



AFRICA CENTER OF EXCELLENCE FOR WATER MANAGEMENT
ADDIS ABABA UNIVERSITY



**LAND USE/LAND COVER CHANGE IN THE CATCHMENT OF
NORTHERN BAY OF LAKE TANA, ETHIOPIA: RAMIFICATION FOR
WATER QUALITY AND PLANKTON COMMUNITY STRUCTURE**

By:

FLIPOS ENGDW WOLDEAB

A PhD dissertation submitted to the Africa Center of Excellence for Water Management,
Addis Ababa University in partial fulfilment of the requirements for The Degree of Doctor of
Philosophy in Water Management (Aquatic Ecosystems Management)

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Addis Ababa, Ethiopia

Africa Center of Excellence for Water Management

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DECLARATION

I, Flipos Engdaw Woldeab (GSR/5582/14), hereby declare that this dissertation entitled “**Land use/land cover change in the catchment of northern bay of Lake Tana, Ethiopia: ramification for water quality and plankton community structure**” has been developed by me and has not been submitted to any other institution for the award of any academic qualification. The content of the dissertation has not been plagiarized, and where works of other researchers have been used, they have been appropriately cited and duly acknowledge.

Candidate’s Full Name: Flipos Engdaw Woldeab (GSR/5582/14)

Signature _____

Date _____



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A Ph.D DISSERTATION SUBMITTED TO

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APPROVED BY THE BOARD OF EXAMINERS

This is to certify that we have read this PhD dissertation research and that in our opinion; it is fully adequate, in scope and quality, as a PhD dissertation for the Degree of Doctor of Philosophy in Water Management (Aquatic Ecosystems Management).

Main Advisor: Professor Tadesse Fetahi Signature_____ Date 16 July 2025

Co-advisor: Dr. Demeke Kifle Signature_____ Date 16 July 2025

External Examiner: Prof. Zinabu Gebremariam Signature_____ Date 16 July 2025

Internal Examiner: Prof. Brook Lemma Signature_____ Date 16 July 2025

Chairperson: Prof. Abebe Getahun Signature_____ Date 16 July 2025

DEDICATION

To my elder brothers **TADESSE** and **DAGNEW ENGDAW**! Nothing can ever take away a
love the heart holds. Forever in my hearts!

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LIST OF ABBREVIATIONS

ANOVA	Analysis of Variance
APHA	American Public Health Association
CANOCO	Canonical Community Ordination
Chl- <i>a</i>	Chlorophyll a
CSA	Central Statistical Agency
DCA	Deterended Correspondence Analysis
DO	Dissolved Oxygen
EC	Electrical Conductivity
ENMA	Ethiopian National Meteorology Agency
ES	Ethiopian Standards
FAO	Food and Agriculture Organization
<i>H'</i>	Shannon-Weiver index
m.a.s.l.	Meter above sea level
NH ₄ ⁺	Ammonia
NO ₂ -	Nitrite
NO ₃ ⁻	Nitrate
NTU	Nephelometric Turbidity Unit
PCA	Principal Component Analysis
S _d	Secchi depth
SD	Standard Deviation
SiO ₂	Silica (reactive)
SRP	Soluble reactive phosphate-phosphorus
TN	Total Nitrogen
TP	Total Phosphorus
TSI	Trophic State Index
USEPA	United States Environmental Protection Authority
WHO	World Health Organization

LIST OF PUBLICATIONS

1.1. Articles published in peer-reviewed Journals that form this PhD dissertation

- I. Engdaw, F., Fetahi, T., Kiflie, D. (2024). Land use/land cover dynamics in the northern watershed of Lake Tana: implications for water quality. *Front. Environ. Sci.* 12: 1426789. <https://doi.org/10.3389/fenvs.2024.1426789>
- II. Engdaw F, Fetahi T, Kifle D. (2025). Increasing anthropogenic stressors influenced the water quality and shifted trophic status of northern Lake Tana gulf, Ethiopia. *Heliyon*. 11(1): e41162. <https://doi.org/10.1016/j.heliyon.2024.e41162>
- III. Engdaw, F., Kifle, D., Fetahi, T. (2025). Taxonomic and functional groups of phytoplankton in the northern gulf of Lake Tana, Ethiopia. *J. of Freshwater Ecol.* 40: 2506588 <https://doi.org/10.1080/02705060.2025.2506588>
- IV. Engdaw, F., Kifle, D., Fetahi, T. (2025). Water quality parameters influence temporal dynamics of zooplankton community structure in a tropical highland lake, Lake Tana, Ethiopia. *Marine Poll. Bul.* 218: 118239. <https://doi.org/10.1016/j.marpolbul.2025.118239>

1.2. Papers not considered as part of dissertation but published within the PhD program.

- I. Engdaw, F. Geremew, A. (2024). Broodstock nutrition in Nile tilapia and its implication on reproductive efficiency. *Front. Aquac.* 3: 1281640. <https://doi.org/10.3389/faquc.2024.1281640>
- II. Engdaw, F. (2023). Morphometric relations and diet compositions of Nile tilapia *Oreochromis niloticus* (Linn.1758) in Lake Tana Gorgora gulf, Ethiopia. *Fish. Aquat. Sci.* 26(3): 169-180. <https://doi.org/10.47853/FAS.2023.e14>
- III. Gobeze A, Kaba T, Tefera M, Lijalem T, Legesse M, Engdaw F, et al. (2023). Evaluation of water quality of Angereb reservoir: a chemometrics approach. *App. Water Sci.* 13:103. <https://doi.org/10.1007/s13201-023-01909-2>
- IV. Muluye, T., Fetahi, T., Engdaw, F., Mohammed, A. (2023). Cyanotoxins in African waterbodies: occurrence, adverse effects, and potential risk to animal and human health. *Environ. Geochem. Health.* 45:7519-7542. <https://doi.org/10.1007/s10653-023-01724-3>
- V. Engdaw F, Hein T, Beneberu G. (2022). Heavy metal distribution in surface water and sediment of Megech River, a tributary of Lake Tana, Ethiopia. *Sustainability*.14: 2791. <https://doi.org/10.3390/su14052791>

Abstract

Rapid population growth-induced industrialization, urbanization, deforestation, settlement expansion and habitat destruction has emerged as critical drivers of land use/land cover (LU/LC) change on a global scale. The uncontrolled LU/LC change and anthropogenic activities, in developing countries like Ethiopia, poses significant environmental threats to water quality and nutrient enrichment. The excessive enrichment with these nutrients emanating from intensive anthropogenic activities has substantially impacted water quality and biodiversity of lakes. The development of strategies of restoration or protection of lakes from further degradation necessitates the availability of adequate information on water quality, its environmental drivers and impact on plankton communities. There is, however, paucity of information available on the LU/LC changes, water quality, trophic status, and plankton community structure of the northern Gulf of Lake Tana, Ethiopia. Therefore, this study aimed to assess spatio-temporal changes in water quality, trophic status and plankton community structure in relation to the LU/LC dynamics in the northern Gulf of Lake Tana catchment. ArcGIS software and Landsat imagery were employed to produce maps for LU/LC changes. Samples for the assessment of water quality and plankton community structure were collected from May 2023 to April 2024 from five sampling sites, which were selected based on their proximity to human interference. Physico-chemical parameters were measured *in situ* or determined in collected water samples using standard methods. LU/LC changes were categorized into six classes with an overall accuracy and Kappa coefficient of 92.2% and 0.86%, respectively. The various LU/LC classes were highly correlated with particular water quality parameters. Settlement and cropland expansion were positively correlated with nutrients ($\text{NH}_3^- + \text{NH}_4^+ \text{-N}$), $\text{NO}_3\text{-N}$, soluble reactive phosphate-phosphorus (SRP), and total phosphorus (TP). Forestation was positively correlated with indicator of good water quality such as dissolved oxygen (DO) and pH, while it was negatively

correlated with temperature, electrical conductivity (EC), SRP, and NO₃-N demonstrating a strong linkage between LU/LC and water quality. There were significant ($p < 0.05$) variations in various water quality parameters such as temperature, DO, EC and turbidity between the dry and rainy seasons. The mean concentrations of nutrients such as NO₃-N (0.48 mg/l), SRP (1.2 mg/l), total nitrogen (TN, 1.27 mg/l), and TP (2.1 mg/l) showed significant spatio-temporal variations, with higher concentrations occurring at the Megech and Dablo sites during the rainy season. The concentrations of SRP, TP, and TN, were slightly higher than the USEPA's threshold concentrations for causing eutrophication. A total of 113 phytoplankton species belonging to six taxonomic groups and 11 Reynolds functional groups (RFGs) were identified. The predominant RFGs, which collectively accounted for 97% of the total phytoplankton abundance, were **M**, **B**, **N**, **D**, **Lo**, **P** and **S2**. Other RFGs, such as **J** and **X1-X3**, constituted 2.4% and 0.8% of the total abundance, respectively. RFGs exhibited significant association with EC, pH, turbidity, and some algal growth limiting nutrients. Forty-five zooplankton species belonging to three groups (17 Rotifera, 14 Copepoda, and 13 Cladocerans) were identified. Rotifers were the most species-rich group, with *Brachionus angularis*, *B. caudatus*, *B. falcatus*, *B. havanaensis*, and *Keratella tropica* being the most abundant. Significant ($p < 0.05$) spatial variations in zooplankton diversity, evenness, and dominance were recorded. Shannon index values ranged from $H' = 2.98$ of the GOR site to $H' = 3.49$ of the MEG site. Evenness values varied between 0.52 and 0.74, indicating less uniform distribution. In summary, the findings indicated that LU/LC changes and spatio-temporal water quality variations significantly affect phytoplankton and zooplankton community structure and functions, subsequently impacting the overall health of the ecosystem. Thus, ongoing monitoring and intervention are imperative to avert future ecological imbalances and facilitate the restoration of Lake Tana.

Key words/phrases: Anthropogenic activity, LU/LC change, nutrient enrichment, plankton community structure, trophic state, water quality

CHAPTER 1

1. Introduction

1.1. Background of the study

Freshwater ecosystems, particularly lakes provide irreplaceable ecological and economic goods and services for numerous communities (Schallenberg *et al.*, 2013; Obiero *et al.*, 2020; Xiong *et al.*, 2020). They provide unique habitats that support high level of biodiversity, harbor fishery production and play a vital role in flood and erosion control. Freshwater ecosystems also regulate microclimate and make significant contribution to carbon sequestration, ground water recharge; water purification and recreation services (Vári *et al.*, 2022). Despite their importance, vulnerability to the effects of multiple stressors such as frequent disturbance from anthropogenic activities, hydrological variations and introduction of invasive species make these water bodies among the most threatened ecosystems (Xiong *et al.*, 2020). Recent studies indicated that, the surface area of freshwater ecosystems declined globally by 64% in 2011 compared to 1997 (Vári *et al.*, 2022). Moreover, the freshwater species have declined by 76%, while vertebrate populations declined by 52% from 1970-2010 demonstrating the adverse effects of anthropogenic activities on aquatic ecosystem stability (Natugonza *et al.*, 2016; Obiero *et al.*, 2020).

Notwithstanding their immense services, African freshwater bodies, which are known as biodiversity hotspots and recognized for their huge fisheries potentials, have lost their ecological integrity and faced serious biodiversity impoverishment (Obiero *et al.*, 2020; Makwinja *et al.*, 2021). For instance, recent reports from International Union for Conservation of Nature and Natural Resources (IUCN) revealed the alarming decline by 76% of endemic freshwater species of Lake Victoria (Njiru *et al.*, 2014). Similarly, the recent climate change induced drought,

which was manifested in a rise in temperature, lake level fluctuation and change in rainfall pattern, exacerbated biodiversity loss (Obiero *et al.*, 2020).

During the last few decades, severe anthropogenic-induced stressors coupled with natural and hydrological factors have become the major causes of the poor ecological integrity of the aquatic ecosystems (Gebremedhin *et al.*, 2018; Gobezie *et al.*, 2023; Engdaw *et al.*, 2025). For instance, rapid population growth, urbanization (Lanckriet *et al.*, 2017; Alemu *et al.*, 2017), land use/ land cover changes (LULC) (Hassen and Assen, 2017; Engdaw *et al.*, 2024), human settlement, activities associated with industrialization, such as discharge of untreated effluents (Legesse *et al.*, 2016; Berhanu *et al.*, 2018), and poor solid waste management activities (Goshu and Aynalem, 2017; Gebremedhin *et al.*, 2018); combined with natural and hydrological variations (Paerl *et al.*, 2016; Lanckriet *et al.*, 2017) are currently threatening the water quality and hence the biodiversity of the systems through their profound effect of excessive nutrient enrichment and erratic water level fluctuations (Fetahi, 2019; Wagaw *et al.*, 2021; Engdaw *et al.*, 2022; Mohammed *et al.*, 2024).

Furthermore, fertilizers applied on farmlands have led to widespread external loading of nutrients and subsequent eutrophication (Alemu *et al.*, 2017; Akale *et al.*, 2018; Fetahi, 2019; Gobezie *et al.*, 2023). Therefore, recent shifts in the trophic state of many lakes from oligotrophic to eutrophic or hypertrophic conditions might have caused water quality deterioration and loss of biodiversity, which are associated with poor ecosystem integrity (Tibebe *et al.*, 2019).

Excessive nutrient enrichment in aquatic ecosystems affects the structural characteristics of communities and individual species (Lopes *et al.*, 2011) as well as functional energy transfer and nutrient cycling attributes (Sarneel *et al.*, 2010). For instance, nitrogen enrichment causes

reduction in species richness, change in species composition (Datri *et al.*, 2015), loss of key species with knock on effects on drinking water quality (Halliday *et al.*, 2014). Enrichment with phosphorus, which is derived mainly from fertilizers, septic tank discharges, human excrement and detergents causes toxic algal blooms (Carpenter, 2005). Moreover, excess concentrations of essential growth limiting nutrients in aquatic systems lead to water quality deterioration with increased cost of treatment (Datri *et al.*, 2015), eutrophication (Carpenter, 2005) loss of biodiversity, disrupted food web structure, high organic matter accumulation and greenhouse gas emissions leading to climate change (Pearl *et al.*, 2016; Yun and An, 2016; Costa *et al.*, 2018).

Spatio-temporal variations in nutrient availability and loading significantly influence abundance and distribution of aquatic communities. It also causes accelerated eutrophication of aquatic ecosystems with consequent problems of water quality deterioration. Moreover, physico-chemical water quality parameters including dissolved oxygen (DO), temperature and salinity influence the transformations of nutrients from one form to another. Finally, they alter the dimension of availability in usable form resulting in community structure dynamics (Pearl *et al.*, 2016).

Consequently, the above mentioned pressures coupled with various anthropogenic and hydrological drivers, significantly affect plankton community composition and structure. Numerous studies have indicated that heterogeneity in distribution of phytoplankton at different scales result from eutrophication and toxic cyanobacterial blooms (Mihaljevic *et al.*, 2014; Zou *et al.*, 2022). Additionally, physical factors for example, light, temperature, wind stress, turbulent mixing and balance of inlet/outlet discharge causes spatial heterogeneity of phytoplankton mainly in shallow water bodies (Reynold, 2002; Ogato and Kifle, 2017). Similarly, alterations of physico-chemical parameters such as temperature and dissolved oxygen, hydrological and

anthropogenic influences on aquatic systems (Imoobe and Akoma, 2009; Fetahi *et al.*, 2011a; Effendi *et al.*, 2016) cause pronounced changes in the abundance, distribution and community composition of zooplankton. Hence, they lead to progressive degradation of biodiversity (Xiong *et al.*, 2020; Mequanent *et al.*, 2022). The pattern of plankton observed in the monitoring of aquatic systems is an indicator of long term ecological changes (Mihaljecic *et al.*, 2015). Therefore, for better understanding, and enhancing traditional classification approach by grouping together those species with similar tolerance and sensitivity to various ecological properties is crucial (Allende *et al.*, 2019).

Ethiopia is endowed with a large number of inland freshwater bodies including 12 major river basins with mean annual flow of 122 billion meter cube and more than 36 lakes, with surface area of about 7,500 km² (FAO, 2015; Gebremedhin *et al.*, 2018; Engdaw *et al.*, 2022). Besides their widespread use as sources of water for drinking, irrigation, and hydroelectric generation, fisheries production, and their importance in flood and erosion control, these aquatic systems sustained diverse and productive floral and faunal communities (Schallenberg *et al.*, 2013; Goshu and Aynalem, 2017; Gebremedhin *et al.*, 2018; Kassa *et al.*, 2021).

Limnological expeditions to Ethiopian inland freshwaters have been conducted since 1930 (Omer-Cooper, 1930; Talling and Talling, 1965; Hecky and Fee. 1981). These studies primarily focused on physico-chemical water quality parameters, plankton composition, biomass, and production in the rift-valley lakes due to their proximity to Addis Ababa University where foreign scientists were stationed and potential for comparative analysis (Mengistou *et al.*, 1991; Dadebo, 2000). However, this resulted in the paucity of scientific information on high altitude/highland (>1800 m a.s.l) lakes such as Lake Tana, which are remotely located and difficult to access. Limnological studies on these aquatic systems are crucial for determining the

suitability of their water for particular domestic or industrial use, as well as for monitoring, conservation, and sustainable exploitation of their resources.

By far, Lake Tana is the largest freshwater lake in Ethiopia accounting for 50% of the freshwater lake area of the country (Goshu and Aynalem, 2017; Engdaw *et al.*, 2025). It is the main source of Blue Nile River that sustains the entire nations of Sudan and Egypt (Fenta *et al.*, 2018). Thus, the quantity and quality of its water is critical for millions of people and peaceful neighborhood. The lake is Africa's top 32 with global rank of 250 for conservation priority (Teshome *et al.*, 2021; Yasuda *et al.*, 2022). Lake Tana is vital component of the environment and biodiversity heritage where livelihood depends on the lake's numerous ecosystem services (Wendie *et al.*, 2007; Gitaw *et al.*, 2020; Kassa *et al.*, 2021; Dersseh *et al.*, 2022). Furthermore, the lake plays a crucial role in local climate regulation and was recently designated as a UNESCO-biosphere reserve in 2015 for its extended ecological and socioeconomic services (Goshu and Aynalem, 2017; Teshome *et al.*, 2021).

1.2. Statement of the problem

As a source of River Abbay (Blue Nile), Lake Tana received great attention from researchers many years ago (Nagelkerke, 1997; de Graaf, 2003; Dejen, 2002; Kebede *et al.*, 2006; Wondie *et al.*, 2007; Akoma and Imoobe, 2008; Kassa, 2021; Goshu and Aynalem 2017; Gebremedhin *et al.*, 2018; Getachew *et al.*, 2021; Ayele and Atlabachew, 2021; Dersseh *et al.*, 2022). Despite the concerted efforts made to investigate various aspects of the lake, more research emphasis was given to the fisheries aspect of the lake especially, to the ecology and taxonomy of the barbs' species flock (Nagelkerke, 1997; de Graaf, 2003; Dejen *et al.*, 2002). Besides, the limnological studies were conducted in the Bahir-Dar gulf of the lake. Other northern and north-eastern gulfs

such as the Gorgora, Delgi, Kunzila and Lemba gulfs were marginalized in the studies due to their remote location, access and logistic constraints.

In the last two decades, Lake Tana's ecosystem consistently changed due to severe stress from anthropogenic activities, hydrological variations, climatic changes and other natural factors (Wendie *et al.*, 2007; Paerl *et al.*, 2016; Goshu and Aynalem, 2017; Kebedew *et al.*, 2020; Mequanint and Mingist, 2019; Kahsay *et al.*, 2022). Rapid urbanization and industrialization increase the amount of domestic and industrial untreated wastewater. This wastewater is discharged directly into the lake or its tributaries, adversely affecting the water quality and biota of the lake (Gebremedhin *et al.*, 2018; Tibebe *et al.*, 2019; Engdaw *et al.*, 2022). Agriculture is considered as a major non-point source of pollution and the intensive agriculture in the region with serious encroachment of the lake shore and excessive applications of fertilizers and pesticides result in nutrient pollution, persistent heavy metal and organo-chloride chemical pollution of water bodies that damage the ecosystems health (Tadesse *et al.*, 2019; Fetahi, 2019; Kassa *et al.*, 2021). Furthermore, land use/ land cover change including real-state construction, extensive deforestation, poor soil-water conservation practice, human settlement, shoreline modification through hotel and resort development, river continuum disruption due to dam construction (Megech, Rib, and Koga), infestation with the invasive alien macrophyte water hyacinth, over-exploitation of fish resources and applications of illegal fishing gears, water abstraction for various development activities are currently threatening the water quality, biodiversity and interactions of whole-lake ecosystem (Mequanint and Mengist, 2019; Goshu and Aynalem, 2017; Alemu *et al.*, 2018; Akale *et al.*, 2018; Tibebe *et al.*, 2019; Kassa *et al.*, 2021; Engdaw *et al.*, 2022; Dersseh *et al.*, 2022).

The water quality of Lake Tana is deteriorating due to severe anthropogenic activities which could affect the plankton community structure distribution and biomass, and consequently the food web structure and function of the ecosystem. These changes may affect the endemic and world heritage of *Labeobarbs* fish flock. Nutrient enrichment and other pollution may pose serious public health risks. Moreover, the current status of the lake particularly in the northern and north-eastern parts has not been documented, which may have been affected by the recent developments in the region. Therefore, the present study aims to document the land use/land cover changes in the watershed of Lake Tana's northern gulf, its water quality and plankton community structure for monitoring and management interventions.

1.3. Research questions

1. Are there significant LU/LC changes in the northern Lake Tana catchment between 1993 to 2022?
2. Is there any spatio-temporal variability in physico-chemical water quality parameters of Lake Tana?
3. Do phytoplankton community structure and biomass vary seasonally and spatially?
4. Is there spatio-temporal variability in zooplankton community structure?

1.4. Objectives

1.4.1. General Objective

- ✚ To assess the spatial and temporal variations in plankton community structure, and biomass in relation to physico-chemical water quality parameters in Lake Tana's northern Gulf and Land Use/land cover changes in its watershed.

1.4.2. Specific objectives

- ✚ To assess the land use land cover changes of northern Lake Tana catchment from 1990-2022
- ✚ To examine spatial and temporal variations in physico-chemical water quality parameters and trophic status of northern Lake Tana gulf using tropical models.
- ✚ To assess spatio-temporal variations in phytoplankton community structure, and biomass in Lake Tana's northern Gulf.
- ✚ To assess spatio-temporal variation in zooplankton community structure in the northern gulf of Lake Tana.

1.5. Hypothesis

- ✚ There are significant LU/LC changes in the northern catchment of Lake Tana due to rapid population growth and urbanization.
- ✚ Physico-chemical water quality parameters show spatio-temporal variations and their levels exceed permissible limits of various standards (EEPA, USEPA, ECE, WHO) for aquatic life.
- ✚ Community structure and biomass of the plankton assemblages in Lake Tana vary spatio-temporally.

1.6. Significance of the study

The present study will address the knowledge gap regarding land use/land cover change, trophic state, diversity, and community structure of plankton communities in relation with nutrient dynamics in the northern gulf of Lake Tana. By providing clear and comprehensive information, this study will serve as a valuable reference for future research on the ecosystem dynamics following the construction of recent mega projects such as Dine for Ethiopia project, hotel and resort development, and Megech dam construction. It will also assist in the development of a strategic and sustainable management plan to maintain a clean ecosystem. As the riparian community relies on the lake water for domestic use, understanding the water quality is crucial for identifying potential public health concerns. The study will provide a detailed analysis of the species composition and diversity of planktons, as well as the impacts of land use/land cover change and trophic state on the lake. This study will be beneficial for policy makers, hydrologists, and fishery officers in the development and implementation of watershed management systems and the prevention of water pollution. Furthermore, the study will make recommendations for intervention measures to mitigate the impacts of both point and non-point sources.

1.7. Scope of the study

This scope of this study includes a comprehensive evaluation of land use and land cover dynamics from 1993 to 2022, as well as spatio-temporal nutrient dynamics, specifically TN, TP, SRP, Si, NH_4 , NO_3 and NO_2 and record the community structure of plankton in Lake Tana. Furthermore, the study elucidates the trophic state of the lake using a model for tropical aquatic systems. Representative water and plankton samples were collected from both the littoral and pelagic zones, along with measurements of key physico-chemical parameters such as temperature, pH, DO, EC and turbidity which were measured on site. It should be noted that the study only

covers a small area of the lake and may not accurately represent the overall status of the lake. To provide context, the results of this study were compared to previous studies conducted at different sites within the same lake, as well as to various studies from around the world. Although more data was collected seasonally, the frequency of sampling was limited to four occasions due to logistical and financial constraints.

1.8. Limitation of the study

Other water quality parameters such as toxic heavy metals, organochloride compounds, persistent pesticides, and biological contaminants are not included with frequent sampling covering large area of the lake due to time, financial constraint, and making the project feasible within the study period.

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CHAPTER 2

Land use/land cover dynamics in the Northern watershed of Lake Tana: implications for water quality

This chapter is based on the following published paper:

Flipos Engdaw, Tadesse Fetahi, and Demeke Kifle (2024). Land use/land cover dynamics in the northern watershed of Lake Tana: implications for water quality. *Front. Environ. Sci.* 12:

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Abstract

Rapid population growth-induced urbanization, industrialization, deforestation, settlement expansion and habitat destruction are significant drivers of LU/LC change at global scale. The uncontrolled LU/LC change, specifically in developing countries like Ethiopia, poses significant environmental threats to water quality, biodiversity, and ecosystem services. This study, therefore, examined LU/LC trends in the Lake Tana catchment, Ethiopia from 1993 to 2022, analyzed the impact of changes on water quality and identified major driving factors for policy input. ArcGIS software and Landsat imagery were employed to produce maps for 10 years. Water quality data were obtained from primary and secondary sources from 2002 to 2022. The relationship between LU/LC changes and water quality was examined using Pearson's correlation and principal component analysis (PCA). Results revealed significant ($p < 0.05$) spatio-temporal variations in water quality parameters, with almost all parameters showing a substantial increase from 2002 to 2012 and some declined in 2022. LU/LC changes were categorized into six classes with an overall accuracy and Kappa coefficient of 92.2% and 0.86%,

respectively. Various LU/LC classes were highly correlated with a particular water quality parameter. Settlement and cropland expansion were positively correlated with nutrients such as ($\text{NH}_3^- + \text{NH}_4^+ - \text{N}$), $\text{NO}_3 - \text{N}$, SRP and TP. Forestation was positively correlated with good water quality such as DO and pH, while they were negatively correlated with temperature, EC, SRP and $\text{NO}_3 - \text{N}$ demonstrating a strong linkage between LU/LC and water quality. These findings therefore, validate the significance of prioritizing LU/LC changes for sustainable water quality and ecosystem service improvement in the Lake Tana catchment.

Key Words: green legacy, land use/land cover, population growth, settlement, water quality

2.1. Introduction

A study on land use/land cover (LU/LC) is crucial for understanding global environmental patterns and ecological trends in natural landscapes (Wolde-Yohannes *et al.*, 2018; Matlhodi *et al.*, 2019; Gashaye *et al.*, 2023). It serves as a foundation for sustainable natural resource management through the development of evidence-based environmental monitoring policies (Lu *et al.*, 2004; Mengistie *et al.*, 2013; Cotter *et al.*, 2014; Gashaw and Fentahun, 2014). Rapid population growth-induced urbanization, industrialization, deforestation, soil erosion, desertification, shortage of arable land, and habitat destruction are significant drivers of LU/LC change at global scale (Muriuki *et al.*, 2011; Biazin and Sterk, 2013; Mengistie *et al.*, 2015; Beyene and Minale, 2023). The intricate and dynamic alterations in LU/LC at various spatial and temporal scales have implications on environmental and ecosystem services (Gashaw *et al.*, 2018; Bufebo and Elias, 2021; Gashaye *et al.*, 2023). For example, LU/LC alteration significantly affects the landscape, ecological imbalance and ecosystem services (Hassen and Assen, 2017).

During the last 30 years, significant global LU/LC change has been observed with the vast expansion of pastureland and cropland at the expense of forest and grassland (Lambin *et al.*, 2003; Hassen and Assen, 2017). However, the direction and magnitude of the change is not uniform across the world. For instance, global rate of deforestation from 1990-2005 was estimated at about 13 million ha/yr (FAO, 2005; Garedew *et al.*, 2009). In the temperate regions forested areas increased by 3×10^6 ha/yr, while in the tropics, forest cover and grassland declined by 12×10^6 ha/yr (Gashaw *et al.*, 2018). Between 2015 and 2020, an estimated 6 million hectares of natural forest area was degraded globally, with the majority occurring in tropical regions (Matlhodi *et al.*, 2019; FAO, 2020; Gashaye *et al.*, 2023). Deforestation remains a primary pressure on LU/LC that substantially affecting global environmental change. Similarly, population growth-induced higher urbanization rate has been reported in many developing countries than developed nations (Beyene and Minale, 2023). Only 28.8% of world's population lived in urban centers in 1950 compared to 56% in 2020, and projected to 70% in 2050 (Avis, 2016; Fitawok *et al.*, 2020). Hence, unplanned and uncontrolled LU/LC changes particularly in sub-Saharan Africa remains a primary pressure that affect water quality and ecosystem service (Beyene and Minale, 2023).

Ethiopia has one of Africa's fastest growing populations, with the total number of inhabitants expected to reach 133.5 million by 2030 (UN, 2018). LU/LC change is manifested in the expansion of deforestation, settlement and croplands (Ariti *et al.*, 2015; Garedew *et al.*, 2009; Hassen and Assen, 2017; Gashaw *et al.*, 2018; Bogale, 2020; Bufebo and Elias, 2021; Getachew *et al.*, 2021; Gashaye *et al.*, 2023). The country had the second-highest rate of deforestation in Africa, with rates of 0.94% for 1990-2000 and 0.97% for 2000-2005 (Garedew *et al.*, 2009). Cultivated land has expanded in the Andassa watershed of the Blue Nile basin between 1985-

2015 resulting in decline of forest, shrub-land and grassland (Gashaw *et al.*, 2018). However, opposing trends of LU/LC changes have also been reported (Wolde-Yohannes *et al.*, 2018). For example, natural forest regrowth has been reported in the uplands of Bela-Welleh catchment (Nyssen *et al.*, 2009). A similar extent of LU/LC change has been reported from the Lake Tana area, which is the focus of the current study. Recently, an increase in settlement by 22% and a substantial decline in cropland coverage by 11% were also reported from the north-eastern catchment of Lake Tana (Gashaye *et al.*, 2023). Further expansion of grassland and riparian vegetation cover in Gish Abbay watershed was also recorded (Bewuketu and Solomon, 2013). LU/LC change cause regional water quality deterioration, habitat destruction, biodiversity loss and a decline in ecosystem integrity (Bewuketu and Solomon, 2013; Kindu *et al.*, 2013; Ariti *et al.*, 2015; Minta *et al.*, 2018; Matlhodi *et al.*, 2019 Gashaye *et al.*, 2023).

Though Ethiopia is endowed with large inland freshwater bodies that provide ecological and ecosystem services essential to the livelihoods and economic revenue to the region (Goshu and Aynalem, 2017; Gebremedhin *et al.*, 2018; Wondie, 2018; Assefa *et al.*, 2021; Wubneh *et al.*, 2022), the lakes and the ecosystems within their catchment are under stress from severe LU/LC change (Alemu *et al.*, 2017; Hassen and Assen, 2017; Moisa *et al.*, 2022). The interdependence between LU/LC, water quality and the health of aquatic ecosystems is reported in many parts of Ethiopia (Hassen and Assen, 2017; Gashaw *et al.*, 2018; Serur and Adi, 2022; Gashaye *et al.*, 2023; Gule *et al.*, 2023; Muche *et al.*, 2023). Correspondingly, the water quality of several lakes is deteriorating ranging from eutrophic to hyper-eutrophic condition due to degraded catchment areas (Fetahi, 2019). Lake Tana, Ethiopia is not exceptional. Recent study indicated substantial LU/LC change in the northern Lake Tana catchment, such as conversion of natural land cover to settlement and agricultural land (Gashaw *et al.*, 2018; Abuhay *et al.*, 2023). Agricultural

expansion, charcoal production, firewood harvesting and real-state construction have rapidly increased significantly affecting the forest and grazing lands (Gebremichael *et al.*, 2013; Woldesenbet *et al.*, 2017; Muche *et al.*, 2023). These human-induced change in LU/LC leads to water quality deterioration and biodiversity impoverishment (Muriuki *et al.*, 2011; Gashaw *et al.*, 2018; Abuhay *et al.*, 2023).

Lake Tana is the largest lake in Ethiopia, and the source of Blue Nile River, which sustains Ethiopia, Sudan and Egypt (Fenta *et al.*, 2018; Minale, 2019; Yasuda *et al.*, 2022). Thus, the quantity and quality of its water is critical for millions of people and peaceful neighborhood. However, these aquatic ecosystems are heavily affected by LU/LC change (Hassen and Assen, 2017; Abuhay *et al.*, 2023).

To reverse deforestation, land degradation, water quality deterioration and biodiversity impoverishment triggered by the severe LU/LC modifications, the Ethiopian government has recently launched a green legacy initiative through reforestation and sustainable forest management programs (Lambert and Deyganto, 2023; Muche *et al.*, 2023). With an emphasis on sustainable development and combating drought-induced desertification, the initiative aimed to restore degraded lands, conserve and improve water resource management (ECRGE, 2015). The northern watershed of Lake Tana, where croplands and Afromontane forest have been converted as a result of settlement, is known for having mosaic of various LU/LC types. Despite the concerted efforts being made to avert the situation, there is paucity of information revealing the trend of LU/LC dynamics and its impact on water quality of northern Lake Tana watershed. Therefore, this study examined the trends in LU/LC changes in the Lake Tana catchment between 1993 and 2022. The study also documented the historical water quality parameters and compare with the contemporary data. We analyzed the impact of LU/LC change on water quality

and pinpoint the major driving factors on the catchment. We hypothesized that better LU/LC improves the water quality and ecosystem services. The study will also aims to identify the major driving factors affecting water quality within the catchment for policy and decision makers.

2.2. Materials and Methods

2.2.1. Study area

Lake Tana, the main source of Blue Nile River and Grand Ethiopian Renaissance Dam (GERD), is the largest lake in Ethiopia, which accounts for 50% of the freshwater lake area of the country without considering the recently built reservoirs (Goshu and Aynalem, 2017; Wolde-Yohannes *et al.*, 2018; Wondie, 2018). The lake is located on the basaltic plateau of north-western highlands of Ethiopia, between 12°10'N and 37°23'E and at an altitude of 1800 m a.s.l. (Fig. 2.1). Lake Tana is among Africa's top 32, with global rank of 250 for conservation priority (Teshome *et al.*, 2021). The total surface area of the lake during dry and wet seasons ranged from 3,050 to 3,200 km² with maximum and mean depths of 14 and 8 m respectively (Wondim, 2016; Goshu and Aynalem, 2017). Lake Tana is a vital component of the environment and biodiversity heritage, where the livelihood of local people is highly dependent on its numerous ecosystem services (Derseh *et al.*, 2019). Furthermore, the lake plays a crucial role in local climate regulation and was designated as a UNESCO-biosphere reserve in 2015 for its extended ecological and socioeconomic services (Teshome *et al.*, 2021; Sewunet *et al.*, 2022).

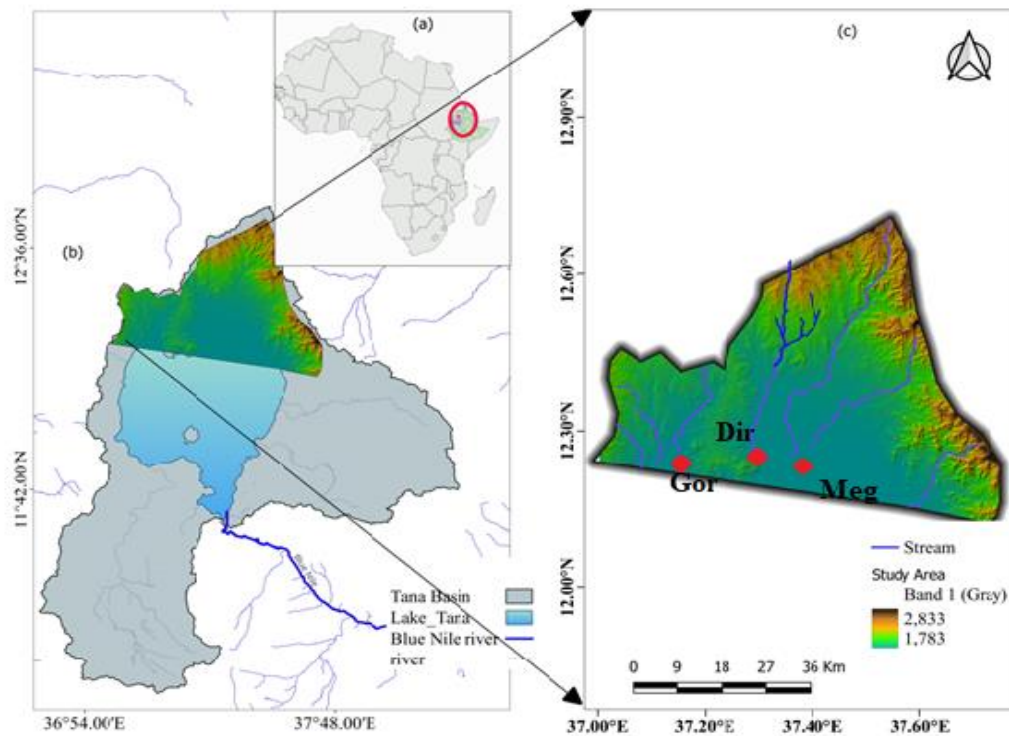


Fig. 2.1. Location map showing Lake Tana catchment and the northern Lake Tana study area with increasing elevation (green to red) and streams and major sampling sites (Gor = Gorgora; Dir = Dirma, Meg = Megech).

Lake Tana's northern watershed is typical of Ethiopian highlands with mountainous topography and extensive agriculture. It is part of the north and central Gondar zone of the Amhara regional state (Fig. 2.1). The watershed extends up to Simen mountain national park, with an altitude of 3,500 to 4,000 m a.s.l. (Alemu *et al.*, 2017). The study area consists of varying topographic features including high plateau, hills, plains and valleys. According to the Ethiopian National Meteorological Service Agency (NMA, 2024), the study area had monthly rain-falls ranged between 89 to 343 mm, with mean monthly maximum temperature of 23.4 to 32.5°C and mean monthly minimum temperature of 12.2-14.5°C. The region's climate is 'tropical highland monsoon' with a major rainy season between June and September and minor rainy season from

March to April. Maximum monthly rainfall is recorded during July/August where it reaches 250-330 mm, while mean annual rainfall of the region is above 1,280 mm (Uhlenbrook *et al.*, 2010; Abebe *et al.*, 2017). Mean annual humidity and evapotranspiration ranged from 44-80 g/m³ and 773 mm, respectively. The catchment includes the head waters of Angereb, Megech and Dirma Rivers that drain through Dembia plain before they flow into Lake Tana. These rivers are also listed among the lake's major tributaries and primary water sources for Gondar city and surrounding areas (Abebe and Kebede, 2017).

Currently, Lake Tana watershed is facing multiple stressors to continue as a resilient ecosystem. The recent government-led green legacy initiative that endorses forestation, conservation and restoration of water resource has emerged as the main land cover driver. Furthermore, rapid population growth-induced agricultural expansion towards lake shore, river banks, wetlands and preserved forests has caused alarming land cover change (Wubie *et al.*, 2016). Consequently, accelerated soil erosion attributed to steep slopes and poor soil conservation practices, has resulted in loss of vegetation cover and hence occurrence of deforestation (Nyssen *et al.*, 2009). Moreover, population triggered built-up and settlement, urbanization, climate alteration and small scale extensive agriculture make the region vulnerable to severe environmental degradation (Sewunet *et al.*, 2022). Siltation resulted from poor soil conservation and farming practices, habitat and shoreline modification due to hotel and resort construction, river continuum disruption due to dam construction (Engdaw *et al.*, 2022), proliferation of exotic macrophytes such as water hyacinth (Derseh *et al.*, 2019), excessive agrochemical application to improve yield (Akale *et al.*, 2018), improper waste discharge and water abstraction have become significant problems hampering sustainable lake watershed management and led to water quality deterioration (Goshu and Aynalem, 2017; Gebremedhin *et al.*, 2018; Sewunet *et al.*, 2022).

2.2.2. Water Quality Data collection and analysis

Despite the lack of regularly collected consistent historical data on the watershed and surface water of Lake Tana, an attempt was made to acquire data of more than two decades from primary and secondary sources. Abbay basin water quality laboratory (ABWQL), an organization concerned with the monitoring of Lake Tana and its major tributaries provided a monthly water quality data on Dirma and Megech Rivers, which are among the major tributaries of the northern gulf of the lake and Lake Tana basin. The data was collected at monitoring (gauging) stations located near the river mouth (before the confluence point) and data for the Gorgora site was collected at the gulf station which is closer to the old port. For the missing dataset (for the months of December to March of 2003 and 2004), additional secondary data were obtained from published articles (Goshu *et al.*, 2017; Womber *et al.*, 2022), unpublished thesis, government reports of Ethiopian central statistical agency (CSA) and environmental protection agency (EPA). Appraisals of secondary data sources were associated with seasonal representations focusing on the northern gulf of the lake and consideration of our parameters of interest (physico-chemical). Temperature (Temp), pH, dissolved oxygen (DO), turbidity, total dissolved solids (TDS), total suspended solids (TSS), alkalinity, electrical conductivity (EC), total hardness, sulphate (SO_4^-), ammonia ($\text{NH}_3 + \text{NH}_4\text{-N}$), nitrate ($\text{NO}_3\text{-N}$), nitrite ($\text{NO}_2\text{-N}$), soluble reactive phosphate-phosphorus (SRP) and total phosphorus (TP) were our parameters of interest, which were assumed to adequately describe the lake's physico-chemical water quality.

Data was obtained through field measurements and laboratory analysis. *In situ* measurements of some parameters including Temp ($^{\circ}\text{C}$), DO (mg l^{-1}) and EC (K_{25} , $\mu\text{S cm}^{-1}$) were made using a multi-meter probe (HQ40d, model10115 HACH). pH was recorded using a portable probe (HANNA, HI991301), while turbidity (NTU) was measured using a turbidity meter (Wag-

WT3020). TDS (mg l^{-1}) was measured using a water proof TDS meter (HANNA, HI98130). For nutrient analysis, a composite sample was produced from replicated water samples collected across the width of Megech and Dirma river mouths, 30 cm away from the left and right bank of the river (Engdaw *et al.*, 2022). Furthermore, water samples in the lake basin were collected from three sampling points at each of the three different sampling sites. All the collected water samples of a site were mixed in a single container to produce one composite sample. The composite water samples were filtered using 0.47 micrometer pore size glass fiber filter paper (GF/F), and used for the analyses of SRP, $\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$, $\text{NH}_3 + \text{NH}_4^+\text{-N}$ and SO_4^- . Unfiltered water samples were used for the analysis of total alkalinity, total hardness and TP. All samples were processed and analyzed according to the methods outlined in APHA (1999).

Soluble reactive phosphate-phosphorus (SRP) was analyzed using the ascorbic acid method, while TP was determined by the ascorbic Acid method after persulfate digestion. Nitrate-nitrogen was determined using the sodium-salicylate method, while $\text{NO}_2\text{-N}$ was quantified colorimetrically as reddish purple azo dye by coupling diazotized sulfanilamide with N-Naphthyl-(1)-ethylendiamine-dihydrochloride. Ammonia nitrogen was measured by the phenate method, while sulfate was estimated using turbidimetric method. Iron was determined using ICPOES. Total alkalinity was estimated by titrating 100 ml of unfiltered water sample using 0.1 N HCl as a titrant and a bromocresol green-methyl red mixed indicator, and calculated according to Wetzel and Likens (2001). Total hardness was determined by titrating 25 ml of sample using EDTA and buffer solution (APHA, 1999).

2.2.3. Data acquisition and pre-processing

LU/LC dynamics analysis involved the collection of primary and secondary data. In order to have a better understanding of the primary land use inside cities and towns, an exploratory field

survey was conducted in search of ground truth data on the northern watershed of Lake Tana and riparian towns such as Gondar, Gorgora, Koladiba and Chuahit. All land cover classes were identified and documented across each town and watershed. Each sampling site's GPS coordinates were also noted and recorded. The land cover data for this study were classified using satellite information on the years 1993 to 2022, which was acquired and processed using Google earth Engine, a platform for online pre-processing of satellite imagery, enhancement and classification based on machine learning algorithms (Gorelick *et al.*, 2017; Gule *et al.*, 2023). To minimize visual clutter and reduce the effect of cloud cover, we obtained images taken during the dry season of the watershed, which ran from November to January. Images used to map and identify the various land cover classes were obtained for the years 1993, 2002, 2013 and 2022. Landsat data used for mapping land cover, which were selected based on their constant temporal coverage and geographical resolution, are indicated in Table 2.1.

Table 2.1. Description of spatial dataset used for mapping land cover

Satellite data	Acquisition date	Bands	Spatial resolution
Landsat 8	15 November - 01 December 2023	Thermal bands (2)	30 m
	01 December – 15 December 2023	SWIR (2)	
		NIR	
Landsat 5 TM	01 January – 15 February 2024	Thermal band (1)	30 m
	16 February – 05 March	SWIR (2)	
	10 March – 20 March	NIR	
	21 March – 31 March	Visible bands (4)	

2.2.4. Image processing and land cover change

Satellite image processing was used to extract attributes of the target images at several phases, such as image correction, enhancement, cloud removal, subset/masking and band composition/stacking using ENVI software version 5.3. To evade significant distortion of image

statistics and overall accuracy, image pre-processing involved setting background pixel values using ENVIROI tool and masking cloud cover using (QA) band were performed. Seamless techniques were also implemented to mosaic images and color correction was applied based on histogram matching technique. Processed composite satellite images for the years 1993, 2002, 2013 and 2022 were classified independently using ArcGIS 10.5's Random Trees classification algorithm. Thus, LU/LC time series maps were produced. Classes were demarcated based on FAO's land cover classification system (LCCS) (FAO, 2016), as illustrated in Table 2.2.

Table 2.2. Descriptions of LU/LC classes identified in the Northern watershed of Lake Tana

Class	LCCS standard	User class definition
Bare land	Bare areas, rocks, cliffs, no dominant activity, lack artificial cover	Fallow land, exposed rocks and sandy areas
Cropland	Cultivated and managed areas used for annual and perennial crop, vegetables and fruit production	Cultivated terrestrial areas farms, reclaimed lands
Forest	Land covered with dense evergreen natural and semi-natural terrestrial vegetation's, forest and mixed plantations	Grasses, shrubs, trees
Grassland	Areas covered with grass used for grazing	Grazing land,
Settlements	Built-up areas, artificial surface and associated areas	Built-up structure, rural residential & constructed sites
Water	Natural and artificial water bodies	Lakes, artificial dams, wetlands & rivers

Geometric compatibility of images from different time classes is crucial for determining temporal changes and hence, ENVI SPEAR Ortho-rectification Wizard was used to rectify the images. A topographical map at 1:50,000 scale was used to geometrically correct the images using the WGS 1984 spatial coordinate system. Following the identification of Ground control

point (GCPs) using satellite image and topography maps, the linear geometric correction function was applied.

2.2.5. Image classification and accuracy assessment

Accurate data on LU/LC change can be obtained from multiband raster satellite images, which are valuable for monitoring LU/LC change and its impact on environmental alterations (Li *et al.*, 2014; Obubu *et al.*, 2022). Thus, several core principles including radiometric, spectral, spatial differentiation and digital band combinations were applied for image analysis and interpretation. In the present study, pixel based supervised classification technique that provides better extraction of LU/LC features based on Random Forest classifier was adopted (Obubu *et al.*, 2022).

The accuracy of LU/LC classification was assessed using error matrices and Kappa analysis (Congalton, 1991; Gule *et al.*, 2023; Muche *et al.*, 2023). The Landsat 8 image of 2022 was used to build the error matrix for the land cover classification, using 30% of the ground truth data points collected from the field as a reference data in 2022. Various land cover types were verified using Google Earth data points from previous years. The accuracy of each land cover class was appraised by comparing the results of the land cover classification to the ground observation using confusion matrix. The confusion matrix was then used to compute producer and user accuracy and the Kappa statistics for each class (Congalton, 1991).

To ensure the accuracy of the classification, 40 sampling points were created for each LU/LC class employing a stratified random sampling technique; hence, 240 sampling points were generated. Therefore, essential components of accuracy assessment including user accuracy (errors of commission), producer accuracy (errors of omission), total accuracy and Kappa index were obtained (Congalton, 1991; Obubu *et al.*, 2022). Total accuracy was computed by dividing

the number of correctly identified pixels in the image by the total number of pixels used throughout the classification process.

$$\text{Overall accuracy} = \sum_{i=1}^r X_{ii}$$

Where X_{ii} = diagonal elements in the error matrix; X = total number of samples in the error matrix

User accuracy was determined by dividing the number of samples of each class in the map by the number of correctly classified points in each class image (Matlhodi *et al.*, 2019).

$$\text{User accuracy} = \frac{CaU}{Ca + U} \times 100$$

Where, CaU is total number of correct classifications of a particular map; $Ca+U$ is total number of pixels classified in a particular map class

Kappa coefficient of the raster layer of the study area were determined using the statistical equation

$$\text{Kappa coefficient } (\check{K}) = \frac{n \sum_{i=1}^r X_{ii} - \sum_{i=1}^r (X_{i+} + X_{+i})}{n^2 - \sum_{i=1}^r (X_{i+} + X_{+i})}$$

Where, r = number of rows in the error matrix; X_{ii} = number of observations in row i and column i ; X_{i+} and X_{+i} = marginal totals of row i and column i respectively; and n = total number of observations (sample/pixels).

When compared to a random classification, the Kappa statistic indicates how much superior one classification is over another (Gule *et al.*, 2023). After the evaluation of the accuracy of LU/LC maps, magnitudes and pattern of the LU/LC changes were assessed by computing percent and rate of change between 1992 and 2002; 2002 and 2012; and 2012 and 2022 (Gashaw *et al.*, 2018; Abuhay *et al.*, 2023).

$$\text{Percent of change} = \frac{A - B}{B}$$

$$\text{Rate of change} = \frac{A - B}{C}$$

Where, A = recent area of the LU/LC class (Km²); B = earlier area of coverage of the LU/LC class (km²); C = the time interval between A and B years

2.3. Data analysis

Using IBM SPSS Version 25 and R-studio, the data recorded from both field and laboratory analyses were summarized using descriptive statistics such as mean and standard deviation. After the data was checked for normality using Shapiro-Wilk test, both parametric and non-parametric (Kruskal-Wallis ANOVA) tests were made to compare the temporal variations in water quality parameters and LU/LC changes. To determine the association between LU/LC change and water quality parameters, Pearson correlation coefficient was used. Principal component analysis (PCA) using Spearman's correlation coefficient was performed to identify the most important variable in determining the temporal variations between water quality variable and various LU/LC types.

2.4. Results

2.4.1. Water quality

The overall spatial and temporal physico-chemical water quality parameters for the northern gulf of Lake Tana and its main northern tributaries obtained from various sources are summarized in Table 2.3. Results indicate that there were significant ($p < 0.05$) spatio-temporal variations in several water quality parameters from 2002 to 2022.

Mean surface water temperature ranged from 20.9°C in 2002 to 24.5°C in 2022, with the lowest value recorded at Megech River mouth in 2002 and the highest at Gorgora gulf in 2022.

Throughout the study period, pH level measured in the northern gulf of Lake Tana was slightly alkaline ranging from 8.0 at Megech in 2002 to 9.0 at Gorgora in 2022. The Kruskal-Wallis non-parametric ANOVA showed that concentrations of DO have significantly declined ($p < 0.05$) over the course of the study from 8.03 at Gorgora in 2002 to 5.3 mg/l at Dirma in 2022.

EC, turbidity, TDS, total alkalinity and total hardness fluctuated considerably and rose significantly ($p < 0.05$) at all sites from their levels in 2002 to those in 2012 (Table 2.3; Fig. 2.2). The same parameters declined in 2022 at one or more sites although TDS and Total alkalinity increased to levels higher than those recorded in 2012 at Gorgora gulf. Apparently, significant spatial variations were observed between sites. For example, the highest EC was measured at Dirma River mouth ($480 \mu\text{S cm}^{-1}$) followed by those at Megech ($435.5 \mu\text{S cm}^{-1}$) and Gorgora gulf ($189.7 \mu\text{S cm}^{-1}$). Turbidity, which is a measure of water clarity showed a wide range of values varying from 24.7 to 200 NTU and differed significantly ($p < 0.05$) between sites. The highest turbidity was measured during the rainy seasons in Megech (200 ± 21.6 NTU) and Dirma (177.5 ± 28.1 NTU) river mouths while lowest values (24.7 ± 4.2 NTU) were recorded during dry months at Gorgora gulf.

Table 2.3. Mean $\pm$ SD (n= 36; i.e. 12 per year) of water quality parameters of northern Lake Tana and tributaries (* = significantly different (p<0.05) temporally).

Year	2002			2012			2022		
Parameter	Meg	Dir	Gor	Meg	Dir	Gor	Meg	Dir	Gor
Temp ($^{\circ}$ C)	22.9 $\pm$ 1.2*	23.1 $\pm$ 2.3	23.3 $\pm$ 1.2	23.3 $\pm$ 1.2	21.7 $\pm$ 3.5	22.6 $\pm$ 1.3	20.9 $\pm$ 2.2*	23.5 $\pm$ 1.3	24.5 $\pm$ 0.8*
pH	8 $\pm$ 0.2	8.03 $\pm$ 0.3	8.2 $\pm$ 0.23	8.3 $\pm$ 0.3	8.1 $\pm$ 0.3	8.2 $\pm$ 0.5	8.3 $\pm$ 0.4	8.3 $\pm$ 0.5	9 $\pm$ 0.5*
DO (mg/l)	7.95 $\pm$ 0.3*	7.95 $\pm$ 0.3*	8.03 $\pm$ 0.2*	5.4 $\pm$ 0.6*	5.6 $\pm$ 0.3	6.9 $\pm$ 0.3*	5.7 $\pm$ 0.1	5.3 $\pm$ 0.3*	7.5 $\pm$ 0.5
TDS (mg/l)	89.7 $\pm$ 6	98.2 $\pm$ 2.3	80.3 $\pm$ 4.8	147.5 $\pm$ 29	235 $\pm$ 177*	64 $\pm$ 11.4	91.1 $\pm$ 4.1	83.2 $\pm$ 4.9	79.8 $\pm$ 9.5
EC (μ S cm $^{-1}$)	182.3 $\pm$ 8.6	179.6 $\pm$ 12.9	126.9 $\pm$ 5.2	292.5 $\pm$ 60.7*	480 $\pm$ 36.7*	140.7 $\pm$ 11.4	435.5 $\pm$ 18.2	393.5 $\pm$ 13.6	166.7 $\pm$ 7.5
Turbidity (NTU)	86 $\pm$ 5.8	84.3 $\pm$ 6.2	24.7 $\pm$ 4.2*	173 $\pm$ 62.4	177.5 $\pm$ 28.1	52.6 $\pm$ 32.9*	200 $\pm$ 21.6	75.6 $\pm$ 38.3	42.8 $\pm$ 5.7*
Total alkalinity (meq/l)	175.2 $\pm$ 17.7	170.3 $\pm$ 14.6	61.1 $\pm$ 7.3*	245.8 $\pm$ 30.2*	616.3 $\pm$ 65.2*	109.3 $\pm$ 20.3	622.5 $\pm$ 38.9*	494.8 $\pm$ 18.7*	139.4 $\pm$ 23.8
TSS (mg/l)	18.2 $\pm$ 2.3	19.7 $\pm$ 1.2*	9.3 $\pm$ 1.1	21.1 $\pm$ 2.2	26.1 $\pm$ 0.3*	13.3 $\pm$ 1.3	39.3 $\pm$ 6.6	43.3 $\pm$ 6.3*	25.4 $\pm$ 3.3
Total hardness (mg/l)	36 $\pm$ 6.1	33.9 $\pm$ 6.1	23.6 $\pm$ 4.2	315 $\pm$ 100.7*	501 $\pm$ 86.1*	100.9 $\pm$ 20.9	155.8 $\pm$ 66*	125 $\pm$ 19.7*	80.2 $\pm$ 5.5
NO $_3^-$ (mg/l)	0.02 $\pm$ 0.0	0.09 $\pm$ 0.01	0.02 $\pm$ 0.0	0.05 $\pm$ 0.05	0.17 $\pm$ 0.3	0.02 $\pm$ 0.01	0.68 $\pm$ 0.13	0.61 $\pm$ 0.35	0.68 $\pm$ 0.11
NH $_3$ (mg/l)	0.28 $\pm$ 0.0	0.34 $\pm$ 0.03	0.16 $\pm$ 0.03	0.05 $\pm$ 0.03	0.29 $\pm$ 0.4	0.04 $\pm$ 0.02	2.1 $\pm$ 0.6*	2.3 $\pm$ 0.2*	1.9 $\pm$ 0.3*
NO $_2^-$ (mg/l)	0.79 $\pm$ 0.1	1.4 $\pm$ 0.84*	0.55 $\pm$ 0.05	0.39 $\pm$ 0.1	0.32 $\pm$ 0.29	0.15 $\pm$ 0.13	3.15 $\pm$ 0.6*	3.28 $\pm$ 0.1*	1.62 $\pm$ 0.2*
SRP (mg/l)	0.93 $\pm$ 0.1	1.1 $\pm$ 0.23	0.21 $\pm$ 0.09	1.31 $\pm$ 0.7*	3.6 $\pm$ 1.7*	0.29 $\pm$ 0.17	3.5 $\pm$ 1.3*	5.1 $\pm$ 1.8*	1.2 $\pm$ 0.32
TP	1.4 $\pm$ 0.2	1.7 $\pm$ 0.21	0.51 $\pm$ 0.14	1.82 $\pm$ 0.6*	4.9 $\pm$ 1.1*	0.53 $\pm$ 0.3	5.2 $\pm$ 1.9*	6.2 $\pm$ 0.3*	1.7 $\pm$ 0.4
SO $_4^-$ (mg/l)	11.8 $\pm$ 2.2*	10.8 $\pm$ 2.2*	8.5 $\pm$ 1.3*	22 $\pm$ 21.6*	106.7 $\pm$ 18.2*	14.1 $\pm$ 2.3*	110.5 $\pm$ 21.6*	256.3 $\pm$ 59*	35.7 $\pm$ 13.1*
Fe (mg/l)	0.21 $\pm$ 0.0	0.13 $\pm$ 0.02	0.11 $\pm$ 0.02	0.65 $\pm$ 0.01	3.92 $\pm$ 0.84*	0.87 $\pm$ 0.7	1.94 $\pm$ 0.89*	1.41 $\pm$ 0.05*	0.45 $\pm$ 0.03*

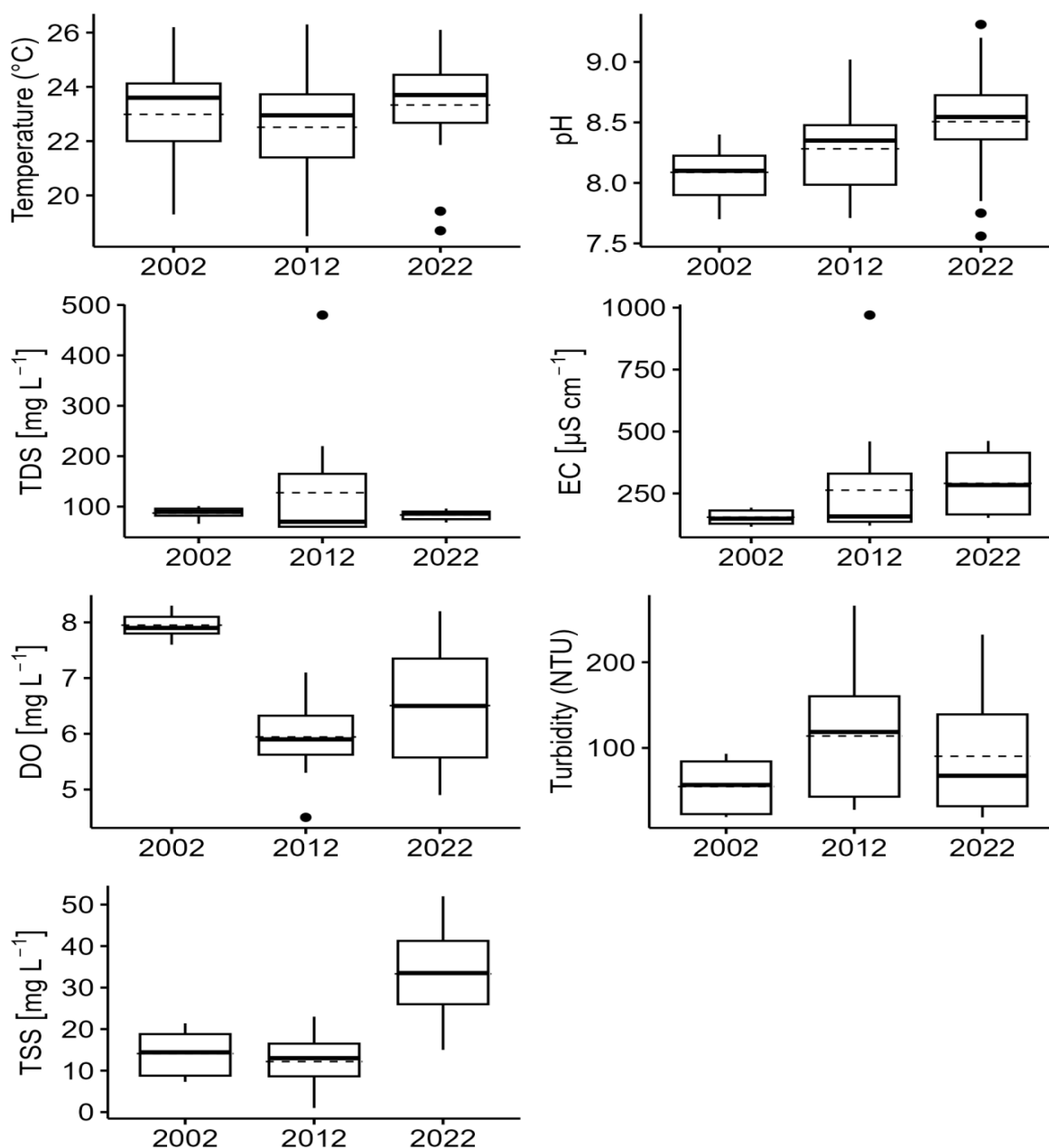


Fig. 2.2. Temporal variations in water quality parameters of the Northern gulf of Lake Tana and its tributaries Megech and Dirma Rivers (n=36, dots represent outlier data).

Concentrations of nutrients measured in the water samples of the study sites are indicated in Table 2.3 and Fig 2.3. Concentrations of TP, SRP, ammonia and NO₂-N were significantly different both spatially and temporally, whereby higher concentrations were recorded during the

rainy seasons. Compared with tributary rivers, relatively lower concentrations of nutrients were measured at Gorgora gulf and littoral region of Lake Tana.

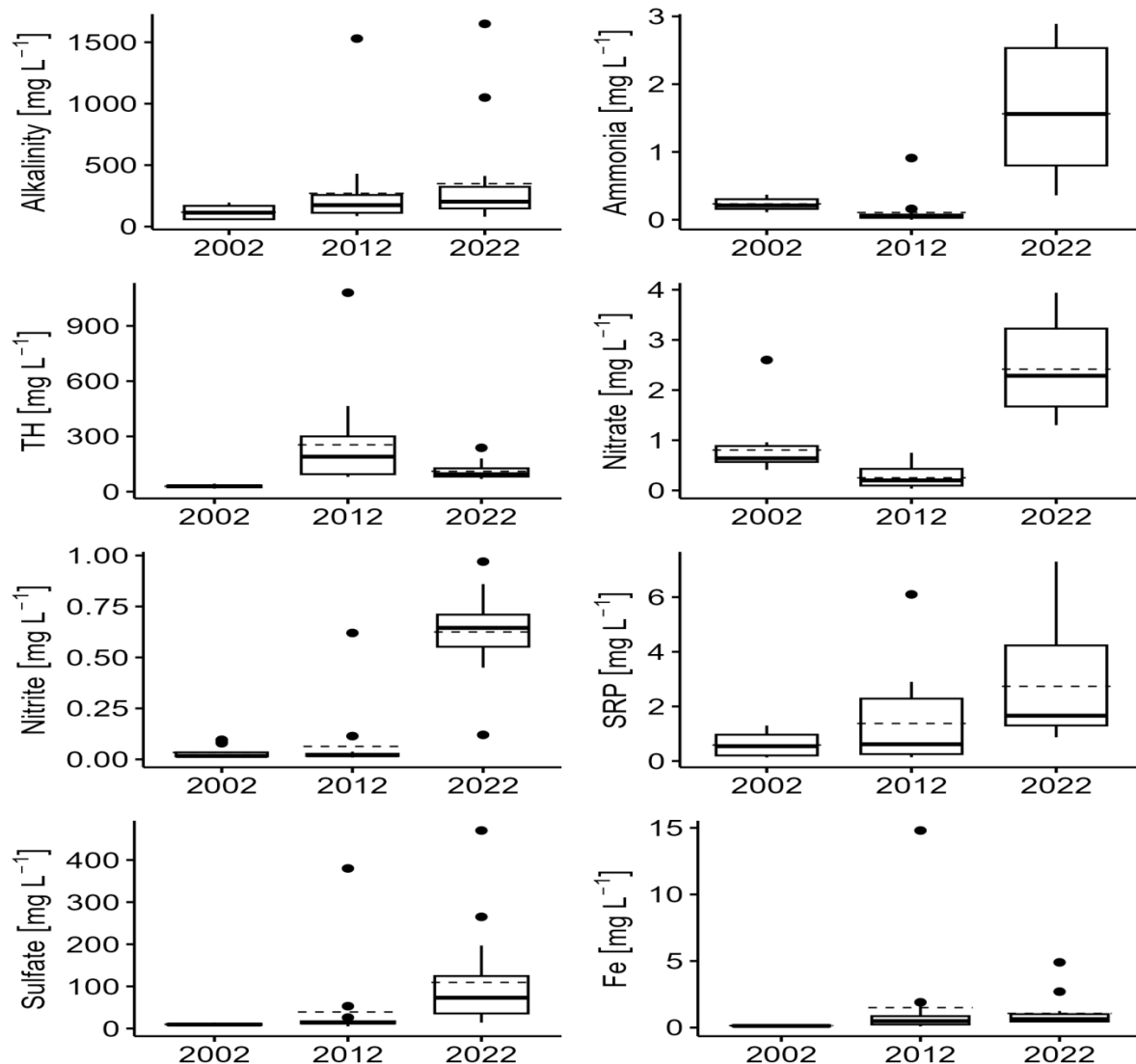


Fig. 2.3. Temporal variations in levels of nutrients, total alkalinity and total hardness measured in the Northern gulf of Lake Tana and its tributaries (n=36, dots represent outlier data).

2.4.2. Land use/land cover maps, and changes

Land cover maps obtained from the image classification indicated a remarkable change over the past three decades (Fig. 2.4). Six different land cover classes (bare land, crop land, forest, grass

land, settlement and water body) were identified. For the years 1993, 2002, 2013 and 2022, the classification results verified remarkable overall accuracy of 95.7%, 92.2%, 93.1% and 93% respectively. Accuracy values for the individual land cover classes shown in Table 2.4 revealed that, the maps are sufficiently accurate for further analysis. The accuracy of land cover classification was determined using error matrices, overall accuracy and Kappa statistics. The overall Kappa statistics ranged from 0.86 to 0.91%, while producer accuracy ranged from 87.5 to 100%. Significant variation ($p < 0.05$) was observed between accuracy of users and producers. Supervised classification had the highest accuracy. In 2022, the overall accuracy was 92.2% with Kappa statistic of 0.861%. Accuracy among users and producers was consistently high ranging from 80-100%.

At the beginning of the study period, the landscape was dominated principally by cropland that accounted for 71.4% of the total area (315672 ha), followed by water body 16.3% (72187.8 ha), forest 8.2% (36148.6 ha) and grassland 3.04% (13453.3 ha), whereas settlement (0.8%) and bare-land (0.3%) accounted for very low land cover areas. The LU/LC analysis revealed that between 1993 and 2002, cropland declined considerably by 46.6% and the area of water body remained more or less constant. However, other land cover classes such as forest, grassland and settlement showed progressive expansion by 0.9%, 0.16 and 0.224% respectively (Table 2.5).

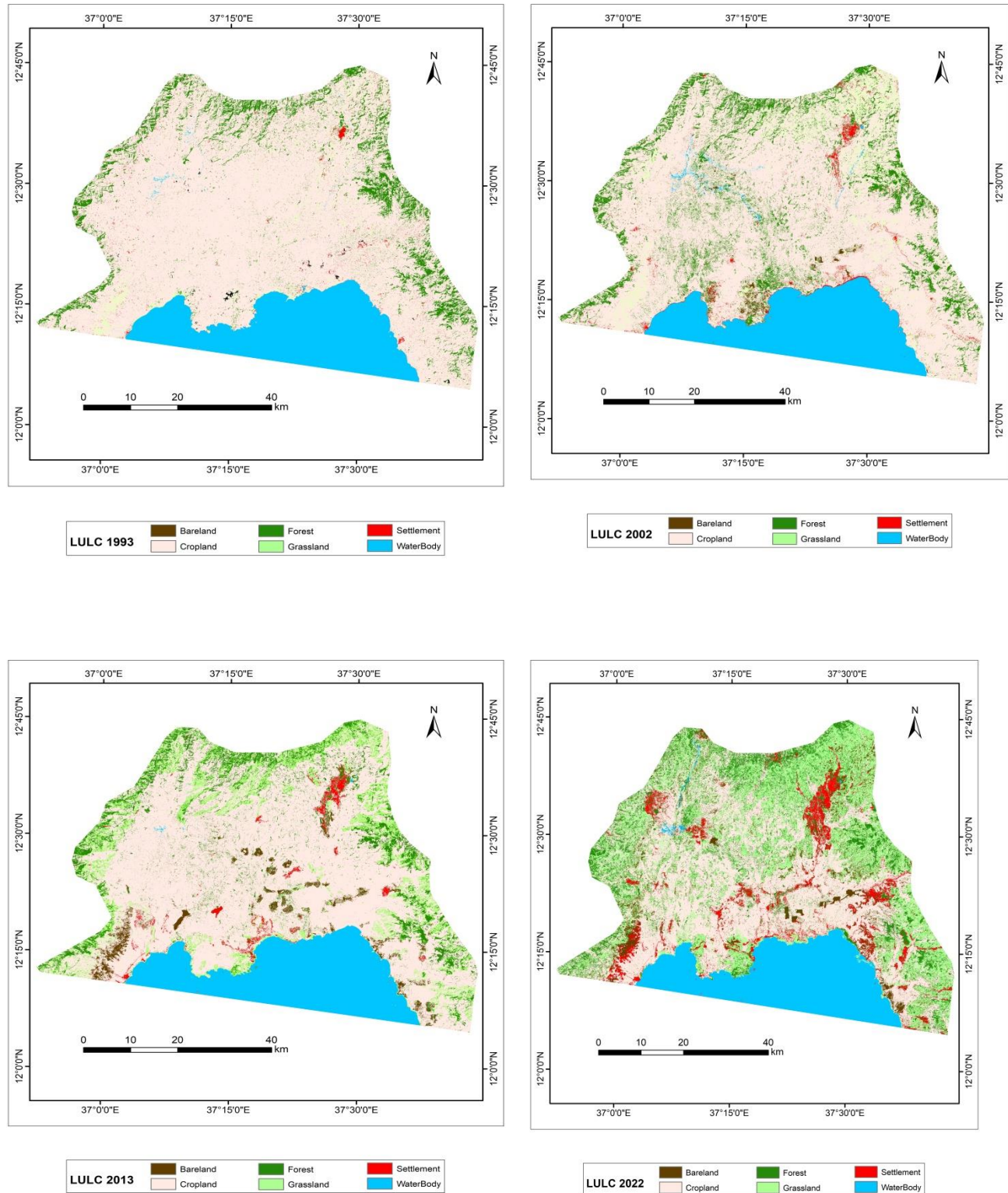


Fig. 2.4. Land use land cover changes of Northern Lake Tana catchment from 1993 to 2022

The trends and rates of change in land use types varied markedly through the study period intervals. For instance, in the year between 2002 and 2013, grassland, settlement and bare land expanded significantly ($p < 0.05$) with 0.6%, 1.8% and 2.9 % respectively. Conversely, other land cover types such as cropland and forest cover declined by 38% and 0.45%, respectively (Table 2.5). Furthermore, forest cover declined from 11.1% in 2002 to 4.1 % in 2013 at a rate of -0.63% due to anthropogenic pressure, while settlement expansion accelerated dramatically since 2002 and reached 7.8% (34461.8 ha) of the total area in 2013.

In the study period between 2013 to 2022, a substantial decline in crop land cover (by 23.1%) was observed, while a large increase in grass land from 8.5% (37459.6 ha) to 21.5% (94973.4 ha), in settlement from 7.8% (34461.8 ha) to 8.9% (39708.6) and in forest from 4.1% (18308.2 ha) to 11.6% (51438 ha) was recorded. Despite the fact that there was conversion of land cover from one use to another, the image classification clearly indicated conversion of crop land cover into grass-land, forest and settlement (Fig. 2.4).

Table 2.4. Accuracy assessment results

	1993		2002		2012		2022	
Land cover class	User accuracy	Producer accuracy	User accuracy	Producer accuracy	User accuracy	Producer accuracy	User accuracy	Producer accuracy
Water body	87.23	97.62	86.79	100	89.3	95.13	85.22	91.34
Grass land	88.89	100	80.77	87.5	91.2	97.42	91.31	97.21
Forest	86.96	95.24	88.89	69.57	88.3	93.21	93.22	96.17
Bare land	100	100	100	100	96.4	100	100	100
Settlement	100	100	100	100	100	100	100	100
Agricultural land	99.43	95.05	96.18	93.7	94.2	96.14	92.21	95.31
Overall accuracy	95.70		92.19		93.11		93.01	
Kappa statistics	0.9101		0.8608		0.8801		0.8703	

Table 2.5. Area per land covers class of Northern Lake Tana catchment in 1993, 2002, 2013 and 2022

	1993		2002		2013		2022		Overall from 1993-2022		
Land cover	Area (ha)	(%)	Area (ha)	(%)	Area (ha)	(%)	Area (ha)	(%)	Area (ha)	% of change	Rate of change
Water body	72187.8	16.3	71972	16.3	72079.9	16.3	74129.7	16.8	+1941.9	+0.004	+64.7
Settlement	3552.6	0.8	12327.4	2.8	34461.8	7.8	39708.6	8.9	+36156	+0.082	+1205.2
Grass land	13453.3	3.0	23501.2	5.3	37459.6	8.5	94973.4	21.5	+81520.1	+0.184	+2717.3
Forest	36148.6	8.2	48959.4	11.1	18308.2	4.1	51438.8	11.6	+15290.2	+0.035	+509.7
Crop land	315672	71.4	281636	63.6	264722	59.8	171016	38.7	-144656	-0.33	-4821.9
Bare land	1304.6	0.3	3922	0.9	15287.3	3.5	11052.3	2.5	+9747.8	+0.022	+324.9
Total	442318.8	100	442318.8	100	442318.8	100	442318.8	100	442318.8		

The analysis of various LULC changes indicated a sharp decline in cropland cover from 71.4 to 38.7%, while grass land and forest showed a steady but significant rise in total area coverage from 3% to 21.5% and 8.2% to 11.6%, respectively. However, settlement increased remarkably from 0.8% to 8.9%.

2.4.3. Impacts of LU/LC change on water quality of Northern Lake Tana gulf

All the crucial water quality parameters obtained from historical data, field and laboratory measurements were correlated with the identified land cover classes using Pearson correlation (Table 2.6). A correlation coefficient greater than ± 0.68 represented strong positive or negative correlation between variables (Schober *et al.*, 2018). Highly significant association was observed between various water quality parameters and land cover changes. For instance, settlement expansion has substantial positive association with turbidity, alkalinity and nutrients such as $\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$, SRP, TP and $\text{NH}_3\text{-N} + \text{NH}_4^+\text{-N}$. This implies that, as the amount of settlement and built-up area increased, nutrient enrichment, turbidity and other water quality parameters led to gradual water quality deterioration. Conversely, substantial decline in cropland cover showed a strong negative correlation with various water quality parameters including pH (-0.928), EC (-0.956), turbidity (-0.755), TSS (-0.975), $\text{NH}_3\text{-N}$ (-0.98), SRP (-0.97) and TP (-0.961).

Table 2.6. Correlation matrix of LULC change and water quality parameters

Water Quality Parameters	Water body	Settlement	Grassland	Forest	Crop land	Bare land
Temp ($^{\circ}\text{C}$)	0.757	0.038	0.658	0.951*	-0.69	-0.498
pH	-0.387	0.942	-0.043	0.067	-0.928*	0.621
DO (mg/l)	-0.283	-0.895*	-0.415	0.672	0.376	-0.295
TDS (mg/l)	-0.525	0.265	-0.4	0.997**	0.439	0.436

EC	-0.689	0.756	0.784	-0.262	-0.956**	0.842
Turbidity	-0.519	0.831*	0.296	-0.761	-0.755	0.997**
Alkalinity (mg/l)	-0.489	0.987**	0.074	-0.115	-0.485	0.753
TSS (mg/l)	-0.492	0.584	0.565	0.623	-0.975**	0.064
Total hardness	-0.116	0.648	0.25	-0.909*	0.018	0.953*
NO ₃ ⁻ (mg/l)	-0.963	0.758	0.915	0.732	-0.841*	-0.084
NO ₂ ⁻ (mg/l)	-0.994	0.759*	0.696	0.421	-0.498	0.298
NH ₃ ⁺ (mg/l)	-0.752	0.902**	0.807	0.606	-0.98**	0.086
SRP (mg/l)	-0.942	0.888*	0.718	0.202	-0.97**	0.508
TP (mg/l)	-0.93	0.903	0.972**	0.569	-0.961**	0.537
SO ₄ ⁻ (mg/l)	0.297	0.839	0.194	0.295	-0.589	0.423
Fe (mg/l)	0.283	0.895	0.415	-0.172	-0.676	0.995**

Principal component analysis (PCA) was conducted to identify the variables of overriding importance to the observed variations. The output of PCA depicted that, the first two components (PC1 and PC2) together explained about 81.2% of the total variations with PC1 contributing 53.4% and PC2 contributing 27.8% to the total variation. The loading plot (Fig. 2.5) indicated that water quality parameters such as turbidity, hardness and Fe were impacted by bare land; while most parameters including EC, alkalinity, SO₄⁻, TP, SRP and pH were highly impacted by settlement.

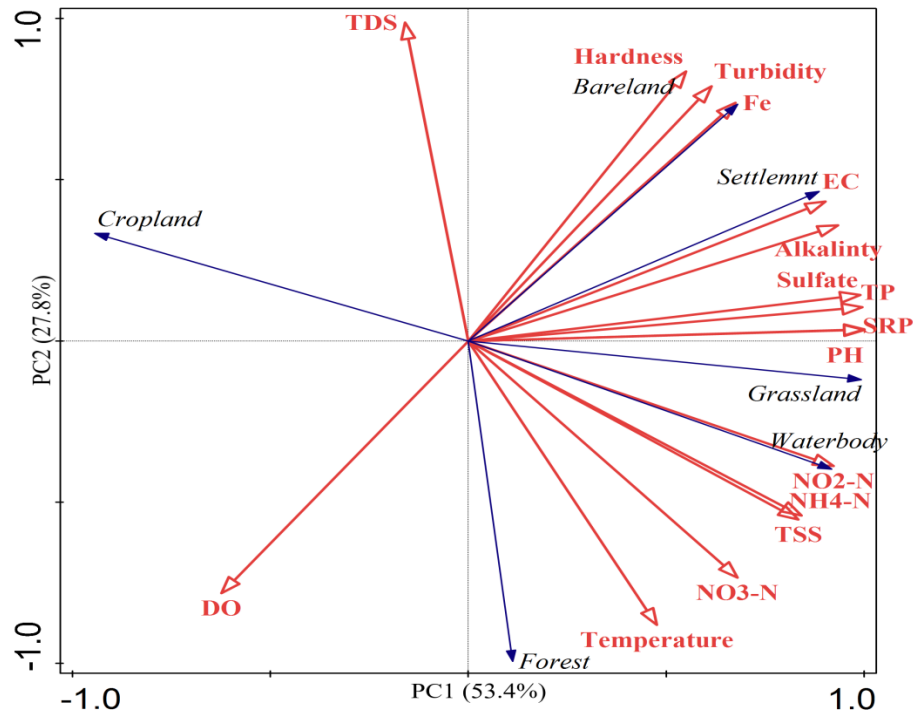


Fig. 2.5. Principal component analysis biplot indicating the relationship between water quality parameters and LULC classes

2.5. Discussion

2.5.1. Water quality parameters

Change in water quality parameters such as temperature, DO, EC, pH and nutrients influence the ecological, biogeochemical and integrity of aquatic systems (Alemu *et al.*, 2017; Engdaw *et al.*, 2022). Hence, health of aquatic systems is dependent on physico-chemical parameters. Consequently, the recent LU/LC modification particularly settlement expansion and increased socio-economic developments impacted water quality (Matlala, 2023).

In the present study, mean surface water temperature varied from 20.9 to 24.5°C with increasing trend from 2002 to 2022. In shallow lakes like Lake Tana, surface water temperature corresponds to air temperature (Shah *et al.*, 2019). Thus, values reported during the study period were within the normal environmental temperature range of 21-27°C in the area and below the permissible

upper limit of WHO (30°C; Abebe and Kebede, 2017; WHO, 2017; Gobezie *et al.*, 2023). Similarly, the recorded maximum temperature fell within the normal range of 25-35°C for natural inland water bodies in the tropics (Egun and Ogiesoba-Eguakun, 2018). The observed spatio-temporal variations could be associated with the time of sampling, drought-induced climate change and canopy cover of vegetation (Abebe *et al.*, 2017; Engdaw *et al.*, 2022).

The pH level measured in the northern gulf of Lake Tana was slightly alkaline, but is still within the tolerable limits set in various guidelines for drinking (6.5-9.2), irrigation and livestock watering (6.5-9.0) purposes (DWAF, 1996; USEPA, 2011; WHO, 2017). Higher pH values of the tributaries and the lake indicate the dissociation of HCO_3^- and CO_3^{2-} and production of OH^- at equilibrium (Toufeek and Korium, 2009; Gobezie *et al.*, 2023). Furthermore, pH in aquatic systems is highly associated with corrosive nature of the water and is highly impacted by various biological processes such as photosynthesis and respiration (Shah *et al.*, 2019; Rahman *et al.*, 2021). Higher pH is ultimately caused by increased photosynthetic removal of carbon dioxide. The current results are broadly similar to those previously reported by Goshu and Aynalem (2017; 8.2) and Engdaw *et al.* (2022; 8.7); for the littoral area of Lake Tana.

DO in the present study significantly varied between 5.3 and 8.03 mg/l among the 10 year intervals of the study period. The gradual decline in DO particularly from that of the river mouth of Megech to that of Dirma could be attributed to the severe anthropogenic activity in the catchment, which probably resulted in the introduction of oxygen-demanding domestic and industrial wastes and surface runoffs from agriculture lands that promoted microbial decomposition (Alemu *et al.*, 2017; Engdaw *et al.*, 2022; Gobezie *et al.*, 2023). Lower anthropogenic pollution from point and diffuse sources may have contributed to the higher concentrations of DO recorded in the early 2000s (Akale *et al.*, 2018). Moreover, the recent

enormous proliferation of the macrophyte water hyacinth, which forms a dense mat in the littoral regions of Gorgora gulf blocking light penetration, may have caused decreased submerged growth and cessation of underwater photosynthesis by phytoplankton leading to oxygen-depleted condition (Dersseh *et al.*, 2019; Sewunet *et al.*, 2022). The DO measured in the present study is slightly lower than the earlier records of 6.6 to 8.2 mg/l (Akoma and Imoobe, 2009; Wondim, 2016; Kassa *et al.*, 2020). It is also below WHO's acceptable lower limit of 6.5-8.5 for aquatic life survival (WHO, 2017) at Dirma and Megech sites.

Elevated level of EC recorded in this study is ascribed to the raised levels of dissolved solids and metallic ions originating from improper domestic and industrial waste discharge (Akale *et al.*, 2018). Similarly, activities associated with agriculture, excavation and sand mining practices that dissolve the uppermost clay particles in the catchment are linked to greater EC (Gobezie *et al.*, 2023). The lower EC level recorded in the littoral region of Lake Tana is possibly due to absorption and filtration capacity of water hyacinth and other aquatic macrophytes (Ratan and Verma, 2014; Dersseh *et al.*, 2019; Kassa *et al.*, 2020). EC values recorded in the present study fall below the limit of 2500 $\mu\text{S cm}^{-1}$ for freshwater (WHO, 2017). The higher turbidity values of the present study may possibly be due to runoff from agricultural lands with poor farming practice and meager soil conservation measures that promote uppermost soil erosion (Akinbile and Omoniyi, 2018; Engdaw *et al.*, 2022; Gobezie *et al.*, 2023). Further, dissolved solids, suspended organic matter, algal bloom, dense assemblage of microbial communities and minerals contribute to higher turbidity, resulting in declined light penetration through the water column (Liu *et al.*, 2020; Mekuria *et al.*, 2021; Gule *et al.*, 2023). Thus, higher turbidity values recorded in this study may seriously impact effectiveness of disinfection through enhanced pathogenic bacteria growth and could impose health risk (Daghara *et al.*, 2019). The lowest

turbidity value measured at the littoral region of Lake Tana gulf at Gorgora seems to be associated with dilution by the huge volume of lake water and filtration by riparian macrophytes. Turbidity measured in this study is consistent with the reports of Wondim (2016) and Kassa *et al.* (2020). However, it does not conform to WHO (2017) standard of 5 NTU for drinking water.

Higher values of algal growth limiting nutrients recorded in this study are associated with the severe anthropogenic activities including sewage and industrial effluents from the nearby highly populated and urbanized cities of Gondar, Azezo, Koladiba and Chuahit; which have poor wastewater drainage systems (Alemu *et al.*, 2017; Engdaw *et al.*, 2022). The strong associations of domestic wastewater and industrial effluents with nutrient enrichment were also reported by several studies (Marti *et al.*, 2006; Akale *et al.*, 2018; Kassa *et al.*, 2020). In Ethiopia, more than 80% of wastewater generated from household and industries is directly discharged into nearby aquatic systems without treatment (Mekonen, 2012). Correspondingly, Megech and Dirma Rivers and their tributaries are also used as waste dumping sites.

Moreover, point and non-point sources of pollutants originating from animal fattening and husbandry, phosphate containing detergents used for washing and bathing, runoff from intensive agriculture in the Dembia plain substantially may have contributed to nutrient enrichment (Li *et al.*, 2014; Alemu *et al.*, 2017; Fetahi, 2019). For instance, in the northern catchment of Lake Tana, farmers apply excessive amounts of fertilizers and pesticides on farm lands to improve yield and alleviate food insecurity. Incidentally, annual fertilizer consumption of the country grew alarmingly from 3500 ton in 1970 to 1.92 million tons in 2020 (Akale *et al.*, 2018; Fetahi, 2019; CSA, 2021; Asegid *et al.*, 2022). However, the poor soil conservation measures coupled with traditional farming technique resulted in the fertilizer to be washed away eventually entering aquatic systems through runoffs.

The lower concentrations of nutrients measured at Gorgora gulf and littoral region of Lake Tana might be due to uptake by algal communities, bacteria and by the highly proliferating macrophyte water hyacinth; or adsorbed onto the sediment (Gerhardt *et al.*, 2010). The current mean values are higher than those reported previously by Wondim (2016), Alemu *et al.* (2017) and Derseh *et al.* (2019). The increasing trend in concentration can probably be ascribed to nutrient enrichment by domestic, industrial and agricultural activities. In general, freshwater systems respond to excess nutrient enrichment in various ways. According to Tibebe *et al.* (2019), TP above 0.01 mg/l is enough to cause eutrophication. Hence, nutrient enrichment had an inevitable contribution to the recent infestation of water hyacinth and eutrophication of Lake Tana.

2.5.2. Land use land cover map change

In the present study, the overall accuracy, which varied between 92.2 and 95.7% and Kappa coefficient of 0.86-0.91%, revealed highest level of reliability. A very good Kappa coefficient value should be between 0.70 and 0.85% (Franklin and Wulder, 2002; Garedew *et al.*, 2009). Hence, the classified images in this study exhibited a mean Kappa coefficient of 0.88%, indicating a strong association between ground truth data and categorized images (Fitawok *et al.*, 2020; Muche *et al.*, 2023). The finding of this study is in line with the above standard and results of other studies (Gashaye *et al.*, 2023; Gule *et al.*, 2023). Over all accuracy of LU/LC study could be affected by post-classification correction and difference in the quality of images.

During the last 30 years, significant LU/LC modification was observed in the study area as a result of settlement and forest expansion. At the beginning of this study, the landscape was dominated by crop land and gradually declined from 71.4% to 38.7%. The observed conversion of crop land in to other land cover types is a common phenomenon in central and northern

Ethiopian highlands (Amha *et al.*, 2013). The primary driver of these changes was the rise in local population which necessitated the availability of more land for settlement and grazing (Table 2.5; Ariti *et al.*, 2015; Minta *et al.*, 2018). During the last three decades, fourfold population increase was recorded in the region escalating expansion of formal and informal built-up at the expense of cropland (Fitawok *et al.*, 2020). Similarly, substantial conversion of crop land to grass land to improve soil fertility was also reported in the northwest of Lake Tana (Gashaye *et al.*, 2023).

In the second 10-year period of the study (2002-2013), considerable expansion of settlement, bare land and grass land at the expense of forest land was recorded. The observed expansion of settlement was associated with rapid population growth and government-favored urbanization (Garedew *et al.*, 2009; Gashaye *et al.*, 2023), while the expansion in grass land cover was attributed to animal husbandry, grazing, rehabilitation, soil fertility and biodiversity conservation measures (Minta *et al.*, 2018; Beyene and Minale, 2023). Similar findings were also reported from Somado catchment and the northwestern watershed of Lake Tana (Fasika *et al.*, 2019; Gashaye *et al.*, 2023). Apparently, the sharp decline in forest cover from 11.1% in 2002 to 4.1 % in 2013 at a rate of -0.63% could be associated with human-induced destruction of woodlands and shrub lands for various purposes including charcoal production, construction, timber and animal grazing (Minta *et al.*, 2018; Obubu *et al.*, 2022). Similar findings of declining forest land cover during the same period have been reported from various parts of the country (Kindu *et al.*, 2013; Minta *et al.*, 2018; Muche *et al.*, 2023). These studies elucidated the fact that alarming population growth and subsequent urbanization have led to the expansion of settlement at the expense of forest causing resource overexploitation. Expansion of bare land was however, associated with drought, increase in built-up and urbanization. Moreover, throughout the study

period, the irregularities observed in the trend of dynamic settlement could be due to the reduced backyard used for other purposes such as grazing and crop production. Rapid population growth-induced urbanization and settlement were also reported for many parts of the country between 2002 and 2013 (Minta *et al.*, 2018; Gule *et al.*, 2023).

From 1993 to 2022, large-scale crop land conversion to pasture, forest or grassland was recorded, and this is related to the medium-long term fallow practice applied to replenish soil fertility. Another crucial element in the change in LU/LC is rapid conversion of crop land into forest land (Table 2.5; Fig. 2.6). The increased interest of smallholder farmers in withdrawing massive farmland area and allocating it for plantation could be linked to the expansion of eucalyptus plantation in response to rapid population growth that caused lack of wood for fuel and housing (Minta *et al.*, 2018). Furthermore, eucalyptus plantation became a viable economic alternative due to dwindling agronomic productivity brought on by inadequate soil-water conservation measures and low production, even in the face of massive agro-chemical application to farmlands.

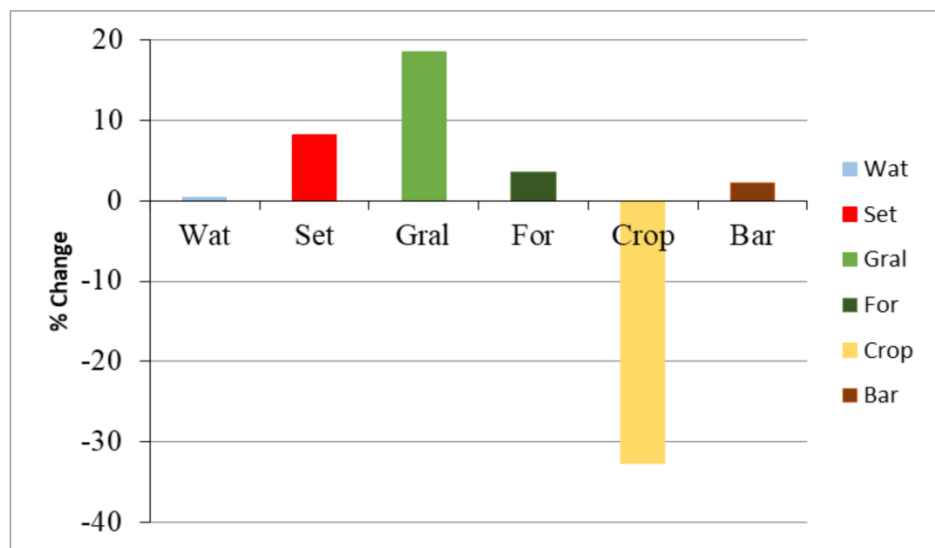


Fig. 2.6. Percent of change in LU/LC of northern Lake Tana catchment from 1993-2022

The recent significant expansion of forest particularly during the period of 2013-2022 could be associated with the recent nationwide green legacy campaign to combat environmental degradation, deforestation, soil erosion and climate variability (Beyene and Shumetie, 2023; Lambert and Deyganto, 2023). This finding agrees well with the reports of Hamad (2022), and Razzaq *et al.* (2023), that evidenced the positive impact of green legacy in mitigating climate change, averting land cover change through ecosystem restoration, and fostering environmental stewardship across the world. Subsequently, the role of green legacy initiative in promoting healthy environment in Ethiopia is significant (Fentaw *et al.*, 2022).

2.5.3. Impact of LU/LC change on water quality

LU/LC change has a significant impact on water quality. The results of the present study also elucidated the strong association of water quality deterioration and increased urbanization, and built-up area and decreased agricultural land in the northern watershed of Lake Tana. This is attributable to the fact that expansion of settlement is a response to rapid population growth, urbanization and industrialization that led to increased quantity and reduced quality of domestic and industrial effluents (Assegid *et al.*, 2022; Gule *et al.*, 2023). The present results are well in line with several studies that reported exacerbation of water pollution due to increased domestic and industrial discharge quantity (Fetahi, 2019; Akale *et al.*, 2018; Angassa *et al.*, 2020). Furthermore, Tibebe *et al.* (2019) and Engdaw *et al.* (2022) pointed out that free-style waste dumping activities i.e. about 80% of waste is dumped and discharged into nearby aquatic systems without treatment.

Increase in the expansion of forest and grassland was positively associated with water quality improvement. This could be ascribed to nutrient absorption and uptake by macrophytes, grasslands and forest vegetation (Gerhardt *et al.*, 2010; Kassa *et al.*, 2020). Conversely, decline

in cropland showed a strong negative correlation with various water quality parameters. Hence, decrease in cropland expansion considerably increases point and nonpoint sources of pollutants through intensified use of agrochemicals, which hampered water quality. Numerous studies have disclosed that excessive applications of fertilizers accelerate water quality deterioration. For example in the northern catchment of Lake Tana, farmers apply 100 kg of fertilizer (Urea or DAP) per hectare of crop land, which is above the recommended amount of 30-40 kg/ha (Akale *et al.*, 2018; Fetahi, 2019; Getachew *et al.*, 2021). Conversely, expansion of grassland and forest showed a negative correlation with temperature, EC, TDS turbidity, hardness, $\text{NH}_3\text{-N} + \text{NH}_4\text{-N}$, and others. This finding indicates the considerable role of vegetation cover in regulating local microclimate, improving water quality and mitigating pollution (Derseh *et al.*, 2019; Sewunet *et al.*, 2022).

2.6. Conclusions and recommendation

The results of the present study obviously demonstrated the strong linkage between LU/LC changes and water quality. Over the study period, water quality in the northern gulf of Lake Tana was exacerbated due to rapid population growth and urbanization-driven settlement. Point and diffuse sources of pollutants emanating from cropland, settlement and other allochthonous sources have increased nutrient concentrations in the water. However, the recent green legacy campaign has significantly changed the land cover of the catchment, with increased forest and grassland cover at the expense of cropland. Integrated LU/LC change management, shoreline protection, modernized agronomic activity and strengthening the forestation program will improve the water quality, ecology and climate of the catchment.

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CHAPTER 3

Increasing anthropogenic stressors influenced the water quality and shifted trophic status of northern Lake Tana Gulf, Ethiopia

This chapter is based on the following published paper:

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Abstract

Nutrients in an aquatic system determine productivity, integrity and ecological status of the aquatic system. However, the excessive enrichment of these nutrients emanating from severe anthropogenic activity has substantially impacted water quality and biodiversity. There is diminutive information available on the water quality and trophic status of the northern Gulf of Lake Tana, Ethiopia due to accessibility difficulties. Therefore, this study aimed to assess the spatio-temporal changes in water quality and trophic status of Lake Tana, specifically the northern Gulf. Sampling sites were selected based on their proximity to human interference and samples were collected from five sampling points between May 2023 to April 2024. The physico-chemical parameters were measured *in situ* or determined in water samples using standard methods. There were significant ($p < 0.05$) variations in temperature, dissolved oxygen (DO), electrical conductivity (EC) and turbidity between the dry and rainy seasons. The mean concentrations of nutrients such as $\text{NO}_3\text{-N}$ (0.48 mg/l), soluble reactive phosphorus (SRP) (1.2 mg/l) total nitrogen (TN) (1.27 mg/l), and total phosphorus (TP) (2.1 mg/l), showed significant spatio-temporal variations, with higher concentrations recorded at the Megech and Dablo sites

during the rainy seasons. The depth-wise distributions of the parameters exhibited a consistent pattern, with a slight increase in the levels of EC, turbidity, TP, and TN, and a decrease in temperature, DO and NO₃-N. The concentrations of SRP, TP, and TN, were slightly higher than the USEPA's threshold concentrations for causing eutrophication. The shift in the trophic status of Lake Tana from oligo-mesotrophic to eutrophic is likely due to nutrient enrichment associated with anthropogenic stressors. Therefore, continuous monitoring would help in identifying as early as possible to take mitigation measures to avert future ecological imbalances in the lake.

Key words: anthropogenic stressor, eutrophication, nutrient enrichment, pollution, trophic status, water quality

3.1. Introduction

Water quality is crucial for human health, ecosystem integrity, and sustainable exploitation of aquatic resources (Akale *et al.*, 2018; Engdaw *et al.*, 2022; Matlala, 2023). However, rapid contamination of aquatic systems with various pollutants has become a major global public health and ecological concern (Akele *et al.*, 2016). Influx of pollutants from natural geogenic and anthropogenic sources, atmospheric deposition, leaching of rocks, industrial effluents (Rabalais, 2002; Carpenter, 2005; Pizarroa *et al.*, 2010; Inamdar *et al.*, 2015), and high volume of municipal sewage triggered by rapid population growth and agricultural malpractices (Gebremedhin *et al.*, 2018; Sewunet *et al.*, 2022), have all contributed to reduced water quality and the consequent adverse environmental impacts (Goshu and Aynalem, 2017; Ebner *et al.*, 2022).

In many developing sub-Saharan-African countries like Ethiopia, freshwater is under serious threat and is the most poorly managed resource (Matlala, 2023; Assegid *et al.*, 2022). Rapid population growth-induced settlement expansion, urbanization, and industrial development have

led to a significant increase in the amount of wastewater discharged into aquatic systems (Alemu *et al.*, 2017; Lanckriet *et al.*, 2017). Consequently, about 3.2 million inhabitants lack access to safe drinking water and proper sanitation, highlighting the need for better resource management (Emama *et al.*, 2010; CSA, 2017; Gebremedhin *et al.*, 2018; Eludoyin and Olanrewaju, 2021; Ebner *et al.*, 2022).

Ethiopia is endowed with over 7000 km² of inland freshwater bodies, with a mean annual discharge of 122 billion cubic meters (FAO, 2015; Fetahi, 2019; Engdaw *et al.*, 2022). These water bodies are home to numerous ecological and economic goods and services (Gebremedhin *et al.*, 2018; Kassa *et al.*, 2021; Dersseh *et al.*, 2022; Sewunet *et al.*, 2022; Mohammed *et al.*, 2024). However, the country is recognized as having the lowest access to basic water and hygienic services due to pollution emanating from reckless anthropogenic activities and poor watershed management (Gule *et al.*, 2023). Recent studies elucidated that, only 6.3% of Ethiopian households have adequate sanitation facility, and 66% have safe drinking water (FAO, 2015). A significant portion of the population (62.7%) relies on unimproved water sources (Berhanu and Hailu, 2015), with untreated water from lakes, rivers, wells and springs being commonly used for drinking and domestic purposes (Alemu *et al.*, 2015). The main threats to water bodies in Ethiopia include vulnerability to multiple stressors, such as frequent disturbance from anthropogenic activities, hydrological variations and the introduction of invasive species (Dersseh *et al.*, 2022; Vári *et al.*, 2022; Ma *et al.*, 2024), leading to a decline in ecological integrity, biodiversity impoverishment, and water quality deterioration (Tibebe *et al.*, 2019; Engdaw *et al.*, 2022; Matlala, 2023).

Lake Tana, is the largest lake in Ethiopia, accounting for 50% of the freshwater lake area of the country (Vijverberg *et al.*, 2009; Tibebe *et al.*, 2019; Sewunet *et al.*, 2022; Goshu *et al.*, 2023;

Engdaw *et al.*, 2024). It is the main source of the Blue Nile River, which sustains the entire Sudan and Egypt (Fenta *et al.*, 2018). The lake is among Africa's top 32, with a global rank of 250 for conservation priority, and was designated as a UNESCO-biosphere reserve in 2015 (Teshome *et al.*, 2021; Yasuda *et al.*, 2022). Lake Tana is a vital component of the environment and biodiversity heritage, and livelihood is highly dependent on its numerous ecosystem services (Vijverberg *et al.*, 2009; Goshu and Aynalem, 2017; Dersseh *et al.*, 2022). However, the lake ecosystem has been constantly changing in the last two decades due to various stressors, including anthropogenic activities, hydrological variations, climatic changes and natural factors (Wondie *et al.*, 2007; Goshu and Aynalem, 2017; Kebedew *et al.*, 2020). As a result, water pollution has been exacerbated alarmingly, disrupting the natural ecological balance of the lake. Rapid population-growth, urbanization, land use/land cover changes (LU/LC), deforestation, poor soil-water conservation practices and human settlement have contributed to this problem (Hassen and Assen, 2017; Gitaw *et al.*, 2020; Dersseh *et al.*, 2022; Engdaw *et al.*, 2024).

Additionally, shoreline modification through hotel and resort development, continuum disruption of major tributaries (Megech, Rib and Koga) due to dam construction and proliferation of invasive macrophyte water hyacinth, combined with poor solid waste management and water level fluctuations (Alemu *et al.*, 2017; Fenta *et al.*, 2018), are currently threatening the water quality and hence biodiversity of the lake through their profound effect of excessive nutrient enrichment and trophic state alteration (Tadesse *et al.*, 2018; Fetahi, 2019; Kassa *et al.*, 2021; Goshu *et al.*, 2023). Intensification of agricultural practices in the Dembia plane of the northern catchment, with increased agrochemical applications and serious shoreline encroachment, has resulted in the external loading of nutrients and subsequent eutrophication (Emama *et al.*, 2010; Alemu *et al.*, 2017; Goshu and Aynalem, 2017; Fetahi, 2019; Assegid *et al.*, 2022). Elevated

influx of algal growth limiting nutrients such as nitrogen (N), phosphorus (P), silica (Si) and their derivatives from various anthropogenic sources and internal cycling can result in eutrophication, increased primary productivity, anoxic hypolimnion, and shift in the trophic state of the aquatic system (Carpenter, 2005; Fetahi, 2019; Matlala, 2023).

Since many years ago, Lake Tana has received considerable attention from researchers, with studies focusing on various aspects of the lake (Nagelkerke, 1997; de Graaf, 2004; Wondie *et al.*, 2007; Vijverberg *et al.*, 2009; Emama *et al.*, 2010; Dejen *et al.*, 2017; Goshu and Aynalem, 2017; Fenta *et al.*, 2018; Gebremedhin *et al.*, 2018; Gitaw *et al.*, 2020; Kebedew *et al.*, 2020; Dersseh *et al.*, 2022; Yasuda *et al.*, 2022; Goshu *et al.*, 2023). However, these studies have mainly emphasized the taxonomy and ecology of fishes, with limited attention given to limnological studies in the southern gulf of the lake. Other gulfs, particularly northern gulfs of the lake such as Gorgora, Delgi, Kunzila and Lemba were neglected due to their remote location, limited access and logistic constraints. No study examined the spatio-temporal variability in nutrient concentrations and trophic status of Lake Tana's northern gulf, which the recent developmental activities (urbanization, industrialization, effluent discharge, agricultural intensification) and alarming land use/land cover change have impacted. Therefore, this study aimed to assess spatio-temporal changes in the physico-chemical parameters and trophic state of the northern gulf and identify the major factors contributing to its adverse effects on biodiversity and ecosystem services.

3.1. Materials and methods

3.1.1. Sampling site selection

The field investigation study collected samples from five different sampling sites located in the littoral and open water zones of the lake (Fig. 3.1). The sites in the littoral region were covered by various emergent and floating macrophyte species, such as water hyacinth (*Eichhornia crassipes*), *Phragmites* and *Typha* species. Thus, sampling was conducted at about 50 to 100 m away from the vegetation towards the pelagic region. Sampling sites were chosen based on their proximity to human interference and macrophyte invasion. The first site was the open water (OP), which was free from anthropogenic influence and considered as a reference site. The second site was the Megech (MEG) river mouth, the fourth major tributary of the lake and has a mean annual discharge of $196 \times 10^6 \text{ m}^3 \text{ y}^{-1}$ (Abebe and Kebede, 2017; Alemu *et al.*, 2017). This river carries domestic, industrial, and agricultural effluents from Gondar and other riparian towns. Sampling was conducted at the confluence point. The third site was Dablo (DAB), closer to the intensive agriculture of Dembia woreda where farmers apply excess amount of fertilizers to improve yield. At this site, two seasonal rivers (Kersit and Nedit) passing through the agricultural areas drain into the lake during the rainy season. The fourth site was the Gorgora (GOR) gulf near Gorgora town. This site is home to churches, a large resort on the lake's shore, and various streams that carry domestic waste from the town draining directly into the lake. The final site was a water hyacinth-infested (WHI) site, where the macrophyte forms a dense mat on the lake's surface.

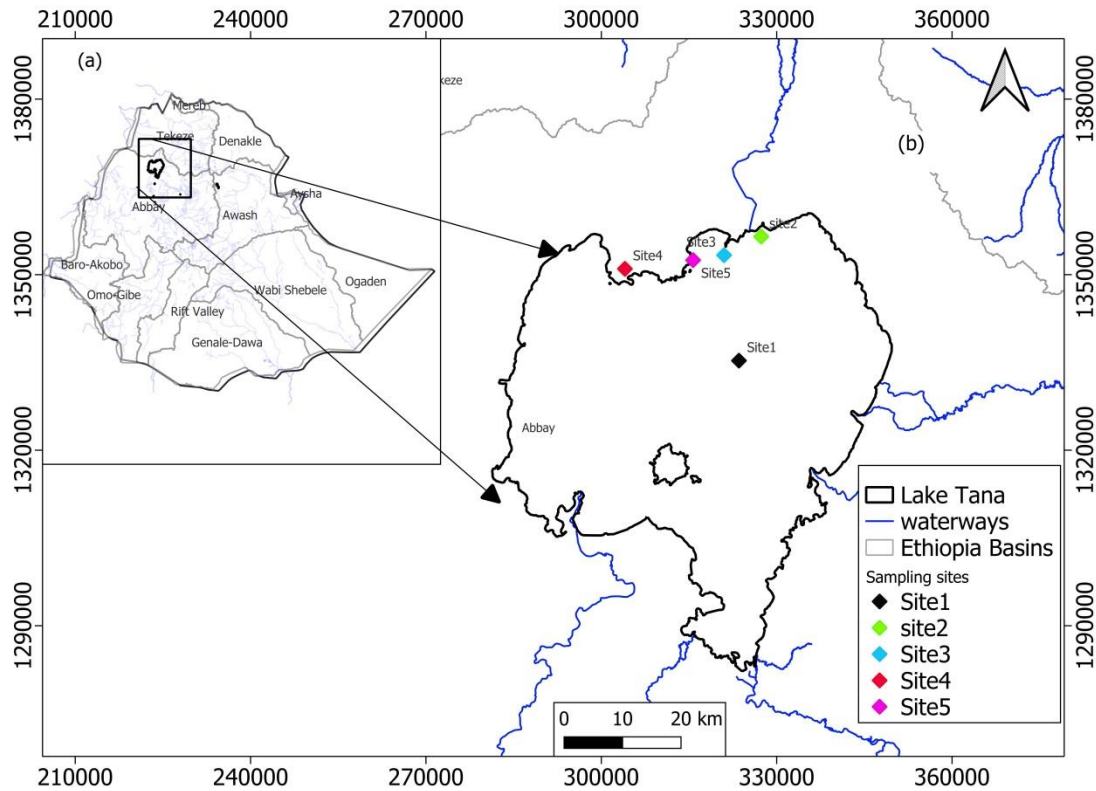


Fig. 3.1. Map of the Lake Tana indicating sampling sites (Site1 = Open water (OP), Site2 = Megech river mouth (MEG), Site3 = Dablo (DAB), Site4 = Gorgora (GOR), Site5 = Water hyacinth infested site (WHI))

3.1.2. In situ measurements

In-situ measurements were made using a multi-parameter probe (HANNA model HI991301) to measure physico-chemical water quality parameters such as dissolved oxygen (DO, mg/l), temperature (Temp, °C), and electrical conductivity (EC, K_{25} , $\mu\text{S}/\text{cm}$). pH levels were measured using a pH meter (HANNA, model HI98186), while turbidity (Tur, NTU) was measured using a turbidity meter (Wag-WT3020) prior to water samples collection. Water transparency, Secchi depth (Z_{SD} , cm) was estimated by lowering a 20 cm diameter circular Secchi disc and used to estimate euphotic depth (Z_{eu} , cm) (Borowiak and Borowiak, 2016). The water depth at each site

was determined using a handheld echo-sounder (Plastimo, Echotest II). Similarly, depth profile measurements were conducted every 1 m from surface to 7 m in the OP site.

3.1.3. Water sampling and laboratory analysis

Sampling was conducted seasonally from May 2023 to April 2024, including pre-rainy season (April-June), main-rainy season (July-September), post-rainy season (October-December), and dry season (January-March).

Water samples were collected from the open water's surface and various depths (up to 7 m) using 2-L capacity van Dorn bottle sampler (Thermo Fisher Scientific Inc., Waltham, MA, USA, horizontal model). All the collected water samples were mixed in equal proportions in a single container to produce one composite sample. One liter of the composite samples was kept in a cool box and transported to Seyoum Mengitou Limnology laboratory of Addis Ababa University. The samples were filtered using 0.47µm pore size glass fiber filter paper (GF/F) for the analyses of soluble reactive phosphate-phosphorus (SRP), nitrate ($\text{NO}_3\text{-N}$), nitrite ($\text{NO}_2\text{-N}$), ammonia + ammonium nitrogen ($\text{NH}_3 + \text{NH}_4^+$, hereafter ammonia) and silica (SiO_2), whereas unfiltered water samples were used for the analysis of total phosphorus (TP) and total nitrogen (TN). The samples were processed and analyzed according to APHA (1999).

In the laboratory, Total suspended solids (TSS) were estimated gravimetrically by filtering 100 ml of water through a pre-weighed 0.47 µm GF/F filter paper pre-dried at 105 °C. The filter paper with residue was then dried for approximately 4 hours at 105 °C, and TSS was calculated using the formula given in APHA (1999). Similarly, total dissolved solids (TDS) were determined by drying the filtrate of 100 ml water in an oven using a pre-dried Erlenmeyer flask. The concentration was then calculated according to APHA (1999). Total alkalinity was

determined by titration of unfiltered water samples using 0.01N HCl as a titrant and bromocresol green-methyl red mixed indicator (APHA, 1999).

3.1.4. Nutrient analysis

The concentrations of TP after persulfate digestion and SRP were determined using the ascorbic acid method as outlined in APHA (1999). Nitrate-nitrogen levels were determined using the sodium-salicylate method, while nitrite-nitrogen was measured colorimetrically as reddish-purple azo dye by coupling diazotized sulfanilamide with N-Naphthyl-(1)-ethylendiamine-dihydrochloride. Ammonia-nitrogen was quantified using the phenate method and total nitrogen was determined using the persulfate method. Soluble reactive silica (SiO₂) was determined by the ammonium molybdosilicate method.

3.1.5. Chlorophyll *a* concentration

Chlorophyll-*a* (*Chl-a*) concentration was determined by filtering the appropriate volume (150-200 ml) of lake water sample through 0.47µm GF/F filter paper. The filter and residue were folded in half with the seston inside, then covered with aluminium foil and stored at -20 °C in a freezer overnight. The residue and the filter were then homogenised using tissue grinder at approximately 5000 rpm for about 1 minute with 5 ml of 90% aqueous acetone. The samples were transferred into centrifuge tubes by rinsing the grinder with 90% acetone. The extract was kept in the dark for 24 hrs at 4°C. Then, the extract was centrifuged for 10 min at 3000 rpm and decanted into a clean test tube. Light absorbance of the *Chl-a* extract was measured with a spectrophotometer at 750 nm and 665 nm (Talling and Driver, 1963).

$$Chl - a (\mu g l^{-1}) = \frac{(11.9(E_{665} - E_{750}) \times V1)}{(V2 \times L)} \quad (1)$$

Where: 11.9 = is the absorption coefficient for *Chl-a*, V1 = volume of extract in ml, V2 = volume of the filtered water sample in liter, L = light path length of cuvette in cm, E665, E750 = optical density or absorbance of the sample at 665 and 750 nm.

3.1.6. Trophic state index

Trophic state index (TSI) of Lake Tana was determined using the equation developed for tropical and subtropical aquatic systems (Lamparelli, 2004; Cunha *et al.*, 2013). The TSI developed by Lamparelli (2004), was based on total lake phosphorus concentration (TP), chlorophyll-*a* (*Chl-a*) levels within the lake, and Secchi disk depth (SDD). The following formula was adopted to determine Lake Tana's TSI.

Total phosphorus (TP) TSI (**TSI_{TP}**)

$$TSI_{TP} = 10 \left[6 - \left(\frac{1.77 - 0.42 \ln TP}{\ln 2} \right) \right] \quad (2)$$

Chlorophyll-*a* TSI (**TSI_{chl}**)

$$TSI_{chl} = 10 \left[6 - \left(\frac{0.92 - 0.34 \ln Chla}{\ln 2} \right) \right] \quad (3)$$

Secchi Disk Depth TSI (**TSI_{SD}**)

$$TSI_{SDD} = 10 \left[6 - \left(\frac{0.64 + \ln SD}{\ln 2} \right) \right] \quad (4)$$

Overall Trophic State Index (**TSI_c**) was calculated as

$$TSIC = \frac{TSITP + TSIChl + TSISDD}{3} \quad (5)$$

3.1.7. Data analysis

Values recorded for all parameters were analyzed using R-software and summarized using descriptive statistics. After the data was checked for normality using Shapiro-Wilk test, non-parametric (Kruskal-Wallis) ANOVA was computed to determine spatio-temporal significant difference ($p < 0.05$) among sites and between seasons. Furthermore, Pearson correlation and

principal component analysis (PCA) were employed using CANOCO-5 to assess the association between the measured water quality parameters and to determine the parameters of overriding importance to the observed variations, respectively.

3.2. Results

3.2.1. Temporal variations in water quality parameters

The overall spatial and temporal variations in physico-chemical water quality parameters of the northern gulf of Lake Tana are summarized in Tables 3.1 and 3.2. Most parameters showed clear temporal variations between the dry and rainy seasons. The highest mean surface water temperature (24.7 ± 1.06 °C) was recorded in the dry season, while the lowest (21.4 ± 0.28 °C) was documented for the rainy season. The mean pH value of the lake was slightly alkaline and ranged from 8.6 to 9.7, with the highest (9.5 ± 0.23) and the lowest (8.9 ± 0.2) values corresponding to post rainy season and pre-rainy seasons, respectively. There was a slight seasonal decline in the mean concentrations of DO, whereby the highest mean DO (7.2 ± 0.65 mg/l) was recorded for the post-rainy season, while the lowest (6.4 ± 0.37 mg/l) was noted for the main-rainy season. The Kruskal-Wallis non-parametric ANOVA indicated that EC recorded during the dry and post-rainy seasons (145.8 ± 10.6 and 148.9 ± 13.4 μ S/cm, respectively) significantly differed ($p < 0.05$) from EC measured during the pre-rainy and rainy seasons (185.4 ± 20.5 and 191.8 ± 40.4 μ S/cm), respectively. Turbidity measured in all seasons showed a wide range of values (7.6 to 391.7 NTU). There was considerable seasonal variation ($p < 0.05$) in mean turbidity, with the highest (124.2 ± 98.2 NTU) and the lowest (10.1 ± 1.74 NTU) values corresponding to the rainy and dry seasons, respectively. There was also significant positive association between EC and turbidity with precipitation ($p < 0.001$). Mean concentrations of TDS did not showed significant variations ($p > 0.05$) among all seasons, while TSS measured during the main-rainy season differed significantly from those of other seasons. The lake's Secchi depth

(Z_{SD}) differed significantly among seasons, whereby the shallowest depth was determined during the rainy season and the deepest during the dry season (Table 3.1).

Table 3.1. Temporal variations in physico-chemical water quality parameters of northern gulf of Lake Tana (n = 12)

Parameter	Pre-rainy	Main-rainy	Post-rainy	Dry	Range
Temp (°C)	21.4 ± 0.76*	21.4 ± 0.28	22.1 ± 1.2	24.7 ± 1.06*	20.3 – 25.1
pH	8.9 ± 0.2	9.0 ± 0.42	9.5 ± 0.23	9.1 ± 0.49	8.0 - 10.0
DO (mg L ⁻¹)	6.6 ± 0.22	6.4 ± 0.37	7.2 ± 0.65	7.0 ± 0.26	5.7 - 7.8
EC (µS cm ⁻¹)	185.4 ± 20.5*	191.8 ± 40.4*	148.9 ± 13.4*	145.8 ± 10.6*	112.1 - 283.7
Tur (NTU)	50.2 ± 60.4*	124.2 ± 98.2*	14.0 ± 12.4*	10.1 ± 1.74*	7.6 - 391.7
TDS (mg/l)	0.07 ± 0.001	0.07 ± 0.005	0.05 ± 0.0	0.06 ± 0.003	0.05 - 0.07
TSS (mg/l)	0.031 ± 0.003	0.11 ± 0.003	0.01 ± 0.004	0.001 ± 0.004	0.00 - 0.16
Secchi (cm)	85 ± 3*	73 ± 4*	172 ± 2*	206 ± 3*	53 - 117
Alk (meq/l)	90 ± 21.2	196.2 ± 38*	169.6 ± 43.3*	43.5 ± 21.9	30 - 240
NO ₂ (mg/l)	0.04 ± 0.002	0.038 ± 0.001	0.06 ± 0.002	0.083 ± 0.00	0.029 - 0.13
NO ₃ (mg/l)	0.43 ± 0.053	0.48 ± 0.06	0.36 ± 0.05	0.21 ± 0.19	0.12 - 0.93
Ammonia (mg/l)	0.72 ± 0.031*	0.5 ± 0.024	0.6 ± 0.03	0.3 ± 0.013*	0.2 - 1.6
SRP (mg/l)	0.76 ± 0.04	1.2 ± 0.14*	0.36 ± 0.05	0.15 ± 0.06*	0.13 – 1.25
SiO ₂ (mg/l)	0.35 ± 0.04	0.34 ± 0.05	1.41 ± 0.7*	0.33 ± 0.003	0.28 - 1.57
TN (mg/l)	1.26 ± 0.15	1.27 ± 0.15	1.11 ± 0.2	0.83 ± 0.14*	0.56 - 2.43
TP (mg/l)	1.5 ± 0.04	2.1 ± 0.04	1.75 ± 0.04	0.86 ± 0.042*	0.85 – 2.11

*Significantly different at $p < 0.05$

Concentrations of algal growth limiting nutrients also showed considerable temporal variations between dry and main-rainy seasons. The highest mean concentrations of nutrients such as NO₃-

N (0.48 mg l^{-1}), SRP (1.2 mg l^{-1}) TN (1.27 mg l^{-1}) and TP (2.1 mg l^{-1}) were observed during the rainy season revealing their strong significant ($p < 0.05$) association with rainfall, while lowest mean values were recorded during the dry season. However, the temporal variations between post-rainy and pre-rainy seasons were not always significant.

3.2.2. Spatial variations in water quality parameters

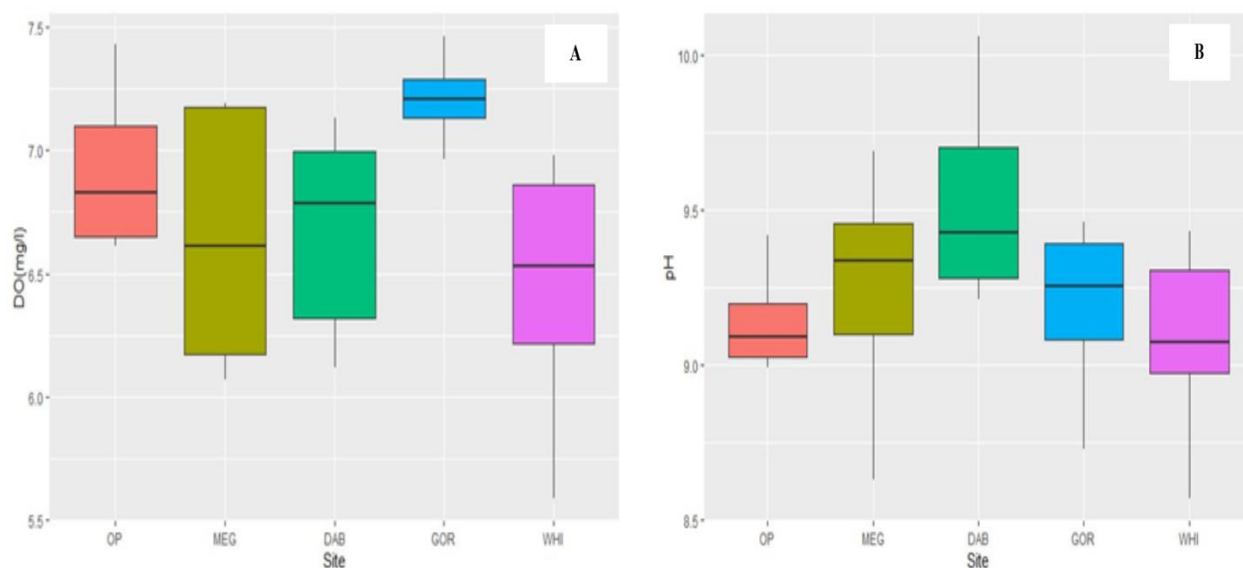
Spatial variations in physico-chemical water quality parameters of the northern gulf of Lake Tana are shown in Table 3.2 and Fig. 3.2. The lowest temperature level of 23.2°C was recorded in the WHI site. In contrast, the temperature levels of other sites did not showed significant spatial variations ($p > 0.05$). The highest mean pH level of 9.3 was recorded in the DAB site, while the lowest mean value of 9.1 was documented for OP and WHI sites.

Table 3.2. Spatial variation in physico-chemical water quality parameters of northern gulf Lake Tana (n = 12)

Parameter	OP	MEG	DAB	GOR	WHI
Temp ($^\circ\text{C}$)	24.4 ± 0.8	24.4 ± 0.6	24.3 ± 1.3	24.5 ± 1.0	23.2 ± 1.3
pH	9.1 ± 0.18	9.2 ± 0.46	9.3 ± 0.57	9.2 ± 0.28	9.1 ± 0.3
DO (mg/l)	7.1 ± 0.5	7.0 ± 0.3	6.9 ± 0.3	7.2 ± 0.3	6.6 ± 0.7
EC ($\mu\text{S/cm}$)	158.1 ± 14.9	205 ± 60.3	186 ± 48.2	157.1 ± 18.7	128.8 ± 20.1
Tur (NTU)	27.2 ± 26.8	171.0 ± 176.6	87 ± 103.8	48.4 ± 59.3	55.2 ± 62.1
TDS (mg/l)	0.06 ± 0.001	0.07 ± 0.001	0.06 ± 0.001	0.06 ± 0.001	0.06 ± 0.001
TSS (mg/l)	0.031 ± 0.006	0.08 ± 0.007	0.061 ± 0.002	0.05 ± 0.007	0.038 ± 0.003
Secchi (cm)	173 ± 32	65 ± 23	53 ± 17	82 ± 16	--
Alk (meq/l)	107.3 ± 6.5	116.3 ± 7.2	167.5 ± 8.1	165 ± 8.3	138.7 ± 5.3
NO ₂ (mg/l)	0.061 ± 0.006	0.055 ± 0.001	0.047 ± 0.001	0.049 ± 0.001	0.054 ± 0.001

NO ₃ (mg/l)	0.31 ± 0.12	0.56 ± 0.17	0.5 ± 0.07	0.35 ± 0.05	0.26 ± 0.05
Ammonia (mg/l)	0.5 ± 0.02	0.6 ± 0.04	0.48 ± 0.02	0.5 ± 0.02	0.7 ± 0.04
SRP (mg/l)	0.47 ± 0.06	1.14 ± 0.06	1.2 ± 0.09	0.78 ± 0.09	0.4 ± 0.06
SiO ₂ (mg/l)	0.6 ± 0.18	0.7 ± 0.09	0.64 ± 0.25	0.6 ± 0.2	0.63 ± 0.23
TN (mg/l)	1.02 ± 0.17	1.42 ± 0.16	1.25 ± 0.16	1.05 ± 0.13	1.32 ± 0.15
TP (mg/l)	1.43 ± 0.06	2.35 ± 0.03	2.3 ± 0.05	1.78 ± 0.04	1.89 ± 0.05

Mean maximum DO concentration of 7.2 mg/l was recorded for the GOR site and the lowest value of 6.6 mg/l for WHI site. A significant difference ($p < 0.05$) was observed in mean levels of EC and turbidity among sites, whereby the highest values of EC and turbidity (205 $\mu\text{S}/\text{cm}$ and 171 NTU, respectively) were recorded for MEG site. The lowest mean EC value of 128.8 $\mu\text{S}/\text{cm}$ and turbidity value of 27.2 NTU were recorded for WHI and OP sites, respectively. The lowest mean SiO₂ (0.6 mg/l), TN (10.2 mg/l) and TP (1.43 mg/l) were recorded for OP site, while the highest mean values of 0.7, 1.42 and 2.35 mg/l were registered respectively for MEG site.



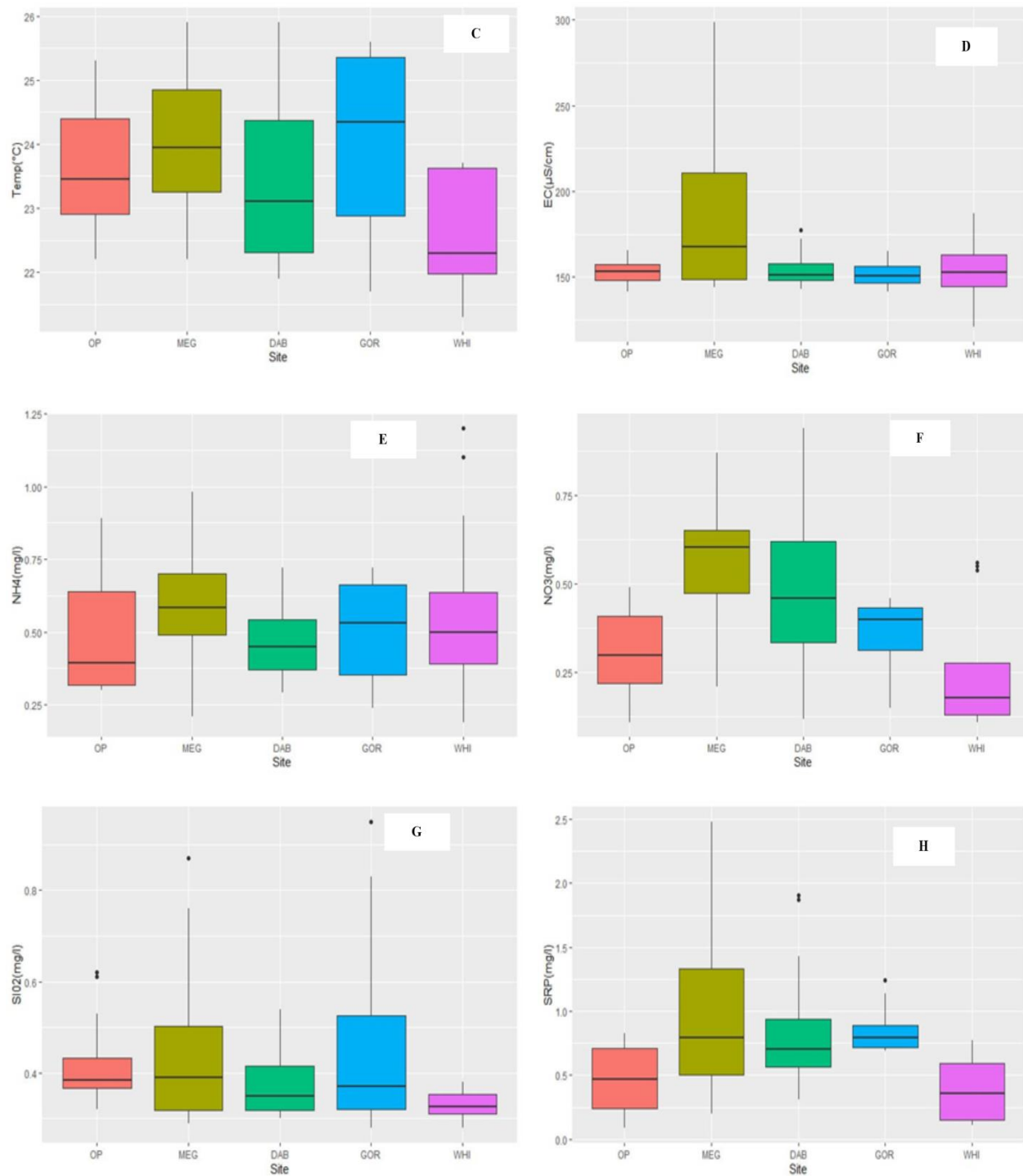


Fig. 3.2. Spatial patterns of physico-chemical parameters in Lake Tana, northern gulf (n = 12; A) DO = dissolved oxygen; B) pH; C) Temp = temperature; D) EC = electrical conductivity; E)

NH₄ = Ammonium; F) NO₃ = Nitrate; G) SiO₂ = soluble reactive silica; H) SRP = soluble reactive phosphorus; dots represent outliers).

3.2.3. Depth profiles of physico-chemical parameters

The vertical distributions of physico-chemical parameters in Lake Tana are illustrated in Fig. 3.3 and 3.4. Levels of DO and temperature exhibited a consistent pattern, with a slight decrease below the four-meter depth (Fig. 3.3. A-E). Rainy and post-rainy seasons showed superficial thermal stratification, with a 1.6 °C difference between surface and bottom temperatures, while pre-rainy and dry seasons exhibited homogenous thermal conditions. The vertical temperature distribution showed significant positive correlations ($p < 0.05$) with NO₃⁻ and negative association with EC, TSS and turbidity. The concentration of DO in the surface water remained relatively steady, indicating that the water column was well-oxygenated. Despite this, DO levels declined after the 4 m depth. A notable positive correlation ($p < 0.05$) was also observed between DO, temperature, and NO₃⁻, while EC, turbidity, and TP showed negative association. Except the pre-rainy season, levels of EC and turbidity steadily rose towards the bottom surface. Turbidity was significantly higher in the bottom during the rainy season, while it did not vary substantially ($p > 0.05$) in other seasons. The vertical distribution of pH was inconsistent throughout the water column. Generally, the pH recorded in the trophogenic zone was relatively higher than in the tropholytic zone (Fig. 3.3 A-E).

Vertical distributions of the major algal growth limiting nutrients are displayed in Fig. 3.4 (A-D). Ammonia was unevenly distributed throughout the water column. At the same time, nitrate levels increased slightly with depth in all seasons except the dry season (Fig. 3.4-B). Unlike that of TP, SRP concentration slightly decreased with increasing depth in all seasons except the pre-rainy season. Furthermore, TP was positively associated with EC and SRP, while it was

negatively correlated with DO. The Kruskal-Wallis ANOVA indicated the absence of significant difference ($p < 0.05$) in SRP, ammonia and silica levels between the surface and the bottom.

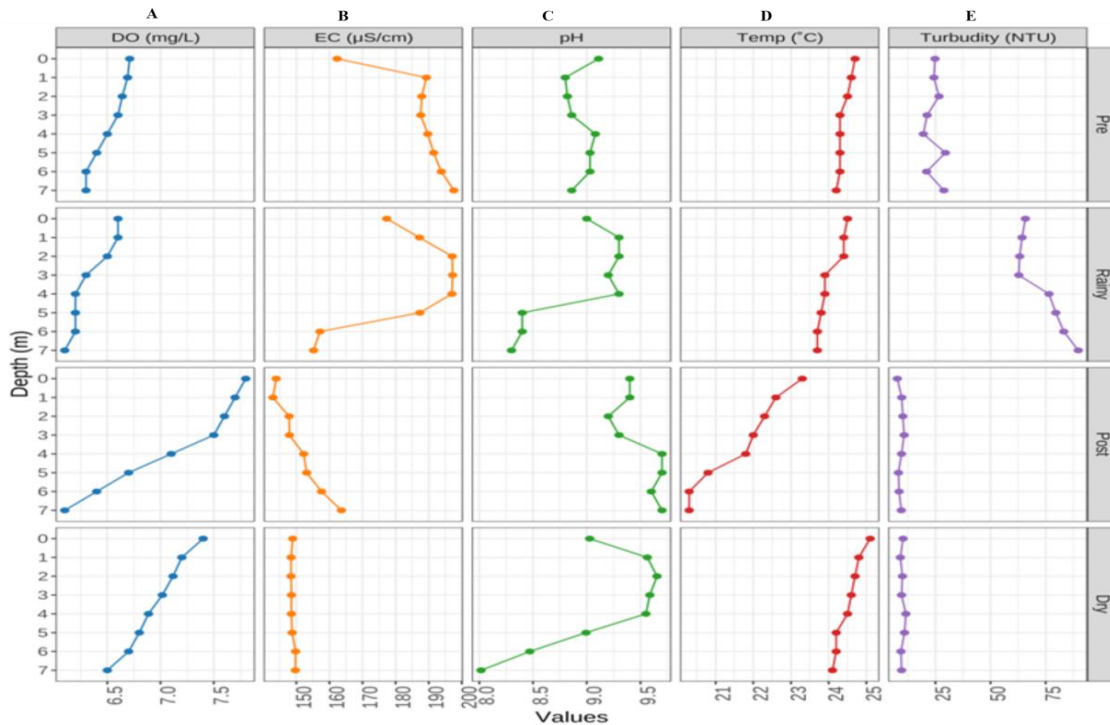


Fig. 3.3. Depth profiles of selected physico-chemical water quality parameters in Lake Tana's, northern gulf (n=12; A) DO = dissolved oxygen; B) EC = electrical conductivity; C) pH; D) Temp = temperature; E) Turbidity).

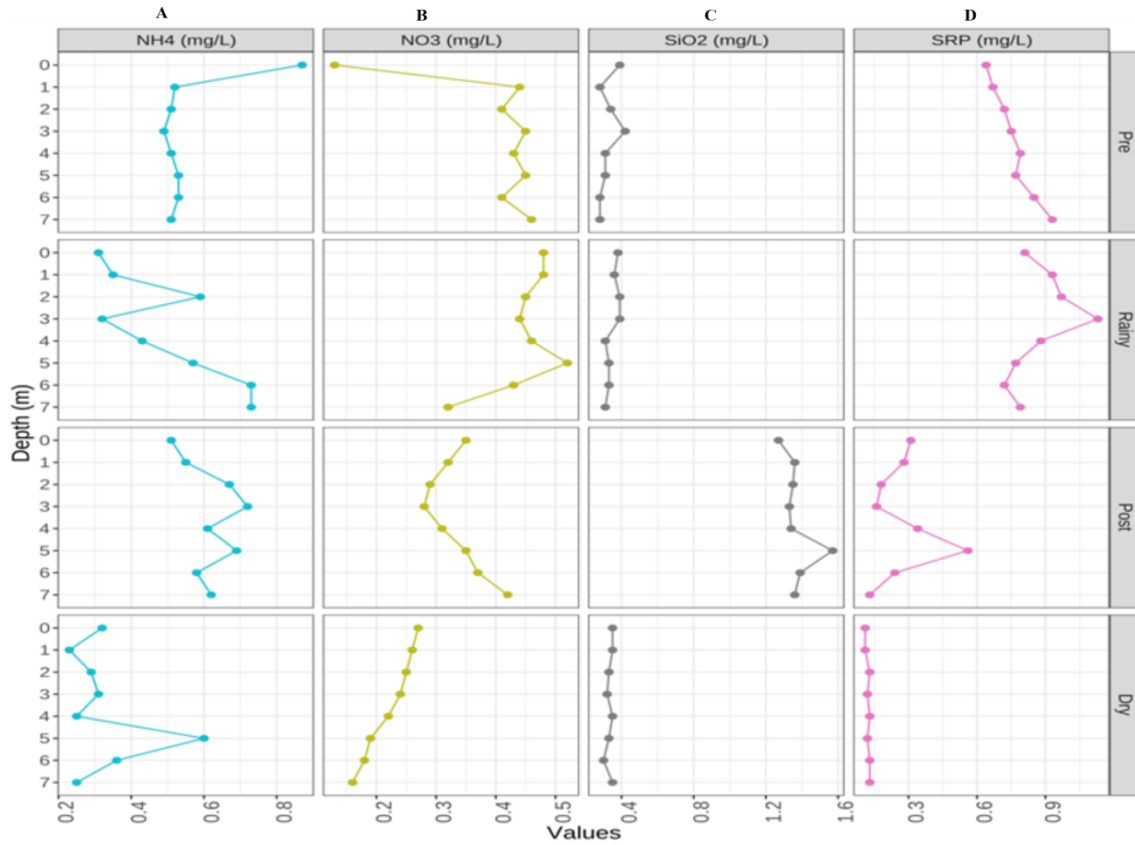


Fig. 3.4. Depth profiles of selected nutrients in Lake Tana, northern gulf (n=12; A) NH_4 = ammonia; B) NO_3 = nitrate; C) SRP = soluble reactive phosphorus; D) SiO_2 = soluble reactive silica).

3.2.4. Chlorophyll-*a* concentration and trophic state index

Chlorophyll-*a* concentration in the northern gulf of Lake Tana showed significant ($p < 0.05$) spatio-temporal variability. In the post-rainy season (Oct-Dec), the mean maximum concentration of Chlorophyll-*a* ($103.0 \mu\text{g/l}$) was recorded in the lake shore of GOR site, followed by high values of $71 \mu\text{g/l}$ in the DAB and $63.1 \mu\text{g/l}$ in the OP sites. Other sites including MEG ($59.3 \mu\text{g/l}$) and WHI ($18.1 \mu\text{g/l}$) exhibited relatively lower mean chlorophyll-*a* concentrations.

The trophic status of Lake Tana was determined using the model developed by Lamparelli (2004), which was found to be suitable for tropical aquatic systems. The model presupposes a strong interdependence among SDD, TP and Chl-*a* levels. The analytical results revealed that the trophic status of Lake Tana varied ranging from mesotrophic to eutrophic conditions (Table 3.3).

Table 3.3. Seasonal trophic state index estimated for Lake Tana according to Lamparelli (2004)

Values used for Lake Tana							
Models	Parameters	TSI	Pre-rainy	Rainy	Post-rainy	Dry	Overall
Lamparelli, 2004	SD	TSISD	55.94	58.14	45.78	43.17	50.76
	TP	TSITP	59.95	59.97	59.96	59.92	59.95
	Chl- <i>a</i>	TSIChl- <i>a</i>	67.63	66.75	69.46	67.05	67.7
	Total	TSILamp	61.17	61.62	58.4	56.71	59.47

3.2.5. Principal Component Analysis

Principal component analysis (PCA) was conducted by combining all the physico-chemical parameters to reduce number of variables and infer most discriminating parameter. The output of the PCA indicated that the first two components explained about 86.2% of the total variation (PC1 = 33.7 and PC2 = 52.5%). The score plot of PC indicated spatial heterogeneity of water samples separating clusters of MEG, DAB and GOR sites in the rainy season, while WHI and OP were almost overlapping in all seasons (Fig. 3.5). In the present study, we considered a loading value of $|\geq 0.75|$ as most important to the variation. Thus, SRP, NO₃, EC, turbidity, temperature and TDS showed highest loading towards PC1 indicating their influence in the rainy

habitat quality, biodiversity, and organism reproduction and survival. This can lead to ecosystem dysfunction and adverse effects on human health (Alemu *et al.*, 2017; Lucas *et al.*, 2022).

This study aimed to assess the water quality of Lake Tana and provide evidence-based measures to prevent its degradation. The physico-chemical water quality parameters showed significant spatio-temporal variations, with temperature ranging from 20.3 to 25.1 °C. The highest mean temperature of 24.7 ± 1.06 °C was recorded during the dry season, while the lowest mean was 21.4 ± 0.28 °C during the rainy season, with no evidence of stratification. These temperature values are consistent with the normal environmental temperature range of 21-27°C (Dejen *et al.*, 2017; Kassa *et al.*, 2021; Engdaw *et al.*, 2022). The maximum temperature recorded was within the range of 25-35°C for natural inland water bodies (Egun and Ogiesoba-Eguakun, 2018), and below the WHO's permissible limit of 30°C for aquatic life (WHO, 2017; Gobezie *et al.*, 2023). The observed variation could be attributed to seasonal differences in canopy and cloud cover, and sampling time. This finding is in agreement with previous studies (Table 3.4) (Leidonald *et al.*, 2018; Kassa *et al.*, 2021; Engdaw *et al.*, 2022), and other Ethiopian rift valley lakes such as Ziway (23.9 °C) and Hawassa (22.4°C) (Tibebe *et al.*, 2019).

Lake Tana's water has a slightly alkaline pH range of 8.0 to 10.0. However, the mean maximum pH of 9.5 ± 0.23 is outside the acceptable ranges of Ethiopian (6.5-8.5) and other standards for domestic use, such as drinking (6.5-9.2), animal watering, and irrigation (6.5-9.0) (ES, 2001; USEPA, 2011; WHO, 2017). It is also higher than other local water bodies, like Lake Chamo (8.84), Hawassa (8.66), and Ziway (8.65) (Tibebe *et al.*, 2019). The highest pH observed in this study is likely due to the presence of CO_3^{2-} and HCO_3^- ions from the massive calcite and dolomite deposits in the bedrock of the catchment area, as well as production of OH^- at equilibrium (Potasznik *et al.*, 2014; Leiva-Tafur *et al.*, 2022; Gobezie *et al.*, 2023). High pH

levels in aquatic ecosystems are often linked to increased photosynthetic removal of CO₂ (Shah *et al.*, 2019), which could explain the higher levels recorded during the post-rainy season. Decreased pH towards the lake bottom during dry and rainy seasons could be associated with microbial decomposition that increases CO₂ concentration towards equilibrium resulting in acidic water (Leidonald *et al.*, 2018).

The study revealed significant spatio-temporal variations in DO, EC and turbidity. DO measurements in the littoral regions (GOR and WHI) and river mouth sites (MEG and DAB) were consistently lower than those in the OP site. This could be ascribed to the severe anthropogenic stressors in the catchment including, the entrance of oxygen-demanding domestic and industrial waste, oxygen depletion by respiration, and high internal decomposition (Engdaw *et al.*, 2022; Gobezie *et al.*, 2023). Reduced diffusion, cessation of underwater photosynthesis by phytoplankton below the tropogenic zone, and lack of wind-generated turbulence leading to deoxygenation of the water column are among the major drivers for the decline of DO with depth (Leidnolad *et al.*, 2018; Leiva-Tafur *et al.*, 2022). The lowest DO recorded in WHI is likely due to canopy cover of water hyacinth, which forms a dense mat and blocks light penetration (Dersseh *et al.*, 2022). The mean DO reported in this study falls within the standards of 6.5-8.5 mg/l (WHO, 2017).

Turbidity and EC are tracers for the runoff event and largely depend on impervious surface and connectivity of stream channel. They showed significant seasonality to precipitation, with variability primarily attributed to non-point sources like runoff and internal mixing. The trend for both EC and turbidity increased towards the bottom of the lake, possibly due to sediment suspension by wave and thermal action (Dersseh *et al.*, 2022). Higher values were recorded during rainy and pre rainy seasons, while lowest values were during dry and post-rainy seasons.

The complete mixing of the lake in the pre-rainy season and increased runoff input from tributaries passing through agricultural lands with meager soil conservation and farming practices in the rainy season contributed to the elevated EC and turbidity (Akale *et al.*, 2018; Engdaw *et al.*, 2022).

Table 3.4. Comparison of physico-chemical parameters of Lake Tana with previous findings (DO, TDS, TP, SRP, TN, NO₃, NO₂, NH₄, SiO₂ and Alk are in mg/l; EC = µs/cm; Tur = NTU; -- = values not reported).

Temp	pH	DO	EC	Tur	SRP	TP	NO ₂ -	NO ₃ -	NH ₄	TN	SiO ₂	Reference
24	8.5	7.3	187.6	24	1.03	--	--	0.55	--	--	--	[36]
22.5	8.3	6.6	156.3	--	0.22			2.31		2.4		[63]
25.5	8.2	8.2	--	199	1.03	2.6	0.15	0.2	0.8	1.8	0.65	[67]
24.2	7.2	5.9	157.7	15.7	0.41	0.45	0.01	1.3	0.22		13.9	[65]
23.6	9.26	7.82	175	550	3.75		1.3	13.8	0.88		32.6	[64]
25.5	8.9	6.8	164.2	--	0.34	1.0	--	1.53	--	--	--	[21]
22.7	9.0	6.7	197.9	199.7	0.69	1.48	0.07	0.53	0.9	1.5	0.93	Current study

Anthropogenic activities such as excavation, and sand mining together with algal development and suspension of organic matter may also contributed to the increased EC and turbidity of Lake Tana (Liu *et al.*, 2020; Gule *et al.*, 2023; Gobezie *et al.*, 2023). The lowest EC and turbidity

values were observed in the WHI areas, possibly due to absorption and filtration capacity of the macrophytes (Goshu and Aynalem, 2017; Engdaw *et al.*, 2022). The highest EC ($283.7 \mu\text{S cm}^{-1}$) falls below the WHO's permissible limit of $2500 \mu\text{S/cm}$ for freshwater, while the lowest turbidity (7.6 NTU) exceeds the drinking water standard of 5 NTU (WHO, 2017; Kassa *et al.*, 2021; Gobezie *et al.*, 2023).

Nutrient enrichment substantially impacts water quality, integrity and ecological status of aquatic systems (Durand *et al.*, 2011; Ma *et al.*, 2024; Mohammed *et al.*, 2024). The results of this study highlighted the implications of elevated nutrient concentrations on the recent eutrophication, proliferation of alien macrophytes and domestic water use (Tantu *et al.*, 2024). However, various factors including DO, pH, temperature, sediment texture, depth, and redox potential influence the transformation of nutrients, altering stoichiometry and dimension of availability in usable form (Durand *et al.*, 2011; Inamdar *et al.*, 2015). In the northern gulf of Lake Tana, spatially non-uniform but temporal variations among nutrients were recorded. Concentrations of NO_3^- , SRP and TP were higher during the rainy season at MEG and DAB sites, mainly due to surface runoff, intensive anthropogenic activities such as applications of fertilizers, sewage and industrial effluents from nearby highly populated towns (Gondar, Koladiba and Chuahit) with poor wastewater treatment and drainage systems (Alemu *et al.*, 2017; CSA, 2017; Dersseh *et al.*, 2022; Mohammed *et al.*, 2024; Tantu *et al.*, 2024). Domestic wastewater and agricultural land use have been linked to increased nutrient content of Lake Tana (Akale *et al.*, 2018; Tibebe *et al.*, 2019).

The study found that the concentrations of NO_3^- in Lake Tana ranged from 0.26 to 0.56 mg/l at the WHI and MEG sites, respectively. The mean NO_3^- (0.4 mg/l) concentrations observed in this study is considerably higher than the values reported by Mohammed *et al.* (2024) for Lake

Ardibo (0.0036 mg/l), Degefu and Schagerl (2015), for Lake Wonch (0.0047 mg/l), Fetahi *et al.* (2014) for Lake Hayq (0.0042 mg/l), and Tibebe *et al.* (2019) for Lake Tana (0.0076 mg/l). Higher NO_3^- concentrations are attributed to animal husbandry byproducts, surface runoff, sewage, and domestic and industrial wastewater effluents (Inamdar *et al.*, 2015). Akale *et al.* (2018) have highlighted the inevitable contribution of poor soil-water conservation and backward farming practices in washing applied fertilizers. Conversely, lower concentrations of NO_3^- were measured during the dry (0.26 mg/l) and post-rainy (0.36 mg/l) seasons, possibly due to uptake by phytoplankton and submerged aquatic macrophytes and denitrification. The present finding is consistent with the reports of (Gogoi *et al.*, 2019; Mohammed *et al.*, 2024). Ammonia concentration in the northern gulf of Lake Tana varied between 0.48 mg/l at DAB and 0.7 mg/l at WHI sites (Table 3.2). The highest mean concentration (0.72 mg/l) was observed during the pre-rainy season, while the minimum (0.3 mg/l) value was recorded during the dry season (Table 3.1), where DO was relatively higher, and could favor nitrification (Strokal *et al.*, 2016). The observed rise in ammonia concentrations could be associated with the high pH value of 8.7 during the pre-rainy season, which increases the proportion of unionized ammonia in the environment (Mohammed *et al.*, 2024). Ammonia concentration higher than 0.05 mg/l indicates poor water quality and anthropogenic contamination. Thus, the present ammonia level potential threatens human and animal health.

Concentrations of TP ranged from 1.43 to 2.35 mg/l at OP and MEG while SRP ranged from 0.4 to 1.2 mg/l at WHI and DAB sites, respectively. The observed mean TP and SRP concentrations are higher than values reported previously for Lake Tana (1.73 and 0.88 mg/l) by Tibebe *et al.* (2019), for Lake Hayq (0.067 and 0.029 mg/l) by Fetahi *et al.* (2014), and for Lake Ardibo (0.069 and 0.024 mg/l) by Mohammed *et al.* (2024). The increasing trend in concentrations of

TP and SRP, especially in DAB and MEG sites, could be attributed to nutrient enrichment from intense anthropogenic activities and fertilizer inputs from non-point allochthonous sources rich in organic matter. Additionally, the presence of phytoplankton, zooplankton, detritus, animal waste, and other macrophyte components may have contributed to the elevated TP level in the lake. Similarly, higher concentrations of TP and SRP were also reported in water bodies closer to agricultural land use due to elevated applications of phosphate containing fertilizer (100 kg of DAP/ha), which easily leaches in the form of PO_4^- to nearby aquatic systems (Akale *et al.*, 2018; Fetahi, 2019; Tibebe *et al.*, 2019). Further, animal excretions, phosphate-containing detergents, and runoff from pastureland contribute significantly to phosphorus enrichment (Li *et al.*, 2014; Alemu *et al.*, 2017). TP and SRP showed significant spatio-temporal variations with maximum concentrations recorded during pre-rainy and rainy seasons at the river mouth and littoral areas of the lake. Alemu *et al.* (2017) elucidated the distinct temporal variation in TP and SRP concentrations, with higher values recorded during the rainy season. However, the substantial decline in the concentration of SRP at OP and WHI site could be associated with uptake by algae and aquatic macrophytes or adsorption onto fine sediment particles, especially during anoxic condition (Kassa *et al.*, 2021; Sishu *et al.*, 2021; Mohammed *et al.*, 2024).

Silica (SiO_2), an essential nutrient that limits diatom growth (Fetahi *et al.*, 2014), showed no significant spatio-temporal variations except during the post-rainy season. This study's mean silica level (0.63 mg/l) is lower than other Ethiopian lakes. For example Mohammed *et al.* (2024) reported 7.57 mg/l for Lake Ardibo, Lemma *et al.* (2016) documented 81.4 mg/l for Lake Abijata and 1.7 mg/l for Lake Chamo. Like nitrogen and phosphorus, silica in aquatic systems determines diatom abundance and distributions. The observed lower concentrations of silica could be due to decreased geogenic deposition. Silica concentrations lower than 0.5 mg/l limit

diatom growth and abundance (Mohammed *et al.*, 2024), suggesting that the Lake Tana ecosystem can support this group of algae.

Trophic status index (TSI) indicates nutrient enrichment, eutrophication, biotic integrity, and chemistry of aquatic systems (Lucas *et al.*, 2022). In this study, we used water transparency, Chl-*a*, and TP to determine the TSI of Lake Tana, specifically its northern gulf. We applied the models developed by (Lamparelli, 2004; Cunha *et al.*, 2013) for tropical aquatic systems. According to the organization for economic development and cooperation (OECD) (OECD, 1982), a SD value falling within the $1.5 < x < 3$ m range indicates a eutrophic lake. Despite the lower SD values recorded in this study due to seasonal fluctuations, i.e. during the pre-rainy (mixing) and rainy seasons (rain turbulence, runoff and cloud cover), the TSI of Lake Tana based on SD value falls in the range for eutrophic lakes. Profound soil loss due to erosion in the Lake Tana catchment ranged 12-27 t/ha/yr (Ahmed *et al.*, 2020), significantly influencing the TSI. The Kruskal-Wallis ANOVA showed significant ($p < 0.05$) temporal variation in SD, with higher value recorded in the dry and lower in the rainy seasons. Consequently, Lake Tana can be classified as a productive lake with low euphotic zone. This finding contradicts with the report by Tibebe *et al.* (2019), which categorized Lake Tana as a hyper-eutrophic lake. TSI of Lake Tana based on the mean annual TP concentration (59.95) falls in the category of eutrophic ($59 < \text{TSI} < 64$). This result also contradicts the findings of (Wondie *et al.*, 2007; Tibebe *et al.*, 2019) who reported the TSI of Lake Tana as oligo-mesotrophic and hypertrophic lake, respectively. Various findings also illustrated a TP concentration > 35 µg/l as a threshold for eutrophication (OECD, 1982; Fetahi *et al.*, 2014; Mohammed *et al.*, 2024). Based on Chl-*a* concentration, which is an index of algal biomass, a lake with TSI that fall within the range of $52 < \text{TSI} < 59$ is categorized as mesotrophic lake. Therefore, TSI of Lake Tana based on Chl-*a*

makes it a hypertrophic lake. This finding corroborates with report of (Tibebe *et al.*, 2019). Restrictions of sampling sites exposed to severe anthropogenic stressor that resulted in algal blooming could have influenced the result. Compared to other Ethiopian lakes, Lake Tana showed lower productivity in terms of Chl-*a*, possibly due to seasonal grazing by zooplankton, competition with water hyacinth for nutrient and light, and siltation-induced turbidity that block light or cause sinking. Overall TSI value of Lake Tana falls in the category of eutrophic condition.

3.4. Conclusions and recommendation

Most of the physico-chemical parameters recorded for the northern gulf of the lake showed significant spatio-temporal variations. Industrial and municipal effluents and surplus agrochemical inputs are assumed to be the major causes of nutrient enrichment in Lake Tana. The lake's trophic status has been changed from oligotrophic to eutrophic condition indicating temporal variability. Despite spatio-temporal variability, mean concentrations of nutrients recorded for Lake Tana exceeded USEPA threshold concentrations for potentially causing eutrophication. Therefore, integrated watershed management, delineation of buffer zone, waste management and continuous monitoring are essential to avert future ecological imbalance of the lake. Moreover, creating awareness about the need to exercise modest fertilizer application and introduction of waste treatment ponds to resorts will have a substantial impact the sustainable use and management of the lake.

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CHAPTER 4

Taxonomic and functional groups of phytoplankton in the northern gulf of Lake Tana, Ethiopia

This chapter is based on the following published paper:

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Abstract

Water quality of Lake Tana has been deteriorating due to nutrient enrichment associated with various anthropogenic stressors including domestic and industrial effluents, and agricultural runoff. Changes in the ecological health of a lake are better understood by studying the community structure of phytoplankton as they react quickly to environmental changes. Human-induced stressors can lead to changes in the taxonomic composition and Reynolds functional groups (RFGs) of phytoplankton resulting in biodiversity impoverishment. RFGs are more effective in detecting stressor-induced alterations in phytoplankton compared to taxonomy-based assemblages as the former reflect common adaptive or functional traits. Information on RFGs of phytoplankton in the northern gulf of Lake Tana, Ethiopia, is however non-existent. Therefore, this study aimed to assess the spatio-temporal changes in phytoplankton community structure and identify the variables influencing the dynamics of RFGs in the lake. Samples were collected from five sampling sites between May 2023 and April 2024. A total of 113 phytoplankton species belonging to six taxonomic groups and 11 RFGs were identified. The predominant RFGs which collectively accounted for 97% of the total phytoplankton abundance were **M**, **B**, **N**, **D**,

Lo, P and **S2**. Other RFGs, such as **J** and **X1-X3**, constituted 2.4% and 0.8% of the total abundance, respectively. Redundancy analysis indicated significant association of RFGs with electrical conductivity, pH, turbidity, and some algal growth limiting nutrients. This suggests that both biotic and anthropogenic stressors have a substantial impact on RFGs. Therefore, continuous monitoring would help in identifying mitigation measures essential to prevent future ecological imbalances in the lake.

Key words: anthropogenic stressor, phytoplankton, RFGs, taxonomy, water quality

4.1. Introduction

Phytoplankton is an extremely diverse polyphyletic group of primarily photoautotrophic organisms, which fuel food webs and drive biogeochemical cycling (Fetahi *et al.* 2011; Rousseaux and Gregg 2011; Kahsay *et al.* 2022; Bai *et al.* 2023; Walbara *et al.* 2024; Wang *et al.* 2024). The community structure and dynamics of phytoplankton have significant effects on the stability and services of aquatic ecosystems (Wang *et al.*, 2021). The conventional method of monitoring structure of phytoplankton communities involved determination of their taxonomic composition (Mischke *et al.*, 2011). Although the accuracy of morphological taxonomic classification of phytoplankton has recently been improved by genetic and molecular approaches, ecologists have often been dissatisfied with the taxonomic assemblages as they are unable to reflect the observed ecological functions (Nasselli-Flores *et al.* 2007; Salmaso and Padisák 2007), and due to the inherent complications when working with these tiny individuals (Bai *et al.* 2023; Wang *et al.* 2024). Additionally, the overwhelming abundance of phytoplankton species results in significant workloads, challenging researchers' competence (Fang *et al.* 2022). As a result, traditional taxonomic schemes of classification often fail to accurately depict the health of aquatic ecosystems and reflect the observed ecological functions (Nasselli-Flores *et al.* 2007; Salmaso and Padisák 2007; Wang *et al.* 2024; Wu *et al.* 2024). Furthermore, conventional

taxonomic studies are not suitable for comparing remote habitats with distinct species compositions and similar functional traits (Etisa *et al.* 2024; Lee *et al.* 2024). In ecosystems where there is no well-established groundwork, the interplay between the food chain, climatic conditions, and hydro-chemical factors complicates the accurate assessment of the prevailing phytoplankton species within the community, making the assessment much more difficult (Liu *et al.* 2023; Wang *et al.*, 2024). Results of a taxonomic study on phytoplankton are not sufficient to describe the perceived ecological functions (Padisák, 2007) and a taxonomic unit may also contain phytoplankton species that have different ecological and functional properties (Padisák and Reynolds 1998).

Phytoplankton assemblages are shaped largely by local conditions (Beisner *et al.* 2016). Thus, the local conditions should favor groups of species, which are not necessarily taxonomically related, that exhibit similar adaptive features (Webb *et al.* 2002). This shows that phytoplankton functional groups are well predictable from changes in environmental variables suggesting their greater suitability as water quality indicators (Estifanos *et al.*, 2022). In an effort to clearly understand ecosystem functioning, phytoplankton ecologists have, therefore, recently been classifying species based on their shared structural and functional traits (Reynolds *et al.*, 2002; Abonyi *et al.* 2012; Bai *et al.* 2023; Liu *et al.* 2024), as environmental conditions select for or against traits, rather than species (Borics *et al.* 2016; Lee *et al.* 2024; Hou *et al.* 2025). Since the initial attempt of Reynolds (1980), numerous efforts have improved the concept of classification into functional groups (FGs), resulting in the evolution from 34 distinct codes to 39 FGs used to classify phytoplankton based on identifiable traits. As a result, RFGs are frequently used to evaluate the state of aquatic systems.

There is a wealth of information on the functional group-based ecological assessment of phytoplankton in various lakes and reservoirs around the world (Reynold *et al.* 2002; Nasselli-Flores *et al.* 2007; Salmaso and Padisak 2007; Alves-de-Souza *et al.* 2008; Gebrehiwot *et al.* 2017; Estifanos *et al.* 2022; Lee *et al.* 2024; Zhao *et al.* 2025). Due to its broad applicability and solid ecological foundation, FG classification is widely used to evaluate the ecological status of aquatic systems globally (Wang *et al.* 2024; Hou *et al.* 2025). It can be used in conjunction with water quality and trophic state index (TSI) to provide a comprehensive assessment of the environment. However, this method is limited by the use of incorrect classification codes and lack of habitat preference for certain species (Abonyi *et al.* 2012).

Among the East African great lakes, Lake Tana, the subject of the present study, is the third largest in the Nile River basin, and the largest lake in Ethiopia. Its ecosystem services including hydropower generation, capture fishery, and water supply for agricultural (irrigation), industrial, and domestic use, are irreplaceable and make the lake a vital component of the environment of the region (Wondie and Mengistou 2017; Gitaw *et al.* 2020; Engdaw *et al.* 2024). However, during the last two decades, the Lake Tana ecosystem has undergone unprecedented changes due to multiple stressors, such as climate change, hydrological alterations, urbanization and industrialization-induced waste effluent, siltation, nutrient enrichment, and severe anthropogenic-induced land use/land cover changes, shoreline modifications, and excessive water abstraction (Goshu and Aynalem 2017; Dersseh *et al.* 2022; Kahsay *et al.* 2022; Swunet *et al.* 2022). These threats have triggered a series of ecosystem changes, encompassing water quality deterioration, biodiversity impoverishment, eutrophication, and overall ecosystem dysfunction, which is reflected in the decline in its ecosystem service (Goshu and Aynalem 2017; Kassa *et al.* 2021; Engdaw *et al.* 2025). It is essential to exploit this lake without affecting

its water quality and quantity to ensure sustainable use and peaceful coexistence. Ensuring sustainable and effective exploitation of this aquatic resource requires the development of appropriate strategies of protection and restoration, which necessitate the generation of adequate information on water quality (Sewunet *et al.* 2022). Although water quality can be assessed using physico-chemical parameters (Tibebe *et al.* 2019; Kassa *et al.* 2021; Engdaw *et al.* 2022; Gobezie *et al.* 2023; Engdaw *et al.* 2025), the most direct and effective indicator of the ecological health of a lake is the status of its living systems (Reynolds *et al.* 2002; Gebrehiwot *et al.* 2017; Estifanos *et al.* 2022; Etisa *et al.* 2024; Kahsay *et al.* 2022; Welbara *et al.* 2024).

Phytoplankton are the most commonly employed bio-indicators of anthropogenic disturbances due to their quick reaction to wide range of stressors (Domingues and Galvão, 2007; Baloloy *et al.* 2016; Mohammed *et al.* 2023), which enables them to provide early warning signals of environmental changes. In addition to their fast response to environmental changes, their short generation time, and wide range of phenotypic diversity make them exquisite bio-indicators of long-term ecological changes in aquatic systems (Crossetti *et al.* 2013; Mihaljevic *et al.* 2015; Liu *et al.* 2024). Comprehensive understanding of the spatio-temporal dynamics of phytoplankton in freshwater ecosystems has now become a common practice (Fetahi *et al.*, 2014; Ogato and Kifle, 2017; Lee *et al.*, 2024; Zhao *et al.*, 2025).

While numerous studies have been conducted on the lake's ecology (Wondie and Osondu 2008; Wondie and Mengistou 2017; Gitaw *et al.* 2020; Kassa *et al.* 2021; Dersseh *et al.* 2022; Kahsay *et al.* 2022; Engdaw *et al.* 2024), most focused on the southern gulf, neglecting the northern gulf probably due to its remote location, limited access, and logistical constraints (Engdaw *et al.* 2025). Therefore, there is limited information on phytoplankton ecology of the lake, particularly in relation to their FG classification. The aim of this study is therefore, to investigate the spatio-

temporal dynamics of the taxonomic and functional composition of the phytoplankton assemblages in Lake Tana's northern gulf.

4.2. Materials and methods

4.2.1. Phytoplankton sampling

Phytoplankton samples were obtained by vertically hauling 30 µm mesh size plankton net, with a 42 cm opening and 58 cm length. Phytoplankton samples were also collected by vertical and horizontal towing of 15 µm mesh size plankton net, and placed in polyethylene-terephthalate bottles. Additionally, a 2-liter capacity Van Dorn bottle sampler (Thermo Fisher Scientific Inc., Waltham, MA, USA, horizontal model) was used to collect phytoplankton samples in the pelagic region. The vertical hauling depth in the littoral sites ranged from 0.5 to 1 m, while it was up to 1 m plus the euphotic depth deep in the pelagic region. All samples were dispensed into brown bottles and preserved with Lugol's iodine solution by adding the preservative until their color turned into that of weak tea (Hötzel and Croome, 1999; Estifanos *et al.*, 2022; Etisa *et al.*, 2024). They were then stored under dark condition until transported to limnology laboratory.

4.2.2. Identification and enumeration of phytoplankton

Identification of phytoplankton taxa to the lowest possible taxonomic level using both preserved and fresh samples was conducted under an inverted microscope using various taxonomic literatures (e.g. Hindák, 1992; Komárek and Komárková, 2003; Krammer, 2003; Bellinger and Sigeo, 2010; Arguelles *et al.*, 2014). Names of taxonomic author and phyla are according to Guiry and Guiry (2013). Enumeration of phytoplankton was also done using preserved samples, which were concentrated by sedimentation. A 1000 ml composite water sample was sedimented in a 1-liter capacity cylinder by allowing it to stand for a period of 6 hours per cm height of the sedimentation chamber under dark conditions (Lund *et al.*, 1958). After sedimentation, approximately 90% of the water was siphoned off without disturbing the settled algae. A 1 ml

subsample was then taken from the concentrated sample and placed into a Sedgewick-Rafter cell for cell counting at a magnification of 400× using an inverted microscope (A KRÜSS Optronic MBL320 model), with a 95% confidence interval and a counting error of 10.33 (Lund *et al.*, 1958; Kruk *et al.*, 2002). The enumeration of phytoplankton taxa was carried out by randomly counting 30-50 grids, and the actual number of cells per meter-cube (m³) of the original sample was calculated according to Hotzel and Croome (1999). For colonial and filamentous algae, the mean number of cells per filament or colony was determined by counting the number of cells in the filament or colony of the sample in the first 30-50 grids (Kruk *et al.*, 2002; Alves-de-Souza *et al.*, 2006; Etisa *et al.*, 2024).

$$Cell.l^{-1} = \frac{n \times GF \times SSF}{\pi \times R^2 D} \quad (2)$$

Where; Cells = cell counts of phytoplankton, n = actual species counts, GF = grid factor, SSF = sub-sample factor, R = radius of the net (m), D = vertical haul depth (m), π = constant (3.14)

Grid factor is the ratio between total numbers of grids in the Sedgewick-Rafter cell divided by the total number of counted grids with cell (eq. 3).

$$GF = \frac{\text{Total nuber of grids}}{\text{counted number of grids with cell}} \quad (3)$$

Subsample factor can be calculated by dividing the total volume of concentrate to the actual concentrated volume of sample used for counting (eq. 4).

$$SSF = \frac{Tv \text{ of concentrate}}{\text{Actual volume of concentrate}} \quad (4)$$

Phytoplankton community species diversity was determined using species richness and diversity indices, which were calculated according to the Shannon-Weaver diversity index (H') (Shannon and Weaver, 1949).

$$H' = \sum_{i=1}^S \rho_i \times \ln \rho_i \quad (5)$$

Where, H' = diversity index, ρ_i = relative contribution of the i^{th} species to the total abundance of all species together, S = number of species in the community.

Species richness of phytoplankton was determined using Margalef's index (D) (Death, 2008).

$$D = \frac{S - 1}{\ln N} \quad (6)$$

Where, D = species richness of phytoplankton, S = number of species in the population and N = total number of individuals collected.

4.2.3. Functional classification of phytoplankton

The functional groups of phytoplankton were classified based on their ecological affinity and concept of adaptive strategy including survival strategy, tolerance and sensitivity to different environmental stressors. Therefore, the various phytoplankton taxa identified in this study were classified into RFGs based on Reynolds *et al.* (2002) and Padisak *et al.* (2009). The abundance corresponding to each functional group was calculated for numerical analysis.

4.2.4. Data analyses

Data of all parameters were summarized and analyzed using SPSS (Ver. 25). Before analysis, the data was checked for normality using the Shapiro-Wilk test and for homogeneity of variances. Spatial and temporal variability in RFGs were analyzed using non-parametric ANOVA (Kruskal-Wallis test). Multivariate analysis of the association between phytoplankton species distribution

and physico-chemical parameters was examined using CANOCO Version 4.5. To determine the appropriate ordination method, Detrended Correspondence Analysis (DCA) of the RFGs and environmental data was first conducted. The gradient lengths were short (<3.0) indicating that the RFGs and environmental variables had linear relationship. Therefore, RDA with manual forward selection was chosen (Leps and Smilauer, 2003). Additionally, Pearson correlation was also used to examine the correlation between phytoplankton and environmental parameters.

4.3. Results

4.3.1. Spatio-temporal variations in phytoplankton community structure

A total of 113 phytoplankton taxa, belonging to six taxonomic groups; Bacillariophyta (diatoms, 41), Chlorophyta (green algae, 42), Cyanophyta (cyanobacteria, 24), Euglenophyta (euglenoids, 4), Cryptophyta (Cryptomonads, 1) and Dinophyta (dinoflagellates, 1), were identified in the composite samples from the northern gulf of Lake Tana (Appendix 1). The major taxonomic groups contributing to the total phytoplankton abundance were green algae (37.2%), diatoms (36.3%), and cyanobacteria (21.2%). The species-poor and rare taxa collectively contributed of 5.3%, of the total abundance. Some species including those of Cryptomonads were sporadic and infrequent, often seen in small numbers. In terms of abundance, the phytoplankton communities were constituted largely by several species, including *Aulacoseira alpigena*, *A. granulata*, and *Amphora ovalis*, from division Bacillariophyta, and *Closterium acutum*, *Staurastrum multinodulosum*, *Staurodesmus triangularis* var *limneticus*, and *Oocystis* from division Chlorophyta. Additionally, Cyanobacterial species such as *Microcystis flos-aquae*, *Dolichospermum circinale*, *Microcystis aeruginosa*, and *Planktolyngbya limnetica* were also found largely across all sites. However, other species including *Cryptomonads*, *Peridinium gatunense*, *Euglena viridis*, *Phacus longicauda*, *Peranema trichophorum*, and *Euglena* sp.

showed a reverse trend, with higher abundance occurring in the littoral sites compared to the pelagic sites.

The overall phytoplankton biomass, estimated as Chl-*a*, showed an increasing trend from littoral (macrophyte-dominated and highly impacted) sites to the pelagic (open water) sites. Significant spatial variation ($p < 0.05$) was exhibited by the total abundance of dominant algal groups, including Chlorophyta, Bacillariophyta, and Cyanophyta with consistent increase from macrophyte-dominated littoral sites to the pelagic one. The mean Chl-*a* level recorded at the OP and GOR sites (87 $\mu\text{g/l}$) during the post-rainy season was significantly higher ($p < 0.05$) than those of the littoral sites, DAB (71 $\mu\text{g/l}$), MEG (59.3 $\mu\text{g/l}$), and WHI (18.1 $\mu\text{g/l}$), while the lowest Chl-*a* concentrations were observed during the rainy and pre-rainy seasons. Conversely, higher Cyanobacterial abundance was recorded at OP site during post-rainy and rainy seasons. Other taxonomic groups of phytoplankton, such as diatoms and euglenoid, showed peak abundance during the pre-rainy and rainy seasons, respectively.

In terms of abundance, the phytoplankton communities showed spatio-temporal changes (Fig. 4.1 and Fig. 4.2), with the highest abundance occurring in the post-rainy season (115×10^6 cells/ m^3), followed by that of the pre-rainy (108×10^6 cells/ m^3) and dry (634×10^4 cells/ m^3) seasons. The lowest abundance (287×10^4 cells/ m^3) was recorded during the rainy season. These results are consistent with the observed Chl-*a* levels with the highest (103 $\mu\text{g/l}$) and lowest (59.3 $\mu\text{g/l}$) values coinciding with the highest and lowest phytoplankton abundance, respectively. However, in the rainy season, Bacillariophyceae accounted for more than half of the total abundance (50.1%), while Chlorophyceae dominated in the other seasons. In the post-rainy season, Chlorophyceae had the highest total abundance (108×10^5 cells/ m^3 , 41.1%) followed by Bacillariophyceae (31.1%) and Cyanophyceae (25.1%). The rest of the taxa had negligible

contribution to the total abundance (Fig. 4.1). Spatially, the order of dominance in phytoplankton abundance was Cyanophyceae > Bacillariophyceae > Chlorophyceae > Eugleneophyceae > Dinophyceae and Cryptophyceae in all sites except the WHI site (Fig. 4.2). The highest abundance levels of Cyanophyceae were observed in the order DAB (55.2%) > OP (52.8) > GOR (51.6%) > MEG (50.5) > WHI (25.8). Bacillariophyceae dominated the OP site with percentage abundance of 55.9%, while their abundance was similar in the other sites. Chlorophyceae had the highest total abundance in the pelagic site, with their abundance significantly decreasing in the littoral areas. During the rainy season, Dinoflagellates, particularly *Peridinium gatunense*, and Euglenoids (*Phacus longicauda*, and *Euglena viridis*) were the least dominant phytoplankton species, contributing 1.9 to 9.9 % of the total abundance.

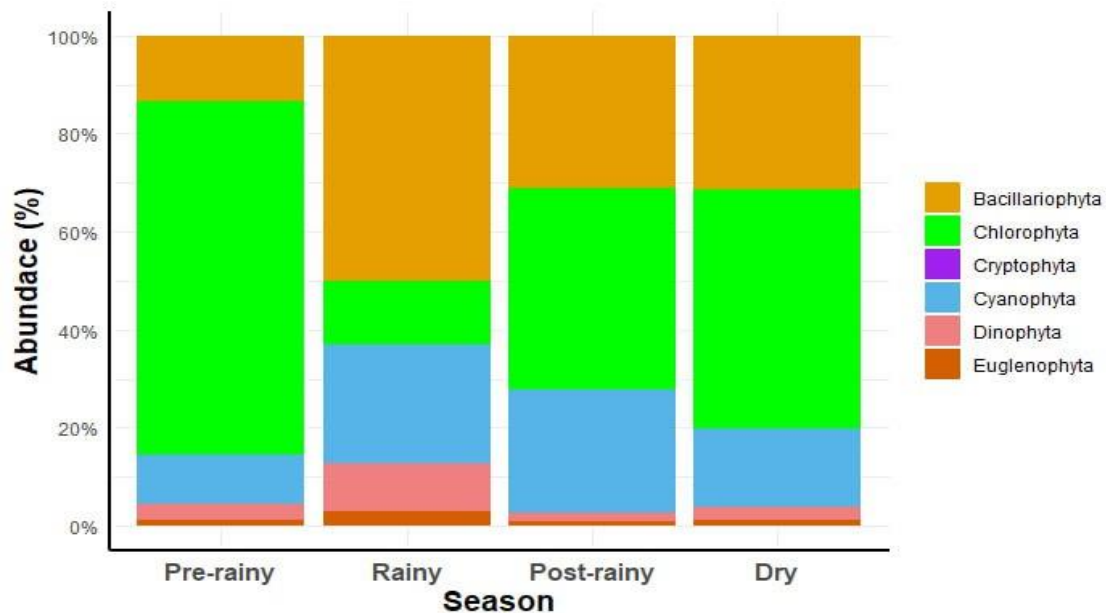


Fig. 4.1. Mean seasonal abundance of phytoplankton groups in the northern gulf of Lake Tana.

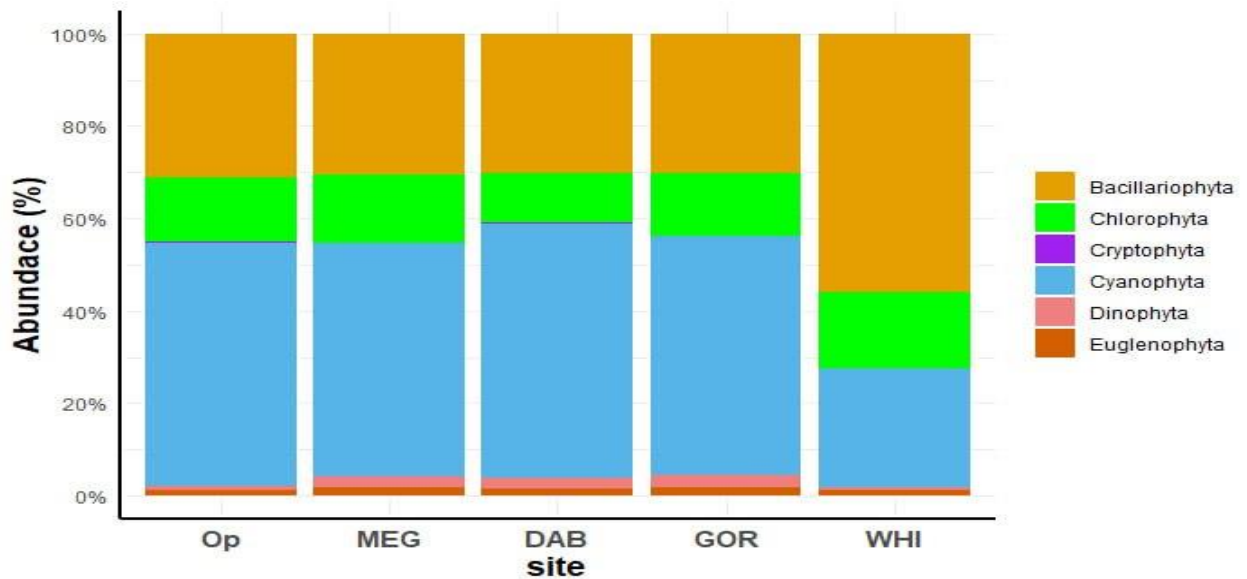


Fig. 4.2. Mean spatial abundance of phytoplankton groups in the northern gulf of Lake Tana

There was a significant ($p < 0.05$) variations in phytoplankton species richness, diversity, evenness, and dominance among sampling sites and seasons (Table 4.2). The highest species richness was observed at the OP site (106), followed by those of GOR (96), MEG (91), DAB (86) and WHI (73) sites. The species richness of phytoplanktons communities decreased in the order Post-rainy>Pre-rainy>Dry>Rainy seasons. The Shannon-Wiener index indicated that the diversity of phytoplankton communities in Lake Tana was higher, with the highest value of $H' = 3.61$ recorded at the OP and GOR sites, and moderate diversity index value of $H' = 3.0$ at the WHI site. However, evenness ranged from 0.13 at the WHI site, indicating less uniform distributions of taxa to 0.7 at the DAB site, revealing even distribution of phytoplankton species. The temporal dynamics of phytoplankton showed notable variations among seasons (Fig. 4.1). The highest species richness (98 species) and evenness (0.34) were observed during the post-rainy season, followed by those of dry, pre-rainy, and rainy seasons. Temporally, diversity index

was most pronounced during the post-rainy season with $H' = 3.61$, while less diversity was noted during the rainy season.

Table 4.1. Spatio-temporal variations in species diversity of phytoplankton in Lake Tana, Ethiopia

Index	Seasonal				Spatial				
	Pre-rainy	Rainy	Post-rainy	Dry	OP	MEG	DAB	GOR	WHI
Shannon_H	3.2	2.9	3.5	3.3	3.61	3.37	3.22	3.61	3.0
Simpson_1-D	0.92	0.89	0.94	0.93	0.95	0.94	0.93	0.96	0.87
Evenness_e^H/S	0.3	0.22	0.34	0.31	0.35	0.32	0.28	0.41	0.24

4.3.2. Reynolds Functional groups (RFGs) of phytoplankton

The identified phytoplankton taxa were qualitatively classified into 11 RFGs, designated by alpha numeric codes (Table 4.2). The most predominant RFGs were **M**, **B**, **N**, **D**, **Lo**, **P** and **S2**, which accounted for 96.7% of the total phytoplankton abundance. **M** accounted for 37% of the total abundance, followed by **B** (23.3%), **N** (10.1%), **D** (7.9%), **Lo** (7.2%), **P** (6.9%), and **S2** (4.3%). Other groups such as **J** and **X1-X3**, constituted 2.4% and 0.8% of the total abundance, respectively.

Table 4.2. Descriptions of the dominant phytoplankton RFGs, their habitat preference, tolerance and sensitivity (Reynold *et al.* 2002; Padisák *et al.* 2009), and their representative taxa in the northern gulf of Lake Tana, Ethiopia.

RFGs	Typical habitat	Representative species in Lake Tana	Tolerance	Sensitivity
B	Vertically mixed, mesotrophic small	<i>Aulacoseira alpigena</i> (Grunow) Krammer,	Light deficiency	pH rise, Si depletion,

	to medium lakes	<i>Aulacoseira granulata</i> var. <i>angustissima</i> , <i>Cyclotella</i> <i>comate</i> , <i>Cyclotella radiosa</i>		stratification
D	Shallow, enriched turbid water	<i>Aulacoseira italica</i> , <i>Phormidium breve</i> Kützing, <i>Nitzschi acicularis</i>	Flushing	Nutrient depletion
N	Mesotrophic epilimnion	<i>Dolichospermum circinale</i> , <i>Cosmarium sexangulare</i> , <i>Staurostrum</i> sp. <i>Staurodesmus</i> <i>incus</i>	Nutrient deficiency	Stratification, pH rise
P	Eutrophic epilimnion	<i>Aulacoseira granulata</i> , <i>Aulacoseira agassizii</i> , <i>Closterium acutum</i> , <i>Closterium</i> <i>limneticum</i>	Mild light and C deficiency	Stratification, Si depletion
S2	Shallow, turbid mixed layers	<i>Cylindrospermopsis</i> sp., <i>Pseudanabaena acicularis</i>	Light deficient conditions	Flushing
X1 – X3	Shallow, mixed layers in enriched condition	<i>Ankistrodesmus falcatus</i> , <i>Staurodesmus triangularis</i> var. <i>limneticus</i>	Stratification, low base status	Filter feeding, nutrient deficiency, mixing, grazing
J	Shallow, enriched lakes	<i>Coelastrum microporum</i> , <i>Pediastrum simplex</i> , <i>P. duplex</i> , <i>Scenedesmus</i> sp., <i>Tetrastrum</i> sp.	High light	Nutrient deficiency
Lo	Summer epilimnion	<i>Merismopedia</i> sp <i>Peridinium</i> <i>gatumense</i> , <i>Phacus longicauda</i>	Segregated nutrients	Prolonged or deep mixing
M	Mixing layer	<i>Microcystis flos-aquae</i> , <i>M.</i> <i>aeruginosa</i> , <i>M. wessenbergii</i>	High insolation	Flushing, low total light

There was significant spatial variation ($p < 0.05$, Tukey HSD test) in the dominant RFGs (**M**, **B**, **N**, **D**, **Lo**, **P**, and **S2**). The **B** and **M** RFGs were more important and had significantly higher abundance at the vertically mixed OP site (Fig. 4.2). Over the course of the study period, larger contributions of *Aulacoseira alpigena*, *A. granulata* var. *angustissima*, *Microcystis aeruginosa*, *M. flos-aquae*, and *M. wesenbergii*, of the RFGs **B** and **M** were recorded. The abundances of **B**, **N**, **P**, and **M** showed marginally significant spatial variations among MEG, DAB, and GOR sites ($p > 0.05$, Tukey HSD test), while all RFGs attained lower abundance at the macrophyte-infested WHI site. Comparatively, the abundance level of **P** was higher at OP than those at MEG, DAB, GOR, and WHI.

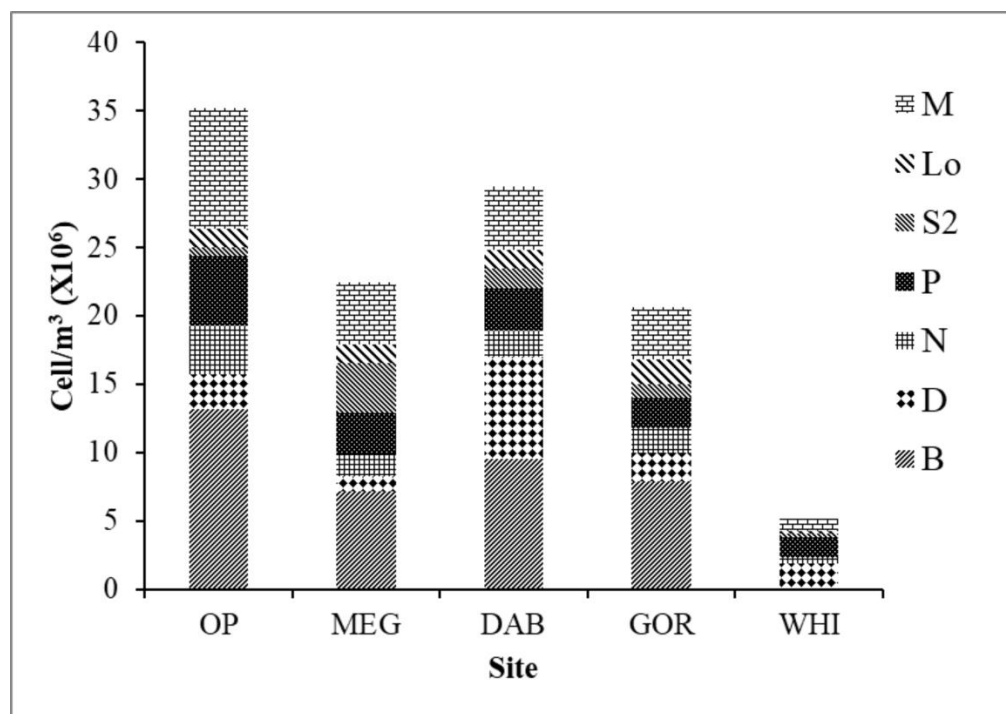


Fig. 4.3. Spatial variations in mean abundance (cells./m³) of RFGs (OP= Open water; MEG= Megech; DAB = Dablo; GOR= Gorgora; WHI = Water hyacinth infested site).

Similarly, significant temporal variations in the abundance of RFGs of phytoplankton were observed, with **M** and **B** dominating in all seasons except the rainy period (Fig. 4.4). During the pre-rainy season, higher contributions were recorded for *Dolichospermum circinale*, *Cosmarium sexangulare*, and *Pseudoanabaena acicularis*, while *Nitzschi acicularis*, *Aulacoseira agassizii*, *Closterium acutum*, and *C. limneticum* were more abundant in the dry and post-rainy seasons. Overall, the rainy season was characterized by lower contributions to total abundance of all RFGs.

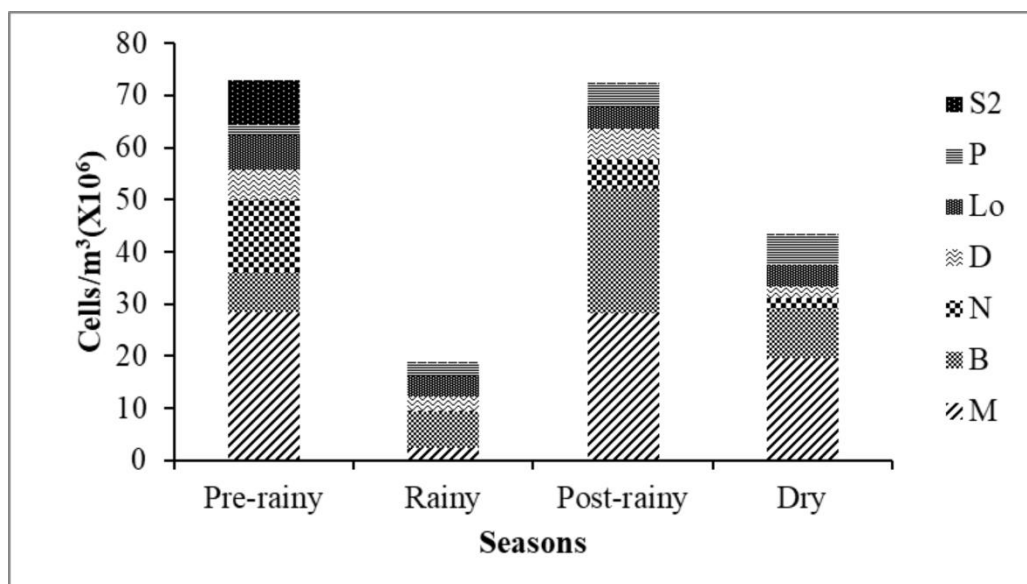


Fig. 4.4. Temporal variations in mean abundance (cells./m³) of RFGs.

4.3.3. Relationships of phytoplankton species with physico-chemical parameters

Redundancy analysis (RDA) was used to illustrate the association between phytoplankton species abundance, RFGs and physico-chemical parameters measured at various sites over the study period. RDA showed that the first two axes collectively explained 86.42 (axis 1 = 72.04% axis 2 = 14.4%) of the total variance in RFGs distribution explained by the environmental factors, revealing the significant ($p < 0.05$) correlation between water quality parameters and

temporal distributions of RFGs (Fig. 4.5). During the rainy season, the predominant RFGs such as **B**, **M**, **Lo** and **N** were highly correlated with various water quality parameters including EC, turbidity, SRP, NO_3^- , and SiO_2 as indicated by axis 1. Similarly, in the pre-rainy season, clear clustering of RFGs (**D**, **S2**, **P**, and **Lo**) with water quality parameters (ammonia and NO_3^-) was also observed, indicating a strong association. In contrast, during the post-rainy and dry seasons, the RFGs **N**, **J**, **P**, **M**, and **X1-X3** showed a strong positive association with DO and temperature towards axis-2. However, the RFGs of the post-rainy and dry seasons exhibited significant but negative correlation with turbidity, Secchi depth, and elevated EC.

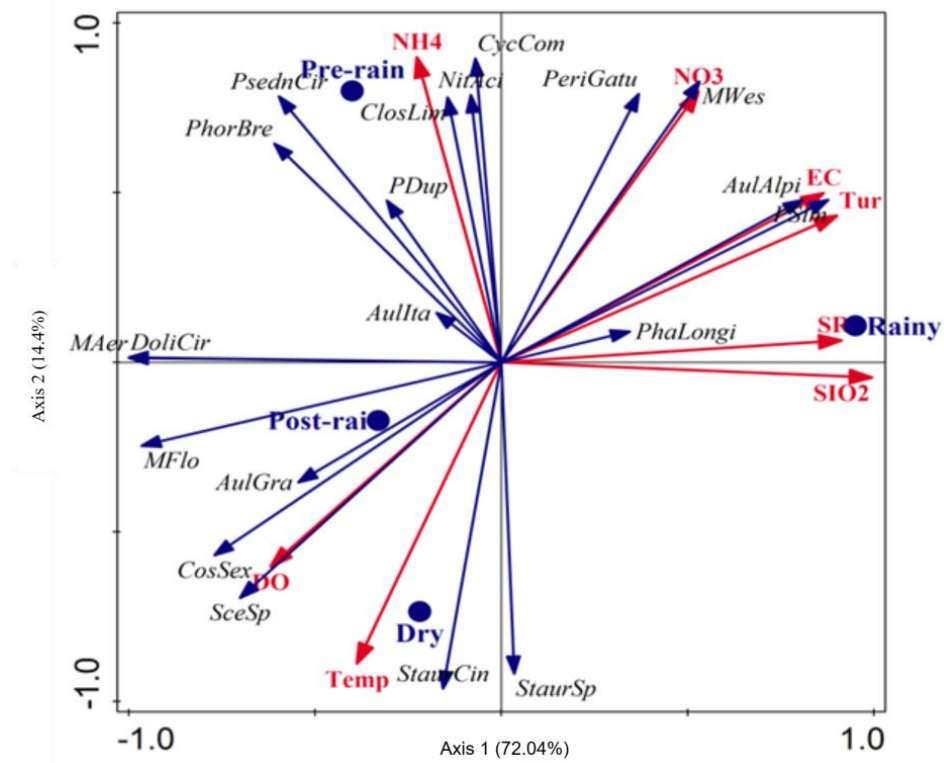


Fig. 4.5. RDA tri-plot indicating the correlation between RFGs (blue arrow) sampling seasons (blue circles), and water quality parameters (red arrows): (DO – dissolved oxygen, Temp – temperature, EC – electrical conductivity, Tur – turbidity, SRP soluble reactive phosphorus, Cyccom – *Cyclotella comate*, PeriGatu – *Peridinium gatunense*, Mwes – *Microcystis*

wesenbergii, Mflo – *M. flos-aquae*, Dolicir – *Dolichospermum circinale*, Phorbre - *Phormidium breve* Kützing, AulAlpi – *Aulacoseira alpigena*, Aulgra – *A. granulata* Phalongi – *Phacus longicauda*, StaurSP – *Staurastrum* sp. Staurcin – *S. cingulum*, Scesp – *Scenedesmus* sp., Cossex – *Cosmarium sexangulare*, , ClosLim – *Closterium limneticum*).

Spatially, the first two axes (axis 1: 63.54%; axis 2: 24.14%) collectively explained 87.68% of the variation in RFGs distribution (Fig. 4.6). The OP site (which has open mixing layer and mesotrophic epilimnion) and the GOR site (which has shallow mixed and turbid layer) were dominated by the RFGs such as **M**, **N** and **S2**, indicating a significant association ($p < 0.05$) with elevated DO, temperature, EC, SRP, and NO_3^- levels. Other RFGs, including **L**, **P**, and **N** were correlated with MEG. However, all RFGs except **B**, showed a strong negative correlation with the macrophyte-infested site.

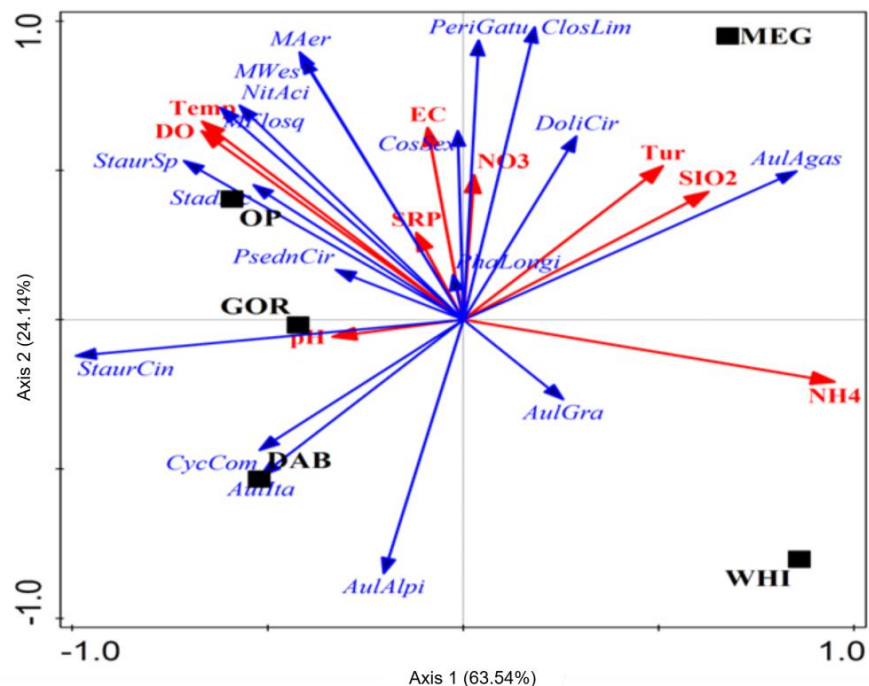


Fig. 4.6. RDA tri-plot indicating the correlation between RGFs (blue arrow), water quality parameters (red arrow), and sampling sites (black squares); OP – open water; MEG – Megech; DAB – Dablo; GOR – Gorgora; WHI – water hyacinth infested site), and water quality parameters (red arrows) (DO – dissolved oxygen, Temp – temperature, list of species is similar to that of Fig 5.5).

4.4. Discussion

Environmental parameters, that changes the water quality parameters, can have a detrimental impact on ecosystem functioning and potentially alter the abundance of phytoplankton communities (Ndah *et al.*, 2022; Bai *et al.* 2023; Ginatullina *et al.* 2023). These changes in water quality parameters, such as temperature, pH, DO, EC, turbidity, and nutrient levels, can significantly affect phytoplankton composition and structure (Gebrehiwot *et al.* 2017; Rodil *et al.*, 2024). Furthermore, when the levels of these water quality parameters surpass the established thresholds, it affects habitat quality, biodiversity, and survival of organisms (Leidonald *et al.* 2018; Mequanent *et al.* 2022).

4.4.1. Spatio-temporal variations in phytoplankton community structure

Phytoplankton communities in Lake Tana exhibited spatial and seasonal variations in local richness and abundance. These differences can be attributed to factors such as nutrient availability, local hydrological alterations, and habitat heterogeneity (Kahsay *et al.* 2022; Yongo *et al.* 2023; Liu *et al.* 2024). Rodil *et al.* (2024) exemplified the susceptibility of phytoplankton dynamics to water quality alteration. The present study revealed that the shallow, less turbid WHI site, and dominated by macrophytes, had fewer phytoplankton taxa compared to the highly turbid, shallower, and nutrient-enriched littoral sites of DAB and MEG. This could be due to the canopy cover of the macrophytes that lowered water temperature and the consequent increased

turbidity, which reduces light availability (Kahsay *et al.* 2022; Wang *et al.* 2024). In aquatic systems, water transparency (turbidity) and total solids (> 300 mg/l) are crucial factors that influence phytoplankton community assemblage (Ding *et al.* 2021; Yongo *et al.* 2023). Our findings are consistent with those of Kahsay *et al.* (2022), who also reported lower phytoplankton species and abundance in turbid and macrophyte-dominated areas. Further, the allelopathic inhibition and the harsh competition for nutrients from other macrophytes may have also affected species abundance. Additionally, lake mixing pattern along with the shallower depth of the lake could result in pronounced turbidity and cause disturbance and flushing of water column which impact phytoplankton assemblage (Bai *et al.* 2023; Liu *et al.* 2024). However, some small-sized, shade-tolerant members of Bacillariophyta and Chlorophyta were found to dominate these areas, particularly at the GOR site.

Higher phytoplankton abundance and richness were recorded for the OP and DAB sites, which are macrophyte-free, and pelagic. The average phytoplankton abundance levels recorded at the OP and GOR sites were 7.7×10^6 and 5.2×10^6 , respectively. This spatial difference in phytoplankton abundance reflects the changes in environmental parameters such as pronounced temperature, and reduced turbidity (Su *et al.* 2017; Kahsay *et al.* 2022; Bai *et al.* 2023). The observed highest phytoplankton abundance at the middle of the lake can also be due to resource availability, longer residence time and light conditions of the open water (Abonyi, *et al.* 2012; Su *et al.* 2017; Liu *et al.* 2024).

Phytoplankton abundance, richness, diversity and evenness are effective metrics of water quality (Ding *et al.* 2021). In this study, phytoplankton abundance and species richness showed distinct seasonality, with the post-rainy season exhibiting the highest levels, followed by those of the pre-rainy, dry and rainy periods (Fig. 4.4). The low levels of phytoplankton during the rainy season

could be associated with precipitation-induced runoff and the consequent turbidity, which resulted in low-light condition (Wondie and Mengistou 2017). Liu *et al.* (2024) elucidated the correlation between the rainy season and the high water level period, resulting in well-mixed and turbid conditions that hinder phytoplankton dynamics. Similarly, the recent siltation in Lake Tana emanating from the agricultural catchment can influence abundance and diversity of phytoplankton through sedimentation and nutrient adsorption (Ding *et al.* 2021; Engdaw *et al.* 2024). However, the higher phytoplankton abundance of the post-rainy season may be explained by the prevalence of favorable climatic and hydrological conditions, water level change as well as improved nutrient levels (Ding *et al.* 2021; Bai *et al.* 2023; Yongo *et al.* 2023). In the pre-rainy season, peak abundance was attained by Cyanobacteria. However, following the replenishment of the lake water at the end of the rainy season, a variety of species emerged, with *Microcystis* spp. initially dominating the phytoplankton assemblage, and then *Aulacoseira* sp. overtaking the leading role. Chlorophyta was also represented by different species, including those of *Closterium*, *Pediastrum*, *Scenedesmus*, and *Staurostrum*. Furthermore, the increased abundance observed during the post-rainy period can be attributed to the stable hydrology, warm temperature, and reduced grazer activity (Bai *et al.* 2023). The present findings are consistent with those of Wondie and Mengistou (2017), and Wu *et al.* (2024) who reported significant temporal differences in phytoplankton assemblages due to mixing pattern, and variations in physico-chemical parameters that influence physiological metabolism of algae enhancing reproduction and growth rate (Su *et al.* 2017). Therefore, this study indicated that phytoplankton diversity and abundance is significantly affected by water quality parameters. Similar findings were also reported from various Ethiopian lakes (Wondie *et al.* 2017; Wagaw *et al.* 2023; Etisa and Mebrat 2024). The ability of cyanobacteria to thrive in all seasons and environments

demonstrates their tolerance to highly-turbid and low-nutrient conditions (Wu *et al.* 2024; Suba *et al.* 2025). As a result, prolonged dominance of Cyanobacteria may lead to harmful algal blooms and the consequent production of cyanotoxins, which can negatively impact ecosystem function and pose health risk (Muluye *et al.* 2023).

Phytoplankton diversity is an important indicator of water quality and ecological health of aquatic systems. Higher diversity is typically associated with less pollution, while a decline in diversity is often correlated with severe pollution (Ding *et al.* 2021). Therefore, it is crucial to assess the long-term trends of phytoplankton assemblages for effective aquatic ecosystem monitoring (Suba *et al.* 2025). A total of 113 phytoplankton species were identified in this study, a figure which is significantly higher than those reported previously for the same lake by Wondie and Osondu (2008), Wondie and Mengistou (2017), and Kahsay *et al.* (2022). Most of the species identified in this study are commonly found in tropical and sub-tropical aquatic systems (Wondie and Mengistou 2017). The three taxonomic groups, Chlorophyta, Bacillariophyta, and Cyanophyta, dominated the phytoplankton community, collectively accounting for 94.7% of the total phytoplankton abundance. This finding is in agreement with the results of Wondie and Mengistou (2017) for Lake Tana, Gebrehiwot *et al.* (2017) for Lake Ziway, Estifanos *et al.* (2022) for Lake Hawassa, and Suba *et al.* (2025) for Lake Victoria.

Throughout the year, the major contributors from Bacillariophyta were *Aulacoseira alpigena*, *A. granulata*, *A. agassizii*, *Cyclotella radios*a, and *Nitzschia acicularis*. During the post-rainy season when the water level and nutrient concentrations reached their seasonal peak, all species were favored, but the phytoplankton community was dominated by *Microcystis aerugenosa*, *M. flos-aquae*, *M. wesenbergii*, *Dolichospermum circinale*, *Planktolyngbya limnetica*, and *Peridinium gatunense*, resulting in eutrophication of the lake. This finding is consistent with the

results of Suba *et al.* (2025), who highlighted the adaptability of these species to eutrophic systems and varying light environments.

4.4.2. Reynolds Functional groups (RFGs) of phytoplankton

RFGs are more effective in identifying stressor-induced alterations in phytoplankton assemblages. They are valid indicators for proper functioning of aquatic systems (Bai *et al.* 2023). Each phytoplankton FGs has specific environmental requirements and adaptive traits that are shared by all its members, regardless of their phylogenetic groups (Reynold *et al.* 2002; Padisák *et al.* 2009; Wang *et al.* 2024). This study observed significant differences in the abundance of RFGs between sites with macrophyte infestations and open water, as well as between rainy and post-rainy seasons. Fig. 4.3 and 4.4 showed the distinct clustering of FGs along the gradient of macrophyte coverage and open water, enhancing comprehensive overview of the community structure by drawing insights on the ecological pattern (Reynolds, 1980).

The dominance of RFGs **M**, **B**, and **S2** at the OP and GOR sites can be attributed to their adaptation to colonize standing water columns and withstand the negative impact of mixing, which disrupts their suspension (Estifanos *et al.* 2022). These RFGs are able to thrive under this condition due to the availability of dissolved nutrients, access to CO₂, and adequate solar radiation. Wang *et al.* (2024) elucidated the strong effect of environmental factors on FGs structure and taxonomic composition. Similarly, Bai *et al.* (2023) reported the predominance of Cyanobacteria as a result of increased NH₄, NO₃⁻ and SRP. Additionally, the presence of gas vesicles and mucilaginous sheaths in **M** (*Microcystis* sp), and **S2** (*Cylindrospermopsis* sp.), provide them with the competitive advantage of remaining in suspension. The dominant diatom species *Aulacoseira granulata* var. *angustissima* (RFG **B**) withstands mixing water column condition and silica depletion (Reynolds *et al.* 2002; Estifanos *et al.* 2022; Liu *et al.* 2024). It

could also be attributed to its extensive morphological variations that broaden its ecological niche. In general, Cyanobacteria overwhelmingly dominated the OP and GOR sites, followed by **B**, which can be associated to the eutrophic status of the lake. This finding corroborates the reports of Gebrehiwot *et al.* (2017) for Lake Ziway and Estifanos *et al.* (2022) for Lake Hawassa, which indicated the adaptation of Cyanobacteria to thrive in both mixed and stratified water columns.

The abundance of RFGs within the macrophyte stands of the WHI site was low and represented by a single group, **P**. In freshwater ecosystems, macrophyte-dominated sites prevent wind-induced mixing and serve as a refuge for numerous zooplankton, which are capable of grazing more than 80% of the phytoplankton (Choi *et al.* 2014; Gebrehiwot *et al.* 2017). However, in the case of Lake Tana, macrophyte cover seems to have influenced phytoplankton dynamics by causing low abundance of zooplankton due to reduced availability of food sources (Imoobe and Christopher, 2009). Thus, the dominance of **P** at the WHI site could be due to their elongated morphology, which resists grazing by small-sized rotifers (Mohammed *et al.* 2023). The lower abundance of RFGs recorded at the WHI site could also be due to competition for nutrients and CO₂ with macrophytes, which also canopy cover that reduces light penetration and diffusion of gases from the atmosphere (Estifanos *et al.* 2022).

4.4.3. Relationships of phytoplankton with physico-chemical parameters

The results of the RDA (Fig. 4.5 and 4.6) illustrate how the RFGs respond to changes in physico-chemical parameters of Lake Tana. The high percentage of variance observed in the relationship between RFGs and environmental parameters suggests that RFGs can be accurately predicted based on changes in these parameters. Therefore, the functional method can be considered as a valuable tool (Estifanos *et al.* 2022). The data reveals a strong positive correlation between RFGs

such as **B**, **M**, **Lo**, and **N** and major driving factors of phytoplankton community assemblage, such as EC, turbidity, SRP, NO_3^- , and SiO_2 . This indicates that phytoplankton dynamics in Lake Tana are primarily controlled by bottom-up (nutrients) rather than top-down (grazer) effects. This is consistent with the findings of Estifanos *et al.* (2022) in Lake Hawassa, which also depicted the insignificant role of zooplankton in shaping functional assemblages of phytoplankton. Similarly, Kruk *et al.* (2016) have also demonstrated the insignificant impact of zooplankton on determining phytoplankton community assemblage in tropical lakes. This could possibly be due to the lack of efficient grazers in the system and the frequent spawning behavior of filter-feeder fishes (e.g. *Oreochromis niloticus*), which exert less grazing on phytoplankton due to migration to breeding grounds and mouth-brooding behavior. During the post-rainy season, RFGs showed a significant negative correlation with Secchi depth and elevated EC. This could be ascribed to the pronounced turbidity of the lake, which hampers light availability, a limiting factor for phytoplankton growth.

A distinct spatial variation in species composition and abundance of phytoplankton RFGs and physico-chemical parameters was observed. The significant association between higher RFGs abundance in the open and shallow mixing layers and mesotrophic epilimnion and environmental parameters (SRP and NO_3^-) is likely due to nutrient enrichment. Conversely, a strong negative correlation was observed between RFGs abundance and the WHI site. This could be due to the allelopathy, nutrient competition, and shading effects of the macrophytes.

4.5. Conclusions and recommendation

The water quality parameters in the northern gulf of Lake Tana exhibit significant spatio-temporal variability due to multiple stressors caused by anthropogenic activities. The main taxonomic divisions of phytoplankton identified in this study were Chlorophyta, Bacillariophyta,

and Cyanophyta, while Cryptophyta, Dinophyta, and euglenophyta were found to be less prevalent. This study highlights the significant impacts of environmental factors and macrophyte stands on the composition and dynamics of the phytoplankton community. The RFGs identified in the northern gulf of Lake Tana were found to have a significant association with water quality parameters such as turbidity and other nutrients that limit algal growth. Therefore, the RFGs method can be utilized as an effective monitoring tool for assessing the ecological status of aquatic systems and their associated water quality parameters.

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CHAPTER 5

Water quality parameters influence temporal dynamics of zooplankton community structure in a tropical highland lake, Lake Tana, Ethiopia

This chapter is based on the following accepted paper:

Flipos Engdaw, Demeke Kifle, Tadesse Fetahi (2025). Water quality parameters influence temporal dynamics of zooplankton community structure in a tropical highland lake, Lake Tana, Ethiopia. *Marien Pol. Bul.* 215(2025)

Abstract

Assessing long-term trends in zooplankton community structure is essential for the monitoring of aquatic systems. However, there is paucity of information on zooplankton community structure in Lake Tana's northern gulf. Therefore, this study aimed to provide baseline information on the spatio-temporal variations in zooplankton assemblages in Lake Tana's northern gulf, focusing on water quality parameters. Data were collected from five sampling sites from May, 2023 to April, 2024. Forty-five zooplankton species belonging to three groups (17 Rotifera, 14 Copepoda, and 13 Cladocerans) were identified. Rotifers were the most species-rich group, with *Brachionus angularis*, *B. caudatus*, *B. falcatus*, *B. havanaensis*, and *Keratella tropica* being the most abundant. Significant ($p < 0.05$) spatial variations in zooplankton diversity, evenness, and dominance were recorded. Shannon index values ranged from $H' = 2.98$ for the GOR site to $H' = 3.49$ for the MEG site. Evenness values varied between 0.52 and 0.74, indicating less uniform distribution. Zooplankton taxa were rare at the water hyacinth-infested site probably due to stress from low light and reduced temperature. Copepods and rotifers showed resilience to survive in the slightly polluted water of the littoral region, while cladocerans preferred the clear

pelagic water. Zooplankton community structure showed a similar pattern across all sites, with minimum (27) and maximum (36) species richness occurring during the rainy and post-rainy seasons, respectively. Generally, the spatio-temporal variations in water quality parameters had implications for zooplankton community structure.

Key words: community, diversity, evenness, indicator, water quality, zooplankton

5.1. Introduction

Zooplankton are essential components of aquatic ecosystems, which play an indispensable structural and functional role by linking primary producers and higher trophic levels (Imoobe and Akoma, 2008; Xiong *et al.*, 2020; Beyene *et al.*, 2022; Ginatullina *et al.*, 2023; Oh *et al.*, 2023). They influence growth, survival, and community structure of numerous aquatic organisms as they rely on zooplankton diet at least at some stage of their lifetime (Balseiro *et al.*, 2023; Mohammed *et al.*, 2023). Zooplankton also play a crucial role in maintaining the health of aquatic systems by controlling eutrophication, biogeochemical cycling of nutrients (nitrogen and phosphorus) and biological self-purification process thereby elucidating the proper functioning of these ecosystems (Krylov *et al.*, 2020; Mequanent *et al.*, 2022). Zooplankton's role in energy transfer makes them a compelling choice for use as bio-indicators (Lan Smith *et al.*, 2019; Musialik-Koszarowska *et al.*, 2019; Kahsay *et al.*, 2022; Ndah *et al.*, 2022). Furthermore, their frequent-responsive nature to multiple environmental stressors, such as changes in hydrology, climate, and water pollution, make them valuable indicators of long-term ecological changes in the monitoring of aquatic systems (Xiong *et al.*, 2020; Boldrocchi *et al.*, 2023). Numerous studies have elucidated the potential application of zooplankton as an ecological and water quality indicator (Kour *et al.*, 2022; Ndah *et al.*, 2022; Rashid and Prakash, 2022; Enawgaw *et al.*, 2023).

Zooplankton community structure is influenced by a variety of factors, including season (rainy and dry periods), hydrology, anthropogenic activity, resource availability (bottom-up), predation pressure (top-down), competition, formation of new habitats, and alterations in physico-chemical parameters such as turbidity, temperature, pH, dissolved oxygen (DO) and water depth (Dejen *et al.*, 2004; Effendi *et al.*, 2016; Musialik-Koszarowska *et al.*, 2019; Krylov *et al.*, 2020; Beyene *et al.*, 2022; Kour *et al.*, 2022; Mohammed *et al.*, 2023). These factors can lead to significant changes in zooplankton composition, ultimately leading to progressive biodiversity impoverishment (Dejen *et al.*, 2004; Xiong *et al.*, 2020; Mequanent *et al.*, 2022). For example, increased soil erosion during the rainy season can cause elevated turbidity, which can limit phytoplankton production, and reduce the algal biomass available for zooplankton consumption (Dejen *et al.*, 2004; Imoobe and Christopher, 2009; Fetahi *et al.*, 2011a). Turbidity influence DO level, and aspects of trophic interactions, such as feeding behavior and vulnerability to predation (Dejen *et al.*, 2004; Beyene *et al.*, 2022). Additionally, predation pressure from zooplanktivorous fish and macroinvertebrates, along with the prominent macrophytes in the littoral zones, can substantially influence the spatial distribution of zooplankton communities (Gebrehiwot *et al.*, 2017; Krylov *et al.*, 2020).

Numerous studies have revealed that excessive nutrient loadings, pharmaceuticals, toxic heavy metals, and microplastic concentrations can have a detrimental impact on zooplankton communities. Consequently, rotifers are used as ecological indicators of freshwater toxicants such as ammonia, lead, and nanoparticles (Xiong *et al.*, 2020), while copepods are used as bio-indicators of contamination in aquatic systems, the effects of climate change, and predation pressure (Musialik-Koszarowska *et al.*, 2019).

Understanding zooplankton communities' structure, particularly their association with physico-chemical water quality parameters, is crucial for assessing the ecological status of aquatic systems (Dejen *et al.*, 2004; Ndah *et al.*, 2022; Mao *et al.*, 2023). Since they occupy a critical position in the aquatic food web, zooplankton can also serve as potential indicators of ecological integrity. Furthermore, their community structure plays a vital role in optimizing lake resource use and enhancing sustainable lake ecosystems management and monitoring (Musialik-Koszarowska *et al.*, 2019; Kahsay *et al.*, 2022). Therefore, sufficient information on spatio-temporal variations in zooplankton community structure is essential to comprehend ecological preference of zooplankton and developing a zooplankton-based water quality monitoring tool.

Several researchers have studied the ecology and community structure of zooplanktons in many Ethiopian inland water systems (Mengistou and Fernando, 1991; Immoibe and Akoma, 2008; Immoibe and Christopher, 2009; Fetahi *et al.*, 2011b; Wondie and Mengistou, 2014; Gebrehiwot *et al.*, 2017; Krylov *et al.*, 2020; Beyene *et al.*, 2022; Kahsay *et al.*, 2022; Mohammed *et al.*, 2023; Wagaw *et al.*, 2023). Most of these studies were conducted in the Rift Valley lakes of the country. However, the northern gulf of Lake Tana has been under-studied due to its remote location, accessibility and logistic constraints rendering the available information scanty.

Lake Tana, the largest lake in Ethiopia, and the third largest lake in the Nile River basin, is a vital component of the environment of the region, providing irreplaceable economic goods and services to local communities (Wondie and Mengistou, 2014; Gitaw *et al.*, 2020; Kassa *et al.*, 2021). The lake is the source of Blue Nile River, upon which millions of peoples from Ethiopia, Egypt, and Sudan depend (Engdaw *et al.*, 2024). It plays a crucial role in regulating the local climate and serves as biodiversity hotspot and home to native and unique *Labeobarbus* species (Engdaw *et al.*, 2024). However, in the last two decades, the Lake Tana ecosystem has

undergone unprecedented changes due to multiple stressors, including hydrological variations, anthropogenic activities, and climatic changes (Goshu and Aynalem, 2017; Dersseh *et al.*, 2022; Engdaw *et al.*, 2024). Like many other East African lakes, Lake Tana is currently experiencing reduced ecological function, which inevitably leads to reduction in diverse ecosystem services. This environmental degradation is related to the input of excess nutrients and sediment through erosion or agricultural malpractices, and increased water demand in the catchment area. In addition, global warming leads to increased heavy rainfall events and dry periods that generally result in eutrophication of the riparian regions and increased water level fluctuations. These changes have had dramatic effects on the entire ecosystem, particularly in the horizontal and vertical flow patterns, which caused loading of nutrients from the sediment and hence rapid algal growth and proliferation of the invasive water hyacinth. Furthermore, rapid land use/land cover changes in the catchment, urbanization and industrialization-induced waste effluent, which led to nutrient enrichment, and shoreline modification for the construction of hotels and resorts, water level fluctuation due to hydrological alterations, dam construction and pronounced water abstraction are currently threatening the water quality, biodiversity and overall ecosystem functioning. This has been documented in various studies (Goshu and Aynalem, 2017; Engdaw *et al.*, 2024). Recent studies have depicted the exacerbation of the water quality of Lake Tana by severe anthropogenic activities (Engdaw *et al.*, 2025), which may have influenced the structure, distribution and biomass of planktonic communities (Dersseh *et al.*, 2022). Despite the concerted efforts made to investigate various aspects of the lake, the emphasis has primarily been on the taxonomy and ecology of fish in the southern Bahir-Dar bay, neglecting the other northern gulfs. While there have been a few cross-sectional qualitative studies, there is lack of detailed research focusing on the relationships between zooplankton communities and physico-chemical water

quality parameters. Furthermore, the taxonomic resolution and accuracy of species identification in these studies are questionable. The effects of season and water quality parameters have also been overlooked. Therefore, it is crucial to investigate the key drivers of zooplankton community structure in the lake and how they respond to seasonal changes in water quality. Thus, this study aimed to document data on physico-chemical water quality parameters and provides baseline information on the temporal dynamics of zooplankton community in Lake Tana's northern gulf.

5.2. Materials and Methods

5.2.1. Zooplankton sampling and analysis

For identification and enumerations of zooplankton, sampling was done seasonally in daylight hours. Zooplankton samples were obtained by vertically hauling 70 μ m mesh size (42 cm opening and 58 cm long) plankton net from 6 m in the OP and 1.5 m in the GOR sites. The vertical hauling depth in other sites ranged from 0.4 to 0.8 m. For qualitative analysis of zooplankton, additional samples were also collected by vertical and horizontal towing of 30 μ m mesh size plankton net. All collected zooplankton samples were preserved with 4% formalin and transported to Addis Ababa University, a limnology laboratory.

5.2.2. Zooplankton identification and counting

Each zooplankton sample was gently mixed and 10 ml of the mixture was then decanted into a glass chamber with a gridded surface positioned beneath a compound microscope. Taking into account the size of the counting chamber and density of zooplankton in the sample, 10 ml of sub-sample was determined to be appropriate for identification and enumeration. Identification of zooplankton taxa to the lowest possible taxonomic level was done based on these keys (Defaye, 1988; Korinek, 1999; Fernando, 2003). In five randomly selected grids, at least 100 individuals of the dominating taxa were counted at a 100 $\times$ magnification. Abundance of zooplankton was computed according to (Edmondson and Winberg, 1971; De-Bernandi, 1984).

$$Ind. m^{-3} = \frac{n \times GF \times SSF}{\pi \times R^2 D} \quad \text{Equation 1}$$

Where; Ind = individual zooplankton, n = actual species counts, GF = grid factor, SSF = sub-sample factor, R = radius of the net (m), D = vertical haul depth (m), π = constant (3.14)

Species diversity of zooplankton community was determined using species richness and diversity indices, which were calculated according to the margalef's index (D) (Death, 2008) and Shannon-Weaver diversity index (H') (Shannon and Weaver, 1949), respectively.

$$H' = \sum_{i=1}^S \rho_i \times \ln \rho_i \quad \text{Equation 2}$$

Where, H' = diversity index, ρ_i = relative contribution of the i^{th} species to the total abundance of all species together, S = number of species in the population.

Species richness of zooplankton was determined using Margalef's index (D) (Death, 2008).

$$D = \frac{S - 1}{\ln N} \quad \text{Equation 3}$$

Where, D = species richness of zooplankton, S = number of species in the population and N = total number of individuals collected.

QBrachionus/Trichocerca (QB/T) index

An attempt was made to determine the trophic state of Lake Tana using the QBrachionus/Trichocerca (QB/T) ratio index (Sládeček, 1983).

$$QB/T = \frac{\text{Number of species of Brachionus}}{\text{Number of species of Trichocerca}}$$

The QB/T index is composed of 3 scales for determining the trophic state of a lake; QB/T = 1 for oligotrophic, QB/T = 1.0-2.0 for mesotrophic, and QB/T > 2 for eutrophic.

Saprobic index

The saprobic index was applied to evaluate the water quality of Lake Tana's northern gulf based on the abundance and composition of zooplankton species, with a focus on rotifers (Sládeček, 1983; Rolauffs *et al.*, 2004).

$$SI = \frac{\sum_{i=1}^n Ai \times Si \times gi}{\sum_{i=1}^n Ai \times gi}$$

Where: *i* represents a taxon, *n* is the number of taxa, *Ai* is the abundance of the taxon, *Si* is the saprobic value of the taxon, and *gi* is identification weight of the taxon (Zahradkova and Soldan, 2013). The saprobic index scale consists of four levels; oligosaprobic (lightly polluted) if SI=1, β-mesosaprobic (moderately polluted) if SI=2, α-mesosaprobic (critically polluted) if SI=3, and polysaprobic (heavily polluted) if SI=4.

5.2.3. Data analyses

The data of all parameters were summarized and analyzed using SPSS (Ver. 25). Before analysis, the data was checked for normality using Shapiro-Wilk test and for homogeneity of variances. Spatial and temporal variability in zooplankton community distribution among sampling sites and seasons were analyzed using Kruskal-Wallis ANOVA. The association between zooplankton species distribution and physico-chemical parameters was examined by Redundancy analysis (RDA) using CANOCO 4.5. To determine the appropriate ordination method, Detrended Correspondence Analysis (DCA) of zooplankton abundance and environmental data was first conducted. The gradient lengths were short (<3.0) indicating that the zooplankton abundance and environmental variables had linear relationship. Therefore, RDA with manual forward selection was chosen (Lepš and Šmilauer 2003). Pearson correlation was also used to examine the correlation of zooplankton abundance and composition with environmental parameters.

5.3. Results

5.3.1. Zooplankton species composition and diversity

Forty-five species of zooplankton belonging to three groups have been identified in water samples of the northern Lake Tana gulf (Table 5.1). Of these, 17 were Rotifers, 14 Copepods, and 13 Cladocerans. All exhibited seasonal dynamics as illustrated in Fig 5.1. Some species were sporadic, typically found in negligible quantities, and were not found in all sites during all seasons. In terms of overall zooplankton abundance, Copepods were the most abundant, accounting for 40.4%, followed by Rotifers at 31.4%, and Cladocerans at 28.2% (Fig. 5.1). Within the Copepoda, Copepodites, *Cyclopoids*, *Mesocyclops aequatorialis* and the endemic *Thermodiaptomus galebi lacustris* were highly prevalent, while species of *Eucyclops* and *Thermocyclops ethiopiensis* were the most abundant taxa. Nauplii, immature Copepodes larvae, which were not identified to the lowest taxonomic level were the highly prevalent taxa occurring frequently in the majority of samples examined. However, adult *Afroscyclops gibsoni*, and *Megacyclops viridis* were not the major contributors to the total zooplankton abundance.

Table 5.1. List of zooplankton species identified from water and net samples from Lake Tana,

Copepoda	Cladocera	Rotifera
<i>Afroscyclops gibsoni</i>	<i>Alona</i> sp.	<i>Brachionus angularis</i>
<i>Cyclopoids</i>	<i>Bosmina longirostris</i>	<i>Brachionus caudatus</i>
<i>Cyclopoid copepodites</i>	<i>Bosmina</i> sp.	<i>Brachionus diversicornis</i>
Copepodites sp.	<i>Chydrous</i> sp.	<i>Brachionus falcatus</i>
<i>Ectocyclops</i> sp.	<i>Ceriodaphnia dubia</i>	<i>Brachionus havanaensis</i>
<i>Ectocyclops rubescens</i>	<i>Ceriodaphnia quadrangula</i>	<i>Brachionus dimidiatus</i>
<i>Eucyclops</i> sp.	<i>Ceriodaphnia cornuta</i>	<i>Euchlanis dilatata</i>

<i>Megacyclops albidus</i>	<i>Daphnia</i> sp.	<i>Filinia pejleri</i>
<i>Microcyclops varicans</i>	<i>Diaphanosoma excisum</i> Sars	<i>Filinia pejleri grandis</i>
<i>Mesocyclops aequatorialis</i>	<i>Diaphanosoma</i> sp.	<i>Filinia longiseta</i>
<i>Megacyclops viridis</i>	<i>Kurzia longirostris</i>	<i>Keratella tropica</i>
<i>Thermodiaptomous galebi</i>	<i>Moina micrura</i>	<i>Keratella quadrata</i>
<i>Thermocyclop ethiopiensis</i>	<i>Moina</i> sp.	<i>Keratella cochlearis</i>
Nauplii		<i>Lecane</i> sp.
		<i>Trichocera similis</i>
		<i>Trichocerca chattoni</i>
		<i>Trichocerca longiseta</i>
		<i>Trichocerca cylinderica</i>

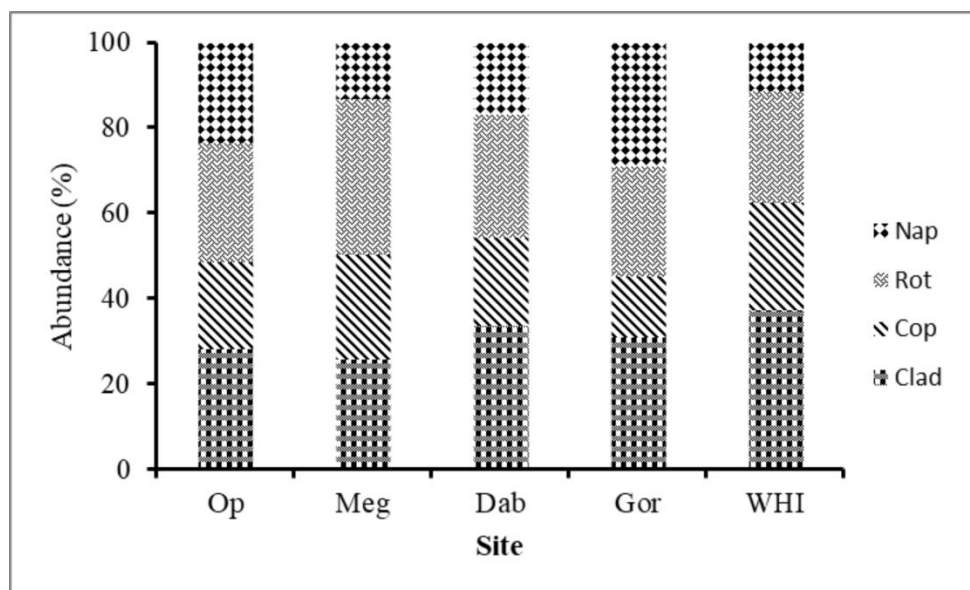


Fig 5.1. Average percentage abundance of zooplankton community in northern Lake Tana Gulf (OP= Open water; Meg= Megeche; Dab = Dablo; Gor = Gorgora; WHI = Water hyacinth infested site)

Rotifers consistently occurred in all sites and seasons, and were the most species-rich and the second most abundant group, accounting for 31.4% of the total abundance of the zooplankton community. *Brachionus angularis*, *B. caudatus*, *B. falcatus*, *B. havanaensis*, *Keratella tropica* and *Trichocera similis*, were the most frequently occurring species making up major portion of the total abundance of the zooplankton community. Less common species, such as *Lecane* sp., *Keratella quadrata*, and *Filinia longiseta*, had the lowest abundance in the zooplankton community.

Cladocerans, which were primarily represented by the members of the family Daphnidae and Bosminidae, were the third most abundant group, making up 28.2% of the total zooplankton abundance. Of the 13 species of Cladocerans, *Bosmina longirostris*, *Diaphanosoma excisum* Sars, *Alona* sp., and *Chydrous* sp., were the most prevalent groups, while, *Kurzia longirostris*, *Moina* sp., and *Leydigia* sp. were rarely found in high numbers.

There were significant ($p < 0.05$, Kruskal-Wallis ANOVA) spatial variations in zooplankton species diversity, evenness, and dominance among sampling sites. The diversity of zooplankton species was categorized as moderate to high. Among the ecological indices, the highest Shannon index of $H' = 3.49$ was recorded at the MEG site, while moderate diversity of $H' = 2.98$ was noted at the GOR site (Table 5.2). Similarly, evenness varied between 0.52 and 0.74 at all sites, indicating less uniform distribution of species. The seasonal dynamics of the zooplankton community structure showed a more or less similar pattern in all sites (Fig 5.2). The minimum species richness (S) value of 27 and maximum of 36 were recorded during the rainy and post-rainy seasons, respectively. Seasonal increase in the abundance of taxa was pronounced during the post-rainy and dry seasons in all sites. For instance, at the OP site, Cladoceran abundance increased from 279,234 in the rainy season to 483,364 Ind./m³ in the post-rainy season; Nauplii

increased from 159,838 to 410,956 Ind./m³, while Rotifera rose from 311,187 to 400,956 Ind./m³ in the rainy and post-rainy seasons, respectively (Fig 5.3).

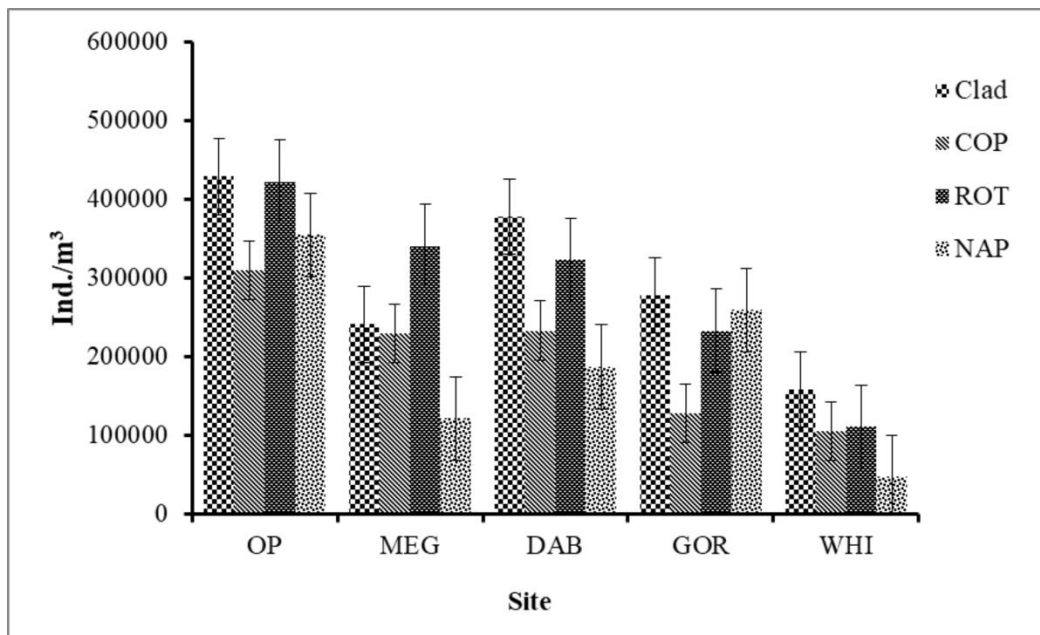


Fig. 5.2. Spatial variation of zooplankton abundance (Ind./m³) in Lake Tana northern gulf (OP= Open water; Meg= Megeche; Dab = Dablo; Gor = Gorgora; WHI = Water hyacinth infested site)

The minimum evenness was recorded during the pre-rainy and rainy seasons. Furthermore, the species diversity of zooplankton showed a moderate range between seasons. Highest species diversity, with $H' = 3.37$ and $H' = 3.33$, was recorded during the post-rainy and dry seasons, while lower diversity of $H' = 2.41$ was noted during the rainy season (Table 5.2).

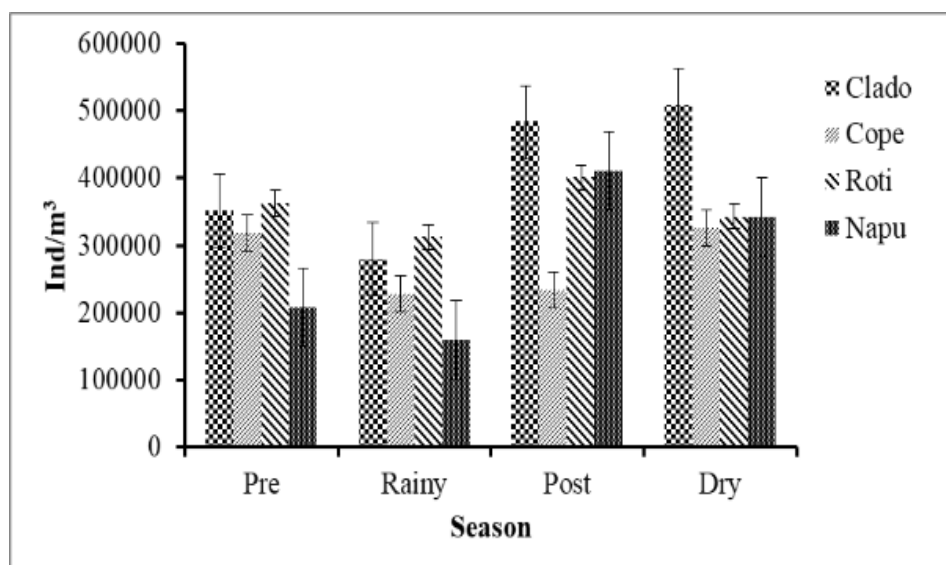


Fig. 5.3. Seasonal variation of zooplankton abundance (ind./m³) in Lake Tana northern gulf (Pre = pre-rainy; Rainy = main rainy; Post = post-rainy)

Table 5.2. Seasonal zooplankton species diversity in Lake Tana, Ethiopia

Index	Seasonal				Spatial				
	Pre-rainy	Rainy	Post-rainy	Dry	OP	MEG	DAB	GOR	WHI
Shannon_H	3.33	3.18	3.37	3.32	3.22	3.46	3.32	2.97	3.31
Simpson_1-D	0.95	0.92	0.95	0.94	0.93	0.96	0.95	0.89	0.95
Evenness_e^H/S	0.65	0.57	0.71	0.69	0.59	0.72	0.71	0.52	0.74

QB/T trophic state and saprobic index

The trophic state index of Lake Tana was determined using Rotifers as an indicator. The mean spatial ratio of *Brachionus* to *Trichocerca* (QB/T) was 3.3, indicating eutrophic condition in the lake. The highest QB/T index value of 6.4 was recorded at DAB, while the lowest was 1.8 at the MEG site. The mean temporal trophic state index of the lake ranged from 2.4 to 2.8, further confirming its eutrophic condition. To evaluate the water quality of Lake Tana, the saprobic

index, which measures state of organic pollution, was calculated. The mean saprobic index of the lake ranged from 1 to 1.5, indicating a β -mesosaprobic or moderately polluted status.

5.3.2. Relationship between water quality parameters with zooplankton and spatio-temporal abundance

The correlation between zooplankton species abundance and water quality parameters at various sites and seasons was examined using redundancy analysis (RDA). RDA ordination revealed that the first two canonical axes explained 72.6% and 17.7% of the total variance in the zooplankton species density, indicating a strong relationship between water quality and seasonal zooplankton distribution (Fig. 5.4). During the pre-rainy season, diverse zooplankton species (including Nauplii, *K. tropica*, *B. longirostris*, *Bosmina* sp., and *C. copepodites*) showed a strong positive association with various water quality parameters including DO and temperature, as indicated by axis 1. In contrast, during the dry season, members of Cladocera such as *M. micrura*, and *D. excisum* became dominant and were strongly and directly correlated with temperature. Conversely, during the rainy season, different species of Copepoda and Rotifera (*A. gibsoni*, Cyclopoids, *F. pejeleris*) were dominant and negatively correlated with temperature. *B. havanaensis*, *K. quadrata*, and *B. caudatus* which were dominant during the pre-rainy season were directly and strongly correlated with pH. With the lowest population size during the rainy season in the sub-littoral zone and with the highest population size of the dry season in the pelagic zone, cladocerans showed conspicuous seasonality and habitat preference.

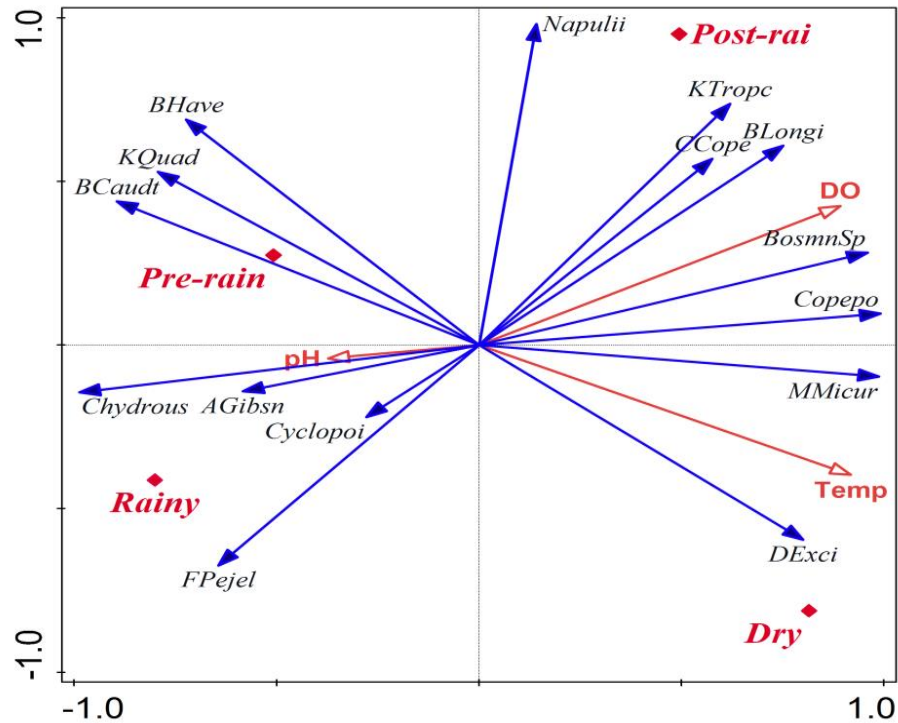


Fig. 5.4. RDA tri-plot indicating the correlation between zooplankton species (blue arrow) sampling seasons (red diamond), and water quality parameters (red arrows) (DO – dissolved oxygen, Temp – temperature; DExci – *Diphanosoma excisum*; MMicur – *Moina micrura*; Copepo – Copepodites; Bosmnsp – *Bosmina* sp.; CCopec – *Cyclopoid copepodites*; BLongi – *Bosmina longirostris*; KQuad – *Keratella quadrata*; KTropc – *K. tropica*; Bhave – *Brachionus havanaensis*; BCaudt – *B. caudatus*; AGibsn – *Afrocylops gibsoni*; Cyclopoi – *Cyclopoids copepodites*; FPejel – *Filinia pejleri*)

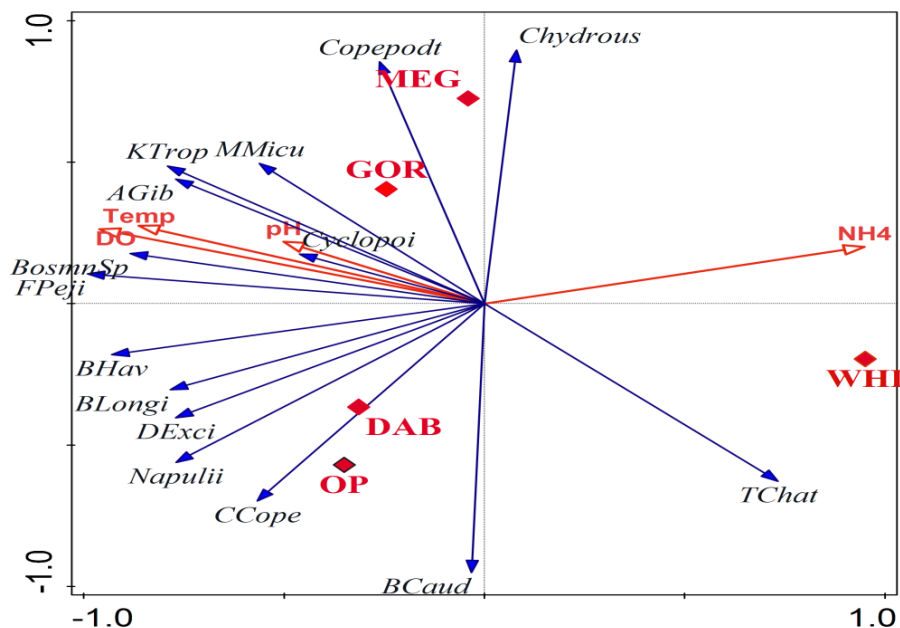


Fig. 5.5. RDA tri-plot indicating the correlation between zooplankton species (blue arrow) sampling sites (red diamond), and water quality parameters (red arrows) (DO – dissolved oxygen, Temp – temperature; abbreviations are similar with Fig. 5.4)

Furthermore, there was spatial variation in both the species richness and abundance of zooplankton taxa among the sampling sites. The abundance of all zooplankton species was significantly lower at WHI site compared to those of DAB and MEG sites ($p < 0.05$), while at the OP and GOR sites species richness was significantly higher ($p < 0.001$) than those of other sites. The RDA ordination (Fig. 5.5) also demonstrated a strong positive and significant correlation between zooplankton species abundance and water quality parameters, with *C. copepodites*, Nauplii, *D. excisum*, *B. longirostris*, and *B. havanaensis* dominating the OP and DAB sites. Similarly, *K. tropica*, Copepodites, *A. gibsoni* and *Chydrous* showed a strong association with DO, temperature and pH at GOR and MEG sites, but negatively correlated with ammonium.

5.4. Discussion

Environmental contaminants, particularly changes in water quality parameters, can have a profound impact on biological processes at various levels within an ecosystem. This can potentially alter taxonomic communities and abundance through direct cause and effect relationships (Ndah *et al.*, 2022; Mequanent *et al.*, 2022; Ginatullina *et al.*, 2023). Changes in water quality parameters, such as temperature, DO, EC, turbidity, and nutrient levels, have the potential to greatly influence the community structure of aquatic organisms (Kour, *et al.*, 2022). For instance, when concentrations of these water quality parameters exceed the established standards, it can directly impact habitat quality, biodiversity, reproduction, and survival of organisms (Leidonald *et al.*, 2018; Mequanent *et al.*, 2022).

5.4.1. Zooplankton species composition and diversity

Assessing the long-term trends and conditions of zooplankton assemblages and taxonomic structure is essential for the monitoring of aquatic systems (Dejen *et al.*, 2004). In this study, a total of 45 zooplankton species were identified, a figure which is higher than those documented in previous reports from various Ethiopian water bodies (Imoobe and Akoma, 2008; Dejen *et al.*, 2004; Fetahi *et al.*, 2011b; Wondie and Mengistou, 2014; Gebrehiwot *et al.*, 2017; Mequanent *et al.*, 2022; Mohammed *et al.*, 2023), although they are still lower than that (57) reported by Kahsay *et al.* (2022). In accordance with previous studies (Dejen *et al.*, 2004; Imoobe and Akoma, 2008, Imoobe and Christopher, 2009; Kahsay *et al.*, 2022), the zooplankton taxa identified in this study consisted mainly of tropical and few temperate species. These co-occurrence of temperate and tropical species is likely due to the lower water temperature of the lake (20.3-25.1 °C), which is a result of its higher elevation (> 1800 m above sea level).

In terms of zooplankton species richness, rotifers consistently dominated tropical and sub-tropical aquatic systems, followed by copepods and cladocerans (Krylov *et al.*, 2020; Mohammed *et al.*, 2023). The abundance of large-sized cladocerans, such as *D. tropicum*, *M. micrura*, *Moina* sp., and *C. dubia*, was lower mainly due to size-based selective grazing by zooplanktivorous fish species and mixing events (Fetahi *et al.*, 2011b; Vijverberg *et al.*, 2014; Enawugaw *et al.*, 2023). This noticeable impact of predation on large-sized zooplankton was previously explored in the study of Fetahi *et al.* (2011b), which reported a decline in the abundance of large-sized cladocerans after the introduction of Tilapia in Lake Hayq. Additionally, the decreased abundance of cladocerans in this study can be attributed to increased turbidity and decreased temperature. Vijverberg *et al.* (2014) and Beyene *et al.* (2022) have also discussed the vulnerability of various zooplankton taxa, particularly cladocerans, to turbidity. This is due to the natural inability of cladocerans to separate their food items from other non-food particles. Furthermore, dissolved and suspended silt (clay) particles can have a detrimental effect on cladocerans by obstructing their feeding apparatus (Enawugaw *et al.*, 2023). The lower abundance of cladocerans observed in this study is consistent with previous reports (Fetahi *et al.*, 2011b; Vijverberg *et al.*, 2014; Enawugaw *et al.*, 2023; Mohammed *et al.*, 2023). In contrast, Dejen *et al.* (2004) reported an increase in the diversity of cladocerans due to their high dispersal rate, which is associated with the production of resting eggs (ephippia) that can survive unfavorable conditions.

In the northern gulf of Lake Tana, Copepods were found to be the most abundant group, followed by Rotifers. This trend has also been observed in other East-African aquatic systems, such as Lake Hawassa (Mengistou and Fernando, 1991; Beyene *et al.*, 2022), Lake Hayq (Fetahi *et al.*, 2011), Lake Ardibo (Mohammed *et al.*, 2023), and Lake Victoria (Nagupula *et al.*, 2010),

and the present study lake, Lake Tana (Dejen *et al.*, 2004). The dominant species in Lake Tana's zooplankton community were Cyclopoids, *C. copepodites*, and *T. ethiopiensis*. This is consistent with the results of previous studies made by Wondie and Mengistou, (2006), Fetahi *et al.* (2011b), and Kahsay *et al.* (2022), which also reported the dominance of copepods in aquatic ecosystems they investigated. The widespread distributions of copepods can be attributed to their diverse feeding habits and distinct body morphology that allows them to prey with the least energy expenditure (Mohammed *et al.*, 2023).

Furthermore, immature small-sized Nauplii which are known to devour phytoplankton, dominated the Copepoda community throughout the sampling periods. This may be because of their moderate size, which allows them to escape large-sized predators and make them less preferred diets.

5.4.2. Effect of water quality on zooplankton community structure

This study revealed notably significant ($p < 0.05$) spatial and temporal disparity in zooplankton species richness and abundance among sampling sites and seasons. This could be attributed to changes in water quality and other related variables (Dejen *et al.*, 2004; Sousa *et al.*, 2008; Beyene *et al.*, 2022). Mequanent *et al.* (2022) and Musialik-Koszarowska *et al.* (2019) also indicated that zooplankton diversity can rapidly change due to climatic conditions and their short life span of days to weeks. There was a significant difference in zooplankton abundance ($p < 0.05$) between the WHI and other littoral sites of DAB and MEG, while OP and GOR sites had relatively higher zooplankton abundance. Ginatullina *et al.* (2023) also disclosed the strong positive association of zooplankton abundance with depth. Littoral areas, such as DAB, GOR, and MEG, were dominated by small-sized rotifers and cladocerans. The observed increase in zooplankton dominance in the littoral area is likely due to the availability of food resources and a

decline in the grazing pressure from visual predator pressure (Imoobe and Chirstopher, 2009; Mohammed *et al.*, 2023). In contrast, numerous studies have reported lower zooplankton abundance and species richness in the littoral areas, due to elevated siltation, and turbidity, which can hinder the feeding potentials of zooplankton (Fetahi *et al.*, 2011b; Beyene *et al.*, 2022). Furthermore, the decline in rotifers abundance in the pelagic area could be attributed to Copepoda and zooplanktivorous fish predation and adverse mixing conditions (Imoobe and Akoma, 2008; Gebrehiwot *et al.*, 2017). Dejen *et al.* (2004) reported the substantial impact of predation by *Oreochromis niloticus* and *Clarias gariepinus* on zooplankton abundance. They also indicated the strong negative correlation between turbidity, resulting from precipitation-associated runoff, and zooplankton assemblage in Lake Tana. The enormous inflows from the major tributaries passing through the agricultural lands and streams from the catchment substantially increased turbidity during the rainy season, resulting in changes in species composition (Mequanent *et al.*, 2022; Mohammed *et al.*, 2023).

The results of the RDA (Fig. 5.4 and Fig. 5.5) show how the zooplankton assemblage responded to changes in water quality of the lake. The data revealed a strong positive correlation between DO levels and temperature with zooplankton species richness. This is consistent with previous studies by Dejen *et al.* (2004) and Beyene *et al.* (2022), which argued that DO plays a critical steering role in influencing zooplankton populations, particularly at its lower concentrations where it can reduce their survival. Despite the tolerance of some zooplankton taxa such as *Thermocyclops* sp. and some cladocerans to survive in low oxygen (3 mg/l) condition, massive death is commonly observed when the DO level drops (Fetahi *et al.*, 2011b).

Distinct seasonal variations in species composition and abundance of zooplankton were observed in the northern gulf of Lake Tana (Fig. 5.4 and Fig. 5.5). During the post-rainy season (October –

December), there was a considerable increase in species diversity and richness, with the decreasing order of Cladocerans > Nauplii > Rotifers > Copepods. This coincided with the highest concentration of chlorophyll-*a* recorded for the same study site during the present study period, indicating a positive correlation between algal biomass and zooplankton abundance (Beyene *et al.*, 2022; Mohammed *et al.*, 2023). The dry season (January-March) was also characterized by higher zooplankton abundance and richness, with the decreasing order of Cladocerans > Rotifers = Nauplii > Copepods (Fig. 5.3). This could be attributed to factors such as increased water temperature that improved metabolism, algal biomass, availability of sufficient food, optimal climatic conditions, reduced nutrient levels, and higher water transparency (Musialik-Koszarowska *et al.*, 2019; Beyene *et al.*, 2022; Mohammed *et al.*, 2023). There was a strong positive correlation between temperature and DO during both post-rainy and dry seasons, which resulted in higher zooplankton abundance and richness. On the other hand, pH was positively associated with the pre-rainy and rainy seasons, during which lower abundance and richness were observed. The decline in zooplankton abundance during these seasons may have resulted from elevated pH levels (as most zooplankton are highly sensitive to slight fluctuations), inadequate diet, and reduced water clarity (Gebrehiwot *et al.*, 2017). Dejen *et al.* (2004) also reported the direct detrimental impact of inadequate food resources, which lead to reduced zooplankton fecundity.

It is evident that certain zooplankton taxa have a species specific habitat preference, with the highest density occurring in the littoral and sub-littoral regions where water clarity is improved (Dejen *et al.*, 2004). However, our study revealed that several zooplankton taxa were more abundant in the pelagic (OP) site than in the littoral site with macrophytes (WHI). This substantial difference may be explained by the difference in sampling sites as the littoral site is

dominated by the invasive macrophyte water hyacinth possessing fibrous roots and leaves that reduce water transparency (light penetration), leading to lower temperature. Similarly, Gebrehiwot *et al.* (2017) argued that zooplankton in Lake Ziway had preference for habitats dominated by the macrophytes *Typha* and *Phragmites*. This is likely due to the attachment and feeding opportunities provided by these plants for the non-active small epiphytic zooplankton such as *Alona* and *Chydrous* sp. Large cladocerans, like *Daphnia*, are known to avoid the littoral region, possibly due to the presence of macro-invertebrates and fish predators (Moreira, *et al.*, 2016). However, our results showed a higher density of cladocerans in the pelagic (OP) site, which could be due to drifting by wind-induced wave action. Zooplanktons particularly, Rotifers are crucial ecological indicators. The eutrophic state recorded using QB/T ratio is similar with the trophic state determined using total phosphorus (TP), Secchi depth (SD), and Chlorophyll-a (Sládeček, 1983; Engdaw *et al.*, 2025).

5.5. Conclusions and recommendation

Water quality parameters determined for the northern gulf of Lake Tana showed significant spatio-temporal variability due to severe anthropogenic stressors. Zooplankton species identified in this study consisted mainly of tropical species, and belonged to copepods, rotifers and cladocerans indicating high biodiversity. Rotifers consistently dominated the Lake Tana system while large-sized cladocerans had lower abundance mainly due to size-based selective grazing by zooplanktivorous species and mixing events. The abundance of all zooplankton species was significantly lower at the macrophyte infested sites. Water quality parameters such as turbidity and ammonium showed strong negative correlation with zooplankton abundance, while their positive association with DO, temperature and pH was observed.

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CHAPTER SIX

6. GENERAL CONCLUSIONS AND RECOMMENDATION

6.1. Conclusions

This study evaluated the water quality, and plankton community structure in the northern gulf of Lake Tana and the LU/LC dynamics in its catchment. The study attempted to investigate the effects of natural and anthropogenic drivers on the water quality and hence on the plankton community composition.

The results of this study indicated significant changes in LU/LC in the northern catchment of Lake Tana between 1993 and 2022, which were strongly linked to water quality. The major drivers of LU/LC changes, such as rapid population growth, urbanization, settlement, industrialization, and deforestation, have impacted water quality by increasing point and diffuse sources of pollutants emanating from households, croplands, settlements, and other external sources. However, the recent green legacy campaign has significantly changed the land cover of the catchment, with an increase in forest and grassland cover at the expense of croplands.

Most of the physico-chemical parameters measured in the northern gulf of the lake showed significant spatio-temporal variations due to multiple stressors caused by anthropogenic activities. Industrial and municipal effluents, as well as surplus agrochemical inputs, are believed to be the major causes of nutrient enrichment in Lake Tana.

The lake's trophic status has shifted from oligotrophic to eutrophic, indicating temporal variability. Despite the spatio-temporal variability, the mean nutrient concentrations in Lake Tana exceeded the USEPA threshold for potentially causing eutrophication.

The zooplankton species identified in this study consisted mainly of tropical species, including copepods, rotifers, and cladocerans indicating high biodiversity. Rotifers were consistently dominant in the Lake Tana system, while large-sized cladocerans had lower abundance, mainly due to size-based selective grazing by zooplanktivorous species and mixing events. The abundance of all zooplankton species was significantly lower at sites infested with macrophytes. Water quality parameters such as turbidity and ammonium showed strong negative correlation with zooplankton abundance, while a positive association with DO, temperature and pH was observed.

The main taxonomic divisions of phytoplankton identified in this study were Chlorophyta, Bacillariophyta, and Cyanophyta, while Cryptophyta, Dinophyta, and Euglenophyta were poorly represented in the phytoplankton communities. This study has demonstrated the significant impacts of environmental factors and macrophyte stands on the composition and dynamics of the phytoplankton community. The RFGs identified in the northern gulf of Lake Tana were found to have significant associations with water quality parameters such as turbidity and nutrients that limit algal growth. Therefore, the RFGs method can be used as an effective monitoring tool for assessing the ecological status of aquatic systems and their associated water quality parameters.

6.2. Recommendations

Based on the findings of the present study, the lake has undergone change in its physico-chemical water quality parameters. Specifically, there has been significant increase in the concentrations of algal growth limiting nutrients compared to previous studies. To address this issue, it is recommended to implement participatory integrated watershed management techniques such as buffer zone delineation, shoreline protection, ploughing along the contour, terracing, modernized agronomic activities, and strengthening the forestation program. These measures will not only improve the water quality, but also benefit the ecology and climate of the catchment area.

The study also found that LU/LC changes and anthropogenic activities, such as waste disposal and excessive fertilizer inputs, have contributed to the deterioration of water quality. To address this, it is important to raise awareness among local communities about better farming techniques and the optimum amount of fertilizer to be used. Proper management of solid and liquid wastes is also crucial, and the introduction of constructed wetlands near hotels and lodges around the lake, as well as controlling sewer systems and installing waste treatment ponds in cities within the catchment are mandatory.

The study also revealed the impact of the macrophyte water hyacinth, on the plankton community structure. It is recommended that long-lasting strategy for eradicating water hyacinth be sought-for and implemented before it covers the entire lake.

Apart from the present taxonomic species identification conducted in this study, further holistic research on eDNA, the level of Cyanotoxins, toxic heavy metals, and trophic interactions in the lake is highly recommended.

To avert future ecological imbalances in the lake and maintain its ecological health and ensure sustainable use, continuous regular monitoring and management practice should be put in place.

APPENDIX

Appendix 1. List of Phytoplankton identified in the Northern gulf of Lake Tana

Bacillariophyta	Chlorophyta	Cyanophyta
<i>Amphora ovalis</i>	<i>Actinastrum</i>	<i>Dolichospermum circinale</i>
<i>Asterionella formosa</i>	<i>Ankistrodesmus acicularis</i>	<i>Dolichospermum flos-aquae</i>
<i>Aulacoseira tenella</i>	<i>Ankistrodesmus falcatus</i>	<i>Dolichospermum solitarium</i>
<i>Aulacoseira ambigua</i>	<i>Arthrodesmus incus</i> var. <i>indentatus</i>	<i>Dolichospermum spiroides</i> <i>kelb</i>
<i>Aulacoseira distans</i>	<i>Botryococcus</i> Sp	<i>Chroococcus dispersus</i>
<i>Aulacoseira distans</i> var. <i>alpigena</i>	<i>Closterium acutum</i>	<i>Chreococcus turgidus</i>
<i>Aulacoseira granulata</i> var. <i>angustissima</i>	<i>Closterium diana</i>	<i>Chroccocus limneticus</i>
<i>Aulacoseira granulata</i>	<i>Closterium limneticum</i>	<i>Coelosphaerium</i>
<i>Aulacoseira italica</i>	<i>Closterium</i> Sp1	<i>Cylindrospermopsis</i> Sp
<i>Aulacoseira varians</i>	<i>Coelastrum</i> sp	<i>Eucapsis minuta</i>
<i>Cyclotella atomus</i>	<i>Cosmarium</i> sp	<i>Lyngbya contorta</i>
<i>Cyclotella comata</i>	<i>Cosmarium circulare</i>	<i>Lyngbya limnetic</i>
<i>Cyclotella melosiroides</i>	<i>Cosmarium sexangulare</i>	<i>Merismopedia</i>
<i>Cyclotella radiosa</i>	<i>Crucigenia quadrata</i>	<i>Microcystis aeruginosa</i>
<i>Cymbella</i> Sp	<i>Kirchneriella</i>	<i>Microcystis flos-aquae</i>
<i>Diatoma vulgare</i>	<i>Korshikoviella</i>	<i>Microcystis incerta</i>
<i>Eunotia pectinalis</i>	<i>Monoraphidium contortum</i>	<i>Microcystis viridis</i>
<i>Fragilaria capucina</i>	<i>Oocystis</i> sp	<i>Nostock</i>
<i>Gomphonema italicum</i>	<i>Pediastrum simplex</i>	<i>Oscillatoria limnetica</i>
<i>Gomphonema</i> SP	<i>Pediastrum duplex</i>	<i>Oscillatoria rubescens</i>
<i>Gyrosigma attenuatum</i>	<i>Scenedesmus acuminatus</i>	<i>Planktolyngbya</i> sp.
<i>Navicula cryptocephala</i>	<i>Scenedesmus arcuatus</i>	<i>Planktolyngbya subtilis</i>
<i>Navicula cuspidata</i>	<i>Scenedesmus quadricauda</i>	<i>Aphanizomenon flos-aquae</i>
<i>Navicula pupula</i>	<i>Scenedesmus</i> Sp(denticularis)	<i>Raphidiopsis</i>
<i>Navicula radiosa</i>	<i>Schroederia setigera</i>	
<i>Neidium iridis</i>	<i>Sphaerocystis schroetri</i>	
<i>Nitzschia acicularis</i>	<i>Spirogyra</i>	
<i>Nitzschia linearis</i>	<i>Staurastrum cingulum</i>	
<i>Nitzschia palea</i>	<i>Staurastrum gracile</i>	
<i>Nitzschia reversa</i>	<i>Staurastrum leptocladum</i>	
<i>Nitzschia sigmaidea</i>	<i>Staurastrum leptocladum</i>	
<i>Pinnularia pusilla</i>	<i>Staurastrum sebaldi</i>	
<i>Pinnularia</i> Sp	<i>Staurastrum</i> sp1	
<i>Surirella angustata</i>	<i>Staurastrum</i> sp2	
<i>Surirella linearis</i>	<i>Staurodesmus convergens</i>	

<i>Surirella ovata</i>	<i>Staurodesmus indentatus</i>	
<i>Surirella robusta splendida</i>	<i>Staurodesmus triangularis</i> var. <i>limneticus</i>	
<i>Surirella tenera</i>	<i>Staurodesmus megacanthus</i>	
<i>Synedra acus</i>	<i>Staurodesmus triangular</i>	
<i>Synedra ulna</i>	<i>Tetraedron arthrodesmiforme</i>	
<i>Tabellaria flocculosa</i>	<i>Tetrastrum</i> Sp	
	<i>Volvox</i> Sp	
Euglenoids	Dinoflagelates	Cryptophyta
<i>Euglena</i> sp	<i>Peridinium gatunense</i>	<i>Cryptomonas</i>
<i>Euglena viridis</i>		
<i>Peranema trichophorum</i>		
<i>Phacus longicauda</i>		