankton dominated the diet of fish by volume. In the final size class (>30 cm TL), the contribution of phytoplankton declined while the contribution of macrophytes and detritus increased. The most plausible explanation for this variation in the proportion of different plant materials during the life cycle of the fish could be that adult fish have the wider mouth gapes and the ability to digest macrophytes and detritus in their gut (Tudorancea *et al.*, 1988).

Examination of the diet of *O. niloticus* showed there is a high percentage of detritus in their gut especially in the large size (adult). This indicates that the fish is also a bottom browser, which most likely to be in the littoral zones of the lake. It is a well-established fact that the composition of different food items utilized by *O. niloticus* changes as the fish grows older. Even though the nutritive quality of foods of animal origin consumed by early stages of the fish was high, the energy demands of growing fish cannot be met by particulate feeding on zooplankton (Shipton *et al.*, 2008). The changes of fish in it's feeding on primarily omnivorous diet to herbivorous diet is most probably to energy demands (Alemayehu Negassa and Prabu, 2008).

In L. Kuriftu (Ethiopia), fish and zooplankton are exposed to continuous predator-prey interactions (Brook Lemma *et al.*, 2001), as described for tropical waters by Fernando (1980 and 2002). In this tropical lake, *D. barbata* and *Moina micrura* zooplankton were observed to be the dominating grazers than *Diaphanosoma* sp. and *Ceriodaphnia* sp. The same results have been observed in LTA (See Chapter 5). According to Arcifa *et al.,* (1986) this has probably suppressed these cladocerans to a bare existence of negligible biomass in the no-fish enclosure. The fish-enclosure in L. Kuriftu was dominated by an array of rotifers and once again by *Moina micrura*, this time represented only by small-sized individuals of its population (Brook Lemma *et al.*, 2001). A similar phenomenon has been observed in the LTA of the present study. Based on these observations, perhaps the size factor has apparently played an important role in vertebrate planktivory fish and zooplankton especially the cladoceran relations. This may results in the low contribution of zooplankton in the diet of *O. niloticus* during the adult compared to the juveniles.

In LTA, the contribution of insects in the diet of *O. niloticus* is comparable in the different size classes. It is a better source for the fish <10 cm than the other size classes. The other animal food source including nematodes, ostracods, and fish scales found in the gut of *O. niloticus* during the study, are increasingly declined as the fish get older and bigger. Generally, the contribution of animal sources (zooplankton, insects, nematodes, and ostracods) is relatively high in the smaller size fish while the plant source (phytoplankton, macrophytes, and parts of detritus) are high in the large size fish in the study lake.

Ontogenetic diet shifts in the present study demonstrate that at the earlier stage, *O. niloticus* is most probably omnivorous; its diet depends on zooplankton and phytoplankton in a high proportion. It shifts to herbivores as its size increase. At the adult stage, the diet depended mostly on macrophytes, detritus, and phytoplankton, which are foods of plant origin. Thus, it is concluding that the dietary habit of *O. niloticus* in LTA is, generally, the omnivorous type.

CHAPTER 8: GENERAL CONCLUSION AND RECOMMENDATIONS

8.1 Conclusion

LTA and its ecosystem are not only biodiversity hotspots, but also of utmost economic significance to the local population and habitat of migrating birds. The lake is the only big inland freshwater system in Silttie zone. It is being used for drinking by humans and their livestock, commercial fishing, agricultural irrigation, recreation, and many other services. Thus it is safe to consider that LTA is the lung of the residence of the nearby small village, Gebrie-Ber in particular and the Silttie zone in general.

The results of this study highlighted the water of LTA is ecologically productive in relation to the various physical and chemical factors and biological properties. The findings of the physicochemical parameters obtained in the study indicated that the lake water is fresh, less saline, slightly warm, well oxygenated, turbid, and with the shallow euphotic area. The lake is also supportive of nutrient enrichment. However, it is also an indication of water quality deterioration due to eutrophication. Even if anoxic condition because of eutrophication due to algal bloom is not observed during the current condition, the probability of its occurrence is high since there is much anthropogenic interference underway along the shores and in the surrounding area.

Besides various physical and chemical parameters, several biological communities including plankton (phyto & zoo), benthic macroinvertebrates, and some fish species (e.g. *O. niloticus*) have been considered in the assessments of the ecological productivity and water quality conditions of LTA. The results of the number of plankton taxa, abundance, and their biomass reported in the lake are fairly high. In addition a considerable number of taxa and specimens of benthos assemblages reported from the littoral zone of the lake. Because of this fairly high number and diversity of these biological communities found in the lake in comparison with those of the other rift valley lakes of Ethiopia, LTA could be regarded considerably productive. This makes the lake water suitable for the survival of most of the aquatic life in it and operations of various activities such as fishing, irrigation, poultry, and many other related multidimensional communal uses.

Moreover, studies on photosynthetic productivity suggested that the mean primary productivity of streams and lakes of the world is about 0.6 g C m^{-2} d^{-1} (Melack and Kilham, 1974b; Oduor and Schagerl, 2006, 2007b). The mean daily photosynthetic productivity of LTA is almost tenfold (i.e 5.78 gC m^{-2} d^{-1}) compared to other productive water bodies as stated by Melack and Kilham, 1974b; Oduor and Schagerl, 2006, 2007b. Thus, LTA is most probably considered as productive in nature and a large consumer population, which includes zooplankton, fish, and birds, among others, are supported by this phytoplankton productivity directly and indirectly with well-established food chain extending to the top predators of the system.

But the benthic macroinvertebrates of LTA is widely comprised by pollution tolerant taxa as compared to pollution sensitive taxa. This clearly indicates the water of the lake is considerably polluted with the inflow of organic wastes from settlements, excess nutrients and other agricultural inputs from the farms in the watershed and thus lake water is under environmental stress. This obviously calls for participatory and interdisciplinary actions of watershed management and wise use of aquatic resources with compliance to commonly set national agenda of sustainable use of natural resources.

8.2 Recommendations

This work first and foremost laid down the basic limnological information on LTA and the data contained in here and the publications thereof have laid the foundation for further applied research. In addition, the study pointed out the main threatening factors and possible management options for the lake and its watershed area. There are numerous shreds of evidence of unwise use that threatening LTA. Based on the findings of this study, keen observation in the meantime of data collection during this study, and by communicating with the nearby communities those in the watershed and fishermen, it is observed various problems affecting the ecology of LTA and hence recommended, calling for urgent actions for management are stated below:

(i) Excessive water abstraction. Regardless of the small-size of LTA, an excessive abstraction of the lake water for irrigation activities and many other purposes is taking place. This may challenge the sustainable use of the lake ecosystem because of increasing competition among users of irrigation and water supply to nearby towns (e.g., Kebet, Worabie, etc.). Thus water abstraction of the lake should be planned with the water budget of the system.

(ii) Shoreline modifications. LTA has no buffer zone all around it. The shoreline of the lake is modified up to compete for exposure of the topsoil for irrigation and for the inflow of municipal and domestic wastes from towns and villages in the watershed. This ultimately results in the deterioration of the water quality of the lake. To minimize this effect, therefore, the establishment of the buffer zone for LTA is one of the management options given for this aquatic resource. In addition, depending on the vegetation established, the buffer zone can be a source of forage production for farm operations.

(iii) Sedimentation. Because of great shoreline modification on the watershed, LTA is exposed to an extensive amount of sediment through inflowing rivers and runoffs. The two rivers, Dacha and Badober, bring a lot of silt and other organic wastes from the Gurage highlands and end up into the LTA. This sedimentation could result in filling up the lake basin to changing it into a terrestrial environment with the added impacts of unplanned water abstraction. This was a recent experience witnessed at L. Harmaya, Eastern Ethiopia (Brook Lemma, 2003; Zinabu Tebeje, 2012; Seifemichael Abebe, 2014). Habitat destruction is most probably one of the primary factors involved in causing high sedimentation. Therefore, restoration riparian vegetation and conservation of the existing macrophytes should be a vital aspect for the maintenance and protecting the degraded LTA and its environments.

(iv) The occurrence of eutrophication. Determining the trophic state of LTA in this study suggests the lake is a hypereutrophic state. The main parameters for the increasing of the trophic state of LTA are water turbidity and high concentrations of phosphorous. This high nutrients load results in the existence of eutrophication of LTA. When and if such eutrophication occurs in this lake, the proliferation of undesirable algal blooms, invasive species such as water hyacinth, and others may occur making the water and fishes of the lake unusable to human functions, including livestock drinking because of toxins produced by algae such as Cyanobacteria. Thus controlling intensive agricultural activities which are a nutrient source through runoff and other human pressure on the watershed of LTA is a vital role for the proper function of the lake.

(v) Risk of water quality deterioration: The anthropogenic effects taking place in the watershed of the lake have led to the decline of the water quality and quantity of the LTA and simultaneously threatening local livelihoods. This work has indicated that most of the water quality monitoring approaches (physical, chemical and biological) has shown the deterioration of the lake water in relation to some recommended agencies such as WHO and FAO for certain particular uses. With this trend continuing, the lake water will soon become unfit for irrigation or consumption by humans and domesticated animals. It is, therefore, the order of the time that some wastewater management should be in place. Some monitoring and management options that should take for minimizing and reducing further reduction of the water quality of the LTA include ban of the direct disposal of domestic wastes into the lake, controlling the different agricultural activities that lead nutrient accumulation, reduce various activities such as car washing, watering cattle, swimming, and other related activities that induce the lake water turbidity which directly affect the water quality of LTA.

(vi). Overfishing: There are four fishermen associations namely Shanka, Lenfur, Gora, and Misrak that catch fish (*O. niloticus*) from LTA. In addition, there are other informal fishermen that operate on this lake. All these fishers conduct their business without worried about the impact of overfishing on the lake. The fishing gear they use does not discriminate the sizes of the fish captured, the removal of female fish from breeding grounds, before spawning or nursing females who shelter their young in their mouths, etc. According to the fishery studies in this work, fishers are observed collecting increasingly small fishes. Thus, there is a high risk of losing this valuable food source provided to support the poor with protein supply and small incomes to supplement their financial requirements. If fishing continues in the way it is operated, the fishery practice at LTA will likely collapse in a few years. Therefore, it may be necessary to totally or partially close the fishing practice for a few years to allow the fish sticks to regenerate themselves. Furthermore, it may be necessary to designate a certain area of the lake as a fish sanctuary so fishes can recuperate and refill the system.

The above arguments are not exhaustive but only indicative of the existing challenges that prevail at LTA and some possible mitigation for the proper use of the lake. However, in a general term, to assure the health and normal functioning of this productive ecosystem are negligible. The user communities are not discussing the problems; the community leaders and the zonal administration have not applied any efforts in terms of mutual consultations, mitigating climate impacts and so on. It is therefore observed and suggested that the attention given to emerging problems by stakeholders or any governmental organizations around the lake is far from sufficient. Thus, it is recommended to have continuous monitoring of the lake water to protect it from any further possible pollution and contamination of the lake water and control the abstraction of the same within the annual budget of the lake. Further, these actions may be necessary to bring about the protection and sustainable use of the lake and its watershed:-

a. Empowering local communities to conserve and protect LTA by themselves for themselves: LTA provides free goods and services to the nearby communities getting no returns to make it sustainable. Thus, maintenance of the essential values and function of this lake should also be the major role and responsibility of the community. The nearby local communities are important stakeholders in using and protecting the lake. Thus, conservation of LTA should be the duty of the local communities. They should be made responsible for the participation for the conservation of the lake for the following reasons:

- They are directly attached to LTA hence should undertake conservation measures to handle the lake.
- They provide the information required by making an inventory of the necessary data and maintain socioeconomic data on this aquatic resource.
- They can easily disseminate the information regarding threats and risks of LTA and its watershed to different concerned bodies.
- They help in maintaining ecological balances by wise utilization of the LTA watershed.
- They should collaborate with local and regional administrations in monitoring uses and mitigating problems of this ecosystem.
- They should collaborate with other watersheds so they exchange experiences for improved watershed management.

b. Formation of comprehensive association: The coordination of wetland management requires signing memoranda of understanding between different stakeholders. This would demonstrate the recognition of multi-sectoral interest in wetlands. Dialogues among key government officials of the Silttie Zone, representatives from Office of the Cultural and Tourism of Silttie Zone and Silttie Woreda, the nearby communities, an association of fishermen, and other many stakeholders should come together and discuss the issue of LTA. In addition, there should be one responsible association which controls or governs monitors and manages the overall activities taking place in the watershed area and in the lake itself.

Moreover, to have more comprehensive, clear, and full knowledge of LTA and its watershed, although not exhaustive, it is recommended to address or study these issues in the future:-

- While the temporal limnological monitiring of LTA over the coming years continues in terms of physical, chemical, and biological features, it is recommended that ionic composition, toxic metal contamination, and pesticides polution of LTA should be studied.
- Land use practices, sedimentation rates, water balance, fish production potential, and watershed management of LTA should be studied.
- Study how the local communities could be made responsible to manage their own affairs and sustain the use of the resources prevailing in LTA and its watershed.

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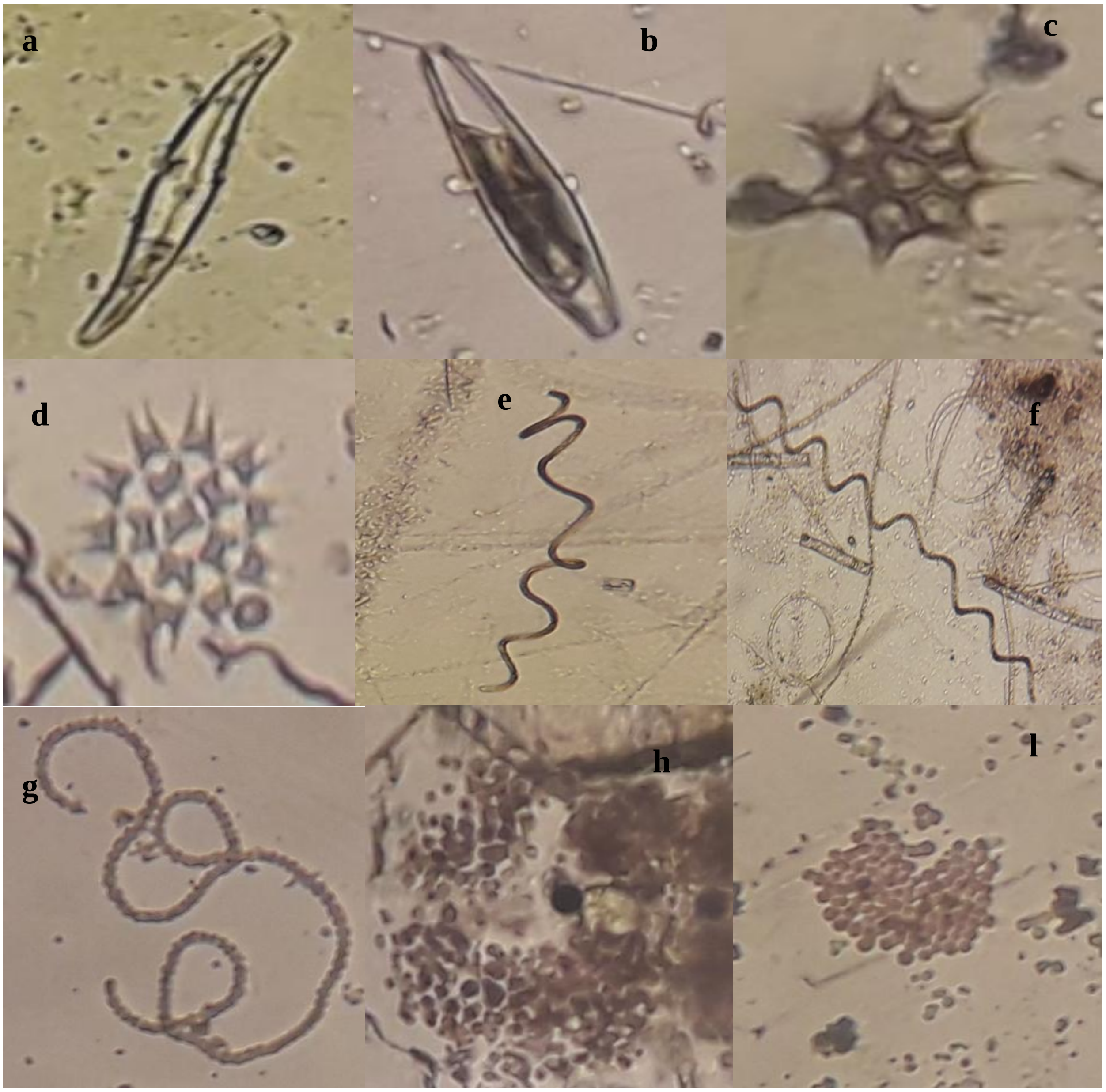
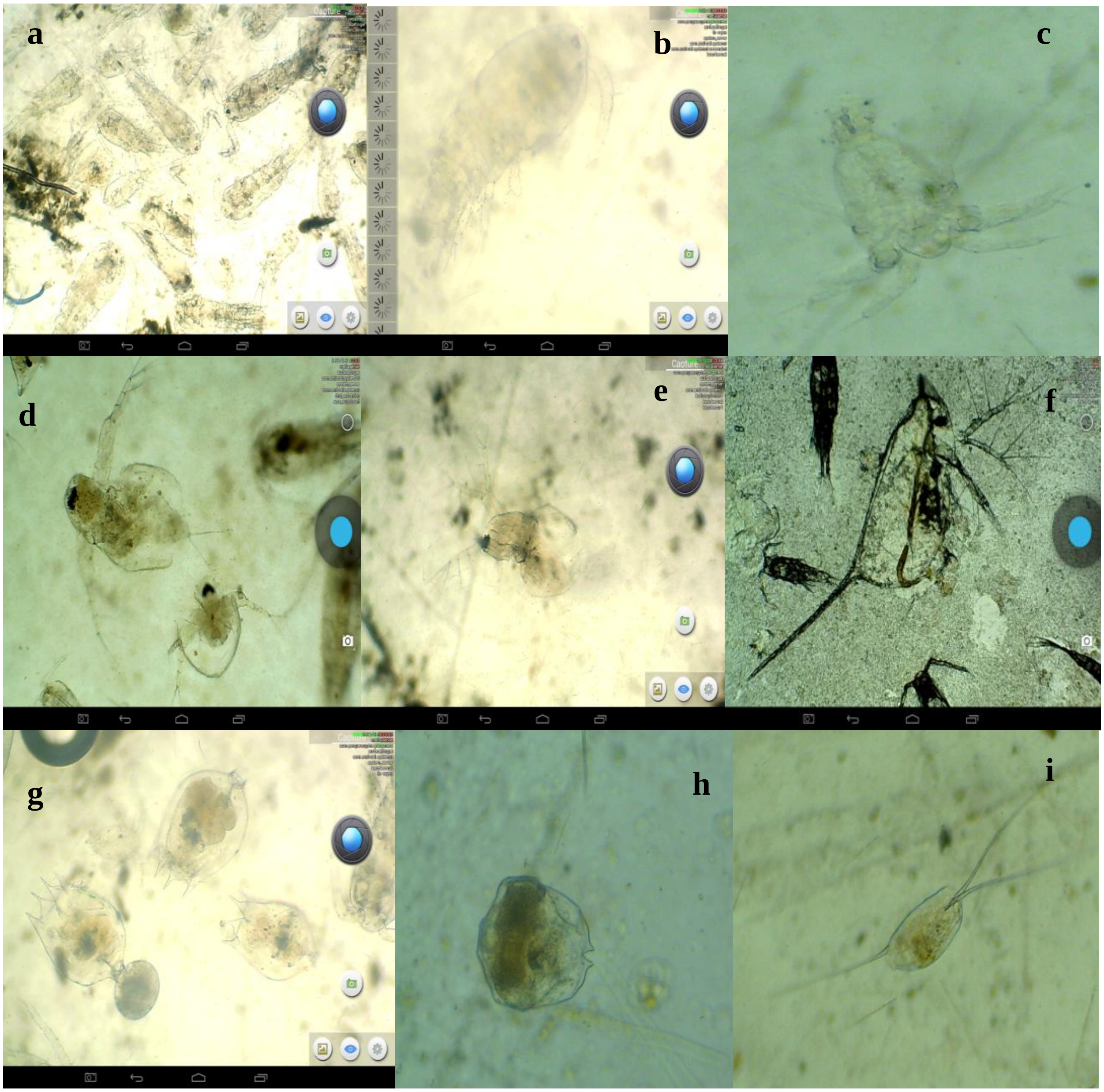
Appendices

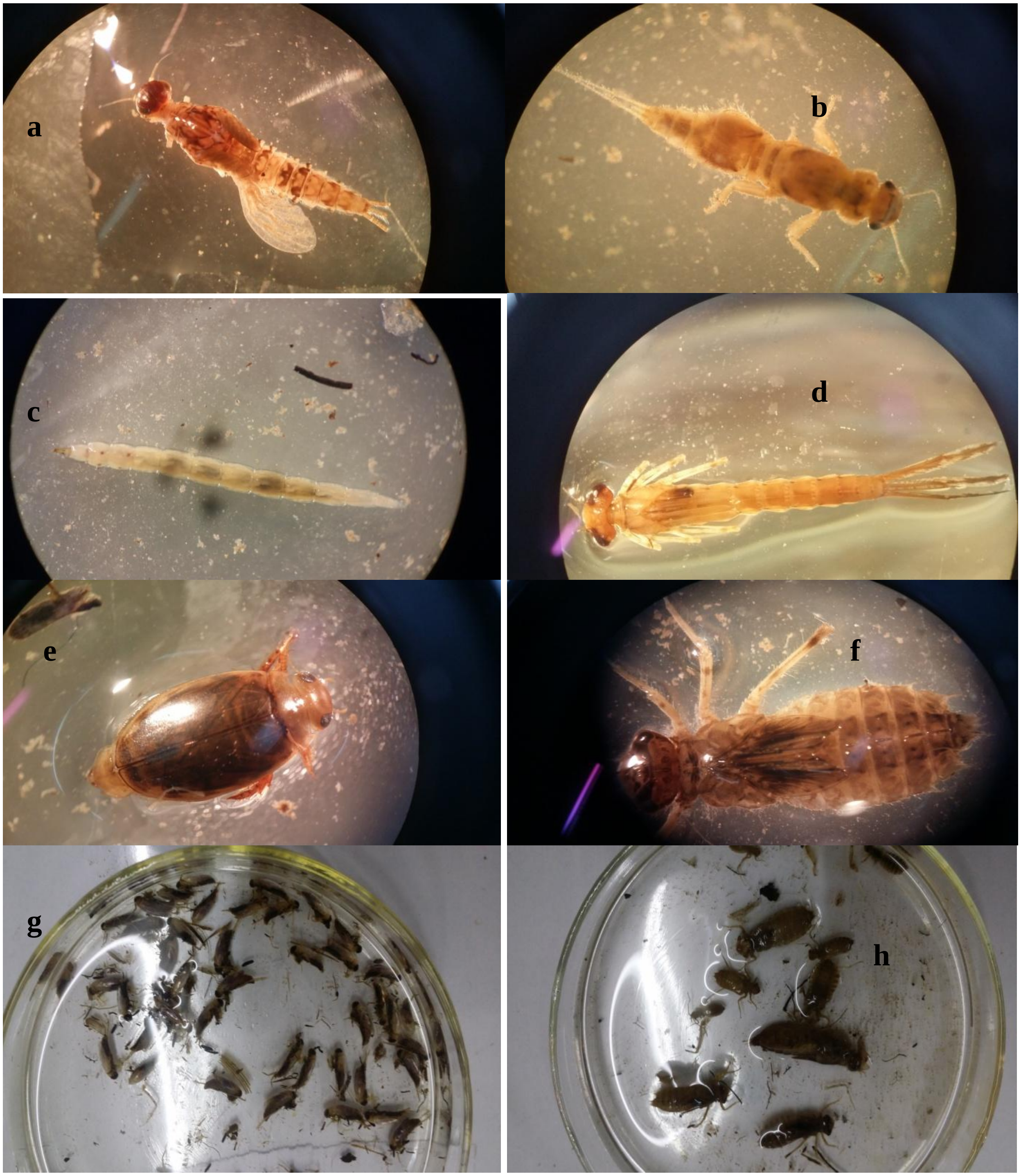
Appendix 1. Some phytoplankton taxa recorded in LTA. a) Gyrosigma sp., b) Cymbella sp. c) *Pediastrum simplex*, d) *Pediastrum duplex*, e-f) *Cylindrospermopsis sp.*, g) *Anabaena circinalis*, h-i) *Microsystis sp.*,



Appendix 2. Major zooplanktons recorded in LTA. a) *Thermocyclops decipiens, b) Mesocyclps aequatorialis*, c) *cyclopoid nauplii d) Moina micrura, e) Diaphanosoma excisum*, f) *Daphnia barbata, g) Keratella tropica*, h) *Brachionus angularis,* i) *Filinia pejleri,*



Appendix 3. Some macroinvertebrates recorded in LTA. a) *Baetidae*, b). *Caenidae*, c). *Ceratopogonidae,* d). *Coenagrionidae,* e). *Dytiscidae,* f). *Libellulidae,* g) *Notonectidae,* h). *Naucorida*



Appendix 4. Major macrophytes observed in LTA. All the macrophytes are growthed in and near the lake water. a-b) *Potamogeton;* covers the lake water during the main rainy period (July and Agust), c-d) *Persicaria senegalensis* (widely found in the nothwest side of the lake where river Dacha is present), e-f) *Cyperus species* (dominated in the northwest and southwest direction of the lake).

Appendix 5. Major anthropogenic activities in and around LTA. Water abstraction for small-scale irigation (a-b) and domostic uses (c-d), grazing on the shore line of the lake (e), car washing (f), bathing and washing (g), wastes from run off via River Dacha into the lake (h-i), pesticides use near the shore line of the lake (j-k), agricultural activities in the shore line of the lake (l-m).


g


h


i


j


k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Appendix 6. The two feeder rivers (Bobodo and Dacha) of LTA. a1-a2). River Bobodo (southern corner of LTA) in the dry month (March) and wet season(July), and b1-b2). River Dacha (norther corner of LTA) in the end of dry month (May) and in main rainy month (August).

a1). River Badober during the dry month (March 2016)

a2). River Badober during the rainy month (July 2016)

b1). River Dacha during the end of dry month (May 2016)

b2). River Dacha during the rainy month (August 2016)