ADDIS ABABA UNIVERSITY

COLLEGE OF NATURAL AND COMPUTATIONAL SCIENCES

DEPARTMENT OF ZOOLOGICAL SCIENCES

Water Quality and phytoplankton community structure in the Southern Gulf of Lake Tana, Ethiopia

by

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List of Abbreviations

Sp. Cond. (K_{25})	Specific conductance
TDS	Total dissolved solid
Sal.	Salinity
DO	Dissolved oxygen
TSS	Total suspended solid
Turb.	Turbidity
SRP	Soluble reactive phosphorus
TP	Total phosphorus
Chl-a	Chlorophyll-a
K. Georgis	Kidus Georgis
WHO	World Health Organization
FAOUN	Food and Agriculture Organization of the United Nations
SD	Standard deviation

ABSTRACT

Water quality and phytoplankton community structure at one offshore (Zege) and three near-shore (Hospital, Resort and K. George) sites in the Southern Gulf of Lake Tana were investigated from Dec. 2015 to Mar. 2016. Surface water temperatures of all sampling sites were within the range of variation reported for most tropical water bodies (20-30°C). Considering the shallowness (mean depth < 10 m) of the sampling sites and their exposure to southerly winds, frequent and complete mixing seems highly likely in the Southern Gulf of Lake Tana. Mean Secchi depth(cm), which was largely a function of total suspended solids, was significantly (at $p < 0.05$) smaller at the relatively deep open water site (Zege, ≈ 83) than at the near-shore sites (mean: 130). The mean levels of pH (<7.3), TDS (< 110 mg L^{-1}), K_{25} (< 160 µS cm^{-1}) and salinity (<0.1 g L^{-1}) showed that the Southern Gulf was a very dilute freshwater with near-neutral pH. Surface water DO (mg L^{-1}), which was primarily a function of temperature and phytoplankton biomass, varied from 4.39 of the K. Georgis site to 6.82 of the Zege site. Mean levels of nitrate varied between 0.875 and 1.257 mg L^{-1}, while those of ammonia ranged from 0.06 of the Zege site to 0.216 mg L^{-1} of the Hospital site. Mean SRP (in mg L^{-1}), which was slightly smaller than TP, ranged from 0.237 of the Hospital site to 0.41 of the Zege site. Although it declined to below or slightly above the level regarded as limiting to diatom growth (<0.3 mg L^{-1}) in the second half of February, silica (mg L^{-1}) was generally at concentrations typical of tropical freshwaters (>10 mg L^{-1}). The phytoplankton communities of all sampling sites were dominated, both in terms of species richness and abundance, by three alga groups, Cyanobacteria (blue-green algae), Bacillariophyceae (diatoms) and Chlorophyceae (green algae). Chlorophyceae was the most species-rich taxonomic group, followed by Bacillariophyceae. *Microcystis flos-aquae, Aulacoseira granulata, and Oedogonium sp.* were quantitatively the most important constituent species of blue-green algae, diatoms and green algae, respectively. The dominance by cyanobacteria constituted primarily by species of the genus *Microcystis*, which may be attributed to turbidity and nutrient availability, is regarded as a sign of eutrophic conditions and poor ecological status of water bodies. Mean Chlorophyll-a biomass (µg L^{-1}) was very low (2.90), with individual observations varying from 0.31 of the K. George site to 8.40 of the Zege site. Cyanobacterial dominance represents a threat to public health, aquatic and terrestrial life. A continuous monitoring program is, therefore, necessary to ensure the protection of public health, aquatic and terrestrial life.

Keywords: Cyanobacteria, nutrient availability, phytoplankton abundance, phytoplankton biomass, turbidity, water quality

1. INTRODUCTION

1.1. BACKGROUND AND JUSTIFICATION

Water is vital to the existence of all living organisms, but this valued resource is increasingly being threatened as human populations grow and demand more water of high quality for domestic purposes and economic activities. The quality of any body of surface or ground water is a function of natural influences and/or human activities. Without human influences, water quality would be determined by weathering of bedrocks, atmospheric processes of evapotranspiration and deposition of dust and salt by wind, natural leaching of organic matter and nutrients from soil, hydrological factors that lead to runoff, and biological processes within the aquatic environment that can alter the physical and chemical properties of water. As a result, water in the natural environment contains many dissolved substances and particulate materials. Dissolved salts and minerals are necessary components of good quality water as they help maintain the health of organisms that rely on the ecosystem under consideration (Wetzel, 2001, Friedl *et al.*, 2004). Water may also contain substances that are harmful to life including metals such as mercury, lead and cadmium, pesticides, organic toxins and radioactive contaminants. Water from natural sources almost always contains living organisms that are integral components of the biogeochemical cycles in aquatic ecosystems (Geneviève and James, 2008).

Water abstraction for domestic use, irrigation and power generation have resulted in considerable reduction in its quantity and consequent shrinkage of lakes and reservoirs. Such agricultural practices as application of fertilizers and pesticides and industrial activities can lead to deterioration in its quality that may impact not only the aquatic ecosystem (i.e., the assemblage of organisms living and interacting within an aquatic environment), but also the availability of safe water for human consumption. It is now generally accepted that aquatic environments cannot be perceived simply as holding tanks that supply water for human activities. Rather, these environments are complex matrices that require

careful use to ensure sustainable ecosystem functioning well into the future. Moreover, the management of aquatic environments requires an understanding of the important linkages between ecosystem properties and the way in which human activities can alter the interplay between the physical, chemical and biological processes that drive ecosystem functioning (Reynolds, 2006). Long-term uncontrolled use of waste water may lead to a buildup of soil salinity, accumulation of toxic chemicals and reduction of soil permeability, and pollution of surface and ground water (FAOUN, 1999).

Providing safe and adequate water to people around the world, and promoting sustainable use of water resources are fundamental objectives of the Millennium Development Goals (Geneviève and James, 2008). The international community has recognized the important links between ecosystem and human health and well-being, particularly as human populations expand and place ever greater pressures on natural environments. However, the ability to properly track progress toward minimizing impacts on natural environments and improving access of humans to safe water depends on the availability of data that document trends in both space and time (Höglander*et al.*, 2013). The availability of water and its physical, chemical, and biological quality affect the ability of aquatic environments to sustain healthy ecosystems: as water quality and quantity are altered, organisms are often adversely affected and ecosystem services may be lost (Reynolds, 2006). Such anthropogenic activities as application of fertilizers on agricultural lands, sewage discharge from a hospital, and prison, hotel constructions, boat transportation, and sanitation are the main human threats to the southern gulf of Lake Tana (Dilnessa Gashaye, 2016).

Eutrophication and the consequent formation of algal blooms have also been observed in the near-shore regions of Lake Tana (Tewodros Taffese, *et al.*, 2014). The degradation of physical and chemical water quality due to human influences is often gradual, and subtle adaptations of aquatic ecosystems to these changes may not always be readily detected until a dramatic shift in ecosystem condition occurs (Geneviève and James, 2008). Phytoplankton (planktonic or suspended

algae), one of the major biological components of aquatic ecosystems, play important roles in water bodies, not only as primary producers and hence as the base of food chains, but also as one of the dependable indicators of lake fertility and environmental changes. Understanding the chemical and physical factors controlling population dynamics of phytoplankton is essential for the understanding of how human activities affect water quality of freshwater ecosystems (Reynolds2006 andLevinton, 2013). Algal composition, diversity, abundance and biomass often measured as chlorophyll-a in aquatic ecosystems are primarily controlled by the light transmission properties of the waters (turbidity) and availability of the nutrients nitrogen and phosphorus (Jensen *et al.*, 1994 and Höglander *et al.*, 2013). Planktonic algae, owing to their generally high reproduction rates and very short life cycles, are valuable indicators of short-term changes in these and other environmental variables (Levinton, 2013). Moreover, algae and aquatic plants, as primary producers, are most directly affected by physical and chemical factors and are sensitive to pollutants, which may not visibly affect other aquatic assemblages, or that may only affect other organisms at higher concentrations (Welch and Jacoby, 2004).

1.2. STATEMENT OF THE PROBLEM

Rapid population growth, urbanization, industrialization, and agricultural practices are altering the water quality of Lake Tana. Human-induced environmental changes in Lake Tana and its catchment area have serious implications for tourism, public health and aquatic life. Changes in the physical and chemical conditions of a lake water are reflected in the taxonomic composition, diversity, abundance and biomass of resident organisms (Kalff, 2002). Adequate information on the chemical and physical factors controlling population dynamics of selected biotic assemblages is essential for the understanding of the human activities that affect water quality of Lake Tana and the extent of damage they have caused to aquatic biota. Planktonic algae are valuable indicators of short-term changes owing to their generally high reproduction rates, very short life cycles, high sensitivity to pollutants, and most

direct exposure to environmental factors as they are primary producers. Changes in species composition, abundance and biomass of phytoplankton in relation to physical and chemical water quality of the Southern Gulf of Lake Tana was, therefore, investigated with a view to evaluate the extent of environmental changes caused by human activities and recommend workable strategies of protection of this valuable aquatic resource.

1.3. RESEARCH QUESTIONS AND OBJECTIVES

1.3.1. Research Questions

- Are there changes in the physico-chemical water quality of the Southern Gulf of Lake Tana?
- What are the overriding physico-chemical parameters related to the observed phytoplankton community structure and biomass?
- How is the phytoplankton community structure (i.e. the species composition, diversity and abundance) and biomass in relation to water quality in the Southern Gulf of Lake Tana?

1.3.2. Research Objectives

1.3.2.1. General Objective

- To investigate phytoplankton community structure in relation to water quality in the Southern Gulf of Lake Tana.

1.3.2.2. Specific objectives

- To assess the changes in physical and chemical water quality of the Southern Gulf of Lake Tana.
- To document the species composition, abundance and biomass of phytoplankton and relate to the physico-chemical water quality in the Southern Gulf of Lake Tana.
- To identify the overriding physico-chemical water quality parameters associated with the observed phytoplankton community structure and biomass.

1.3.3. Hypothesis of the problem

It is hypothesized that there are changes in physico-chemical water quality and phytoplankton community structure in the Southern Gulf of Lake Tana and significant variations among sampling sites due to the impacts of agricultural practices, industrial and domestic influents.

1.3.4. Limitations of the study

The limitation of the present study on Water quality and phytoplankton community structure in the Southern Gulf of Lake Tana is the fact that this research work was done only during the dry period and included a few sampling sites due to financial constraints and logistics problems.

2. RVIEW OF LITERATURE

Water quality of a lake, reservoir, river, etc. is determined by its physical, chemical and biological attributes. An attempt will be made to have a fleeting look at the most commonly tested parameters of water quality in the following paragraphs.

2.1. Physico-chemical water quality parameters

It is very essential to test a body of water before it is used for drinking, domestic, agricultural or industrial purposes. Water must be tested for different physico-chemical parameters. Selection of parameters for testing of water depends upon the intended use. Water does contain different types of floating, dissolved, suspended and microbiological as well as bacteriological impurities. Physical parameters for which water samples are tested include temperature, pH, turbidity, water transparency (Secchi depth) TSS etc., while chemical tests are often performed for inorganic nutrients, dissolved oxygen, alkalinity, salinity (or conductivity), hardness and TDS (Patil *et al.*, 2012).

The following physico-chemical parameters are tested regularly for monitoring water quality (Patil *et al.*, 2012).

2.1.1. Turbidity-Availability of Light

Light is an important variable that controls phytoplankton growth and biomass owing to its effect on the rate of photosynthesis. The optical characteristics of water bodies strongly depend on the optical properties of molecules of water as well as substances dissolved or suspended in it (Kirk, 1994). Underwater light quality rapidly changes with depth because of the spectral selectivity of substances dissolved or suspended in it (Jassby *et al.,* 1999) and the absorption and utilization of light by phototrophic organisms (Kirk, 1994). Sometimes, as a consequence of the high levels of particulate matter, dissolved substances and photosynthetic pigments of algae and plants, the energy supply for photosynthesis in the sub-surface region of a water column becomes insufficient (Kirk, 1994). The energy shortage is furthermore amplified in vertically mixing water columns due to the resuspension of inorganic particles (Tilzer, 1990; Lewis, 1992). The decline in the penetration of light to a body of water is reflected in reduced water transparency (Secchi depth) and hence in the composition of phytoplankton leading to the dominance and persistence of cyanobacteria (Dokulil, 1994; Paerl *et al.*, 2011).

2.1.2. Temperature

Temperature varies locally and over short time-scales, including diel and seasonal cycles, in freshwater aquatic systems (Kalff, 2002). Although water bodies have the ability to buffer against atmospheric temperature extremes, even moderate changes in water temperatures can have serious impacts on aquatic life. Aquatic organisms, from microbes to fish, are dependent on certain temperature ranges for optimal development (APHA, 1992). If temperatures are outside optimal range for a prolonged period of time, organisms are stressed and even death may ensue (USEPA, 1991; Chapman, 1997). Temperature affects the oxygen content of the water, with oxygen levels becoming lower as temperature increases, the rate of photosynthesis by aquatic algae and plants, the metabolic rates of aquatic organisms, and the sensitivity of organisms to toxic wastes, parasites and diseases (USEPA, 1991; Margaleff, 1996; Chapman, 1997; Geneviève and James, 2008). Thermal pollution comes in the form of direct

impacts, such as the discharge of industrial cooling water into aquatic receiving bodies, or indirectly through human activities such as the removal of shading from stream bank vegetation or tall trees found on lake shores or the construction of impoundments (Geneviève and James, 2008).

2.1.3. Dissolved Oxygen (DO)

Oxygen is required for the metabolism of aerobic organisms, and influences chemical reactions. The concentration of oxygen in freshwater has implications for the presence and distribution of organisms, and anoxia can result in the death of aquatic animals. Oxygen is often used as an indicator of water quality such that high concentrations of oxygen usually indicate good water quality. A low DO (less than 2mg/l) would indicate poor water quality and thus would have difficulty in sustaining many sensitive aquatic life (Geneviève and James, 2008). Oxygen enters water through diffusion across the water's surface, by rapid movement such as waterfalls or riffles in streams (aeration), or as a by-product of photosynthesis. The amount of dissolved oxygen gas depends highly on temperature and somewhat on atmospheric pressure. Salinity also influences dissolved oxygen concentrations, such that oxygen is low in highly saline waters and vice versa. The amount of any gas, including oxygen, dissolved in water is inversely proportional to the temperature of the water; as temperature increases, the amount of dissolved oxygen (gas) decreases (Geneviève and James, 2008).

The observed range of dissolved oxygen concentrations reported worldwide is 0 mg/L (anoxic conditions) and 19 mg/L (supersaturated conditions). Super-saturated conditions are caused by algal blooms, with high amounts of algae producing more dissolved oxygen in the aquatic systems (Margaleff, 1996). Anoxic conditions, or periods of zero dissolved oxygen concentration in the water, lead to undesirable odours and chemical toxicity due to the accumulation of such noxious gases as ammonia and hydrogen sulfide until oxic or aerobic conditions develop (Mays, 1996).

2.1.4. pH

Various factors bring about changes in the pH of water. The assimilation of carbon dioxide and bicarbonate during photosynthesis is ultimately responsible for the increase in pH. Higher pH values observed in lakes are associated with the effect of physico-chemical and biological conditions on carbon dioxide, carbonate-bicarbonate equilibrium (Patil *et al.*, 2012). pH is positively correlated with electrical conductance and total alkalinity (Wood and Talling, 1988; Wetzel, 2001; Kalff, 2002).

pH outside the range 6.5 to 8 reduces the biodiversity in a lake because it stresses the physical system of most organisms and can reduce reproduction (USEPA, 1991). Low pH can also allow toxic elements and compounds to become mobile and "available" for uptake by aquatic plants and animals thereby producing conditions that are lethal to aquatic life, particularly to sensitive species (USEPA, 1991). High alkalinity waters are often more biologically productive than low alkalinity waters (Chapman, 1997). Consequently, total alkalinity was once used as an indirect measure of a lake's productivity. In recent years, research has shown that water bodies with low alkalinity levels are more susceptible to the effects of acidic water inputs (acid rain) (Schardt and Ludlow, 2000).

2.1.5. Salinity and Specific conductivity

Specific conductivity measures how well the water conducts an electrical current, a property that is proportional to the concentration of ions in solution and hence to salinity. Thus, saline intrusion into a body of water can be indicated by increased conductivity (Geneviève and James, 2008). The ions responsible for salinity include the major cations calcium(Ca^{2+}), magnesium(Mg^{2+}), sodium(Na^{+}) and potassium(K^{+})) and the major anions carbonate(CO_3^{2-}), bicarbonate (HCO_3^{2-}), sulphate(SO_4^{2-}) and chloride(Cl^{-}). The level of salinity in aquatic systems is important to aquatic plants and animals as species can survive only within certain salinity ranges (Friedl *et al.*, 2004). The level of salinity in a body of water,

therefore, influences the species composition and diversity of its inhabitants (Wetzel, 2001).

2.1. 6. Major inorganic nutrients

Nutrients are required by plants and algae. Major nutrient elements including nitrogen, phosphorus and silicon are required in great amounts, while trace nutrient elements are required in far smaller amounts and include iron, copper and vanadium (Levinton, 2013). Adequate light, sufficient water retention time, and low loss due to grazing will not result in high biomass without sufficient nutrient supply. Nutrients are frequently the key stimulus to high algal biomass, with phosphorus often controlling productivity and causing excess algal biomass in many freshwaters worldwide (USEPA, 1991: APHA, 1992). However, nitrogen can also become important in waters receiving agricultural runoff and/or wastewater with a low N:P ratio and in waters with naturally phosphorus-rich bedrock (Welch, 1992). The directly available forms of N and P are mainly inorganic (NO_3^-, NH_4^+ and PO_4^{3-}), although many algae are able to use organic forms (Darley, 1982). Total nitrogen (TN) and Total Phosphorus (TP) are often good predictors of algal biomass in lakes and reservoirs to a large extent because much of the particulate fraction is live algal biomass. Together with phosphorus, nitrogen in excess amounts can accelerate eutrophication, particularly in tropical waters, causing dramatic increase in aquatic autotrophic growth and changes in types of plants and animals in the lake.

Nitrogen and phosphorus are, therefore, the nutrients limiting the growth and hence productivity of phytoplankton. The transition between N and P limitation for algae in lakes have been defined by the ranges of ambient or cellular N:P ratios (Guildford and Hecky, 2000). If ambient N: P ratios (molar) are greater than 15-17:1, then P can be assumed to be in limiting supply, while N:P ratio less than 9-10:1 imply the limiting role of N (Sakamoto, 1966; Forsberg *et al.*, 1980). In lakes with intermediate ratio, algal growth is nearly balanced with both N and P, and the yield varies with an increase in either nutrient. It has been clearly shown that phosphorus is often the nutrient limiting the growth of

phytoplankton in temperate waters (Schindler, 1977). Nitrogen-limitation of phytoplankton happens to be very common in tropical lakes (Lewis, 1996; Talling and Lemoalle, 1998).

The relative abundance of N and P in lake water has also been suggested to have both quantitative and qualitative effect on phytoplankton community (Downing and McCauley, 1992), with lower N:P ratios favoring the predominance of nitrogen-fixing cyanobacteria (Jensen *et al.*, 1994).

2.2. Biological parameters of water quality

Phytoplankton are one of the biological quality elements used in the EU Water Framework directive (WFD) to assess the ecological status of coastal and transitional waters. Phytoplankton are good indicators of environmental change due to their quick response to changes in environmental pressures such as nutrient availability (Reynolds, 2006). To be fully compliant with the WFD, the parameters biomass, taxonomic composition, abundance (or cover), frequency, and intensity of algal blooms should be included in the assessment system. Today only biomass, measured as chlorophyll *a* and biovolume of autotrophic and mixotrophic species, is used in the Swedish assessment criteria for coastal phytoplankton (Höglander*et al.*, 2013). Phytoplankton community can be described in various ways, for example, by functional groups, species dominance relationships, size-groups, diversity indices, and phytoplankton pigments (Höglander*et al.*, 2013).

The phytoplankton community in a lake may vary both spatially and temporally. Phytoplankton are greatly influenced by various water quality parameters including turbidity (associated with light availability), temperature, and nutrients, which are factors driving their temporal and spatial variability (Höglander*et al*, 2013). Such human activities as shore-line modification and disposal of sewage into water bodies result in increased turbidity of receiving waters thereby impacting phytoplankton community structure (Scheffer, 1998). The association of potentially toxic cyanobacterial blooms with high turbidity in

many lakes is now well-established (Dokulil, 1994: Jensen *et al.*, 1994; Scheffer *et al.*, 1997; Scheffer, 1998). Although individual species have typical temperature preferences, the indirect effects of temperature often seem to be of greater importance than the direct physiological impact (Talling and Lemoalle. 1998). Temperature influences phytoplankton through its effect on the physical stratification of the water column, which affects the availability of both light and nutrients (Wetzel, 2001; Kalff, 2002). Some phytoplankton groups prefer thermally stratified stable water column, while others prefer turbulent well-mixed water column. The fast-growing and silicified diatoms thrive under strong water column mixing conditions, while the motile dinoflagellates flourish in more stratified water columns (Reynolds, 2006).

Changes in nutrient availability can lead to variations in phytoplankton diversity and species composition in aquatic systems. Elevated pH, dissolved oxygen, NH_4^+–N, NO_2-N and silica favored the growth of Cyanophyceae (cyanobacteria) and Chrysophyceae in the tropical ponds of Pindamonhangaba, Brazil (Asiyo, 2003). In eutrophic lakes, cell division occurs rapidly due to the enriched nutrients leading to blooms of cyanobacteria (Wetzel, 2001). The recurrent toxic algal blooms of *Microcystis aeruginosa* in Koka Reservoir, similar to that, which occurred in Lake Chamo in 1978 (Amha Belay and Wood, 1982) andcaused the death of livestock, are attributable to pollution of Koka Reservoir with algal nutrients originating from nearby agricultural lands on which fertilizers were applied (Girma Tilahun, 2006).

3. MATERIALS AND METHODS

3.1. Study area

Lake Tana (Fig. 1) is the largest lake in Ethiopia, with an area of 3150 km^2, located at an altitude of 1830 m. Lake Tana is a crater lake formed two million years ago, due to the volcanic blocking of the Blue Nile River (Dilnessa Gashaye,2016). It has a mean depth of 8m and a maximum depth 14 m. The

catchment area of Lake Tana is about 16,000 km^2. Lake Tana is fed by seven large permanent and 40 small seasonal rivers But, four permanent rivers, namely Gilgel Abbay, Megech River, Gumara River and Ribb River account for 95% of the inflow of the lake, while the Blue Nile is the only outflowing river (Tarekgne Wondmagegne *et al.* 2012).

Lake Tana, which is located in a wide depression of basaltic plateau, is surrounded by wetlands on all sides except in the north east. The lake is bordered by such flood plains as Fogera floodplain in the east, Dembia floodplain in the north and Kunzila floodplain in the south west, which are often flooded during the rainy season and, by steep rocks in the west and North West (Teshale Tadesse, 2014).

The climate of Lake Tana basin is typical of semi-arid regions. The wet period extends from March to September, with the major and minor rainy seasons occurring between June and September and between March and May, respectively. The dry period spans from October to February. There is slightly higher precipitation in the southern and south-eastern parts than in the northern part of the lake catchment area. In general, the southern part of Lake Tana basin is wetter than the western and northern parts of the lake. Although the temperature varies diurnally from 6°C of night time to 30°C of the day time, the mean annual temperature is about 20°C (Teshale Tadesse, 2014).

Lake Tana is a highly turbid lake with low biological productivity, but unique diversity of cyprinid fish. Environmental Changes in Lake Tana and its watershed including eutrophication, associated with various anthropogenic activities that resulted in the destruction of wetlands, have been observed (Tewodros Taffese, *et al.*, 2014). Recently, development of extensive stands of water hyacinth on the shore areas of this sensitive lake, one of the most ecologically dangerous weed infestations, has been reported. Other weeds introduced into Lake Tana include the aquatic fern Azolla species (Wassie Anteneh *et al.*, 2014).

The inappropriate discharge of waste water into Lake Tana and Abay (Blue Nile) River and the associatedhealth risk should be given due attention by the health inspectors and the local inhabitants. Lack of facility for waste water collection and treatment seems to be the most important problem. Furthermore, lack of awareness and carelessness on the part of both the public and sanitation workers andthe weak control or absence ofpenaltyfor offendersseem tohave contributed to thedegradation of the water environment around the fast-growing cities like Bahir Dar (Fesseha Hailu, 1988).

3.2. Sampling Protocol

On the basis of the proximity of agricultural practices and presence of industrial and domestic influents, three near-shore (littoral) sampling sites and one offshore presumably less impacted sampling site were selected in the Southern Gulf of Lake Tana.

Table 1. Description of Sampling sites

Sampling sites (Local names)	Depth(m)	Altitude(m) a.s.l.	Coordinates	
			Latitude	Longitude
Zege	7.946±0.501	1791	11° 39' 45.8" N	037° 21'37.5" E
Hospital	2.244±0.440	1795	11° 36'45.0" N	037° 22' 21.4" E
Resort	1.644±0.360	1790	11° 36' 16.0" N	037° 22' 44.1" E
K. Georgis	1.669±0.230	1797	11° 35' 48.8" N	037° 23' 22.2" E

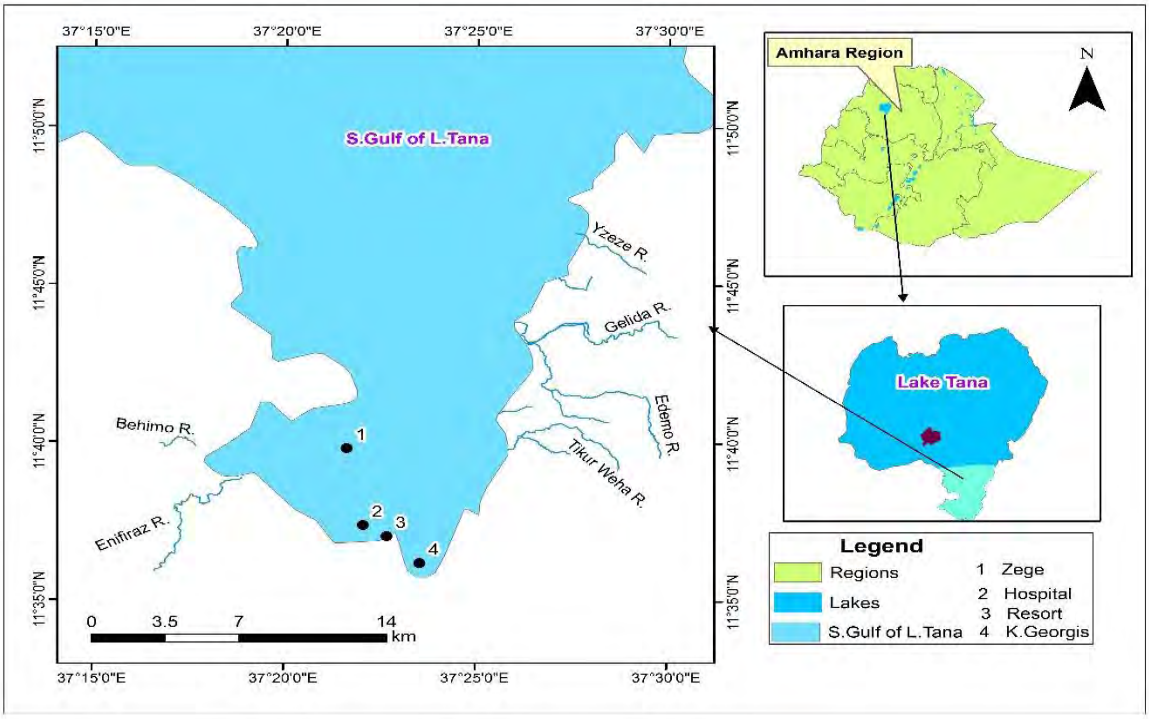

Figure 1. Location map of the Southern Gulf of Lake Tana with sampling sites indicated as closed circles

Zege sampling site is an open water site in Lake Tana, which is a relatively less impacted site. All the other three sites (Hospital, Resort and K. Georgis) are found in the littoral parts of Lake Tana, and are used as damping sites for urban waste water (Table 1). The southern shore of Lake Tana is covered with swamps whose macrophyte vegetation is dominated by papyrus (*Cyperus papyrus*), Hippo grass (*Echinochloa stagnina*), Elephant grass *(Phragmites karka*), Aquatic ferns (*Azolla spp.*), *Typha latifolia*, Waterlilies (*Nymphaea spp.*), and *Ceratophylum sp.*. Especially at the Hospital and Resort sites extensive areas are covered by *Ceratophylum sp. (*Ayalew Wondie, 2006; Wassie Anteneh *et al.*, 2014).

Collection of samples and in situ measurements of physico-chemical parameters were made at the four sampling sites from December, 2015 to March, 2016 at about biweekly intervals during all sampling months except January.

3.3. Measurements of physico-chemical parameters

Water transparency was measured using a Secchi disk of 20 cm diameter. Dissolved oxygen (DO), conductivity, pH, temperature, salinity and total dissolved solids (TDS) were measured *in situ*using YSI 556 multi-probe system. Turbidity was determined photoelectrically using the Palintest Photometer. Total Suspended Solids(TSS) was measured as dry weight of seston filtered onto a glass fiber filter paper (GF/F) pre-dried at 105 ^{o}C and subsequently dried with seston at the same temperature for 1 hr and calculated using the following formula (Estefan *et al.*, 2013);

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

Where:

W_1= Weight of dried clean filter paper (in mg)

W_2= Weight of dried clean filter paper and seston (in mg)

V= Volume of water sample used for measurement (mL)

3.4. Determination of chemical parameters in the laboratory

The concentrations of the following inorganic nutrients were determined using a chemical analysis kit, Palintest transmittance display photometer 5000 (Wagtech International). Ammonia-nitrogen (NH_3– N) was measured using filtered composite samples by the indophenol method. Nitrate and nitrite-nitrogen (NO_3-N and NO_2 –N) were analyzed by the Palintest nitrate test and Palintest Nitricol methods, respectively. Hydrogen Sulphide was determined using a reagent containing diethyl-p-phenylene diamine (DPD) and potassium dichromate with Palintest transmittance display photometer 5000.Phosphate (PO_4 –P) was analyzed by the Palintest Phosphate LR method, while total phosphorus was measured as molybdate reactive phosphate (MRP) after the organically-bound phosphorous was converted to orthophosphate through oxidative hydrolysis with

potassium persulfate (APHA, 1999). Silica (SiO_2) was determined by the molybdosilicate method.

3.5. Species composition and Abundance of phytoplankton

Composite samples of phytoplankton were produced using samples collected from discrete depths distributed within the euphotic zone using van Dorn water sampler. Then, 1 litter aliquot was taken and fixed with Lugol's solution until the sample takes a weak tea colour. The fixed samples were allowed to stand for a period equivalent to a sedimentation rate of 6 hrs per cm height of the sedimentation chamber. After sedimentation, the top 90% of the total sample volume was carefully siphoned off without disturbing the sedimented algae. To facilitate identification, diatoms were cleaned with hydrogen peroxide (Taylor *et a*l., 2007). The remainder was shaken gently and a 1 ml subsample was transferred to a Sedgewick-Rafter counting chamber and allowed to settle before counting (Hotzel, and Croome, 1999). Phytoplankton were identified using appropriate taxonomic literatures on tropical phytoplankton (e.g. Gasse, 1986; Komarek and Kling, 1991; Komárek and Anagnostidis, 2000; Bellinger and Sigee, 2010). The abundance of each species was determined using a Sedgewick-Rafter cell and an inverted microscope (A. Kruss Optronic, Serial number 32010095, Germany) according to Hotzel and Croome (1999). For the filamentous algae, the number of cells per filament of 15 filaments was determined and the mean number of cells per filament for the sample under consideration was calculated. Likewise, the number of cells per colony was first determined for 15 colonies and then the mean number of cells per colony was calculated. The average number of cells per filament or colony was multiplied by the number of filaments or colonies to estimate the abundance of filamentous or colonial taxa. Estimation of phytoplankton abundance was made using the following formula (Hotzel and Croome, 1999).

$$C\ [\text{cells mL}^{-1}] = \frac{N * 1000 mm^3}{A * D * F * \text{Concentration factor}}$$ where,

N = number of cells counted

A = area of grid (mm^2)
D = depth of a grid (Sedgwick-Rafter chamber depth) (mm)
F = number of grids counted.

$$Concentration\ factor = \frac{volume\ of lake\ water\ filtered\ (ml)}{volume of concentrate\ (ml)}$$

3.6. Diversity indices of phytoplankton

The following indices were used to characterize the community structure of phytoplankton (Margurran, 1988).

Diversity- the Shannon's Index, which estimates diversity (H') as

$$H' = -\sum_i Pi \ln(p_i), (i = 1,2,3, \ldots S), 0 \leq H' \leq \infty$$

Where P_i the proportional abundance of the *ith* species = *(n/ N)* and *ln* (*pi*) is the natural log of *pi*, N = the total number of individuals, n = individual species

Species Richness index($S_{Margalef}$)

$$\boldsymbol{S_{Margalef} = \frac{S-1}{lnN}}$$

Where S = number of species in the community, N = total number of individuals in the community

3.7. Chlorophyll-a Biomass of phytoplankton

Appropriate volume of composite samples of phytoplankton was filtered using glass fiber filter papers (GF/F) and frozen immediately. Chlorophyll-a was extracted in 90 % acetone. The pigment extract was centrifuged at 3000 rpm for 10 minutes. The absorbance of the extract was measured with Jenway 6300 UV-VIS spectrophotometer at 665 and 750 nm before and after acidification with 0.1 ml of 1 N HCl (Wetzel and Likens, 2000). Chl-a concentration was calculated according to the following equation;

$$\textbf{Chl-a (µg L}^{-1}\textbf{)} = \frac{\mathbf{26.73[(665b-750b)-(665a-750a)]\times Ve}}{\mathbf{Vf\times Z}}$$ Where

665b and **750b** are absorbance at 665nm and 750nm before Acidification. Respectively.

665αand **750α** are absorbance at 665 and 750 nm after acidification, respectively.

Ve = Volume of extract in ml

Vf = Volume of sample filtered in liter

Z= Path length of the cuvette (1cm)

3.8. Trophic State of the Southern Gulf of Lake Tana

The trophic state index of the Southern Gulf of Lake Tana was estimated according to Carlson (1977). The Trophic State Index(TSI) of Carlson (1977) is a trophic state classification method that is based on total in-lake phosphorus concentration, in-lake Chlorophyll-a (Chl-a) and water transparency (Z_{SD}). The following formulae of Carlson (1977), which are based on the three aforementioned parameters, were used to calculate the Trophic State Index (TSI) of the Southern Gulf of Lake Tana.

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

$$TSI_{SD} = 60 - 14.41 \ln Z_{SD} (m)$$

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

$$TSI_{Chl} = 30.6 + 9.81 \ln \text{Chl-a } (\mu g\ L^{-1})$$

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

$$TSI_{TP} = 4.15 + 14.42 \ln TP\ (\mu g\ L^{-1})$$

Carlson Trophic State Index **(TSI_C)**

$$\mathbf{TSIC} = \frac{\mathbf{TSITP + TSISD + TSIChl}}{\mathbf{3}}$$

TSI was also estimated according to Cunha *et al.* (2013), which developed TSI model for tropical/subtropical reservoirs. The Trophic State Index (TSI) of Cunha *et al.* (2013) is a trophic state classification method that is based on geometric means of total in-lake phosphorus concentration(TP) and Chlorophyll-a (Chl-a).

$$\text{TSItsr} = \frac{\text{TSI(TP)tsr} + \text{TSI(Chl} - \text{a)tsr}}{2}$$

Where

$$\text{TSI(TP)tsr} = 10[6 - \frac{(-0.27637\text{lnTP} + 1.329766)}{\ln 2}]$$

$$\text{TSI(Chl} - \text{a)tsr} = 10[6 - \frac{(-0.2512\text{lnChl} - \text{a} + 0.842257)}{ln2}]$$

Geometric means, $G_{mean} = \sqrt[n]{x_1 x_2 x_3 \cdots x_n}$

3.9. Statistical analysis

Spatial variations of measured physico-chemical parameters were analyzed using one way Analysis of Variance (ANOVA) at 95 % significance level ($P < 0.05$). Causal relationships (correlation) among physico-chemical and biological parameters were assessed and mean numbers and standard errors for each physico-chemical parameter were calculated using a statistical software (SPSS Version 23).

The relationship between the abundance of taxa of phytoplankton species and physico-chemical variables was assessed by using a multivariate analysis tool, Redundancy Analysis (RDA), using CANOCO for windows version 4.5. To determine the suitability of the method used in this analysis, Detrended Correspondence Analysis (DCA) was employed. According to Lepš and Šmilauer (1999), when the length of the longest gradient is less than 3, the species data show linear response to environmental variables. Thus, the linear method of ordination, Redundancy analysis (RDA), is appropriate i.e. since the length of the longest gradient was less than 3, RDA was employed.

4. RESULTS

4.1. Physico-chemical parameters

The water column at the Zege site (7.5-8.6 m) was found to be much deeper than those of the Hospital (1.94-3.2 m), Resort (1.18-2.19m), and K. Georgis (1.27-2 m) sampling sites, with the water levels at all sites declining almost consistently from December, 5 to March, 9.

Secchi depth (Z_{SD}, cm) averaged 83.43, 98.86, 127.71, and 161.86 for the Zege, K. Georgis, Resort and Hospital sampling sites, respectively, with values recorded for the Hospital and Resort sites approaching closely the respective depths of the water columns. Turbidity (NTU) of the water column, which averaged 11.43 and 8.43 for the Hospital (5-14) and Resort (5-17) sites, respectively, was considerably lower than those of the K. Georgis (5-26, mean=15.71) and Zege (17-26, mean=20) sites. Levels of Total Suspended Solids (TSS, mg L^{-1}) varied spatially, with mean values recorded for the Zege (10.367) and K. Georgis (13.643) sampling sites being about 2 to 2.5 times those of the Hospital (4.00) and Resort (4.714) sites, and with the highest peaks of TSS occurring in January at all sampling sites except Hospital and Resort (Fig. 2).

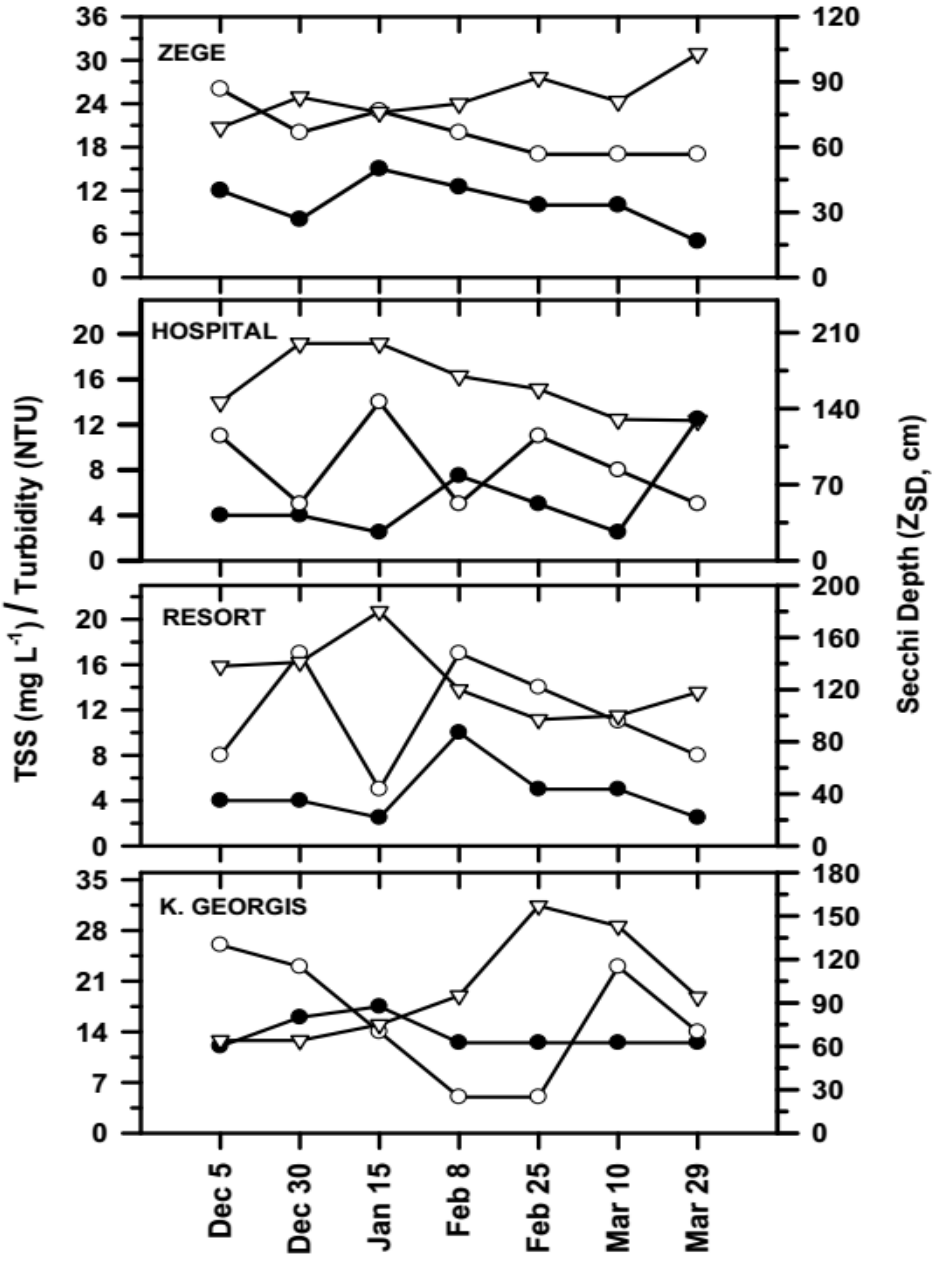

Fig. 2 Variations in Secchi depth (▽—)) in relation toTurbidity(○—) and TSS (—●—) at the four sampling sites of the present study in the Southern Gulf of Lake Tana.

The highest value of TSS (17.5 mg L^{-1})) was recorded at the K. Georgis site, while the lowest (2.5 mg L^{-1}) was observed at the Resort site. Secchi depth was negatively but significantly correlated (at p=0.01) with both TSS and turbidity.

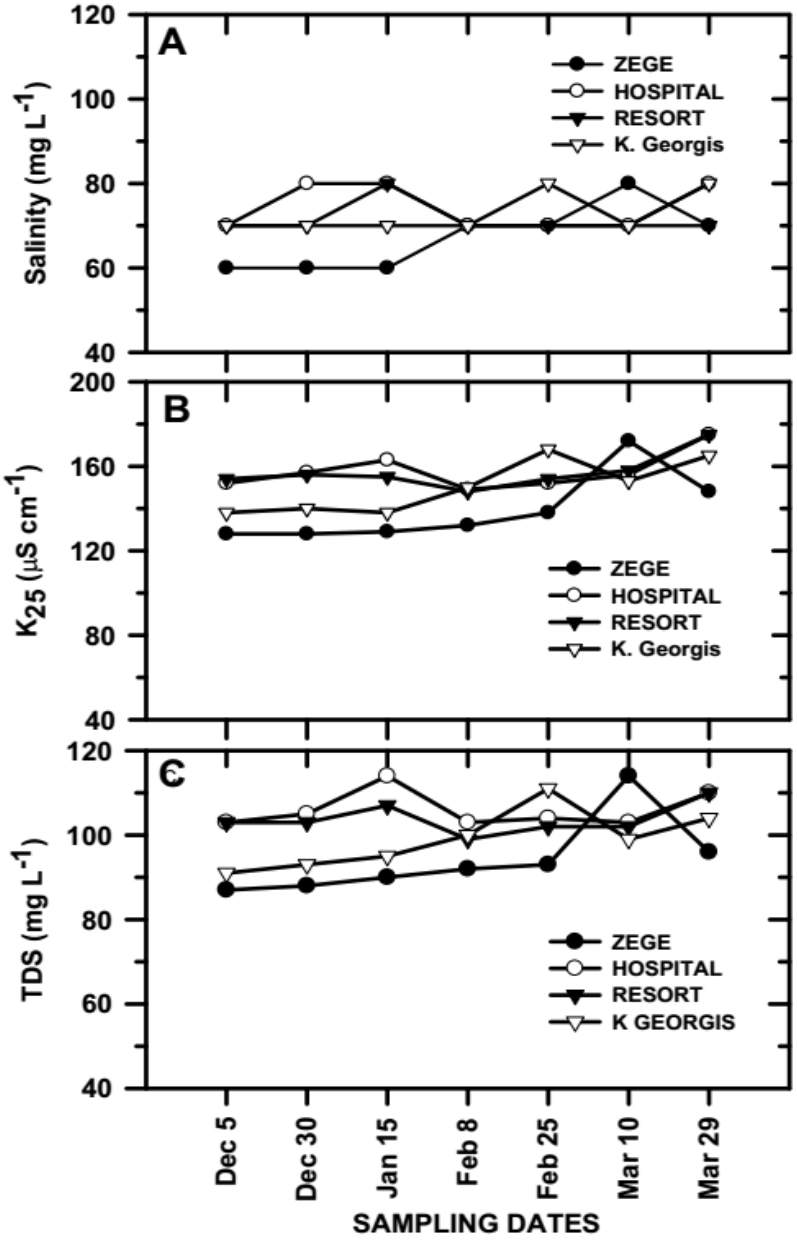

Fig. 3 **Variations in salinity A), Specific conductance (K_{25}, B) and Total Dissolved Solids(TDS, C) at the four sampling sites of the present study in the Southern Gulf of Lake Tana.**

Salinity (g L^{-1}) was exceedingly low (<0.1) at all sampling sites, with the mean value of the Hospital site (0.074) being slightly higher than those of theK. Georgis (0.073), Resort (0.071) and Zege (0.067) sampling sites (Fig. 3). Specific conductance (K_{25}, µS cm^{-1}) exhibited small spatial variations, with the closely similar mean values of the Hospital (157.71) and Resort (157.14) sites exceeding

those of the K. Georgis (150.29) and Zege (139.29) sampling sites. The highest peak of K_{25} occurred in March, 2016 at all sampling sites except K. Georgis coinciding with the maximum TDS values at the Zege and K. Georgis sites (Fig. 3). The mean TDS (mg L^{-1}) value of the Hospital site (106) is slightly higher than those of the Resort (103.71), K. Georgis (99) and Zege (94.29) sites. Salinity was positively and significantly correlated (at p=0.01) with both specific conductance and TDS, while its correlation with pH and Chl-a (at p=0.05) and turbidity (at p=0.01) was negative but statistically significant.

Absolute values of surface water temperature (˚C) recorded for the Southern Gulf of Lake Tana ranged from 21.2 of the first half of January at the Zege sampling site to 27.2 of the second half of March at the K. Georgis site. Although the minimum and maximum values of surface water temperature were recorded in January and March, respectively at all sampling sites, the temperature levels at the K. Georgis site were consistently higher than those of other sampling sites (Fig. 4). Thus, mean surface water temperature averaged 22.86±1.314, (23.211±1.353, 24.037±1.376 and 24.207±1.523˚C at the Zege, Hospital, Resort and K. Georgis sampling sites, respectively.

Concentrations of DO (mg L^{-1}, Fig. 4) at the different sampling sites were broadly similar, with mean values of 6.10,5.97, 5.89 and 5.84 recorded for Zege, Hospital, K. Georgis and Resort sites, respectively. The lowest DO value (4.39) was recorded at the K. Georgis site, while the highest value was observed at the Zege site (6.82) (Figure 4). The lowest concentration of DO was observed in March, 2016at all sampling sites. The concentration of DO was correlated significantly but negatively with temperature (at $p<0.05$) and specific conductance (at $p<0.01$), while its correlation with Chl-a biomass of phytoplankton was positive and statistically significant (at $p<0.01$).

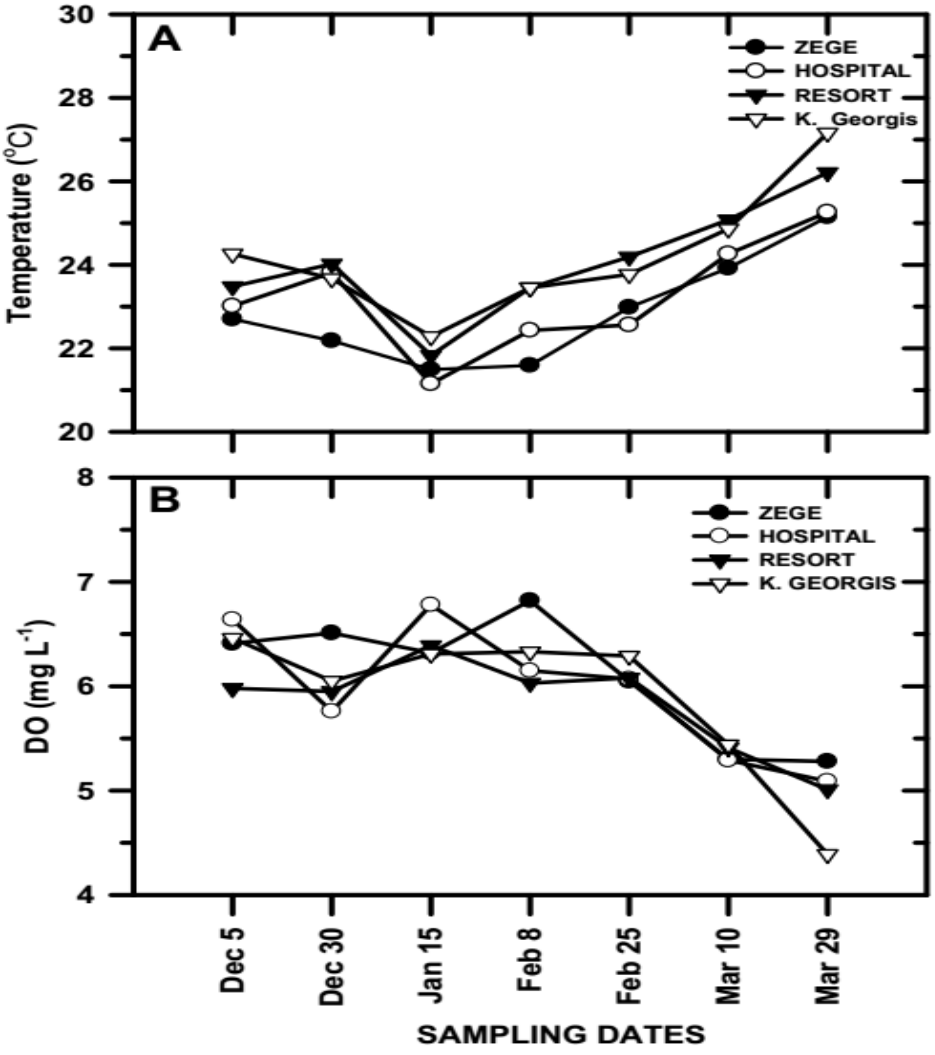

Fig. 4 Variations in the levels of surface water temperature (**A**)and DO(**B**) at the four sampling sites of the present study in the Southern Gulf of Lake Tana.

Other measured chemical parameters including macronutrients exhibited both spatial and temporal variations, although the variations were not statistically significant (at p=0.05). Mean values (mg L^{-1}) of Nitrate-N (NO_3-N) ranged from 0.875 of the Resort site to 1.257 of the Zege site, with individual observations of the Hospital and K. Georgis sites often exceeding 1 mg L^{-1}. Nitrate levels were always above 0.5 mg L^{-1} at all sampling sites except the Resort site where considerably lower concentrations were recorded during the first half of February and end of March (Fig. 5).

Nitrite-N (NO_2-N) concentrations were often below 10 µg L^{-1} at all sampling sites, with generally higher values at the Zege and Hospital sites. Ammonia-N (NH_3 + NH_4^+-N) concentrations varied markedly both spatially and temporally averaging 0.216, 0.110, 0.101, and 0.06 for the Hospital, Resort, K. Georgis and Zege sites, respectively. Levels of ammonia were below the limit of detection of the method of analysis used at all sampling sites in February.

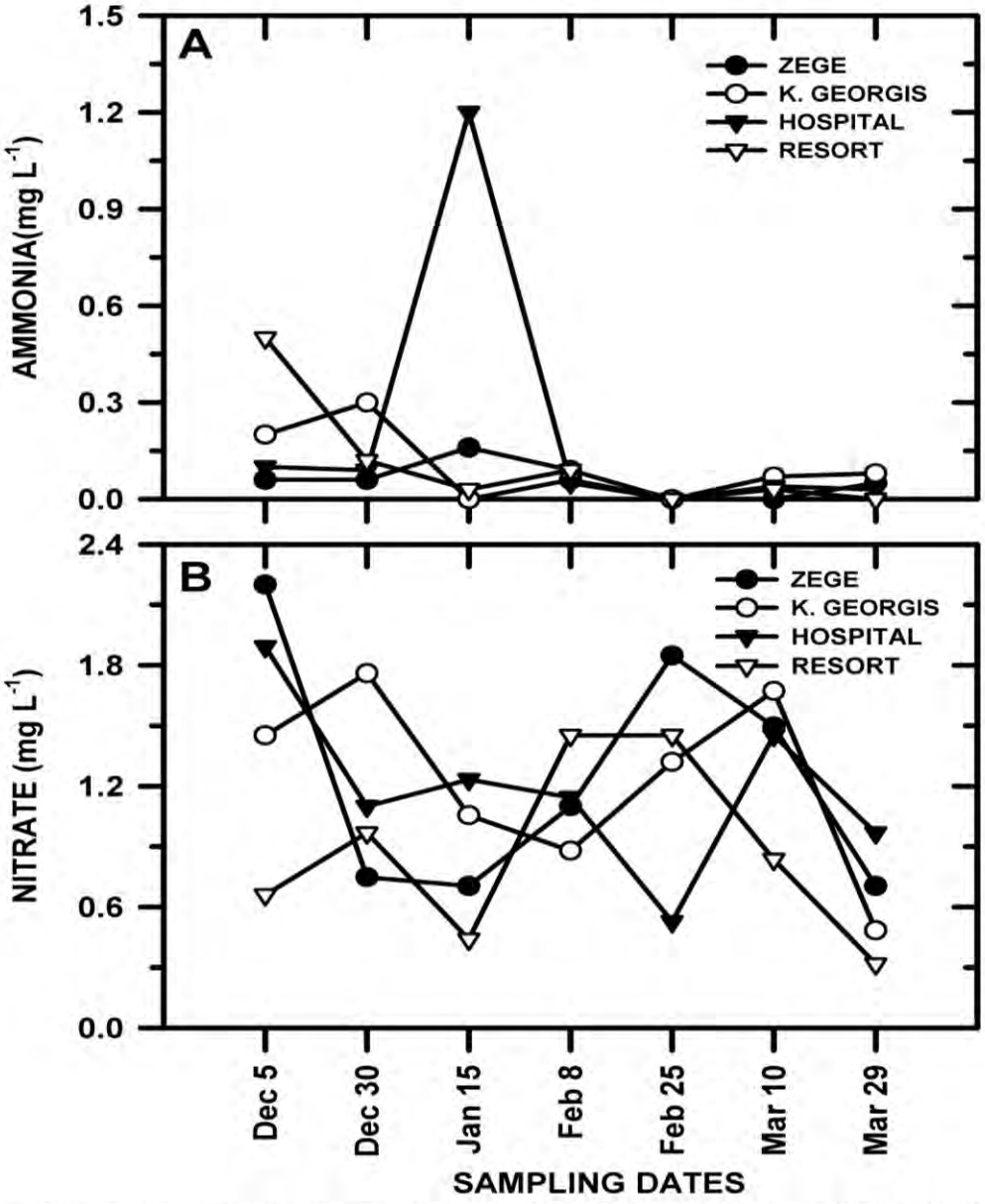

Fig. 5. Variations in the levels of ammonia(A) and nitrate (B) at the four sampling sites of the present study in the Southern Gulf of Lake Tana.

Soluble Reactive Phosphate (SRP, mg L^{-1}) averaged0.407, 0.237, 0.36and 0.390 for the Zege, Hospital, Resort and K. Georgis sites, respectively. Total Phosphorus (TP) varied spatially with a pattern similar to that of SRP (Fig. 6) and with mean values of 0.445, 0.269, 0.4483and 0.410 for the Zege, Hospital, Resort and K. Georgis sites, respectively.

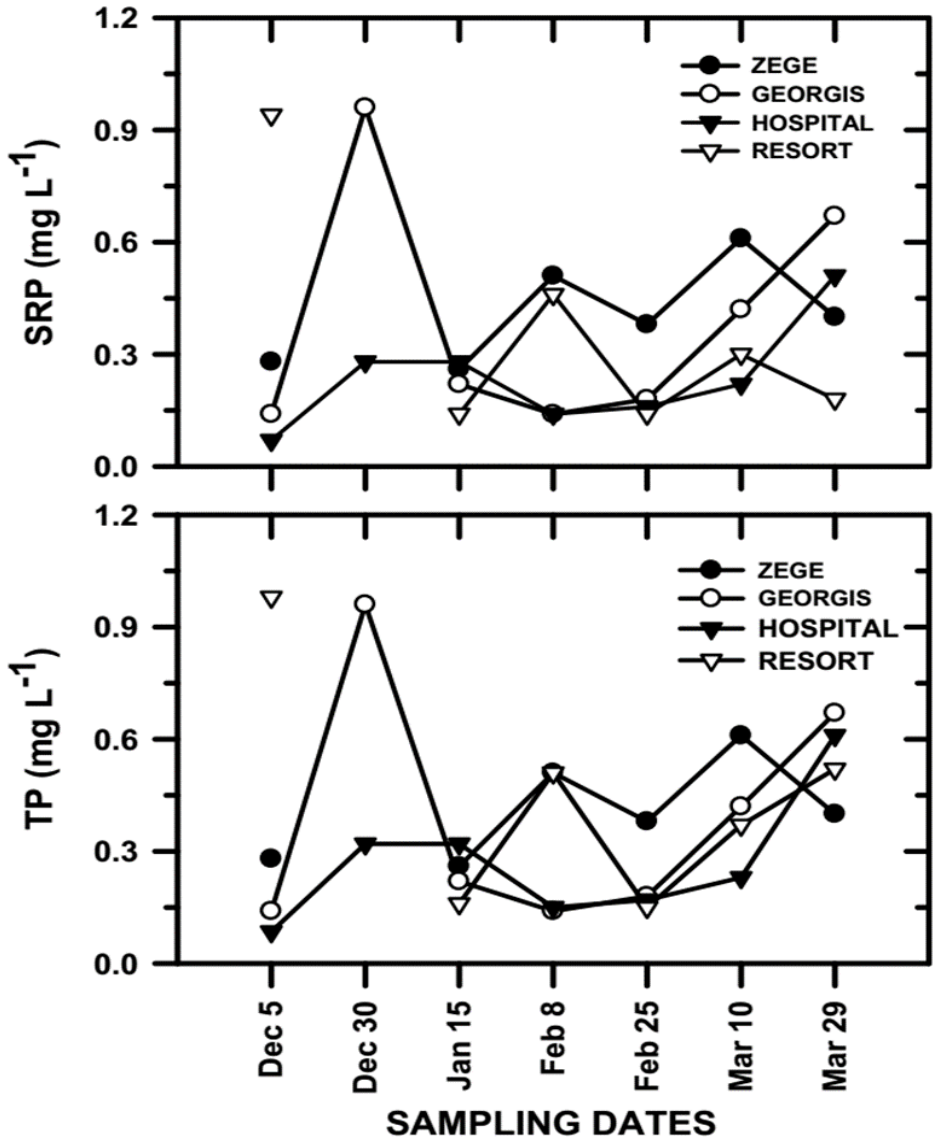

Fig. 6 Variations in the levels of SRP and TP at the four sampling sites of the present study in the Southern Gulf of Lake Tana.

The levels of Silica (SiO_2mg L^{-1}) averaged 16.48, 13.97, 12.28, and 8.97 for the Zege, Resort, K. Georgis and Hospital sites, respectively. The lowest

concentration (0.2 mg L^{-1}) was observed in February at the K. Georgis site, while the highest level (28.6 mg L^{-1}) was recorded in January at the Resort site and in December at K. Georgis site. Silica levels lower than 0.5 mg L^{-1} were recorded at the Hospital and K. Georgis sites in the second half of February during which the minimum silica concentration (2 mg L^{-1}) at the Resort site was also observed coincident with undetectable levels of ammonia at all sampling sites.

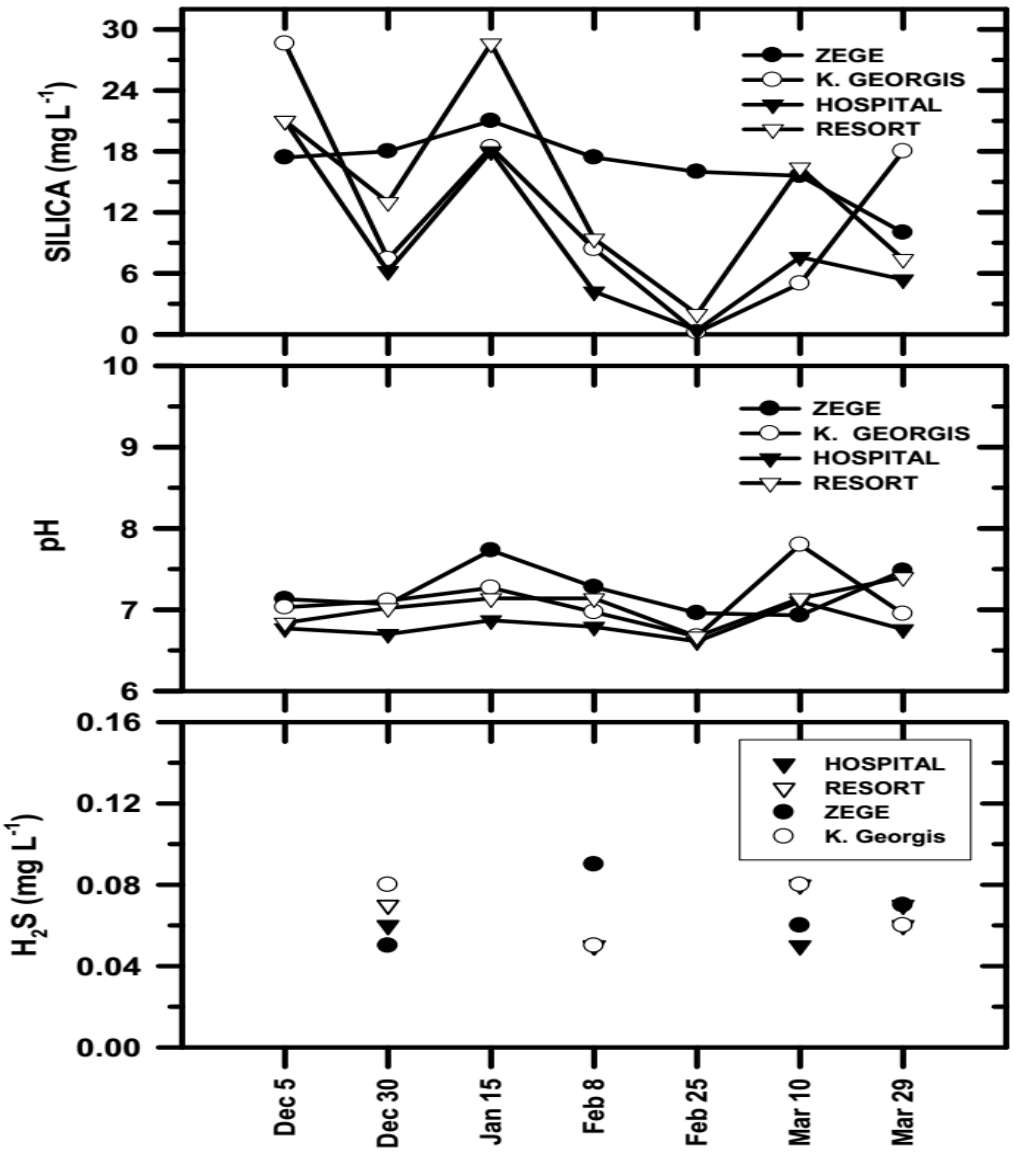

Fig. 7 Variations in the levels of Silica, pH and H_2S at the four sampling sites of the present study in the Southern Gulf of Lake Tana.

The concentrations of SRP and nitrate were correlated negatively with Chl-a, while those of ammonia and silica were correlated positively with Chl-a although statistically significant correlation(atp<0.05) was observed only between silica levels and Chl-a concentrations. H_2S concentration (mg L^{-1}) averaged 0.072 for the Zege, Resort and K. Georgis sites and 0.058 for the Hospital site, with individual observations ranging from 0.053 to 0.085 mg L^{-1}(Fig. 7). The concentration of H_2S was positively and significantly correlated with pH (at $p<0.05$).

pH values averaged 7.23, 7.11, 7.05 and 6.80 for the Zege, K. Georgis, Resort and Hospital sites respectively, with the highest peak occurring in March at all sampling sites. The lowest pH value (6.61) was recorded at the Hospital site, while the maximum was observed at the K. Georgis site (7.80). Although the pH of the lake water at the Hospital site was almost always <7, slightly acidic conditions were also detected at other sampling sites on some occasions (Fig. 7).

4.2. Biological parameters

4.2.1. Species composition and abundance of phytoplankton

Species of phytoplankton encountered in samples collected from the four sampling sitesin the Southern Gulf of Lake Tana are listed in Table 2. A total of 60 phytoplankton species were identified and quantified throughout the sampling period. The species composition of the phytoplankton communities at the four sampling sites were similar. The phytoplankton communities of all the sampling sites were dominated, both in terms of species richness and abundance, by three alga groups, namely Cyanophyceae (Cyanobacteria-blue-green algae), Bacillariophyceae (diatoms) and Chlorophyceae (green algae). The most species-rich group was Chlorophyceae (with 26 species), followed by Bacillariophyceae and Cyanophyceae, with 19 and 8 species, respectively. The other phytoplankton taxa, which constituted the phytoplankton community of the Southern Gulf of Lake Tana included 4 dinoflagellates (Dinophyceae), 2 euglenoids (Euglenophyceae), 1 cryptomonads (Cryptophyceae).

Table 2. List of phytoplankton species identified in samples collected from the four sampling sites of the present study in the Southern Gulf of Lake Tana

Cyanophyeae
Anabaena sp.
Chrococcus turgidus(Kütz.) Nägeli
Microcystis aeruginosa(Kütz). Kütz.
M. flos-aquae (Witt.) Kirch.
Oscillatoria sp.
Planktolyngbya limnetica (Lemmerm.) Komark.
Pseudoanabaena sp.
Synechococcus sp.

Bacillariophyceae
Amphora coffeaeformis (C. Agar.) Kütz.
Aulacoseira distans (Ehr.) Simon.
A. granulata (Ehr.) Simon.
Cyclotella radiosa (Grun.) Lemm.
Cyclotella sp.
Cymbella minuta Hilse
C.ventricosa (C.Agar.) C.Agar.
Diatoma vulgaris Bory
Gomphonema gracile Ehr.
G.cf. grovei M. Schimdt
Meloseira sp.
Navicula cryptocephala Kutz.
Nitzschia filiformis (W. Smith) Van Heur.
*N. palea (*Kütz.) W. Smith
Pinnularia sp.
Rhopalodia gibba(Ehr.) O. Müll.
Rhoicosphenia abreviata (C. Agardh) Lange Bertalot
Synedra ulna (Nitz.) Ehr.

Chlorophyceae
Ankistrodesmus angustus C.Bern.
Ankistrodesmus. Sp.
Chlamydomonas sp.
Chlorella sp.
Closterium acutum Bréb.
Closterium sp.
Cosmarium contractum O. Kirch.
Desmidium swartzii C.Agardh ex Ralfs

Chlorophyceae contd
Eudorina sp.
Haematococcus sp.
Oedogonium sp.
Oocystis eremosphaeria G. M. Smith
O. lacustris Chodat
O. parva West & G.S.West
Oocystis sp.
Pediastrum duplex Meyen
P. simplex Meyen
P. boryanum(Turp.) Menegh.
Schroederia setigera (Schröd.) Lemmer.
Scenedesmus incrassatulus Bohl.S
Selanstrum sp.
Staurodesmus curvatus var. latus (A.M. Scott & Presc.)Teil.
Staurastrum convergens (Ehr.)Menegh.)
Staurastrum gracile Ralf ex Ralfs
Staurastrum longebrachiatum
Staurastrum triangularis var. triangularis A.M. Scott and GrÖnblad

Dinophyceae
Peridinium cinctum(O.F. Mull.) Ehr.
Peridinium gatunense Nyg.
Peridinium volzii Lemm.
Peridinium sp.

Euglenophyceae
Euglena cf. viridis (O.F. Mull.) Ehr.
Phacus acuminatus Stokes

Cryptophyceae
Cryptomonas sp.

Total Phytoplankton abundance at all sampling sites of the present study in the Southern Gulf of Lake Tana exhibited considerable temporal variations with the highest and lowest phytoplankton abundance occurring on different sampling dates (Figs. 8 to 11). The highest peak of total phytoplankton abundance at the Zege site occurred in the first half of February coincident with the highest abundance of diatoms and green algae and the second largest peak of cyanobacteria abundance (Fig. 8).

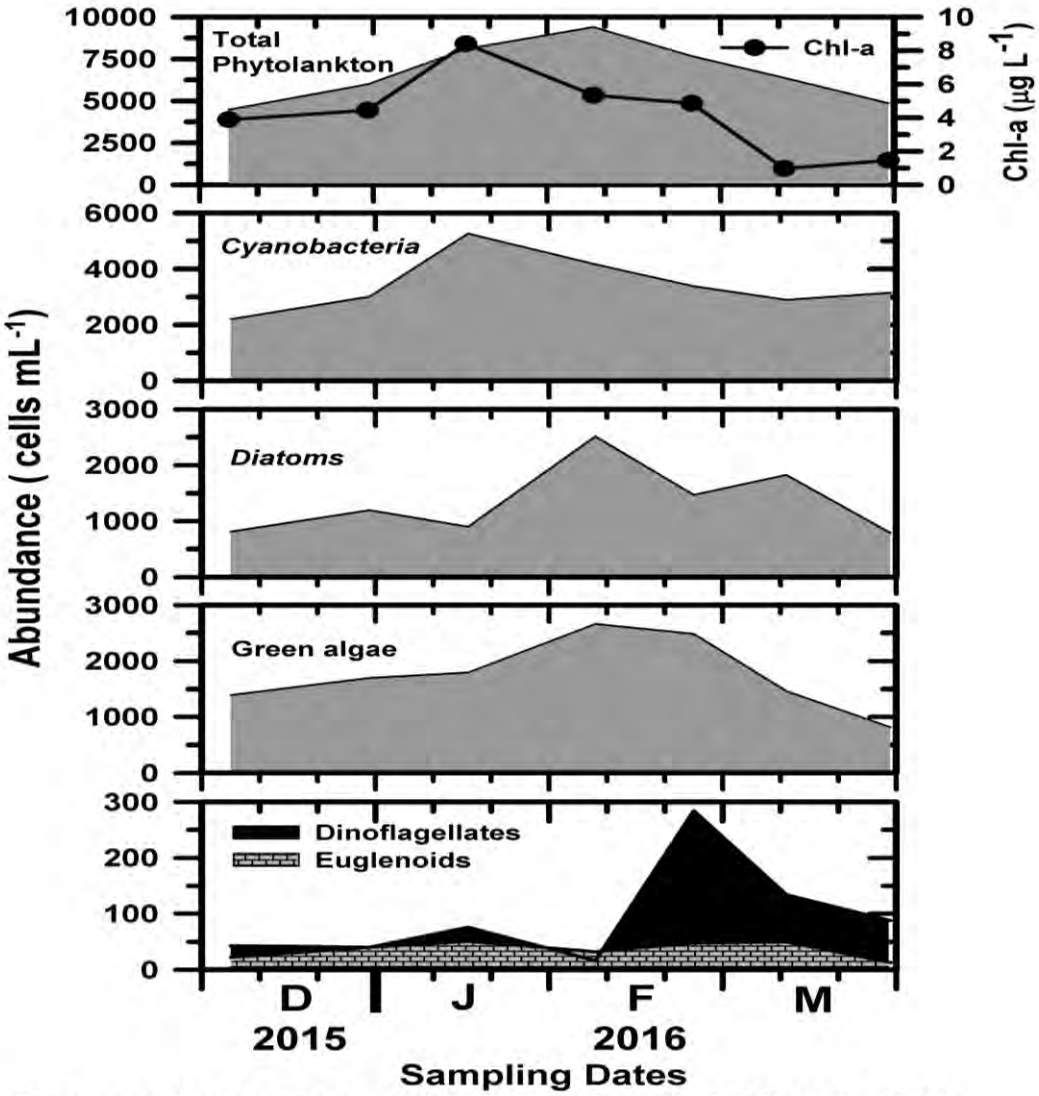

Fig. 8. Temporal variations in the abundance of different algal groups in relation to total phytoplankton abundance (area plot) and Chl-a (line plot) at the Zege Site of the present study in the Southern Gulf of Lake Tana.

At the hospital site (Fig. 9), the highest peak of total phytoplankton abundance was observed in mid-January when cyanobacteria and diatoms were also at their highest peaks of abundance.

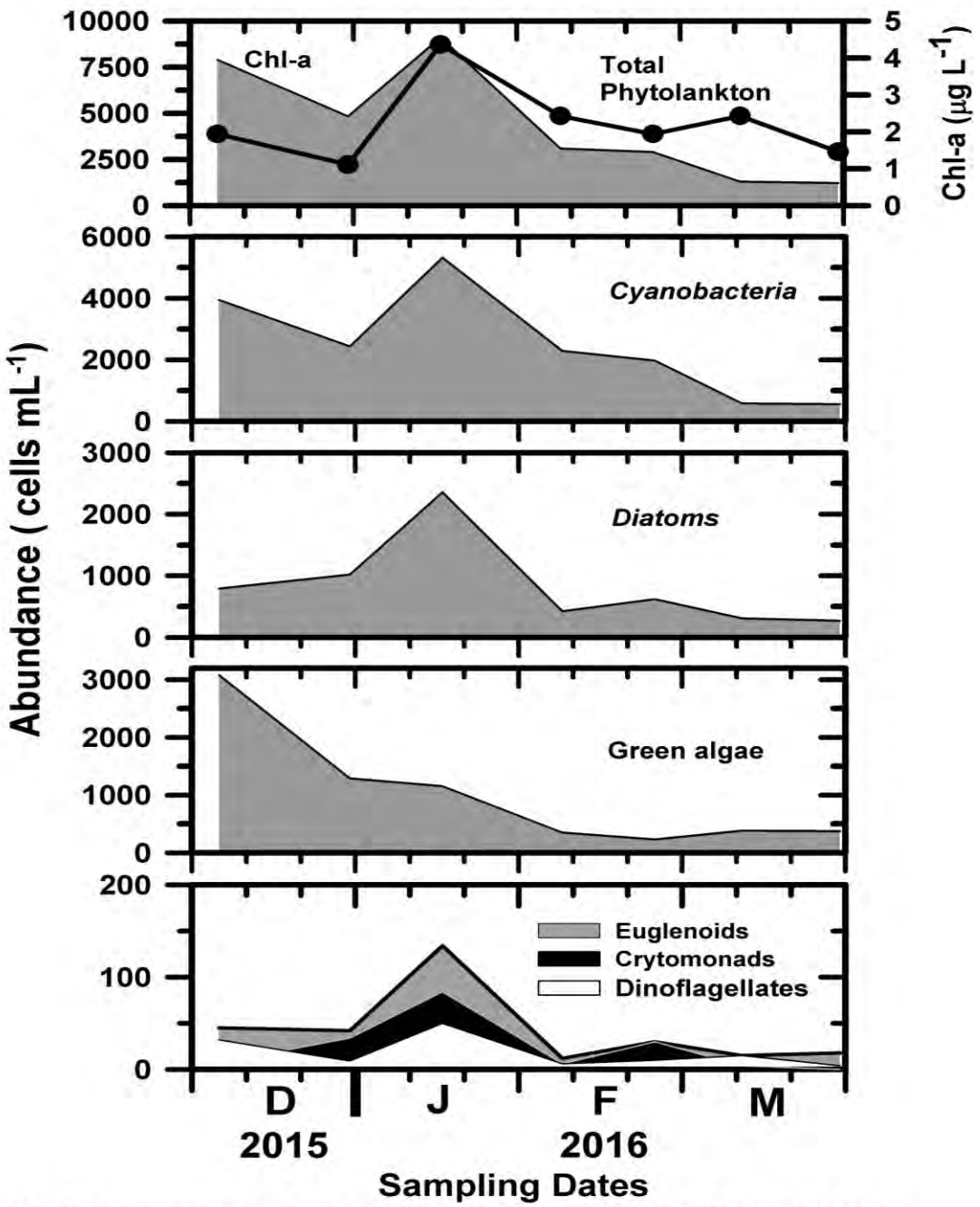

Fig. 9. Temporal variations in the abundance of different algal groups in relation to total phytoplankton abundance(area plot) and Chl-a (line plot) at the Hospital Site of the present study in the Southern Gulf of Lake Tana.

The second highest peak of total phytoplankton abundance at the Hospital site, which coincided with the second largest peak of cyanobacterial abundance and the highest peak of abundance of green algae, was observed at the beginning of January.

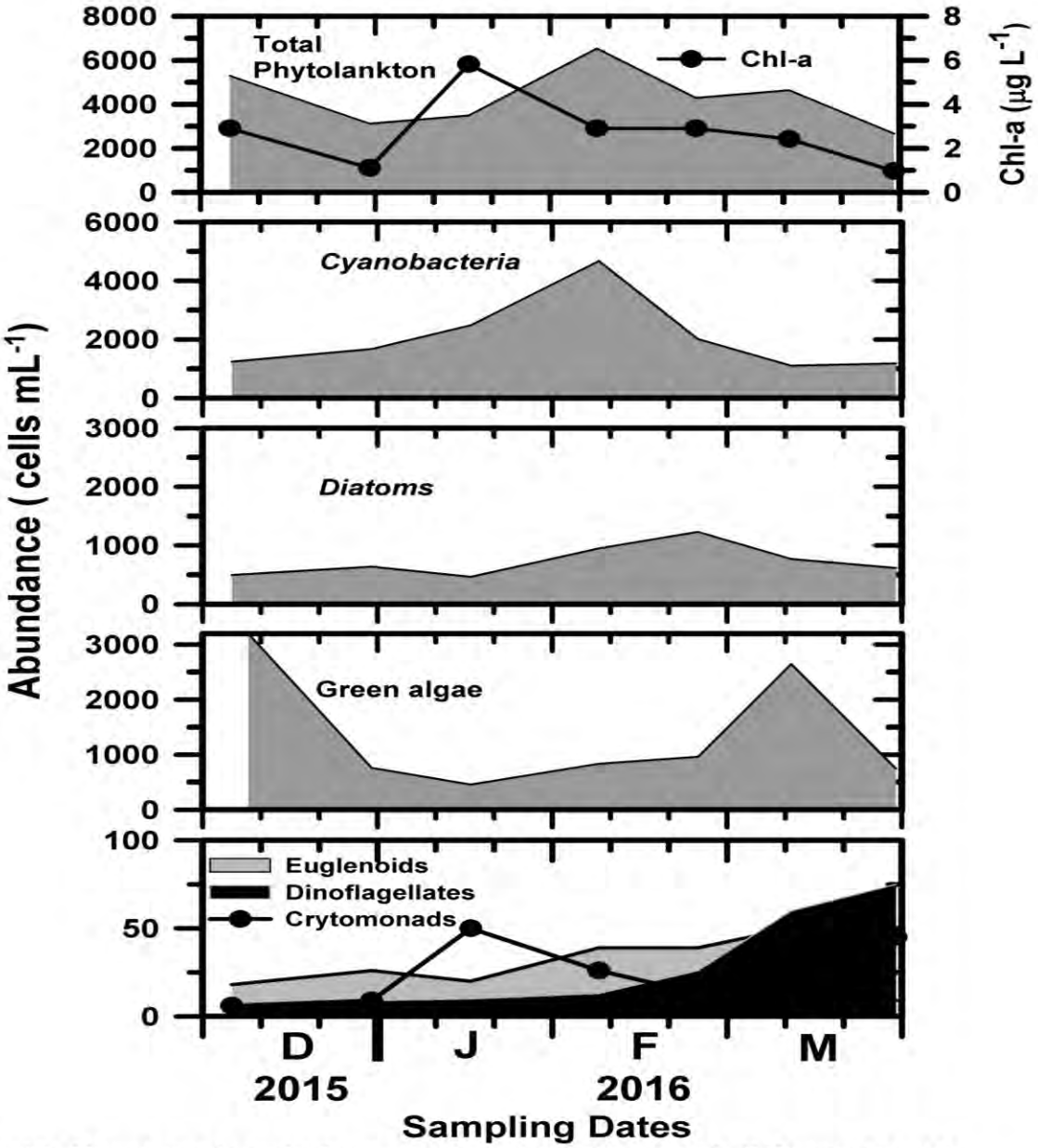

Fig. 10. Temporal variations in the abundance of different algal groups in relation to total phytoplankton abundance (area plot) and Chl-a (line plot) at the Resort Site of the present study in the Southern Gulf of Lake Tana.

The highest total phytoplankton abundance at the Resort site was observed in the first half of February concurrently with the highest peak of cyanobacterial abundance (Fig. 10). At the K. Georgis site, the highest total phytoplankton abundance occurred concomitantly with the highest diatom abundance in the second half of March (Fig. 11).

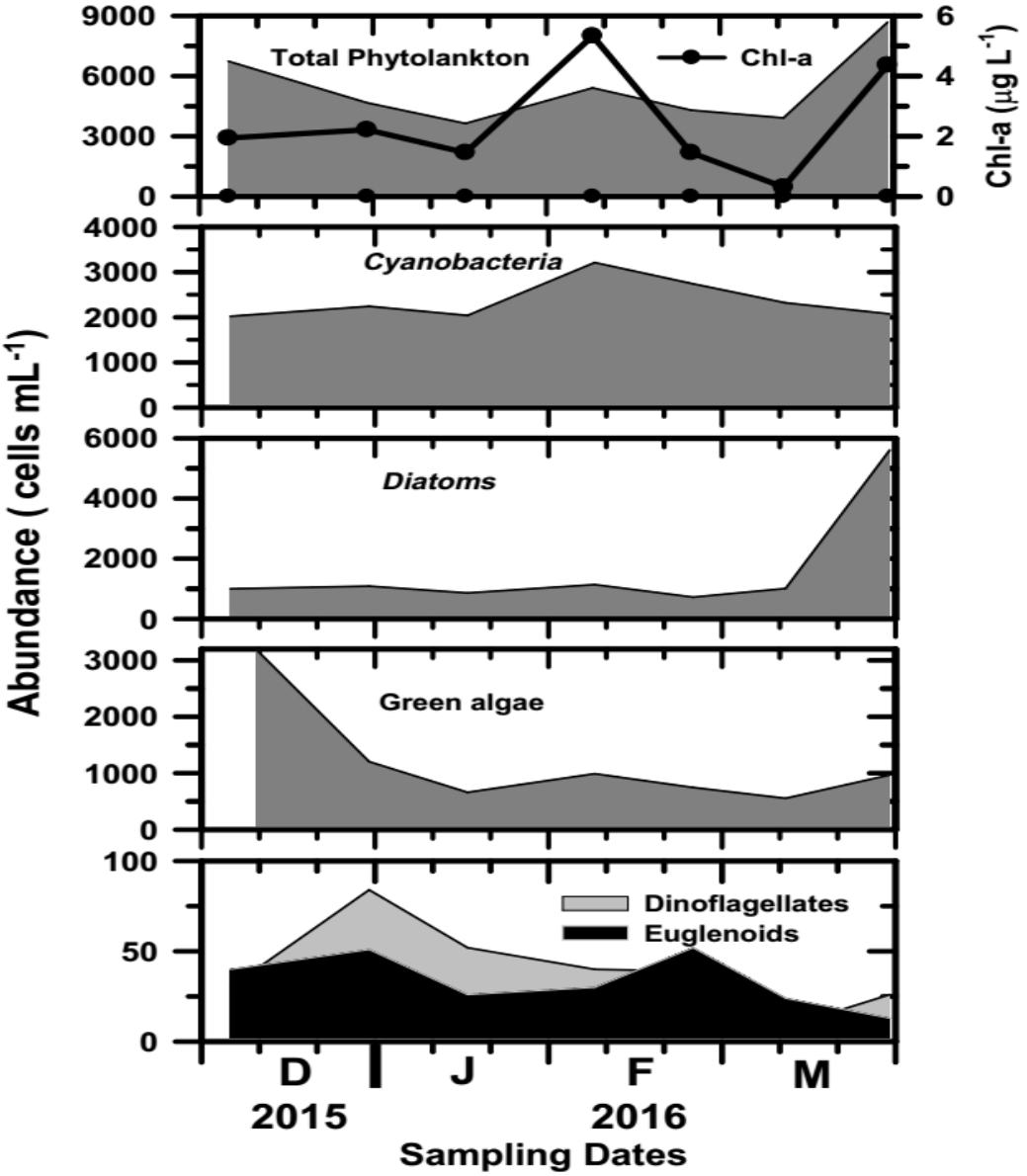

Fig. 11. Temporal variations in the abundance of different algal groups in relation to total phytoplankton abundance(area plot) and Chl-a (line plot) at the K. Georgis Site of the present study in the Southern Gulf of Lake Tana.

Slightly lower peaks of total phytoplankton abundance, which concurred with the highest abundance of green algae, were also observed at the Resort and K. Georgis sites at the beginning of January.

Cyanobacteria was the most important group in terms of abundance with its mean percentage contributions to total phytoplankton abundance varying from about 44.5 at the K. George site to about 56 at the Hospital site (Fig. 12).

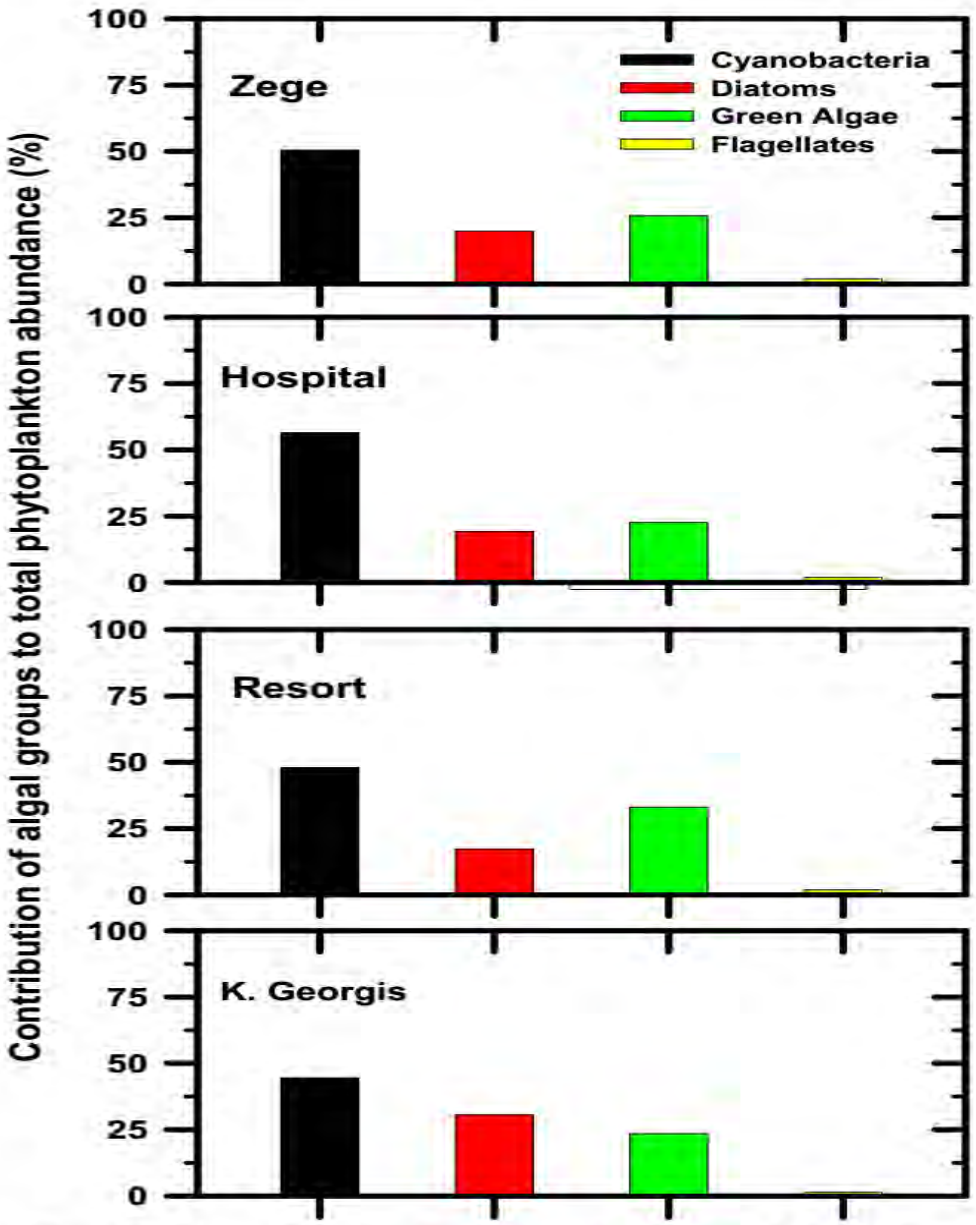

Fig. 12 Mean Percentage contributions of algal groups to total phytoplankton abundance at the different sampling sites of the present study on Southern Gulf of Lake Tana.

The highest cyanobacterial abundance corresponded to the largest peak of phytoplankton biomass measured as Chl-a at all sampling sites except the Resort site. The most dominant cyanobacterial species was *Microcystis flos-aquae* with mean percentage contributions to total cyanobacterial abundance and total phytoplankton abundance ranging from about 78 and 37 to 86 and 46, respectively (Figs. 13 and 14).

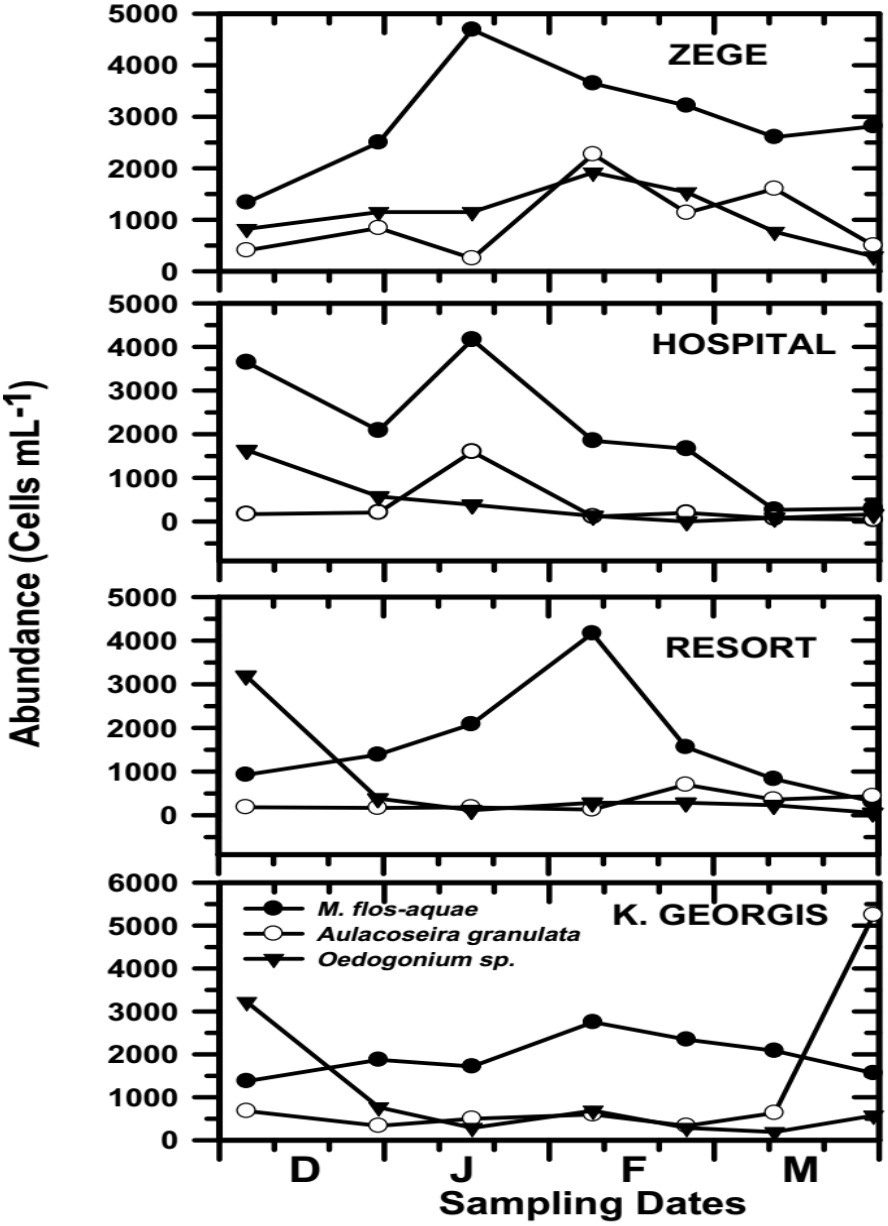

Fig. 13 Variations in the abundance of major taxa of phytoplankton at the four sampling sites of the present study on the Southern Gulf of Lake Tana.

The highest peak of abundance of *Microcystis flos-aquae* occurred in Mid-January atthe Zege and Hospital sites coinciding with the maximum phytoplankton biomass measured as Chl-a concentration. Its largest peaks of abundance at the Resort and K. Georgis sites, however, occurred in the first half of February although its association with the highest Chl-a biomass of phytoplankton was observed only for the K. Georgis site.

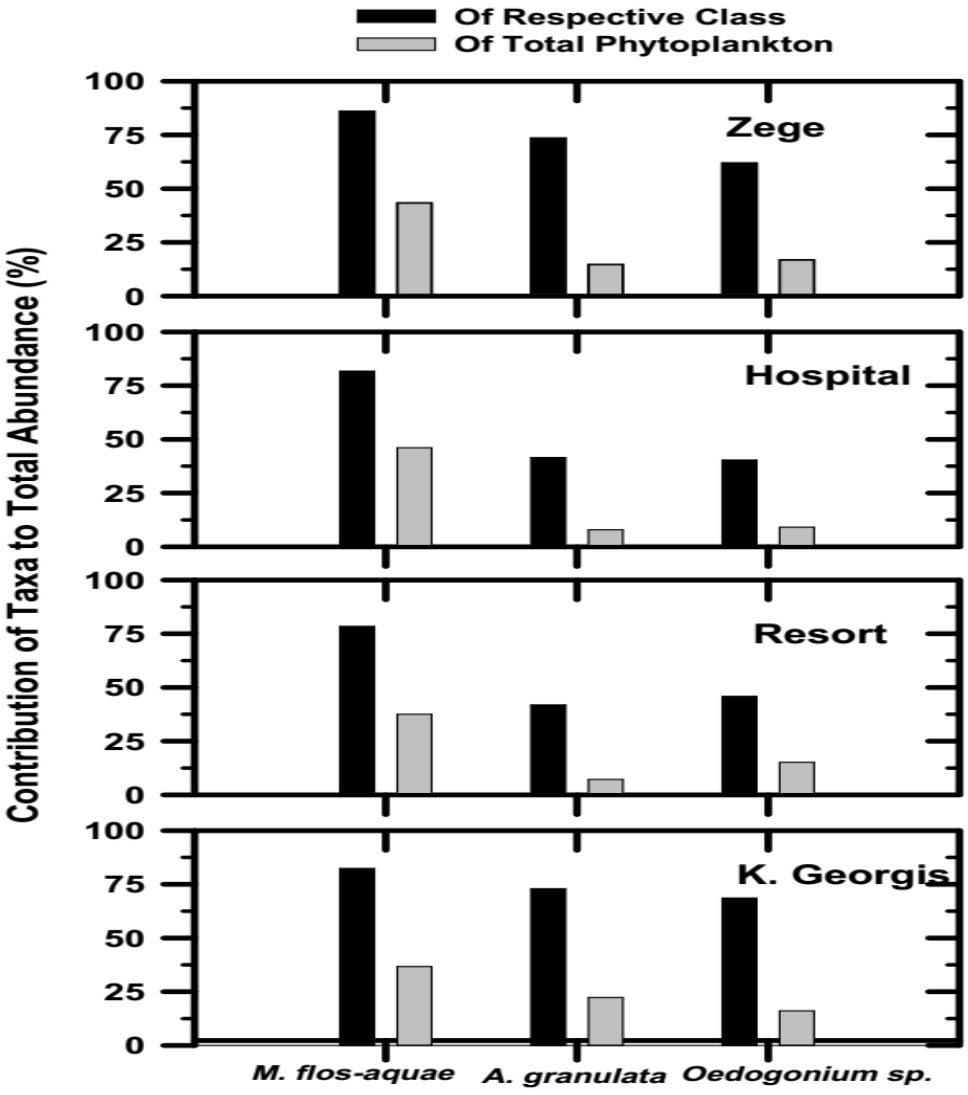

Fig. 14 Mean Percentage contributions of taxa to total abundance of their respective groups and total phytoplankton abundance at the different sampling sites of the present study in the Southern Gulf of Lake Tana.

The second most abundant group was the green algae, which was largely constituted by *Oedogonium* sp. *Oedogonium sp*. was most abundant in January before its population size abruptly declined to markedly low levels of abundance at all sampling sites except the Zege site. At the Zege site, the abundance of *Oedogonium sp*. increased consistently from its low level at the beginning of *December* to its highest peak in the first half of February before it declined consistently to its lowest level at the end of March. The mean percentage contributions of this *Oedogonium sp*.to the abundance of green algae and total phytoplankton ranged from about 40 and 9 to69 and 17, respectively. Other green algae of quantitative importance were the desmids of the genera *Cosmarium, Desmidium, Pediastrum* and *Cosmarium.*

Diatoms whose abundance was primarily constituted by *Aulacoseira granulata*had significant contributions to the total abundance of phytoplankton, with mean contributions ranging from about 17% at the Resort site to about 31% at the K. Georgis site. The largest peak of abundance of diatoms occurred at the four sampling sites on different sampling dates. Although diatoms made considerable contributions to the abundance of the algal flora at all sampling sites of the present study in the Southern Gulf of Lake Tana, an obvious association between peaks of their abundance and Chl-a biomass was observed only at the K. Georgis site at the end of March. Abundance of *Aulacoseira granulata* varied temporally in a pattern, which was closely similar to that of total diatom abundance reflecting its largest contribution to total diatom abundance.

The species-poor microflagellates (euglenoids, dinoflagellates and cryptomonads) had very low representation in the alga flora of the Southern Gulf of Lake Tana with the percentage contribution to total phytoplankton abundance of each group never exceeding 1.5%.

4.2.2. Diversity and Species Richness of Phytoplankton

The Shannon's diversity index (H', Table 3) varied temporally and spatially within somewhat narrow range (0.73-2.69). The highest Shannon diversity index was recorded for the Resort site, while the lowest was observed at the Zege site with

intermediate values for the other two sites (i.e. Zege < K. Georgis <Hospital<Resort). High diversity index values were generally associated with peaks of Chl-a biomass and peaks of abundance of Chlorophyceae or Bacillariophyceae or both. The Shannon diversity index values recorded in the present study are similar to those reported by Rediat Abate (2008) for Hora-Kilole. The species richness index (Margalef diversity, $S_{Margalef}$) varied spatially in a trend similar to that of the Shannon's diversity index. The lowest species richness was recorded for the K. Georgis site, while the highest was observed at theResort site (Table 3).

Table 3. Shannon's diversity index and Species Richness index($S_{Margalef}$)

		Mean	Mean± Std. Deviation	Minimum	Maximum	Sig.
Shannon diversity index	Zege	1.134	1.134 ± 0.39	0.730	1.921	
	Hospital	1.960	1.960 ± 0.24	1.709	2.318	
	Resort	2.043	2.043 ± 0.30	1.826	2.688	0.391
	K. Georgis	1.307	1.307 ± 0.13	1.111	1.467	
	Average	1.610	1.610 ± 0.48	0.730	2.688	
Species richness index	Zege	0.913	0.913 ± 0.24	0.602	1.143	
	Hospital	1.389	1.389 ± 0.17	1.081	1.602	
	Resort	1.479	1.479 ± 0.36	1.166	2.151	0.099
	K. Georgis	1.104	1.104 ± 0.15	0.964	1.331	
	Average	1.221	1.221 ± 0.33	0.602	2.151	

4.2.3. Chl-a biomass of Phytoplankton

Biomass of phytoplankton measured as Chl-a varied both spatially and temporally (Figs. 8 to 11, Fig. 15) although the observed variations were not statistically significant ($p>0.05$). Mean Chl-a concentration at the Zege site was much higher than those of other sampling sites (Fig. 15). The maximum Chl-a

values at all sampling sites except the Resort site were associated with the highest peaks of abundance of cyanobacteria and *Microcystis flos-aquae*.

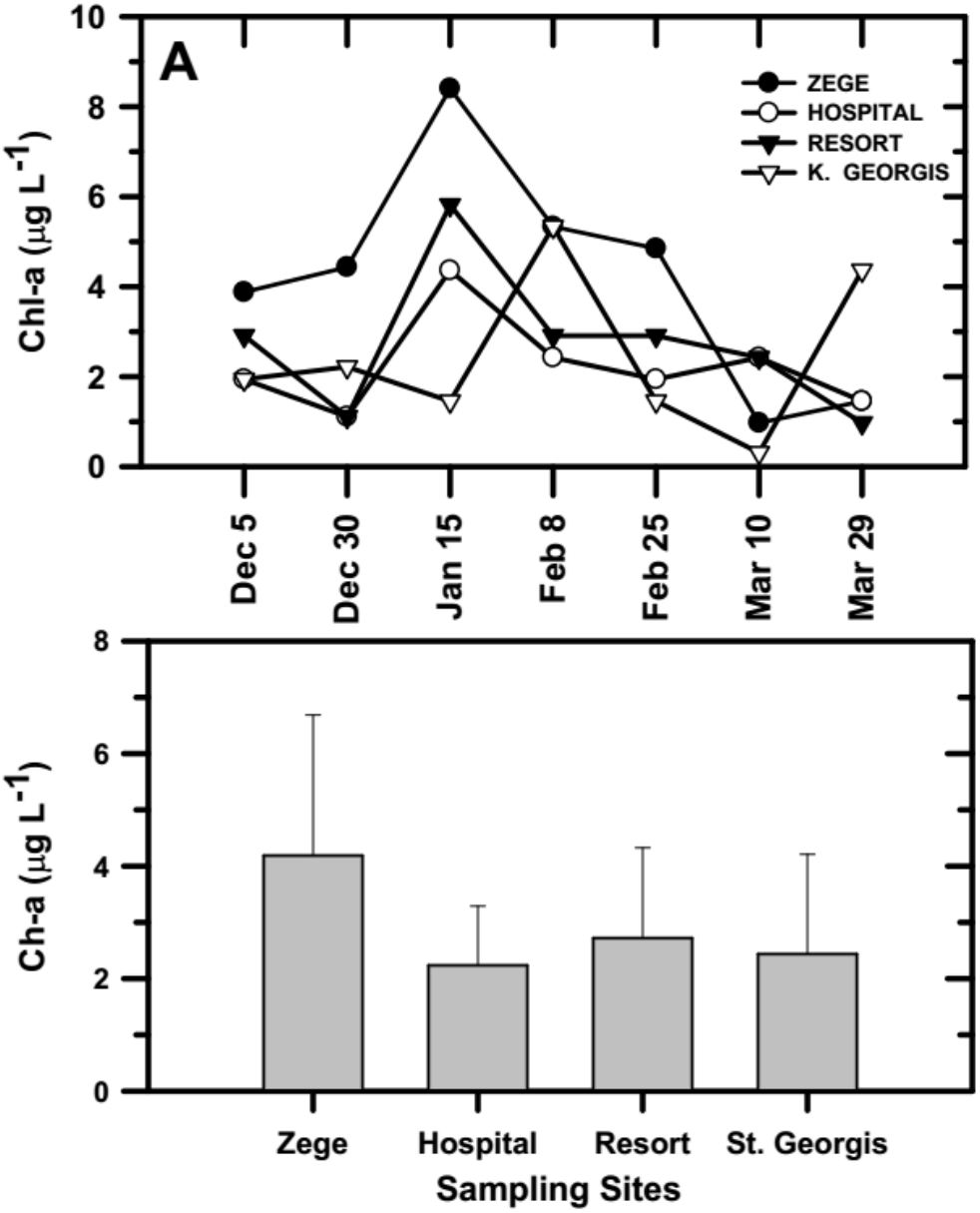

Fig. 15 Variations in the levels of Chl-a: Individual observations(A) and mean values (B) of the four sampling sites of the present study in the Southern Gulf of Lake Tana.

A Chl-a peak, which was slightly lower than the highest peak, was also observed at the end of March at the K. Georgis site coincident with the highest abundance of total phytoplankton, total diatoms and *Aulacoseira granulate* (Fig.11). Chl-a

concentration (all data combined) was negatively but significantly correlated with specific conductivity (at $p<0.01$) and TDS (at $p<0.05$), while its correlation with SiO_2 was positive and statistically significant (at $p<0.01$). Chl-a was also correlated negatively with SRP, TP, NO_3-N, and NO_2-N, although the correlations were not statistically significant ($p>0.05$).

4.3. Trophic State Index

Trophic state index values calculated using Secchi depth (m), total phosphorus (TP) and Chl-a data of all sampling sites based on the models developed for Temperate and Tropical waters are given in Table 4.

Table 4. Trophic State Index values estimated for the Southern Gulf of Lake Tana according to Carlson,1977 and Cunha et al., 2013

Models	Parameters used	Trophic State Index (TSI)	Values estimated for the Southern Gulf of Lake Tana	Trophic state Category
Carlson (1977)	SD	TSI_{SD}	57.62	Slightly eutrophic
	TP	TSI_{TP}	87.15	Hypereutrophic
	Chl-a	TSI_{Chl}	41.04	Slightly mesotrophic
	Total	TSI_{C}	61.93	Eutrophic
Cunha *et al*. (2013)	TP	$TSI\ (TP)_{tsr}$	64.63	Hypereutrophic
	Chl-a	$TSI(Chl)_{tsr}$	59.14	Hypereutrophic
	Total	TSI_{tsr}	61.89	Hypereutrophic

The mean values of TSI_{TP}, TSI_{SD} and TSI_{Chl} were 87.15, 57.62 and 41.04, while TSI_{C} was 61.93 based on Carlson (1977). The TSI_{TP} value suggested that the Southern Gulf was at a hypereutrophic state, while TSI_{SD} and TSI_{chl} implied slightly eutrophic and mesotrophic states, respectively (Table 4). According to Cunha *et al.*, (2013), $TSI\ (TP)_{tsr}$ and $TSI(Chl)_{tsr}$ are 64.63 and 59.14, respectively, while TSI_{tsr} is 61.89. These results seem to suggest the prevalence of hypereutrophic state in the Southern Gulf of Lake Tana.

4.4. Phytoplankton-Environment Relationships

The relationship between phytoplankton abundance/biomass and physico-chemical variables is shown in the ordination biplot (Fig.16). The first axis (horizontal) of the RDA accounted for 78% of the total variance of the biological parameters and showed negative correlation with phosphate (SRP), Ammonia (NH_3 + NH_4^+-N) and TDS. The second axis accounted for 17% of the variance in species–environmental relationship and is positively correlated with TSS, SiO_2, NO_3-N and Chl-a.

Table 5. Results of Redundancy Analysis (RDA) of the relationship between environmental variables and phytoplankton using the first two Axes

Parameter	**Axis1**	**Axis 2**
Eigenvalues	0.780	0.169
% variance of species-environment relation	78	17
TDS	-0.9419	-0.2194
DO	0.8360	-0.5145
TSS	0.6811	0.6919
Phosphate(PO_4-P)	-0.1419	-0.0130
Ammonia(NH_3-N)	-0.6492	-0.3866
Nitrate(NO_3-N)	0.7309	0.0363
Silica(SiO_2)	0.6297	0.0640

The first two ordination axes collectively explained 95% of the variance in phytoplankton abundance/biomass in the Southern Gulf of Lake Tana (Table 5).

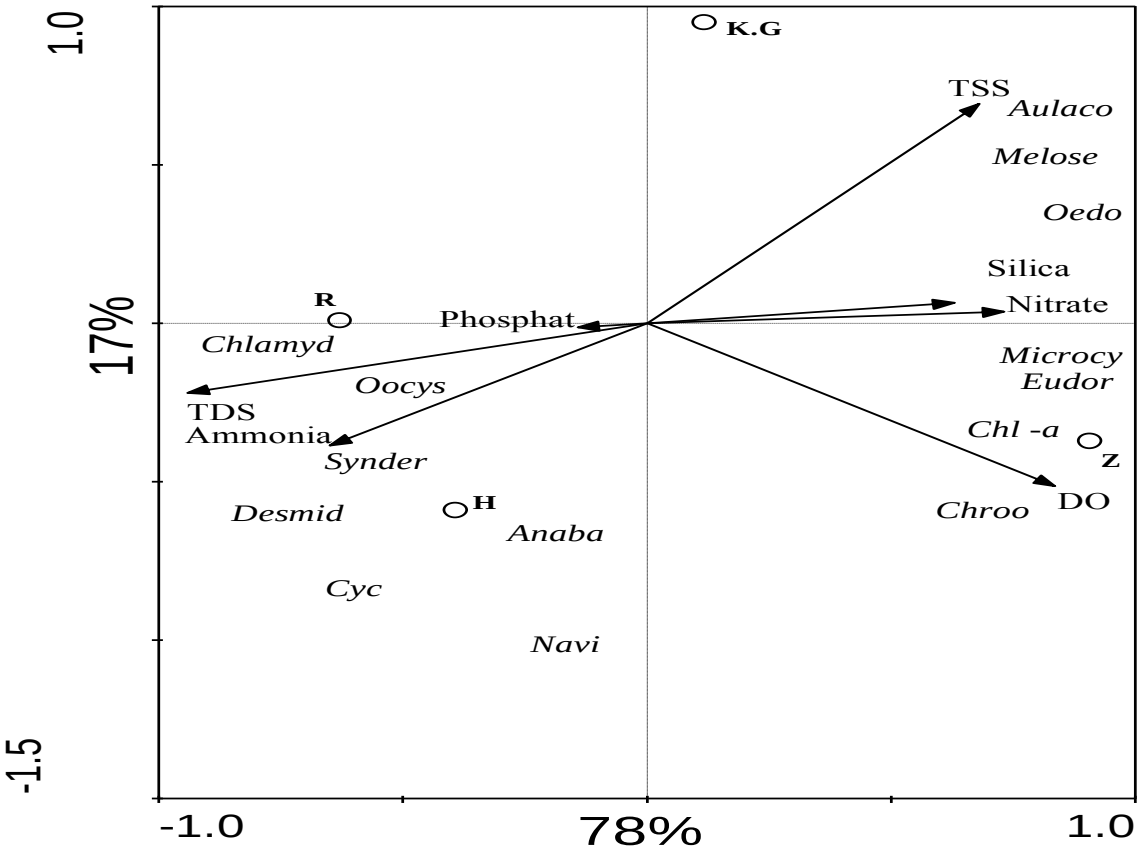

Figure 16. Ordination diagram of Redundancy Analysis (RDA) of the first two ordination axes summarizing the relationship between physico-chemical variables and phytoplankton taxa. (Z = Zege site, H = Hospital site, R = Resort site, K. Georgis site, Oocys = *Oocystis sp.*, Chlamyd = *Chlamydomonas sp.*, Synder = *Syndera sp.*, Desmid = *Desmidium sp.*, Anaba = Anabaena sp., Cyc = *Cycelotella sp.*, Navi = *Navicula* sp., Oedo = *Oedogonium sp.*, Aulaco = *Aulacoseria granulata*, Melose = *Meloseria sp.*, Eudor = *Eudorina sp.*, Microcy = *Microcystis sp.*, Chroo = *Chrococus sp.*, Chl-a = Chlorophyll- a, TSS = Toktal suspended solid, TDS = Total dissolved solids, DO = Dissolved oxygen, Phosphat = Phosphate).

The density of the diatom spp. (i.e. *Aulacoseira granulata*, *Meloseira sp.*, *Cyclotella sp., Navicula sp.)* was positively correlated with the levels of silica. The cyanobacterial spp. (i.e. *Microcystis spp.*, *Chroococus sp.*) and the green algae (i.e. *Eudorina sp.*, *Odeogonium sp.*) were positively correlated with NO_3-N, DOand Chl-a concentrations, and negatively with the levels of phosphate and ammonia. *Chlamydomonas sp.*, *Oocystis sp.*, and *Desmidium sp.* were negatively correlated with ammonia and TDS concentrations.

5. DISCUSSION

5.1. Physico-chemical features

Depths of all sampling sites decreased almost consistently from their maxima in December to their minima in March, presumably in association with the continuously decreasing input from the Feeder Rivers and increasing loss of surface water due to evaporation and water abstraction for various purposes. Mean depth of sampling sites varied between 1.64 of the hospital site and 8.6 of the Zege site, with the depth of the Zege site being significantly ($P<0.05$) deeper than those of the near-shore sites. The very shallow water columns of the near-shore sites are expectable during the months of the present study period (largely dry period) as evaporation losses exceed input via rainfall (Molla and Menelik,2004) and water use for hydroelectric power generation is especially high during the same period (Dec. to April) (Ayalew Wondie *et al.*, 2007).

Mean surface water temperatures exhibited a trend of decreasing levels from the open water site (Zege) to K. Georgis site(Fig.4), which may be attributed to the effect of macrophytes on heat transfer from the near-surface region to deeper parts of the water column through vertical mixing (Wetzel. 2001) orthe difference in the amount of heat transported through inflows**.** Surface water temperatures of all sampling sites were within the range of variation reported for most tropical water bodies (20-30 ºC, Talling and Lemoalle, 1998) including the Ethiopian Rift Valley lakes (Elsabeth Kebede, 1996). They are also similar to those reported previously for the Bahir Dar Gulf of Lake Tana by Akoma (2010) and Flipos and Subramanian (2015). Although depth profiles of temperature and oxygen were not determined in the present study, all the sampling sites can be considered frequently mixing considering their shallowness and frequent exposure to southerly winds of the present study period (Gasse, 1987, cited in Ayalew Wondie, 2007). All sampling sites except the Zege site, which had a mean depth of about 8 m, had water columns shallower than 2 m. Complete mixing is normally frequent in such water bodies as Lake Ziway and Koka Reservoir, which have a maximum depth (Z_{max}) of less than about 15-30 m (Baxter *et al.*, 1665;

Elizabeth Kebede, 1996; Talling and Lemoalle, 1998). Eshete Dejen *et al.* (2004) also reported that Lake Tana was well-mixed with mostly superficial type of thermal stratification- without deep-seated and persistent thermal stratification.

Lake Tana was described as a highly turbid lake with low biological productivity (Tewodros Taffese, *et al.*, 2014). The present data on Secchi depth are not, however, consistent with the above assertion as well as the low transparency reported by Ayalew Wondie *et al.* (2007) for the same lake. Mean values of the Secchi depths recorded in the present study are about 1.5 to 2 times those reported by Ayalew Wondie *et al.* (2007) for the inshore (35.4 cm) and open water (43.3 cm) zones of Lake Tana. The noticeable difference between the two sets of data may have resulted from the spatial separation of the sampling sites of the two studies made on this tremendously large lake. The difference may also be due to the inclusion of data from the rainy seasons in the previous studies.

It is interesting to note that Secchi depths of the relatively deep open water site (Zege) were significantly (at $p < 0.05$) smaller than those of the near-shore sites of the present study. Levels of TSS and Turbidity at the offshore site were also significantly (at $p<0.01$) higher than those of near-shore sites(Fig.2), these contrastswith the results of Ayalew Wondie *et al.* (2007) who reported that water transparency was significantly lower in the inshore zone than in the open water zone (Mann–Whitney U-test: $P < 0.001$). Their results are consistent with the generalization that shallow littoral sites are more vulnerable to wind-induced turbulent mixing and the consequent increased turbidity associated with the resuspension of sedimented particulate materials than the deeper open water sites (Dokulil, 1994). In water bodies fringed with dense macrophyte vegetation, concentrations of suspended particulate matter, which are significantly lower at littoral sites than at open water sites may, however, result from the presumably reduced wave action associated with the shelter from wind afforded by the nearby macrophyte stands. Horppila and Nurminen, (2001) also argued that emergent macrophytes could reduce sediment resuspension thereby decreasing turbidity and TSS content of near-surface waters. TSS and turbidity were also found to be

at relatively lower levels within the stands of emergent macrophytes than at open water sites in Lake Ziway (Mesfin GebreHiwot, Unpublished data).

The highest mean TSS recorded for the K. Georgis site (13.64) may be attributable to the proximity of the sampling site to the terrestrial environment from which there could be increased influx of particulate materials. As Secchi depths much smaller than those recorded in the present study are highly likely during the major rainy season due to the high silt loads from inflowing rivers and the lake bottom through resuspension by turbulent mixing associated with high winds of the same period (Ayalew Wondie *et al.* 2007), mean Secchi depths of the lake would inevitably be low.The wetland through which urban wastewaters enter the lake may have also slowed down the flow of the wastewater and filtered it thereby reducing its TSS content (Nikolić, *et al.*, 2009; Ramsar Convention Secretariat, 2013).The observed highly significant correlation between TSS and Secchi depth indicates that water transparency in the Southern Gulf of Lake Tana was primarily controlled by the concentrations of suspended sediment particles rather than by phytoplankton, Similar results were also reported by Rediat Abate(2008) for Hora-Kilole and Yeshiemebet Major (2016) for Koka Reservoir.

The present results clearly show that the Southern Gulf of Lake Tana is a very dilute freshwater, with conductivity (μS cm^{-1}) levels slightly higher than the upper boundary values of the ranges reported for Geffersa (68-136, Nigatu Ebisa, 2010) and Legedadi (63-131, Adane Sirage, 2006) reservoirs, in Ethiopia. The mean values (μS cm^{-1}) recorded for weed-infested (150) and non-infested (155) sites of the study by Wondie Zelalem (2013) in the northeastern part of Lake Tana are, however, closely similar to those of the present study**.** In his hydrobiological survey of the Bahir Dar Gulf of Lake Tana, Akoma (2010) has, however, reported considerably higher levels of conductivity on some of his sampling dates (152 to 232 μS cm^{-1}), which may be attributable to the spatial separation of his study area from that of the study reported here. There was a general trend of lower conductivity values at the Zege site than at the other

sampling sites, although the difference was not statistically significant ($p > 0.05$). This may be attributable to the influx of dissolved solutes from the surrounding wetlands or urban wastes into the water masses at the Hospital, Resort and K. Georgis sites(Fig.3). The observed low conductivity of the Southern Gulf of Lake Tana, which was also reflected in the low levels of TDS, is an indication of the low salinity of the lake and that of the inflowing rivers and influent wastewaters. The maximum conductivity value of the Southern Gulf of Lake Tana is, however, within the range that drinking water should have (25-250 $\mu S\ cm^{-1}$) (WHO, 1996). According to WHO (2011), the palatability of the water of the present sampling sites, whose mean total dissolved solids (TDS) were less than 600 mg L^{-1}, is considered to be good.

The significant difference between the pH of the Zege site and those of the near-shore sites(Fig.7) is attributable to the extent of photosynthetic removal of carbon dioxide (Reynolds, 2006) and the amount and nature of chemical species introduced through inflows (Kalff, 2000). The slightly alkaline or acidic pH levels recorded for the present sampling sites in Lake Tana have not so far been reported for lakes of the main Ethiopian Rift Valley (Wood and Talling, 1988; Elizabeth Kebede, 1996) or the crater lakes found in and around Debre Zeit (Tudorancea and Taylor, 2002). Similarly, low levels of pH have, however, been reported for the present study lake by different investigators (e.g. Flipos and Subramanian, 2015; Yirga Kebede Wondim and Hassen Muhabaw Mosa, 2015). Wondie Zelalem (2013) also documented slightly acidic pH levels for non-weed-infested sites in the northeastern part of the same lake. The present pH levels of the Southern Gulf of Lake Tana are, however, within the range (6.5-8.5) regarded as being optimal for both potable water as well as for the survival of aquatic organisms (WHO, 2004; Arain *et al.*, 2008).

pH, which can affect the solubility and toxicity of metals in a lake water (Exley, 2003), is influenced by many factors, both natural and man-made, including interactions of lake water with surrounding rocks or influx of wastewater or mining discharges (Warfinge *et al.*, 1995; Engstrom *et al.*, 2000; Boyle, 2008). In

addition, CO_2, the most common cause of acidity in water, can influence pH levels. pH can also be strongly influenced by the organic acids in DOC (Bishop *et al.*, 2004; Erlandsson *et al.*, 2008). The near-shore sites of the present study are covered with dense macrophyte vegetation. High concentrations of dissolved organic carbon emanate from macrophytes, a considerable proportion of which is organic acids (Wetzel, 2001). Because of the continuousoxidative decomposition of the abundant and readily available decomposable organic matter, large quantity of carbon dioxide is generated and accumulated in the macrophyte zone. Urban waste waters, which ultimately find their ways into the lake water, may have acidic constituents. The organic acids of aquatic and terrestrial origin, accumulated carbon dioxide and acidic materials associated with wastewaters can, therefore, result in the slightly acidic conditions observed in the present study considering the low buffering capacity of the lake water implied by its low alkalinity (Wood and Talling, 1988). Based on the inverse relationship they observed between sediment pH and organic matter (r^2 = -0.626), Yirga Kebede Wondim and Hassen Muhabaw Mosa (2015) also argued that decomposition of organic matter releases organic acids into the sediment thereby decreasing its pH and that of the overlying water.

The observed significant correlation between DO concentration and water temperature and Chl-a seem to suggest that the oxygen level in the near-surface water of the Southern Gulf of Lake Tana is primarily a function of water temperature and photosynthetic biomass of phytoplankton. The DO levels (mg L^{-1}), which varied from 4.39 of the K. Georgis site in March, to 6.82 of the Zege site in the first half of February(Fig.4), were almost always above the minimum value (5 mg L^{-1}) recommended for fish and plankton (USEPA, 2008; Yajurvedi, 2008). The minimum DO concentration observed at the K. Georgis site in March concurred with the highest peak of phytoplankton abundance and the second largest peak of Chl-a biomass. The observed lowest level of oxygen may, therefore, have resulted from the reduced solubility of the gas associated with high temperature and its depletion due to oxidative decomposition of

autochthonous (primarily of macrophyte origin) and allochthonous organic matter associated with urban wastewater. Wondie Zelalem (2013) also recorded some oxygen levels lower than 5 mg L^{-1} for weed-infested (4.8) and non-infested (4.6) sampling sites in the northeastern part of the same lake. Dissolved oxygen concentration values as low as 3.5 and 1.66 mg L^{-1} were previously reported for Bahir Dar Gulf of Lake Tana from sampling sites located near Tana Hotel/Shum-Abo Resort and Bahir Dar Resort, respectively by Akoma (2010). The latter site was adjacent to one of the lake wetlands used extensively for livestock grazing, which seems to corroborate the overriding importance of oxidative decomposition of organic matter to oxygen levels in the near-shore sites of Lake Tana. For Welala and Shesher wetlands of Lake Tana Sub-Basin, Tarekgne Wondmagegne*et al.* (2012) reported significantly higher dissolved oxygen levels that varied between 4.7 and 10.4 mg L^{-1} forWelala and Shesher Wetlands, Lake Tana Sub-Basin.

Levels of inorganic nutrients, particularly nitrate, were generally low (< 300 µg L^{-1}) in tropical freshwater lakes including those found in the Ethiopian Rift Valley and the Bishoftu crater lakes (Wood and Talling, 1988; Elizabeth Kebede, *et al.* 1994; Zinabu Gebre-Mariam, *et al.*, 2002; Tudorancea and Taylor, 2002). However, the present mean levels of nitrate, which varied between 0.875 and 1.257 mg L^{-1}, are remarkably high (Fig. 5). Ayalew Wodie *et al.* (2007) reported relatively lower nitrate levels that ranged from about 0.2 to 0.6 mg L^{-1} for the open water of Lake Tana during the major rainy season. A more recent study by Akoma (2010) has, however, recorded levels of nitrate, which are much higher than those observed in the present study, with their mean concentrations varying between 1.92 and 2.61 mg L^{-1} and with some values recorded during the major rainy period surpassing 3 mg L^{-1}. Wondie Zelalem (2013) also reported similarly high mean nitrate levels, which averaged 1.124 and 1.485 mg L^{-1} for weed-infested and non-infested sites in the northeastern part of the same lake. Uncommonly high levels of nitrate have also been reported for a near-shore site in Koka Reservoir (mean: 0.512 mg L^{-1}, Yeshiemebet Major, 2016) and for near-shore and open water sites in Selameko Reservoir, Gondar (mean= 0.71 and

0.758 mg L^{-1}, respectively; Tilahun Adugna Wassie and Ayalew Wondie Melese, 2015). The nitrate concentrations of the present study are below the maximum level tolerable by fish and other aquatic organisms (<5 mg L^{-1}, Chattopadhyay and Banerjee 2007) and the guideline value set to protect public health (< 50 mg L^{-1}, WHO, 1996). The remarkably high levels of nitrate in Lake Tana clearly reflect the extent of environmental degradation to which the lake has been subjected. The concentrations of nitrite were always much lower than those of nitrate and ammonia as they usually are in African lakes (Walmsley and Butty, 1980; Elizabeth Kebede *et al.*, 1994; Talling and Lemoalle, 1998).

Considering the fact that ammonia-nitrogen in well-oxygen shallow waters is usually low due to its prompt oxidation to nitrate and rapid and preferential uptake by phytoplankton (Eppley *et al.* 1969; Liao and Lean, 1978), its present levels (in mg L^{-1}) in the Southern Gulf can be regarded as high. With mean values ranging from 0.06 of the Zege site to 0.216 of the Hospital site (Fig. 5). Although comparison of the present results with those reported previously for the same area of Lake Tana is precluded by the absence of previous data on ammonia-nitrogen, the results documented by Wondie Zelalem (2013) for the weed-infested and non-infested sites in the northeastern part of this large lake (means: 0.175 and 0.15 mg L^{-1}, respectively), and Yeshiemebet Major (2016) for offshore and inshore sites in Koka Reservoir (means: 0.296 and 0.215 mg L^{-1}, respectively) show that high levels of ammonia are possible in well-oxygenated shallow lake and reservoir waters.

SRP (in mg L^{-1}) was also quite high, with mean values ranging from 0.237 of the Hospital site to 0.41 of the Zege site, in comparison with the concentrations recorded for 6 open water sites (< 0.1) in Lake Tana by Ayalew Wondie *et al.* (2007) and those known from most Ethiopian lakes (Zinabu Gebre-Mariam, *et al.*, 2002; Tudorancea and Taylor, 2002). Incredibly high levels of SRP (mg L^{-1}) were also reported by Wondie Zelalem (2013) for weed-infested (0 - 4.4; mean: 0.951) and non-infested (0 - 6.8; mean: 0.65) sites of his investigation in the northeastern part of the same lake. TP of the present study was found to be

slightly higher (with means ranging from 0.269 to 0.448 mg L^{-1}) than SRP at all sampling sites (Fig. 6).

The observed high levels of inorganic nitrogen and phosphorus sources seem to have resulted from their external loading through terrestrial runoff and influent domestic and industrial waste waters as well as internal loading through water column mixing considering the shallowness and exposure to wind action of the present sampling sites. Based on the significant positive correlations they observed between sediment organic matter and available sediment nitrogen and phosphorus in Lake Tana, Yirga Kebede Wondim and Hassen Muhabaw Mosa (2015) also contended that major portions of the nitrate and phosphate found in the lake water emanate from internal loading.

As is the case with many African shallow waters (Talling, 1992), silica concentration in the Southern Gulf was generally much higher than the level (< 0. 3 mg L^{-1}, Reynolds, 2006) at which the possibility of limitation of diatom growth might be suspected, even at the time of the peak abundance of *Aulacoseira granulata* at the St. Georgis site. Owing to the greater mobility of Si in most tropical soils, and the importance of groundwater inputs for many lakes, concentrations over 10 mg SiO_2 L^{-1} are common in African lakes (Talling and Talling, 1965; Talling, 1992) including the Ethiopian freshwater lakes (Wood and Talling, 1988; Demeke Kifle and Amha Belay, 1990; Elizabeth Kebede *et al.*, 1994; Zinabu Gebre Mariam *et al.*, 2002). Wondie Zelalem (2013) has, however, reported silica levels (mg L^{-1}), which were generally uncommonly low for tropical waters from weed-infested (0.04 - 0.56; mean: 0.26) and non-infested (0.16 - 3.6; mean: 0.699) sampling sites in the northeastern part of Lake Tana. The presence or absence of diatoms as major contributors to the algal flora may markedly influence the variation of silica in surface waters (Talling and Lemoalle, 1998). The general presence of silica at levels lower than or close to that regarded as limiting to diatom growth (0.3 mg L^{-1}, Reynolds, 2006) may be attributed to the year-round quantitative importance of diatoms in the northeastern part of Lake Tana (Wondie Zelalem, 2013). A four-fold increase in silicate concentration at 70

m depth relative to surface water was also noted by Baxter and Golobitsh (1971) for Lake Hayq, where diatoms were believed to be a major constituent of the then sparse algal flora. Baxter and Golobitsh (1971) also observed a fall from 20 $mg_{L^{-1}}$ SiO_2in the inflowing Ancherca River to 1 mg L^{-1} in the surface waters of the lake. In the hyposaline crater lake, Lake Babogaya (Pawlo), which is fed by groundwater and where diatoms play a very minor role in the phytoplankton assemblage, a representative concentration at the lake's surface was 77 mgL^{-1} SiO_2 rising to 96 mg L^{-1} in the hypolimnion (Wood *et at.,* 1984). In the second half of February, silica levels in Lake Tana, however, declined to concentrations close to or below that potentially limiting to diatom growth (0.2- 2.0 mg L^{-1}) at the Hospital, Resort and K. Georgis sites although the largest peak of diatom abundance coincided with silica minimum only at the Resort Site (Fig. 7). Furthermore, despite the quantitatively important role of diatoms in the second half of February at the Zege site, silica level remained at a very high level.

5.2. Biological features

The observed dominance of phytoplankton communities in Lake Tana by cyanobacteria (Cyanophyceae), green algae (Chlorophyceae) and diatoms Bacillariophyceae) is a common feature of tropical lakes and has been reported for lakes and reservoirs in Ethiopia including lakes Awassa, Chamo and Ziway (Girma Tilahun, 2006), and Koka reservoir (Yeshiemebet Major, 2016). Similar results were also obtained previously for the Southern Gulf of Lake Tana by Dilnessa Gashaye (2016). Chlorophyceae was the most species-rich taxonomic group, rivaled only by diatoms although most of its constituent species were generally rare and sparsely populated. Similar order of dominance of algal groups in term of species richness was also reported by Flipos and Subramanian (2015) for Lake Tanaat the Bahir Dar Gulf of Tana.These observations are consistent with the generalization that chlorophytes are among the few species-rich taxonomic groups in most tropical and temperate lakes and reservoirs (Reynolds, 2006). It is interesting to note that although Chlorophyceae was also the most

species-rich group in the weed-infested sites of the northeastern part of Lake Tana, this is a trend, which contrasts with that documented for the open water (non-weed-infested) sites of the northeastern part of Lake Tana in which diatoms were found to be the most species-rich and most abundant throughout the study period (Wondie Zelalem, 2013).

The mean values of the Shannon's diversity index (bits/individual) for the present sampling sites in the Southern Gulf varied within a narrow range, from 1.134 of the Zege site to 2.043 of the Resort site (mean= 1.61) (Table. 3). Similarly low values and narrow ranges of the index were also reported for the northeastern part of the same lake (0.47-2.61; mean: 1.87) by Wondie Zelalem (2013) and for Hora-Kilole (1.3-3.11; mean: 2.25) by Rediat Abate (2008). Compared to those of such freshwater bodies as Awassa, Chamo, and Ziway (Elizabeth Kebede, 1996; Girma Tilahun, 2006) of the Ethiopian Rift Valley, the phytoplankton assemblages of the Southern Gulf of Lake Tana had low species diversity, which was also reflected in the estimated values of the Shannon diversity index. Low species diversity of phytoplankton was also reported for the dilute freshwater Crater Lake, Hora-Kilole (Rediat Abate, 2008). The relatively high values of Shannon's diversity index of all sampling sites resulted primarily from the dominance of the species-rich taxonomic groups (Chlorophyceae and Bacillariophyceae), which were at their peaks of abundance. Generally speaking, the diversity index was low, which may be attributed to the predominance of cyanobacteria throughout the study period. Abundance of cyanobacteria often leads to a significantly decreased diversity and an impairment of many ecosystem functions (Paerl and Huisman, 2009) owing to their tendency to suppress other algae through excretion of organic compounds that hinder the growth of other algae (Dokulil and Teubner, 2000). The observed association of high diversity index values with peaks of Chl-a biomass is common since more diverse communities are more likely to include species that are especially effective in capturing resources and converting them into biomass (Loreau, 2001). Ptacnik *et al.* (2008) and Dodson*et al.* (2000) also showed that resource

use and, in turn, biomass production are directly and positively linked to diversity in phytoplankton communities. As has been shown by several investigators (e.g. Sager and Hasler, 1969; Dodson *et al.*, 2000; Reynolds, 2006), high species richness(SR) corresponded to high diversity index and high Chl-a biomass. The increase in phytoplankton SR with increased productivity(biomass)is linked to the more efficient uptake of resources in species-rich communities; either because species use resources in complementary ways (Loreau, 2001; Hooper *et al.*, 2005) or because increased richness increases the likelihood of including highly productive and dominant species (positive selection effects) (Huston, 1997).

The dominance of the phytoplankton community in the lake by cyanobacteria, which was constituted primarily by species of the genus *Microcystis,* is a trend of common prevalence in eutrophic and hypereutrophic tropical lakes and reservoirs (Paerl, 1996). The occurrence of cyanobacteria-dominated phytoplankton assemblages throughout the study period in the Southern Gulf is consistent with the results reported by Flipos and Subramanian (2015) and Dilnessa Gashaye (2016) for the same region of the present study lake (Fig. 8-14). *Microcystis* spp. are potentially toxic because some species produce powerful hepatotoxins called microcystins that initiate cancer and promote tumor formation in the liver of humans and wildlife (Zegura *et al.*, 2003; IARC, 2006). They also produce a surface scum that impedes recreation, reduces aesthetics, lowers dissolved oxygen concentration, and causes taste and odor problems in drinking water (Paerl *et al.*, 2001).

Cyanobacterial dominance, constituted largely by *Microcystis spp.*, is a common event, which has been observed in numerous shallow water bodies worldwide including lakes (Girma Tilahun, 2006; Tadesse Ogato, 2007) and reservoirs (Adane Sirage, 2006; Hadgembes Tesfay, 2007; Tadesse Dejenie *et al.*, 2008) in Ethiopia. Dominance of cyanobacteria is regarded as a sign of eutrophic conditions and poor ecological status of reservoirs and lakes (Paerl *et al.*, 2001). Dominance of cyanobacteria in lakes and reservoirs has been attributed to a

number of environmental variables including low light, high temperature, buoyancy regulation, high phosphate and nitrate levels and immunity to zooplankton grazing (Smith, 1986; Reynolds, 1987; Shapiro, 1990; Gonzalez, 2000). The dominance of cyanobacteria in the present study lake can be attributed to the levels of nitrate and turbidity, which was largely a function of TSS (Fig.2). Buoyancy regulation in *Microcystis spp.* would offset light-limitation, which may occur in turbid environments like Lake Tana and give them a competitive advantage over other algal groups (Dokulil, 1994) and the present results of the RDA seem to suggest the greater importance of inorganic nutrients like nitrate and turbidity, which is a function of TSS, to the predominance of cyanobacteria (Fig 16).

Yeshiemebet Major (2016) underlined the greater importance of Z_{SD}, TSS, temperature and concentration of NO_3 to changes in the abundance of *Microcystis* spp. in Koka Reservoir, which were the major contributors to the cyanobacterial abundance in the present study region of Lake Tana. In addition, the high levels of ammonium-nitrogen must have favored *Microcystis spp.* since ammonium-N pools are known to enhance the growth of non-nitrogen fixing cyanobacteria (Blomqvist *et al.*, 1994). Huszar *et al.* (2000) also reported a similar trend of covariance of cyanobacterial abundance and inorganic nitrogen sources for several tropical productive Brazilian reservoirs.

Among diatoms, *Aulacoseira granulata* was quantitatively the most important species in the Southern Gulf of Lake Tana throughout the study period (Fig. 8-14). Talling (1976, cited in Wood and Talling, 1988) has documented the dominance by *A. granulata* of the phytoplankton community of Lake Tana in the early 1960s. The dominance of this diatom within the Bacillariophyceae was also reported by Ayalew Wondie *et al.* (2007) for the same region of Lake Tana. In the studies made by Hadgembes Tesfay (2007) and Yeshiemebet Major (2016) in Koka Reservoir, the dominance of *A. granulata* was observed during December-February, the part of the year, which corresponds to the present study period. The dominance of *A. granulata* in Koka Reservoir was associated with higher TP, but lower water conductivity and alkalinity (Yeshiemebet Major, 2016), which is consistent with the present chemical condition in the Southern Gulf of Lake

Tana. Although not as abundant as in Koka Reservoir, A. granulata was reported as a common diatom species in the shallow and frequently mixing rift valley Lake Ziway (Elizabeth Kebede, 1996). Although no obvious association between its abundance and silica level was observed for the present study lake, *A. granulata* is a heavily silicified species (Takano *et al.*, 2004), which may rely upon high ambient Si as a requisite for its growth since its distribution in Africa was associated with relatively high silica concentrations (Kilham *et al.*, 1986; Willén, 1991). Thus, the silica level observed in the second half of February was probably limiting to the growth of *A. granulata* at the Hospital, Resort and K. Georgis sites of the present study. The period spanning from December to February is characterized by the prevalence of predominantly southerly winds (Gasse, 1987, cited in Ayalew Wondie *et al.,* 2007) that may result in turbulent water column condition, a physical condition, which seems to favor the persistence and abundance of *A. granulata* in Lake Tana owing to its adaptation to low light conditions and rapid sedimentation under calm water column conditions (Reynolds, 1994).

A. granulata is a common diatom in eutrophic freshwater lakes and reservoirs around the world (Kilham, 1990; Hotzel and Croome 1996).In fact, *A. granulata*is one of the few diatom species that can produce blooms in hypereutrophic freshwater environments (Viera *et al.*, 2008).*A. granulata* was the most frequently occurring taxon (found in 85% of sampled sites) with relative abundance of over 30% at a wide range of both conductivity (85- 389 µS cm^{-1}) and TP (5-65 µg L-1) in Australian freshwater reservoirs (Gale, 2015). With the help of its extracellular polysaccharides, *A. granulata* is known to rapidly form aggregates irrespective of prevailing environmental conditions (e.g. nutrients and temperature, Sandgren, 1988) to allow it to sink and hence avoid being washed out of a lake, even in a polymictic environment, thereby keeping part of its population for seeding by easy resuspension due to turbulence (Viera *et al.*, 2008). This is believed to be the mechanism by which *A. granulata*, with a doubling time close to 6 days, is present around the year in Barra Bonita

Reservoir in Brazil even when the residence time is 2.1 d (Nogueira and Matsumura-Tundisi, 1996, cited in Viera *et al.*, 2008).

A. granulata is, thus, generally regarded as a good indicator of eutrophic water conditions (Nogueira, 2000, Kamenir *et al.*, 2004, Lepistö *et al.*, 2006), since it can easily form predominant populations and even develop to bloom levels (Miyajima *et al.*, 1994, Nakano et al., 1996) in eutrophic waters under suitable physical conditions (e.g. turbulent water column and high temperature).

Among the green algae, *Oedogonium* sp. was the most abundant and persistent, with its abundance decreasing almost consistently with a decline in water column depth at the Hospital, Resort and K. Georgis sites (Fig. 8-14). Catteneo *et al.* (2013) produced a statistical model using climate and hydrological data coupled with remote sensing to assess the hydrological and meteorological determinants of filamentous green algae (FGA) including *Oedogonium* sp. in a lake. They found water depth and water level change to be the most accurate hydrological predictors of FGA occurrence. *Oedogonium* spp. are common in freshwater ecosystems and prefer stagnant waters, such as small lakes and reservoirs (Mrozińska-Weeb, 1976; Pikosz and Messyasz, 2015) although ponds, where water heats up quickly, is a type of ecosystem, which is mostly inhabited by *Oedogonium* species (Mrozińska, 1984). *Oedogonium* was reported to grow abundantly in freshwaters with temperatures at or above 20°C and high nutrient concentrations and near neutral pH (Pikosz and Messyasz, 2015). Thus, the slightly acidic or alkaline pH, high phosphate and nitrate levels and water temperatures higher than 20°C observed in the Southern Gulf of Lake Tana must have favored the proliferation of *Odeogonium* sp.

Species of desmids, which made considerable contributions to the total abundance of green algae throughout the present study period (Table 2), were reported previously for the same lake (Akoma and Imoobe, 2009), and are known for their wide distribution in Ethiopian lakes and reservoirs, with alkalinities and pH of up to 12.5 meq L^{-1} and 9, respectively (Elizabeth Kebede, 1996). The

occurrence of these desmids in the present study region of Lake Tana, characterized by pH levels ranging from 6.6 to 7.7, is consistent with Coesel's (1982) contention that desmids generally prefer slightly acidic water bodies. Based on a culture experiment, Coesel (1993) also gave an upper limit of pH 8.0 for one of the desmids commonly found in this and other Ethiopian lakes (Elizabeth Kebede, 1996), *Closterium acutum*. *Staurastrum gracile* and *Cosmarium spp.,* the constituent spp. of the desmids assemblage in the present study lake, were found to be common to abundant in the saline lakes of Saskatchewan, Canada, with salinity up to 100 g L^{-1}(Hammer *et al.*, 1983) and pH of up to 9.3 (Hammer, 1978). The *Pediastrum* spp., particularly *P. duplex* and *P. simplex,* are cosmopolitan in distribution and occur generally in eutrophic waters (Komárek and Jankovská, 2001).

Because of their susceptibility to environmental disturbance, desmids are very suitable organisms to indicate even apparently negligible changes in their aquatic habitat (Coesel, 1977). During the last few decades, many cases of impoverishment of the desmid flora of previously desmid-rich sites have been reported as a result of eutrophication in freshwaters of Czech Republic (Šťastný, 2009) and the Netherlands (Coesel, 2003).

The rarity of flagellates observed in the Southern Gulf of Lake Tana seems to be characteristic of tropical water bodies including lakes and reservoirs in Ethiopia (Elizabeth Kebede, 1996; Adane Sirage, 2006; Girma Tilahun, 2006; Hadgembes, 2007; Yeshiemebet Major, 2016). According to Wood and Talling (1988), flagellates are more richly represented in Ethiopian lakes with low conductivity and significant concentrations of the divalent cations, Ca^{2+} and Mg^{2+}, whose deficiency inhibits flagellar activity. This seems to be true for such tropical waters as Lake Kinneret, in Israel (Pollinger, 1988) and Cuban reservoirs (Laiz *et al.*, 1993) in which dinoflagellates seasonally dominate and significant concentrations of the divalent cations are present. But, the occurrence of flagellates in lakes with very low concentrations (<0.13 meq L^{-1}) of the divalent cations (e.g. Lake Abijata, Elizabeth Kebede, 1996) is not explicable. As

flagellates are better adapted to calm water column conditions (Reynolds, 2006), frequent mixing of the water column in most Ethiopian lakes and reservoirs seems to be the reason for the paucity of flagellates.

Chl-a biomass (μg L^{-1}) of phytoplankton, which exhibited significant positive correlation with only silica concentration (at $p<0.05$), was low (0.31 to 8.40; mean: 2.90) in comparison to those documented for other freshwater lakes of Ethiopia by Elizabeth Kebede (1996), Girma Tilahun (2006) and Yeshiemebet Major (2016). It is also considerably lower than those reported by Wondie Zelalem (2013) for weed-infested (4.38 to 44.81; mean: 15.42μg L^{-1}) and non-infested (2.92 to 20.68; mean: 9.6μg L^{-1}) sites in the North-eastern part of the same lake.

Based on chemical parameters, Lake Tana was previously described as mesotrophic with low chlorophyll content and primary production by tropical lakes' standard (Ayalew Wondie *et al* 2007, cited in Yirga Kebede Wondim and Hassen Muhabaw, 2015). On the basis of the Carlson's TSI (The model developed for temperate lakes) estimated using the data of all sampling sites or data of only the offshore site, the Southern Gulf of Lake Tana is classified as eutrophic. But, according to the TSI model of Cunha *et al.*, (2013), which was developed for tropical waters, the Southern Gulf of Lake Tana is classified as hypereutrophic. Thus, the model developed for temperate lakes seems to give a better picture of the trophic state of the lake than that developed for tropical reservoirs. Because the present results are of only the dry period, the use of these models for the estimation of the TSI of this lake may not be justifiable. The quantitative importance of such algal taxa as *Microcystis flos-aquae* is, however, suggestive of eutrophic conditions. When classifying lakes, priority is often given to the TSI value associated with chlorophyll-a since it is the most accurate of the three parameters for predicting algal biomass (Fuller and Jodoin, 2016). Any of the three variables, however, can theoretically be used to classify a lake (Savage, 2009; Fuller and Jodoin, 2016). Thus, the Southern Gulf of Lake Tana can be regarded as mesotrophic on the basis of the TSI estimated from mean Chl-a concentration.

6. CONCLUSIONS AND RECOMMENDATIONS

The present results show that the Southern Gulf of Lake Tana is a dilute freshwater with near-neutral pH and high levels of nutrients, particularly nitrate to which human activities associated with agriculture, resort areas and hospital are believed to have contributed. The results also show that phytoplankton biomass in the Southern Gulf is low despite the high levels of inorganic nutrients and moderately high water transparency (mean: ≈ 99 to 162 cm). The Secchi depth of the Southern Gulf, which averaged about 118 cm, suggests that the water column is moderately turbid.

The phytoplankton community is characterized by low species diversity and predominance of cyanobacteria, which was primarily constituted by *Microcystis* spp. Dominance of cyanobacteria is regarded as a sign of eutrophic conditions and poor ecological status of reservoirs and lakes (Paerl, 2011).Mankiewicz-Bocz and Gągała (2012) reported the presence of measurable levels of microcystins associated with the dominance of *Microcystis aeruginosa* in samples collected from sites known as Tana-bay, Tana-bata and Tana-paid of Lake Tana. This is a serious problem in countries like Ethiopia where adequate water quality assessment and monitoring programs are non-existent. Considering the poor medical services and records in this country, health problems associated with consumption of cyanotoxin-contaminated waters can easily go unnoticed or wrongly attributed to other causes. A continuous monitoring program is, therefore, necessary to ensure the protection of public health, aquatic and terrestrial life.

On the basis of the major findings of the research work reported here, the following recommendations are made:

In order to come up with a more complete picture of the phytoplankton community structure and dynamics in Lake Tana, samples should be collected during all seasons and from different parts/regions of the lake.

- Conduction of zooplankton grazing experiments is mandatory to unambiguously resolve the question of phytoplankton control by zooplankton as the variations in biological parameters in the present study were investigated in relation to physico-chemical parameters only.
- The role played by fish in the spatial and temporal variations of the algal flora should also be given due consideration in future investigations.
- Before unwelcome phenomena, which are similar to those of Koka reservoir, take place, investigations aiming at the assessment of cyanotoxins in the lake water should be carried out.
- A closer look at the impact of human activities like irrigation, shoreline modification and disposal of wastewaters, with a view to develop strategies of preventing further degradation of the aquatic ecosystem and conserving its resources, is needed. The extent of release of nutrients from the sediment to the overlying water, implied by a previous study made on the organic content and inorganic nutrients of the lake's sediment, should be investigated in order to unravel the relative importance of the various sources of nutrients. The concerned government body should monitor land use practices around Lake Tana in order to alleviate impacts of anthropogenic origin.

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8. APPENDICES

Appendix A. Physical parameters measured at the sampling sites of the present study in Southern Gulf of Lake Tana

Site	Parameters	units	Month								
			Dec. 5	Dec. 30	Jan. 15	Feb. 8	Feb. 25	Mar. 10	Mar. 29	Mean	Mean±SD
Zege	T°	^{0}C	22.7	22.18	21.49	21.59	22.98	23.92	25.15	22.86	22.86±1.314
	pH		7.13	7.07	7.73	7.28	6.96	6.93	7.48	7.23	7.23±0.293
	Sp. Cond	µScm-1	128	128	129	132	138	172	148	139.29	139.29±16.132
	TDS	mg/l	87	88	90	92	93	114	96	94.286	94.286±9.214
	Sal.	mg/l	60	60	60	70	70	80	70	67.143	67.143±7.559
	DO	mg/l	6.41	6.51	6.32	6.82	6.05	5.3	5.28	6.099	6.099±0.598
	Secchi depth	cm	69	83	76	80	92	81	103	83.429	83.429±11.088
	TSS	mg/l	12	8	15	12.5	10	10	5	10.357	10.357±3.249
	Depth	m	8.6	8.56	8	8	7.5	7.66	7.3	7.946	7.946±0.501
	Turbidity	NTU	26	20	23	20	17	17	17	20.00	20.00±3.464
Hospital	T°	°C	23.01	23.81	21.15	22.43	22.56	24.26	25.26	23.211	23.211±1.353
	pH		6.77	6.7	6.87	6.79	6.61	7.1	6.76	6.8000	6.800±0.155
	Sp. Cond	µScm-1	152	157	163	149	152	156	175	157.714	157.714±8.864
	TDS	mg/l	103	105	114	103	104	103	110	106.000	106.000±4.320
	Sal.	mg/l	70	80	80	70	70	70	80	74.286	74.286±5.345
	DO	mg/l	6.64	5.76	6.78	6.15	6.07	5.29	5.09	5.969	5.969±0.636
	Secchi depth	cm	146	200	200	170	158	130	129	161.857	161.857±29.824
	TSS	mg/l	4	4	2.5	7.5	5	2.5	12.5	5.429	5.429±3.552
	Depth	m	3.2	2.2	2.2	1.94	2.23	2	1.94	2.244	2.244±0.440
	Turbidity	NTU	11	5	14	5	11	8	5	8.429	8.429±3.645
Resort	T°	°C	23.48	24.02	21.83	23.45	24.19	25.08	26.21	24.037	24.037±1.376
	pH		6.84	7.02	7.14	7.14	6.67	7.14	7.4	7.0500	7.050±0.237
	Sp. Cond	µScm-1	154	156	155	148	154	158	175	157.143	157.143±8.454
	TDS	mg/l	103	103	107	99	102	102	110	103.714	103.714±3.638
	Sal.	mg/l	70	70	80	70	70	70	70	71.429	71.429±3.780
	DO	mg/l	5.98	5.95	6.39	6.03	6.08	5.41	5.01	5.836	5.836±0.466
	Secchi depth	cm	138	141	180	120	97	100	118	127.714	127.714±28.523
	TSS	mg/l	4	4	2.5	10	5	5	2.5	4.714	4.714±2.547
	Depth	m	2.19	1.88	1.83	1.71	1.3	1.42	1.18	1.644	1.644±0.360
	Turbidity	NTU	8	17	5	17	14	11	8	11.429	11.429±4.721
K. Georgis	T°	°C	24.26	23.67	22.28	23.45	23.77	24.86	27.16	24.207	24.207±1.523
	pH		7.03	7.11	7.27	6.97	6.67	7.8	6.95	7.114	7.114±0.353
	Sp. Cond	µScm-1	138	140	138	150	168	153	165	150.286	150.286±12.55275
	TDS	mg/l	91	93	95	100	111	99	104	99.000	99.000±6.904

Sal.	mg/l	70	70	70	70	80	70	80	72.857	72.857±4.879
DO	mg/l	6.46	6.05	6.31	6.33	6.29	5.44	4.39	5.896	5.896±0.745
secchi depth	cm	64	64	75	95	157	143	94	98.857	98.857±37.343
TSS	mg/l	12	16	17.5	12.5	12.5	12.5	12.5	13.643	13.643±2.174
Depth	m	2	1.78	1.74	1.76	1.6	1.53	1.27	1.669	1.669±0.230
Turbidity	NTU	26	23	14	5	5	23	14	15.714	15.714±8.635

Appendix B. Chemical parameters measured at the sampling sites of the present study in Southern Gulf of Lake Tana

Site	Parameters	Units	Month								
			Dec.5	Dec.30	Jan. 15	Feb.8	Feb. 25	Mar. 10	Mar. 29	Mean	Mean±SD
Zege	PO_4-P	mg/l	0.28	-	0.26	0.51	0.38	0.61	0.4	0.407	0.407±0.134
	NH_3-N	mg/l	0.06	0.06	0.16	0.09	0	0	0.05	0.060	0.060±0.055
	NO_3-N	mg/l	2.2	0.748	0.704	1.1	1.848	1.496	0.704	1.257	1.257±0.604
	NO_2-N	mg/l	0.019	0.010	0.003	0.003	0	0.059	0	0.014	0.014±0.021
	SiO_2	mg/l	17.4	18	21	17.4	16	15.6	10	16.486	16.486±3.352
	TP	mg/l	0.32	-	0.31	0.55	0.39	0.62	0.48	0.445	0.445±0.126
	H_2S	mg/l	-	0.053	-	0.095	-	0.064	0.074	0.072	0.072±0.018
Hospital	PO_4-P	mg/l	0.07	0.28	0.28	0.14	0.16	0.22	0.51	0.237	0.237±0.142
	NH_3-N	mg/l	0.1	0.09	1.2	0.05	0	0.04	0.03	0.216	0.216±0.435
	NO_3-N	mg/l	1.892	1.1	1.232	1.144	0.528	1.452	0.968	1.188	1.188±0.421
	NO_2-N	mg/l	0.020	0	0	0.003	0	0.063	0	0.012	0.012±0.023
	SiO_2	mg/l	21	6.2	18	4.2	0.4	7.6	5.4	8.971	8.971±7.580
	TP	mg/l	0.085	0.32	0.32	0.15	0.17	0.23	0.61	0.269	0.269±0.174
	H_2S	mg/l		0.064		0.053		0.053	0.064	0.058	0.058±0.006
Resort	PO_4-P	mg/l	0.94	-	0.14	0.46	0.14	0.3	0.18	0.36	0.36±0.310
	NH_3-N	mg/l	0.5	0.12	0.03	0.09	0	0.03	0	0.110	0.110±0.178
	NO_3-N	mg/l	0.66	0.968	0.44	1.452	1.452	0.836	0.3168	0.875	0.875±0.451
	NO_2-N	mg/l	0	0.003	0	0.010	0	0.023	0	0.005	0.005±0.009
	SiO_2	mg/l	21	13	28.6	9.4	2	16.4	7.4	13.971	13.971±8.922
	TP	mg/l	0.98	-	0.16	0.51	0.15	0.37	0.52	0.448	0.448±0.307
	H_2S	mg/l		0.074		0.053		0.085	0.074	0.072	0.072±0.013
K. Georgis	PO_4^{-3}-P	mg/l	0.14	0.96	0.22	0.14	0.18	0.42	0.67	0.390	0.390±0.317
	NH_3-N	mg/l	0.2	0.3	0	0.06	0	0.07	0.08	0.101	0.101±0.110
	NO_3-N	mg/l	1.452	1.76	1.056	0.88	1.32	1.672	0.484	1.232	1.232±0.455
	NO_2-N	mg/l	0.010	0.010	0	0	0	0.030	0	0.007	0.007±0.011
	SiO_2	mg/l	28.6	7.4	18.4	8.4	0.2	5	18	12.286	12.286±9.784
	TP	mg/l	0.15	0.98	0.25	0.15	0.19	0.44	0.71	0.410	0.410±0.323
	H_2S	mg/l		0.085		0.053		0.085	0.064	0.072	0.072±0.016

Appendix C. P- Value of physical and chemical parameters in oneway ANOVA

Parameters	**P-value**	**Parameters**	**P- value**
Temperature	0.237	Phosphate (PO_4-P)	0.574
pH	**0.044***	Ammonia (NH_3-N)	0.672
Sp. conductance	**0.027***	Nitrate (NO_3-N)	0.443
TDS	**0.011***	Nitrite (NO_2-N)	0.769
Salinity	0.12	Silicate (SiO_2)	0.353
DO	0.872	Total phosphorus (TP)	0.520
Secchi depth	**0.001***	Hydrogen sulfide (H_2S)	0.478
TSS	**0.001***		
depth	**0.001***		
Turbidity	**0.004***		

***** is significant at the 0.05 level

Appendix D. Two-tailed Spearman rank Correlation (r) for physico-chemical variables and Chl-a

	Temp.	**PH**	**Sp.Cond.**	**TDS**	**Sal.**	**DO**	**Secchi**	**TSS**	**PO_4-P**	**NH_3-N**	**NO_3-N**	**NO_2-N**	**Sio_2**	**TP**	**H_2S**	**Chl-a**
PH	0.04															
Sp. Cond	**0.567****	-0.345														
TDS	0.27	**-0.439***	**0.941****													
Sal.	0.27	**-0.439***	**0.766****	**0.801****												
DO	**-0.875****	-0.121	**-0.600****	-0.36	-0.32											
Secchi	-0.13	**-0.375***	**0.501****	**0.660****	**0.527****	0.04										
TSS	-0.05	0.287	**-0.392***	**-0.468***	-0.17	0.083	**-0.623****									
PO_4-P	0.04	0.008	-0.049	-0.08	-0.16	-0.004	-0.02	-0.123								
NH_3-N	-0.33	-0.089	0.056	0.205	0.176	0.313	0.316	-0.205	0.032							
NO_3-N	-0.16	-0.085	-0.273	-0.26	-0.15	0.291	-0.19	0.271	-0.116	0.031						
NO_2-N	0.13	0.152	0.117	0.094	-0.04	-0.26	-0.16	-0.07	-0.05	-0.152	**0.417***					
Sio_2	-0.29	0.239	-0.363	-0.3	-0.13	0.305	-0.26	0.026	0.06	0.256	-0.058	0.012				
TP	0.06	0.022	-0.018	-0.06	-0.15	-0.03	-0.01	-0.144	**0.998****	0.025	-0.136	-0.063	0.049			
H_2S	0.09	**0.545***	-0.115	-0.21	-0.03	0.037	-0.27	0.183	0.064	0.318	0.083	-0.101	0.187	0.08		
Chl-a	**-0.514****	0.194	**-0.493****	**-0.377***	-0.29	**0.396***	-0.21	0.168	-0.124	0.18	-0.179	-0.253	**0.463***	-0.141	-0.165	
Turb.	-0.132	**0.517****	**-0.636****	**-0.695****	**-0.545****	0.203	**-0.643****	**0.449***	0.2	0.091	**0.422***	0.17	0.336	0.182	**0.508***	0.156

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

Appendix E. Biomass of phytoplankton as Chlorophyll-a in the Southern Gulf of Lake Tana during the study period

Site	**Parameter**	**unit**	**Month**									
			Dec5	Dec. 30	Jan. 15	Feb.8	Feb. 25	Mar. 10	Mar.29	Sig.	Mean±SE	T. Mean±SD
Zege	Chl-a	$\mu g\ L^{-1}$	3.878	4.432	8.40294	5.333	4.8479	0.969	1.454	0.199	4.188±2.500	
Hospital	Chl-a	$\mu g\ L^{-1}$	1.939	1.108	4.363	2.424	1.939	2.424	1.454		2.236±1.053	
Resort	Chl-a	$\mu g\ L^{-1}$	2.909	1.108	5.817	2.9087	2.9087	2.424	0.969		2.721±1.605	2.896±1.874
K. Georgis	Chl-a	$\mu g\ L^{-1}$	1.939	2.216	1.454	5.333	1.454	0.310	4.363		2.439±1.772	

Appendix F. Abundance of identfied Phytoplankton species at the Zege site during the study period

Site	**Taxa**	Dec. 5 Cell/ml	Dec. 30 Cell/ml	Jan. 15 Cell/ml	Feb. 8 Cell/ml	Feb. 25 Cell/ml	Mar. 10 Cell/ml	Mar. 29 Cell/ml	RA %
Zege	**Cyanophyceae**								**50.44**
	Anabaena sp	143	300	250	167	17	5	6	1.86
	Chrococcus turgidus.	143	40	100	67	34	50	75	1.07
	Microcystis aeruginosa	536	113	188	188	125	188	235	3.29
	Microcystis flos-aquae	1334	2500	4688	3646	3125	2605	2813	43.36
	Oscillatoria sp.	22	10	13	16	17	17	13	0.23
	Planktolyngbya limnetica	15	20	13	0	0	0	0	0.1
	Synechococcus sp.	29	50	25	83	67	34	13	0.63
	Total cell/ml	2207	3013	5264	4167	3385	2899	3155	
	Bacillariophyceae								**19.93**
	Aulacoseira distans	215	120	375	67	67	16	13	0.39
	Aulacoseira granulata	403	840	250	2268	1134	1601	500	14.64
	Cyclotella sp.	22	50	125	16	34	16	50	0.66
	Cymbella ventricosa.	22	40	13	34	50	16	13	0.39
	Diatoma uvlgaris	15	10	13	16	34	33	0	0.25
	Gomphonema gracile	22	20	25	16	16	16	13	0.27
	Navicula cryptocephala	22	40	25	16	67	84	63	0.66
	Rhopalodia gibbia	15	10	13	34	16	16	13	0.25
	Syndera ulna	70	50	50	34	34	16	75	0.69
	Tabellaria sp.	7	20	13	16	16	16	13	0.21
	Total cell/ml	813	1200	902	2517	1468	1830	790	
	Chlorophyceae								**25.79**
	Ankistrodesmus sp.	15	20	38	16	84	16	13	0.42
	Ankistrodesmus angustus	7	10	13	0	84	16	0	0.27
	Chlamydomonas sp.	15	20	13	50	117	167	38	0.88

	Chlorella sp.	71	10	25	50	34	34	13	0.5
	Closterium acutum	21	10	13	67	16	16	0	0.3
	Closterium acutum	7	10	13	16	34	50	50	0.38
	Closterium sp.	21	20	25	16	50	34	25	0.4
	Dictyosphaerium sp.	86	10	38	50	34	16	13	0.52
	Eudorina sp.	86	200	63	100	100	16	13	1.21
	Monoraphdium sp	7	10	13	34	16	16	13	0.22
	Oedogonium sp.	822	1150	1150	1916	1534	767	286	1.6
	Oocystis eremosphaeria	15	60	25	16	34	50	63	0.55
	Oocystis lacustris	7	10	13	34	16	16	38	0.28
	Oocystis parva	36	50	38	100	67	34	25	0.73
	Oocystis sp.	86	20	75	16	34	67	100	0.83
	Pediastrum boryanum	15	10	13	16	16	16	0	0.18
	Schroederia setiera	0		25	16	34	34	25	0.29
	Scendesmus incrassulatus	0	10	38	16	16	16	13	0.23
9	*Selenastrum sp.*	15	10	63	34	67	50	63	0.63
	Staurastrum gracile	15	10	50	34	16	16	13	0.32
	Staurastrum sp.	29	30	38	67	84	16	13	0.58
	Total cell/ml	1391	1700	1795	2664	2487	1463	817	
	Dinophyceae								**1.43**
	Peridinium cinctum	7	10	13	0	67	16	25	0.29
	Peridinium gatunense	21	10	50	16	50	84	50	0.59
	Peridinium sp.	15	20	13	0	167	34	13	0.55
	Total cell/ml	43	40	76	16	284	134	88	
	Euglenophyceae								**0.54**
	Euglena cf. viridis	15	10	13	16	16	16	0	0.18
	Phacus acuminutus	7	30	38	16	32	34	13	0.36
	Total cell/ml	22	40	51	32	48	50	13	

Appendix G. Abundance of identified Phytoplankton species at the Hospital site during the study period

Site	**Taxa**	Dec. 5 Cell/ml	Dec. 30 Cell/ml	Jan. 15 Cell/ml	Feb. 8 Cell/ml	Feb. 25 Cell/ml	Mar. 10 Cell/ml	Mar. 29 Cell/ml	RA %
Hospital	**Cyanophyceae**								**56.34**
	Anabaena sp.	188	84	84	139	50	143	54	2.44
	Chroococcus turgidus.	19	25	34	56	80	18	15	0.81
	Microcystis aeruginosa	63	209	556	93	100	30	24	3.54
	Microcystis flos-aquae	3646	2084	4167	1852	1667	268	298	46.06
	Oscillatoria sp.	7	17	67	6	10	4	4	0.38
	Pseudoanabaena sp.	7	9	334	112	10	72	72	2.03
	Planktolyngbya limnetica	13	17	17	6	10	4	4	0.23
	Synechococcus sp.	0	0	0	6	50	18	15	0.29
	Total cell/ml	3943	2437	5326	2287	1977	578	557	
	Bacillariophyceae								**19.12**
	Amphora coffeaeformis	13	17	34	6	10	29	4	0.37
	Aulacoseira granulata	168	208	1601	112	200	73	44	7.94
	Cyclotella radiosa	25	9	84	34	50	36	29	0.88
	Cyclotella sp.	125	42	50	6	10	11	4	0.82
	Cymbella ventricosa.	50	125	34	6	60	18	47	1.11
	Cymbella tumida	7	34	167	45	30	8	4	0.97
	Diatoma vulgaris	7	17	17	6	10	15	0	0.24
	Gomphonema offine	25	25	17	6	10	0	4	0.29
	Gomphonema cf. grovei	13	9	17	6	30	4	4	0.27
	Meloseira sp.	75	42	67	23	10	15	15	0.81
	Navicula cryptocephala	38	84	34	28	80	29	22	1.04
	Nitzschia palea	7	17	17	6	20	4	4	0.25
	Nitzschia filiformis	38	34	17	6	10	18	61	0.61
	Pinnularia sp.	48	42	17	6	0	0	0	0.37
	Rhapalodia gibba	63	67	50	17	60	8	11	0.91
	Syndera ulna	94	250	134	112	30	40	18	2.23

Total cell/ml	801	1022	2357	425	620	308	271	
Chlorophyceae								**22.61**
Ankistrodesmus angustus	25	25	134	6	20	8	4	0.73
Chlamydomonas sp.	63	59	234	67	50	47	25	1.8
Closterium acutum	13	67	17	6	10	4	4	0.4
Cosmarium cotractum	19	9	17	12	10	4	4	0.25
Desmidium swartzii	1250	209	50	6	10	29	29	5.21
Eudorina sp.	50	34	17	23	10	4	4	0.47
Haematoccus sp.	13	9	84	23	10	25	75	0.79
Oedogonium sp.	1436	575	384	128	0	83	165	9.13
Oocystis parva	19	17	100	6	10	150	15	1.04
Oocystis lacustris.	94	150	50	50	40	8	83	1.56
Scenedesmus sp.	19	25	34	6	10	4	4	0.34
Scenedesmus incrassulatus	25	9	17	6	10	29	15	0.37
Selenastrum sp.	13	25	17	6	10	4	8	0.27
Straurastrum convergens	7	9	17	6	0	0	0	0.13
Straurastrum gracile	19	42	17	12	10	0	4	0.34
Straurastrum triangularis var. triangularis	13	17	34	6	20	4	8	0.34
Total cell/ml	3078	1289	1156	352	230	382	376	
Dinophyceae								**0.42**
Perdinium gatunese	32	9	50	6	10	15	4	0.42
Total cell/ml	32	9	50	6	10	15	4	
Cryptophyceae								**0.54**
Cryptomonas sp.	7	34	84	6	30	4	0	0.54
Total cell/ml	7	34	84	6	30	4	0	
Euglenophyceae								**0.97**
Euglena cf. viridis	7	17	17	6	10	4	0	0.2
Phacus acuminatus	38	25	117	6	20	11	18	0.77
Total cell/ml	45	42	134	12	30	15	18	

Appendix H. Abundance of identfied Phytoplankton species at the Resort site during the study period

Site	**Taxa**	Dec. 5 Cell/ml	Dec. 30 Cell/ml	Jan. 15 Cell/ml	Feb. 8 Cell/ml	Feb. 25 Cell/ml	Mar. 10 Cell/ml	Mar. 29 Cell/ml	RA %
Resort	**Cyanophyceae**								**47.86**
	Anabaena sp.	56	42	150	125	250	100	275	3.32
	Chroococcus turgidus.	12	9	10	25	13	20	10	0.33
	Microcystis aeruginosa	84	112	209	186	125	100	125	3.13
	Microcystis flos-aquae	927	1389	2084	4167	1563	834	313	37.53
	Oscillatoria sp.	12	25	10	25	25	10	3	0.37
	Peseudoanabaena sp.	112	84	10	125	0	0	400	2.43
	Planktolyngbya limnetica	6	9	5	13	0	0	0	0.11
	Synechococcus sp.	23	9	5	13	25	20	6	0.33
	Total cell/ml	1243	1679	2483	4679	2014	1104	1177	
	Bacillariophyceae								**17.2**
	Amphora coffeaeformis	6	9	5	13	13	80	28	0.51
	Aulacoseira granulata	184	168	180	130	700	360	440	7.2
	Cyclotella radiosa	6	9	10	25	63	20	25	0.52
	Cyclotella sp.	45	67	10	13	13	10	6	0.55
	Cymbella tminuta	6	34	5	13	13	20	3	0.31
	Cymbella ventricosa.	17	17	5	138	38	10	20	0.82
	Diatoma vulgaris	6	9	5	13	13	10	6	0.21
	Gomphonema gracile	12	9	5	13	25	10	3	0.26
	Gomphonema cf.grovei	6	9	10	13	25	10	10	0.28
	Aulacoseria distans	6	34	20	300	50	40	10	1.53
	Meloseria sp.	50	68	60	50	50	80	10	1.22
	Navicula cryptocephala	56	68	20	75	38	20	20	0.99
	Nitzschia filiformis	6	9	5	13	13	10	3	0.2
	Nitzschia palea	23	9	35	26	38	20	3	0.51
	Pinularia sp.	6	25	5	26	26	10	3	0.34
	Rhoilosphenia abreviata	17	34	10	38	38	30	10	0.6

Syndera ulna	45	59	75	50	75	30	18	1.17
Total cell/ml	497	637	465	949	1231	770	618	
Chlorophyceae								**33.08**
Ankistrodesmus sp.	28	9	5	13	13	10	75	0.51
Chlamydomonas sp.	12	9	20	26	213	1000	75	4.51
Closterium sp.	6	17	5	13	13	10	3	0.22
Coelastrum sp.	6	9	5	13	13	10	3	0.2
Cosmarium contractum	6	9	5	13	26	80	13	0.51
Desmidium swartzii	50	34	150	63	26	120	25	1.56
Eudorina sp.	23	34	10	26	38	10	3	0.48
Haematoccus sp.	17	9	5	13	13	20	45	0.41
Oedogonium sp.	3195	384	115	288	288	230	58	15.17
Oocystis parva	6	17	50	175	163	1040	268	5.72
Oocystis sp.	112	100	5	75	38	20	75	1.41
Pediastrum boryanum.	12	9	5	13	13	10	3	0.22
Pediastrum simplex	6	9	5	13	13	10	3	0.2
Scendesmus incrassulatus	6	25	5	13	13	20	40	0.41
Scendesmus sp.	6	9	10	13	26	10	6	0.27
Selenastrum sp.	12	42	20	13	13	10	13	0.41
Staurastrum convergens	6	9	5	13	13	10	75	0.44
Staurastrum gracile	17	17	25	26	26	20	3	0.45
Staurastrum longebrachiatum	12	9	10	13	13	30	3	0.3
Total cell/ml	3527	760	460	835	961	2650	744	
Cryptophyceae								**0.53**
Cryptomonas sp.	6	9	50	26	13	10	45	0.53
Total cell/ml	6	9	50	26	13	10	45	
Dinophyceae								**0.66**
Perdinium sp.	6	9	10	13	26	60	75	0.66
Total cell/ml	6	9	10	13	26	60	75	
Euglenophyceae								**0.67**
Euglena cf. viridis	12	9	5	13	26	10	3	0.26

Phacus acuminutus	6	17	15	26	13	40	6	0.41
Total cell/ml	18	26	20	39	39	50	9	

Appendix I. Abundance of identfied Phytoplankton species at the Kidus Georgis site during the study period

Site	**Taxa**	Dec. 5 Cell/ml	Dec. 30 Cell/ml	Jan. 15 Cell/ml	Feb. 8 Cell/ml	Feb. 25 Cell/ml	Mar. 10 Cell/ml	Mar. 29 Cell/ml	RA %
K. Georgis	**Cyanophyceae**								**44.55**
	Anabaena sp.	50	84	125	100	63	42	63	1.41
	Chrococcus turgidus.	20	0	50	20	13	16	25	0.39
	Microcystis aeruginosa	375	250	94	300	282	63	94	3.9
	Microcystis flos-aquae	1375	1875	1719	2750	2344	2084	1563	36.7
	Oscillatoria sp.	30	17	26	10	13	8	25	0.35
	Pseudoanabaena sp.	100	0	0	0	0	83	250	1.2
	Synechococcus sp.	20	0	13	20	13	8	13	0.23
	Total cell/ml	2020	2243	2040	3210	2741	2320	2071	
	Bacillariophyceae								**30.58**
	Amphora coffaeformis	10	17	13	10	13	42	13	0.32
	Amphora sp.	10	17	13	10	13	8	26	0.26
	Aulacoseira distans	40	334	100	240	50	34	50	2.27
	Aulacoseria granulata	680	334	500	600	334	634	5250	22.3
	Cyclotella sp.	30	100	13	30	26	8	13	0.59
	Cymbella sp	20	34	13	40	26	25	50	0.56
	Cymbella turgidula	10	17	26	10	13	8	0	0.22
	Diatoma vulgaris	10	17	13	10	13	16	0	0.21
	Gomphonema gracile	10	17	13	20	13	42	13	0.34
	Gomphonema sp.	10	17	13	10	26	8	13	0.26
	Navicula cryptocephala	50	34	26	10	39	42	13	0.57
	Nitzschia filiformis	10	0	26	10	13	8	26	0.25
	Nitzschia palea	10	17	13	10	26	25	50	0.4
	Pinnularia sp.	10	50	13	10	13	8	0	0.28
	Rhopalodia gibba	30	67	26	50	39	16	13	0.64
	Syndera ulna	60	17	39	70	63	84	75	1.09

Total cell/ml	1000	1089	860	1140	722	1008	5605	
Chlorophyceae								**23.49**
Chlamydomonas sp.	20	17	13	10	113	100	63	0.9
Closterium sp.	10	17	26	10	13	25	13	0.31
Desmidium swartzii	10	34	13	20	26	8	13	0.33
Eudorina sp.	20	50	26	10	13	34	26	0.48
Haematoccus sp.	50	17	13	10	13	16	38	0.42
Oedogonium sp.	3220	767	288	690	288	192	575	16.11
Oocystis parva	20	34	13	40	39	34	75	0.68
Oocystis sp.	130	17	125	80	26	50	25	1.21
Pedastrum duplex	40	17	26	10	13	8	38	0.71
Pedastrum simplex	30	51	26	30	39	8	13	0.34
Scendesmus incrassatules	10	50	13	20	100	34	50	0.74
Schroederia setigera	10	17	13	0	0	25	13	0.21
Selenastrum sp.	30	50	13	20	13	16	26	0.45
Straurastrum convergens	40	17	13	10	13	0	0	0.25
Straurastrum gracile	50	50	39	10	13	13	13	0.5
Staurodesmus curvatus var.latus	10	17	13	30	39	8	26	0.38
Total cell/ml	3650	1205	660	990	748	555	969	
Dinophyceae								**0.75**
Peridinium cinctum	20	17	39	10	13	0	0	0.27
Peridinium sp.	10	67	13	30	26	8	26	0.48
Total cell/ml	30	84	52	40	39	8	26	
Euglenophyceae								**0.63**
Euglena cf. viridis	10	34	13	10	13	8	0	0.24
Phacus acuminutus	30	17	13	20	39	16	13	0.4
Total cell/ml	40	51	26	30	52	24	13	

Appendix J. Relative abundance(RA) of major phytoplankton species within their respective classes in the Southern Gulf of Lake Tana

Site	Taxa	RA (%) species in Class	Site	Taxa	RA (%) species in Class
Zege	**Cyanophyceae**		Resort	**Cyanophyceae**	
	Microcystis flos-aquae	85.97		*Microcystis flos-aquae*	78.43
	Bacillariophyceae			**Bacillariophyceae**	
	Aulacoseira granulata	73.49		*Aulacoseira granulata*	41.84
	Chlorophyceae			**Chlorophyceae**	
	Oedogonium sp.	61.91		*Oedogonium* sp.	45.87
Hospital	**Cyanophyceae**		Kidus Georgis	**Cyanophyceae**	
	Microcystis flos. aquae	81.74		*Microcystis flos. aquae*	82.37
	Bacillariophyceae			**Bacillariophyceae**	
	Aulacoseira granulata	41.49		*Aulacoseria granulata*	72.95
	Chlorophyceae			**Chlorophyceae**	
	Oedogonium sp.	40.38		*Oedogonium* sp.	68.59

Appendix K. Trophic state categories proposed for temperate (Carlson, 1977; Carlson and Simpson, 1996) and tropical (Cunha et al., 2013) waters.

Trophic state		**Carlson and Simpson (1996)***	**Cunha *et al.* (2013)****
Eutrophic	Hypereutrophic	>65	$\geq$59.1
	Strongly eutrophic	62 - 64.9	58.2 - 59.0
	Eutrophic	58 - 61.9	55.8 - 58.1
	Slightly eutrophic	54 - 57.9	
Mesotrophic	Strongly mesotrophic	49 - 53.9	
	Mesotrophic	43 - 48.9	53.2 - 55.7
	Slightly mesotrophic	38 - 42.9	
Oligotrophic	Slightly oligotrophic	33 - 37.9	
	Oligotrophic	26 - 32.9	51.2 - 53.1
	Strongly oligotrophic	20 - 25.9	$\leq$ 51.1

***** based on arithmetic means of Z_{SD}, TP and Chl-a; ******based on geometric means of TP and Chl-a.

Appendix L. P-value of spatial variation of phytoplankton abundance in one-way ANOVA

Site	P - value	Site	P - value
Zege	0.002	Zege(**offshore**)	0.001
Hospital		Hospital, Resort, K. Georgis (**inshore**)	
Resort			
Kidus Georgis			