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Temporal Dynamics of Water Quality and Community Structure and Photosynthetic Production of Phytoplankton in Belbela Reservoir, Ethiopia



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by

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Abstract

The temporal dynamics of the composition, abundance, biomass and photosynthetic productivity of phytoplankton in relation to physico-chemical water quality and zooplankton were studied from September, 2010 to May, 2011 in Belbela reservoir. Water transparency exhibited temporal variation (0.16 m to 0.26 m), which was primarily determined by abiogenic turbidity. The depth profiles of temperature and dissolve oxygen seem to indicate the absence of deep-seated and persistent thermal stratification, which is consistent with the shallowness and complete exposure of the reservoir to wind action. Aggregate chemical parameters were clearly indicative of the very dilute nature of the reservoir water. All nutrients except soluble reactive phosphate were generally at high levels. All chemical parameters including inorganic nutrients exhibited temporal variations with no obvious association with biological variables. The phytoplankton community of the reservoir was primarily constituted by blue-green algae, green algae and diatoms, with the overwhelming dominance of blue-greens whose dominance seemed to be favored by the turbid, turbulent and nutrient-rich water column. The major contributors to the dominance of blue-green algae include the potentially toxic taxa *Cylindrospermopsis*, *Microcystis* and *Planktothrix*. The impact of the rotifer-dominated zooplankton community, on the phytoplankton seemed to have been weakened by the abundant large-sized colonial and filamentous blue-greens, which are not manageable and probably toxic. Chlorophyll a biomass of phytoplankton varied temporally ($20.38 - 68.57 \mu\text{g L}^{-1}$) with its peaks corresponding to those of total abundance of phytoplankton and blue-greens but without any clear causal relationship with inorganic nutrients. The vertically compressed depth profiles of gross photosynthesis exhibited temporal variations in their light-saturated rates (A_{max}) and depths of their occurrence. A_{max} varied from about 226 to 891 $\text{mg O}_2 (\approx 70.5 \text{ to } 278 \text{ mg C}) \text{ m}^{-3} \text{ h}^{-1}$, corresponding to the lowest and highest phytoplankton biomass values although lack of good correlation between the two was evident. The fairly high and positive correlation between A_{max} and the biomass-specific rate at light saturation, P_{max} ($r=0.6938$, $r^2=0.4813$ at $p=0.0563$) provided an explanation for the observed association of high light-saturated rates with low algal biomass. P_{max} ranged from ≈ 5.5 to $24.3 \text{ mg O}_2 (\text{mg Chl a})^{-1} \text{ h}^{-1}$ with most values below 15, but with its maximum value higher than that considered as representative for many African lakes [about $20 \text{ mg O}_2 (\text{mg Chl a})^{-1} \text{ h}^{-1}$] and an upper limit for lakes of the temperate regions. The hourly integral rate of gross photosynthesis (ΣA , $\text{mg O}_2 \text{ m}^{-2} \text{ h}^{-1}$) ranged from 0.112 to 0.510 $\text{g O}_2 (\approx 0.035\text{-}0.159 \text{ g C}) \text{ m}^{-2} \text{ h}^{-1}$ with its variations being primarily due to temporal changes in A_{max} ($r=0.7551$, $r^2 = 0.5701$ at $p= 0.0303$). The present results seem to suggest that the reservoir is at the verge of a seemingly irreversible environmental degradation. There is, therefore, a need for the assessment of cyanotoxins and the impact of human activities like irrigation, shore-line modification and removal of plant cover with a view to develop strategies of preventing further degradation of the aquatic ecosystem and loss of its resources.

1. Introduction

Freshwater is a key natural resource and a limiting factor for the economy of a country. Freshwaters are of immense importance to the development of a country as they are essential for domestic use, industry and agriculture, especially in countries like Ethiopia where agriculture forms the backbone of the economy. Freshwaters including those found in Ethiopia support commercial fisheries on which the livelihood of local inhabitants is partly or entirely dependent. Freshwaters also serve as habitats for a variety of animals including reptiles, birds and mammals, which may be a source of tourist attraction. Freshwaters may, therefore, contribute immensely to poverty reduction in developing countries apart from the ecological and social benefits they provide (FAO, 2000). Thus, without freshwaters of adequate quantity and acceptable quality, sustainable development of a country is not likely.

Freshwaters of the world include reservoirs, which are water bodies formed or modified by human activity for specific purposes with a view to provide a reliable and controllable resource (Thornton *et al.*, 1992). The number of reservoirs in the world has increased tremendously from about 5000 in the 1950s, 30 000 in the 1970s to 40 000 in the 1980s (Naiman and Decamps 1990). Most countries, including those in Africa, have numerous small reservoirs created mainly for domestic and livestock water supply (Worldwide Fund for Nature, 1999). According to a estimate made by McCully(1996), there are about 800,000 small reservoirs worldwide.

Despite their countless uses, nowadays freshwaters and their biodiversity are being threatened by a number of human activities including overexploitation, water pollution, water abstraction, shore-line modification and introduction of invasive alien species (Bartran and Balance, 1996; Soto *et al.*, 1996; Dudgeon *et al.*, 2006). More than two decades have elapsed since human activities started impacting freshwaters including reservoirs in Ethiopia. Several studies (e.g. Fisseha Itana, 1998; Zinabu Gebre-Mariam, 1998; Brook Lemma, 2002; Zinabu Gebre-Mariam and Zerihun Desta, 2002; Zinabu Gebre-Mariam *et al.*, 2002; Seyoum Letta *et al.*, 2003) have shown changes in the limnology of our freshwaters owing to rapid population growth, urbanization and increased agricultural and industrial development practices. These changes, which may have been aggravated by climate change that resulted in increased temperature, and

shifts in precipitation and runoff patterns (Cocich, 2009), represent a serious threat to the health and livelihood of people of this country.

The rapid increase in the demand for more food production and the attendant agricultural development in Ethiopia have demanded the widespread use by farmers of fertilizers, which eventually enter water bodies and pollute them. Enrichment of water bodies with algal nutrients, which originate from fertilizers and enter nearby water bodies through runoff during periods of precipitation (Whatall *et al.*, 2007), often leads to the excessive growth of such nuisance/toxic algae as *Microcystis aeruginosa* in lakes and reservoirs (Lampert and Sommer, 1997). This is a water quality problem to which much public attention has been attracted in Ethiopia. Death of animals around Lake Chamo (Amha Belay and Wood, 1982) and of domestic animals and humans using the water in Koka Reservoir (<http://www.youtube.com/watch?v=eUqgUR4qI98>; <http://www.youtube.com/watch?v=rTUEjL8OhII>) in Ethiopia exemplifies the devastating effect of human-induced changes in freshwaters.

Thus, the assessment and monitoring of water quality of freshwater lakes and reservoirs is imperative in light of the threat chemical pollutants and toxic algae present to aquatic food web dynamics, public health and livestock. Moreover, despite their widespread distribution, small reservoirs are underexploited for recreation, ecotourism, fishery development and biodiversity conservation, particularly in Africa, due to our sparse limnological knowledge (Mwaura, 2003). Belbela reservoir, the subject of the study reported here, was built as a source of water for irrigation, which is now widely practiced in Ethiopia as an obvious response to water scarcity in agricultural areas. Information on the water quality of Belbela reservoir is almost non-existent. The reservoir water is, however, used for watering livestock, sanitation and as the primary source of drinking water supply for local people. It also supports commercial fishery, which is based on introduced fish species primarily *Oreochromis niloticus* (Pers. Observ.).

It is, therefore, essential to investigate the physico-chemical water quality of the reservoir and the composition and production of phytoplankton, which are regarded as good bioindicators of environmental pollution (Danielson, 1998; Apfelbeck, 1999) as they are the first community of organisms affected both quantitatively and qualitatively by external loading of materials (Anna-Lissa and Galina, 1999; Wen-Seng *et al.*, 2004).

2. Literature Review

Reservoirs are water bodies that are formed by humans for purposes of drinking and municipal water supply, industrial cooling water supply, power generation, irrigation, river regulation and flood control, commercial and recreational fisheries, aesthetic recreational uses and waste disposal (Chapman, 1996). Reservoirs show many of the same basic hydrodynamic, chemical and biological characteristics as the natural lakes (Chapman, 1996). Like natural lakes, reservoirs harbour phytoplankton and zooplankton, which form integral parts of the aquatic food web and influence other aspects of the lake including color and clarity of the water and fish production (Tansakul *et al.*, 2008).

The composition and productivity of phytoplankton and other aquatic biota are the result of the interplay of numerous environmental factors. As a result of human activities, the water column conditions in a reservoir may be modified with respect to these environmental variables. In the following sections, cursory treatments will be given to environmental factors that have relevance to phytoplankton dynamics.

One of the physical factors of overruling importance in fresh water ecosystems is light since it drives photosynthetic carbon fixation on which aquatic animals including the zooplankton are ultimately dependent (Kirk, 1994). It also serves as a sensory cue, and heats water and ultimately determines the thermal regime of a reservoir (Dodds, 2002). The quantity and quality of light in the water column of a water body changes because of changes in water transparency—a measure of the vertical extent in a water column to which light penetrates. Secchi discs are the standard tools used for measuring water transparency and monitoring inland waters along with measurements of chlorophyll a and phosphorus, which indicate the current status of the lake and long-term changes in their limnological features. The Secchi disc readings are a semi-quantitative measure of water transparency since a variety of factors such as the time of day, sky and water surface conditions, and differences between observers will give varying depths for the same location (Zipper *et al.*, 2007). The Secchi disk also provides a measure of the amount of suspended inorganic and organic matter in the water (Carlson and Simpson, 1996). Water transparency in a reservoir depends on the turbidity of the reservoir water. Turbidity is a measure of how particles suspended in water affect water clarity. Turbidity depends largely on total

suspended solids constituted by algae, algal detritus or inorganic sediment, which attenuate light and reduce water transparency (Kir, 1994; Tilzer *et al.*, 1995; Jassby *et al.*, 1999; Dodds, 2002). Typically, it increases sharply during and after a rainfall, which causes sediment to be carried into the creek. It is, therefore, an important indicator of suspended sediment and erosion levels (Aporn and Khuantrainrong, 2008). The replacement of forest with agricultural lands and urbanization can have significant effects on watershed hydrology and riparian functions thereby decreasing water quality, which is generally reflected by reduced water transparency (Webb and Walling, 1992). Long-term changes in the composition and concentration of suspended solids can have potential cumulative effects on aquatic ecosystems in a multitude of ways (Newcombe, 1991). Suspended solids probably act as the primary transport mechanism for pollutants and nutrients in streams through flocculation, adsorption, and colloidal action, (Stone and Droppo, 1994), reducing the spawning habitat (Slaney *et al.*, 1977), limiting the ability to find food (Sigler *et al.*, 1984) and increasing gill abrasion of fish (Goldes, 1983). Elevated turbidity also raises water temperature, lower dissolved oxygen and harms fish gills and eggs (Beha, 1997).

Thus, human activities such as shore-line modification and diversion of rivers into lakes lead to the introduction of materials which may be dissolved or suspended in the receiving water. The introduced materials prevent the penetration of adequate light, an important variable that controls phytoplankton growth and biomass. The unavailability of adequate light is not, however, the only reason for poor growth and low photosynthetic productivity of phytoplankton in aquatic ecosystems, The reduction of stratospheric ozone and the resultant increases in UV reaching the planet surface has also been shown to negatively affect phytoplankton primary productivity (Smith *et al.*, 1992).

Another physical factor that affects the composition, growth and productivity of phytoplankton is water temperature, which is influenced by latitude, altitude, season, time of day, air circulation, cloud cover and turbidity of the water column. Temperature affects physical, chemical and biological processes in water bodies and, therefore, the concentration of many variables (Makhlough, 2008). As water temperature increases, the rate of chemical reactions generally increases together with the evaporation and volatilisation of substances from the water (Harrison *et al.*, 2008). Increased temperature also decreases the solubility of gases including the carbon

source of photoautotrophic organisms-CO₂. The metabolic rate of aquatic organisms is also related to temperature, with respiration rates increasing and subsequently leading to increased oxygen consumption and decomposition of organic matter in warm waters (Chapman, 1996). Through its effect on the density of water, temperature also determines the stability of the water column in a reservoir by causing mixing/stratification.

Conductivity is among the chemical factors that influence phytoplankton growth. Electrical conductivity (EC) is a useful indicator of total dissolved solids (TDS) because the conduction of current in an electrolyte solution, is primarily dependent on the concentration of ionic species (Hayashi, 2004). Most natural waters contain dissolved ions (atoms or molecules possessing a charge) derived from the water's interaction with soil, bedrock, atmosphere, and biosphere. As a result of these ions, water is able to conduct electricity. Electrical conductivity (EC) is widely used for monitoring the mixing of fresh water and saline water, separating stream hydrographs, and geophysical mapping of contaminated groundwater (Hayashi, 2004). Significant increases in conductivity may be an indicator that polluting discharges have entered the water. Conductivity is a surrogate for salinity, which influences the osmotic environment of organisms. Abrupt changes in salinity as a result of the introduction of industrial effluents containing high levels of salts can lead to the disappearance of freshwater phytoplankton (Wetzel, 2001).

The taxonomic composition of phytoplankton is also influenced by the alkalinity of water, which is known to select for one or few species in the alkaline soda lakes of East Africa (Wood and Talling, 1988). Alkalinity refers to the equivalent concentration of titratable base present (i.e it is the acid-neutralizing capacities of water). It is mostly taken as an indication of the concentration of carbonate, bicarbonate and hydroxide, but may include contributions from borate, phosphates, silicates and other basic compounds (Reynolds and Osulvan, 2004). A related chemical property of natural water that affects its ability to dissolve minerals and influence chemical reaction is p^H (Champan, 1996). The balance of positive hydrogen ions (H⁺) and negative hydroxide ions (OH⁻) in water determines how acidic or basic the water is. In an aquatic ecosystem, the water's pH is affected by the chemicals discharged by communities and industries. When acid waters (waters with low pH values) come into contact with certain chemicals and metals, they often make them more toxic than normal. Therefore, pH is the master variable in the chemistry of aquatic systems

(Stumm *et al.*, 1981) that affects the kinetics of nutrient uptake and controls the chemical species of most of the nutrient ions algae require (Tilman, 1982). Extent of changes in pH is a function of the alkalinity of the water. Changes in pH in response to the addition of a particular acid or base, are smaller in higher alkalinity waters (i.e. high alkalinity waters are more buffered). The pH of both fresh and saline inland waters is also predictable from alkalinity (Kalf, 2002).

In aquatic systems, phytoplankton abundance and biomass are influenced by nutrient availability. Phosphorus (P), nitrogen (N) and silica (Si) are the basic substrate factors limiting the development of phytoplankton in nature (Correll, 1999; Reynolds, 2006). Concentrations of both inorganic N and P are often below the detection limits of standard methods during periods of most active microbial and algal growth in aquatic ecosystems (Guildford and Hecky, 2000). The primary sources of new nutrients to lakes are terrestrial runoff and atmospheric input (Guildford and Hecky, 2000). Various studies on temperate zone lakes including those of Dillon and Rigler (1974), McCauley (1989) and Champion and Currie (1990) have clearly shown that the variation among lakes in phytoplankton chlorophyll a concentration (*Chl a*) is strongly correlated with total phosphorus (TP) level. Although phosphorus often limits the growth of phytoplankton in temperate lakes (Schindler, 1977), nitrogen limitation of phytoplankton seems to be very common in tropical lakes (Hecky, 1988; Lewis, 1996; Talling and Lemoalle, 1998). The relative abundance of N and P (N:P) in a water body has also been suggested to have both quantitative and qualitative effect on phytoplankton community (Downing and McCauley, 1992; Wang *et al.*, 2008). Low N:P ratios, for example, lead to the dominance of cyanobacteria in various water bodies (Thorton, 1987). The molar ratios between nitrogen, silicon and phosphorus, known as Redfield ratio (Redfield *et al.*, 1963), are believed to be generally constant (N:Si:P = 16:15:1). The consumption of these nutrients by phytoplankton and mineralization of organic nutrients to inorganic forms also occur in the same ratios (Dafner *et al.*, 2001). The range of ambient or cellular N: P ratios have been used to define the transition between N and P limitation for algae (Sakamoto, 1966). If ambient N: P ratios (molar) are greater than 15-17:1, then P can be assumed to be in limiting supply. If the ambient N:P ratio is less than 9-10:1, then N can be assumed to be in limiting supply and 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. Variations in N:P molar ratios

are brought about by different combinations of processes like dissolution, concentration, sedimentation, fixation, and biological transformation (Downing and McCauley, 1992).

Reservoirs may exhibit particularly variable nutrient and light availability, both temporally and spatially (Vanni *et al.*, 2006). The severity and relative importance of nutrient and light limitation can vary seasonally in response to environmental drivers such as allochthonous nutrient inputs, incident solar radiation, thermal stratification and internal nutrient cycling (Vanni *et al.*, 2006). Eventhough tropical water bodies have a smaller seasonal temperature range, greater minimal monthly radiation and higher bottom temperatures, spatial and temporal changes in Chl-a biomass of phytoplankton are primarily controlled by nutrients, light and temperature in the water column (Adame *et al.*, 2008).

Phytoplankton dynamics is regulated not only by changing physical and chemical factors but also by biological factors such as grazing by zooplankton (Greneli and carsson,1999). Most zooplankton are phytoplankton consumers and thus contribute, along with other factors, to the reduction of phytoplankton numbers (Elbert and Shanz, 1989). If its size is large, the zooplankton can be assumed to have a great influence on the phytoplankton community. Since each species of grazer has its own preference for particular food organisms, selection preference has an effect on the composition of the phytoplankton community. Those species of phytoplankton, which are not preferred as food by the zooplankton are able to rapidly become dominant (Wetzel, 2001). Feeding type of tropical zooplankton varies among taxa, season and life stage (Brindle, 2005). For example, zooplankton rotifers are considered opportunistic organisms, showing high adaptive capacity, colonizing rapidly a wide variety of habitats and niches. Generally, zooplankton play an important role in food chains between the primary producers and secondary consumers, such as fish larvae and benthos (Nogrady *et al.*, 1993).

3. Description of the study area

The water body under consideration is a microdam, Belbela Reservior (Fig. 1), established in Ada'a-Liben district in East Showa zone of the Oromia region by damming Belbela river. It is a cascade reservoir created by damming along the course of Belbela river. It is located some 15 km east of Bishoftu (Debre Zeit) and about 55 km southeast of Addis Ababa along the road to chafe

Donsa. It is found between 38° 01' - 40° 04' E longitude and 08° 47' - 09° 00' N latitude. The reservoir is situated on a highland with an average elevation of 2,300 m a.s.l.

Belbela reservoir is one of the two storage dams constructed in 1980 by a Cuban Civil Mission in collaboration with Ethiopian Water Resources Authority (WRDA). The protection works, canals, and on-farm structures for the dam were later constructed by the Ethiopian Water Works Construction Authority (EWWCA) with an objective of irrigating land area to be used by State Farms. The reservoir is supplemented (recharged) by the other storage dam (Wadecha) through hydrological catchments transfers. Some physical features of the Belbela reservoir system are given in Table 1.

In the reservoir region, agriculture is the major source of employment, revenue, export earnings and a means for ensuring food security. As a result, there has been an ever-increasing expansion of irrigation-agricultural development practices in the area although workable management and exploitation strategies are not yet in place (Wakena Totoba, 2006).

The brown soil of the study area is dominated by clay, vertisols or black cotton soil, which is the type of clay mineral believed to have greater importance with respect to soil water storage (Wakena Totoba, 2006). The sloppy part of the land is degraded and exposed to high surface runoff. The poor natural vegetation cover, which is largely constituted by scattered bushes and small shrubs, afford little protection for the land from erosion and degradation. As a consequence, rill and gully erosion are not uncommon in the reservoir area (Wakena Totoba, 2006). This has important implications for the underwater light climate and water quality, which determine the trophic status of the water body under consideration.

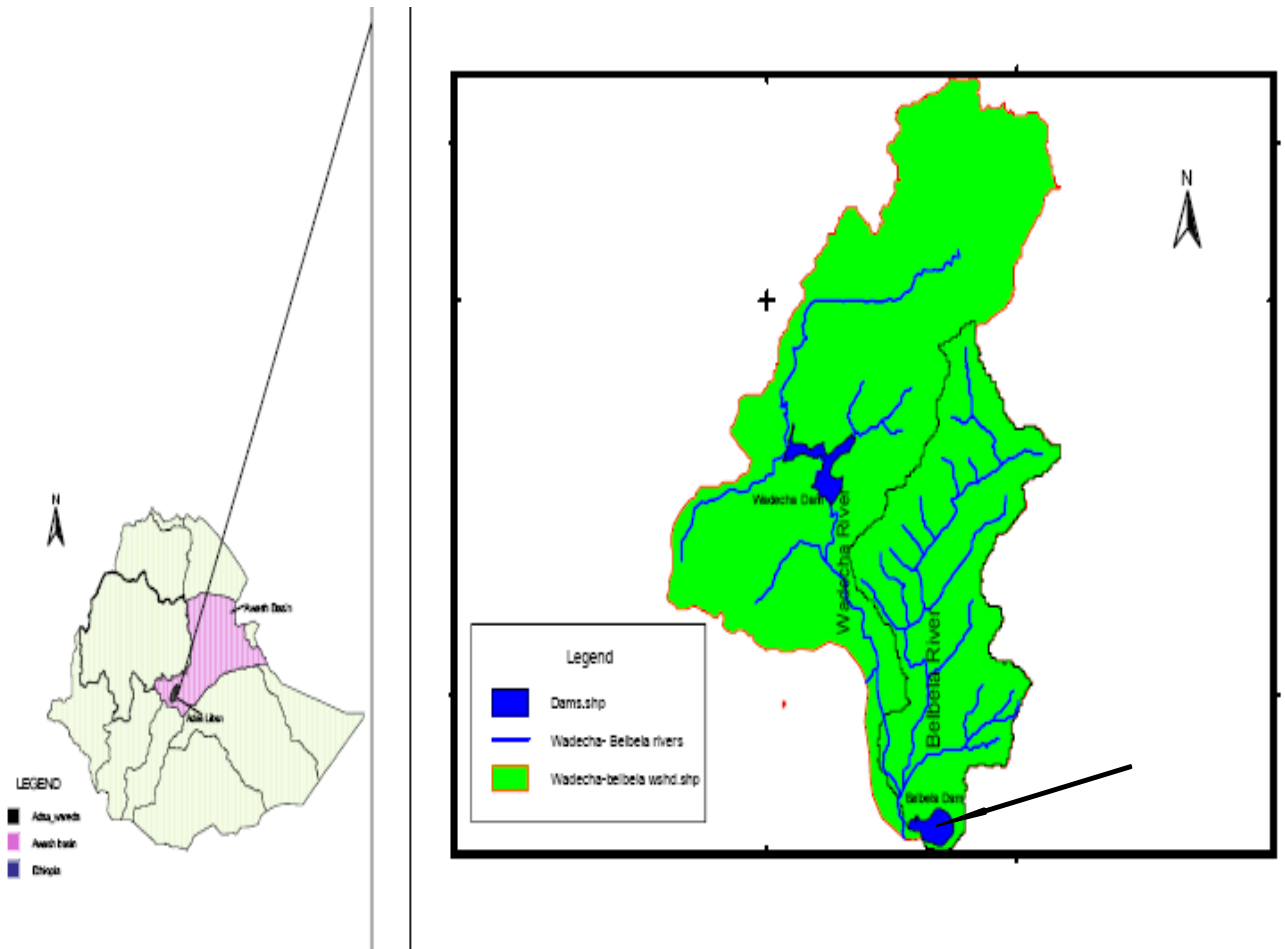


Fig 1. Map showing Belbela reservoir, its drainage system and sampling station (arrow head) (Source :Wakena Totoba, 2006)

The climate in the reservoir area is wet to sub-humid according to the Thornthwaite's system of defining climate or moisture regions (NMSA, 1996, cited in Wakena Totoba, 2006). The Belbela reservoir region has an extended period of wet season (March-September) with mean monthly rainfall varying from 50 to 223 mm. The rainfall of the study region is unimodal with the highest amount of rainfall occurring between June and September and accounting for about 74% of the mean annual precipitation in the catchment areas of the reservoir. The mean annual rainfall obtained by averaging records of 53 years documented by Debre Zeit Air Force Meteorological Station is about 866 mm (Wakena Totoba, 2006). Further Information on other meteorological parameters is also found in Wakena Totoba (2006).

Table 1. Physical features of Belbela reservoir. (Source: Oromia Irrigation Development Authority (OIDA, 2000, in Wakena Totoba, 2006).

Property	Measured values/Description
Type of the dam	Inclined earth fill
Catchment area	85 Km ²
Mean Runoff	21.6 Mm ³
Storing area at HWL	2.10Km ²
Storing volume at HWL	12.10Mm ³
Dead volume at LWL	0.67Mm ³
Useful volume	11.5Mm ³
Dam useful life	30 years
Bed level	1919 m a.m.s.l.
Crest level	1945.75m a.m.s.l.
Embankment length	710m
Free board	5.0m
Spill way- length	98m
-width at crest	15m
Maximum design flood of spillway	387m ³ /sec
Lowest water level (LWL)	1925m a.m.s.l.
Highest water level (HWL)	1940.75m a.m.s.l.
Outlet -Pipe Diameter	0.76m
Maximum discharge	3.87m ³ /sec
Mean discharge	2.4m ³ /sec
Useful volume to be discharged	11,500,000 m ³

Limnological information on Belbela reservoir is almost non-existent. Thus, the physicochemical and biological limnology of this reservoir remains to be explored. The reservoir is known to support flourishing commercial fishery, which is based on introduced fish species primarily *Oreochromis niloticus* (Lemma Abera and fishermen, Pers. Comm.). The reservoir serves as the primary source of drinking water for both local inhabitants and livestock. The reservoir water at the shores is also used for swimming and washing clothes by people living in the vicinity of the reservoir

4. Objectives

4.1. General Objective

- To investigate temporal dynamics of the community structure and productivity of phytoplankton in relation to physico-chemical and biological water quality in Belbela reservoir.

4.2. Specific Objectives

- ❖ To assess the temporal physico-chemical water quality of the reservoir during the study period.
- ❖ To investigate the taxonomic composition of phytoplankton and their grazers, zooplankton.
- ❖ To determine the quantitative importance of major phytoplankton and zooplankton taxa over the study period.
- ❖ To estimate the biomass and photosynthetic production of phytoplankton.
- ❖ To establish causal relationships among phytoplankton composition, biomass and production and physico-chemical and biological variables of the Belbela reservoir.

5. Materials and methods

5.1. Sampling protocol

Before the actual research work started, a reconnaissance survey of the reservoir was conducted. The reconnaissance survey was a fact-finding activity aimed at gathering background information on the reservoir system. During the reconnaissance survey, an offshore sampling station (maximum depth 6m) at about the centre of the reservoir was carefully selected in line with the observations made.

Samples were collected monthly with a Van Dorn bottle sampler from the pre-selected sampling station during the period spanning from September, 2010 to May, 2011. Samples were collected from selected depths distributed within the euphotic zone ($Z_{eu} = 3.5 \times \text{Secchi Depth}$) and mixed in equal proportions to produce composite samples. The composite samples were used for the analysis of inorganic nutrients, identification of phytoplankton taxa, and estimation of phytoplankton biomass as Chlorophyll a (*Chl a*) and photosynthetic production. For the identification and enumeration of zooplankton, separate samples were collected by towing upward from 1.3 m depth up to the reservoir's surface using a tow net (55 μm , Apstein). To avoid unpredictable changes, samples were transported in iced coolers and tests (examinations) were made within a couple of days after collection of samples.

5.2. Measurement of physico-chemical parameters in the field

Secchi depth (water transparency or vertical visibility) was measured with a standard Secchi disc of 20 cm diameter. Secchi depth was determined as the mean of the depth of disappearance and reappearance when the disk was slowly lowered and raised respectively. Photosynthetic Photon Flux Density (PPFD) was measured in $\mu\text{E m}^{-2} \text{sec}^{-1}$ ($\mu\text{mol quanta m}^{-2} \text{sec}^{-1}$) using a spherical sensor (model SPQA-1177) connected to a light meter (model Li-189). The spherical sensor connected to the light meter was lowered to specific depths and readings were taken directly from the instrument. Vertical extinction (attenuation) coefficient of downwelling irradiance (K_d) was calculated by using the Lambert-Beer's formula ($K_d = 1/z \times \ln(I_o/I_z)$) where I_o = Incident light falling on the reservoir's surface, I_z = amount of light received at a given depth, K_d = attenuation (extinction) coefficient. Euphotic depth was approximated as $4.6/K_d$ (where K_d is attenuation coefficient). Turbidity was measured by a portable digital turbidimeter (model HI-93703-11).

Depth profiles of temperature were determined using the combined conductivity-temperature probe (CC-505). Depth profiles of Oxygen were determined by the Azide modification of the Winkler Method (APHA *et al.*, 1999). Conductivity was measured with S-C-T meter (CC-505). Total alkalinity of the reservoir were determined by titration of the sample with 0.01N HCl to a pH of 4.5 using bromocresol green-methyl red indicator solution within a few hours after sample collection according to Wetzel and Likens (2000) and the values were expressed in meq L^{-1} . pH was measured with a portable pH meter (HI- 9024).

5.3. Analyses of Inorganic Nutrients in the laboratory

Concentrations of inorganic nutrients in all monthly samples were determined following the procedures outlined in Mackereth *et al.* (1978), Lind (1979) and American Public Health Association (APHA) *et al.* (1999) . The samples used for the analyses of all nutrients except ammonia + ammonium-nitrogen and total phosphorus (TP) were filtered through Glass Fiber filters (GF/F). Soluble Reactive Phosphate (SRP) and total phosphorus (after persulfate digestion) were determined by the Ascorbic Acid method (APHA *et al.*, 1999). Ammonia + Ammonium-N ($\text{NH}_3 + \text{NH}_4\text{-N}$) was determined by the Indo-phenol Blue method (Tzollas et al 2010) while Nitrate-N ($\text{NO}_3\text{-N}$) was analyzed by the Sodium-Salicylate method (ISO,7150/1(ISO 7150/1, Roberg and Edwards, 1983). Nitrite ($\text{NO}_2\text{-N}$) was determined

colorimetrically as a red azo dye after diazotization with sulphaniamide and coupling with N-1-naphthyl-(1)-ethylendiamine-dihydrochloride (Mackereth *et al.*, 1978). Molybdate-reactive silica was measured by the Molybdosilicate Method (APHA *et al.*, 1999).

5.4. Identification and Estimation of abundance of phytoplankton

Composite samples of phytoplankton, immediately preserved with Lugol's iodine, were examined in counting chambers (Sedgwick-Rafter cells) with an inverted microscope. Both preserved and fresh samples were used for the identification of phytoplankton to the genus or species level using such identification keys as Whitford and Schumacher (1973), Gasse (1986), Komareck and Kling (1991) and Cronberg *et al.* (2000).

Estimation of phytoplankton abundance was made using Lugol's iodine-preserved composite samples. 500 ml aliquots of phytoplankton samples were transferred to measuring cylinders, which were covered with parafilm to prevent evaporation and left in darkness to settle. The samples were allowed to stand for a period equivalent to a sedimentation rate of 4 cm d⁻¹ (Evans, 1982). The reservoir water in the sample kept for sedimentation was then carefully decanted leaving only 40 ml sub-sample at the bottom of the measuring cylinder. Of which 1 ml sub-sample was pipetted into a Sedgwick-Rafter cell and cells of each species of phytoplankton within 12-32 grids (squares) were counted randomly under an inverted microscope (Nikon) at a magnification of 400X. Number of cells in 15 to 20 filaments and colonies were counted and mean cell number per colony or filament of a taxon was determined and employed to estimate the total number of cells of a filamentous or colonial alga encountered in the phytoplankton samples (i.e. no. of filaments or colonies X mean number of cells per colony or filament) to make the comparison meaningful. Cell number of phytoplankton in the collected samples was calculated according to Hotzel and Croome (1999). The counts of phytoplankton species were added together to give total abundance of phytoplankton. The total cell number (cells ml⁻¹) of phytoplankton taxa in the reservoir was estimated according to Hotzel and Croome (1999) and Wetzel and Likens (2000). The cell counts of major phytoplankton species were summed up to give total abundance of phytoplankton.

5.5. Biomass as Chlorophyll a concentration and Photosynthetic Production of phytoplankton

Chlorophyll a biomass or standing stock (not corrected for phaeopigments) of phytoplankton was determined by filtering 300ml samples through Whatman glass fiber filters (GF/F). Pigments of cells retained on filter papers were extracted in 90 % acetone for 24 hrs under cool, dark conditions after homogenization. The extracts were poured into clean centrifuge tubes, firmly stoppered and were centrifuged for 10 minutes at 3000 rpm. The absorbance of pigment extracts was measured at 665 and 750 nm using a UV-visible spectrophotometer (Model Jenway 6405). Chlorophyll a concentration was calculated according to Talling and Driver (1963).

Photosynthetic production of phytoplankton was measured *in situ* as rates of oxygen evolution (quantity of oxygen evolved per unit time) by the Light and Dark bottle technique and the Winkler method of oxygen determination (Mackereth *et al.*, 1978; APHA *et al.*, 1999; Wetzel and Likens, 2000). Composite samples were incubated in 125 ml bottles between 10.00 a.m. and 2.00 p.m. for 3 hrs at depths of 0, 0.25, 0.50, 0.75, 1 and 1.50m. Immediately after incubation, 0.5 ml of each Winkler reagent was added to fix the oxygen in the incubation bottles. In the laboratory, samples were titrated with 0.01N sodium thiosulphate following the procedures outlined in APHA *et al.* (1999) and concentration of oxygen (in $\text{mg L}^{-1} \text{ h}^{-1}$) was calculated according to APHA *et al.* (1999). Conversion of the amount of oxygen evolved to the amount of carbon fixed was made assuming a photosynthetic quotient of 1.2 (Wetzel and Likens, 2000). The volumetric rates of gross photosynthesis ($\text{mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$) were converted to areal rates of gross photosynthesis ($\text{g O}_2 \text{ m}^{-2} \text{ h}^{-1}$) using the Grid enumeration technique (Olson, 1960). Areal (integral) rate of gross photosynthesis was equated to the area enclosed by the curve formed when rates of gross photosynthesis were plotted against depths of incubation. The hourly rates of gross photosynthesis were converted to daily rates by multiplying with the commonly used factors, 10 (hours of sunshine) and 0.9 (effective day length), empirically derived for tropical waters by Talling (1965). Biomass-specific rates of gross photosynthesis (P, in $\text{mg O}_2 (\text{mg Chl } a)^{-1} \text{ h}^{-1}$) were also calculated by dividing volumetric rates of gross photosynthesis by *Chl a* biomass of phytoplankton.

5.6. Identification of zooplankton taxa and determination of their quantitative abundance

Zooplankton samples were preserved with 5% formaldehyde solution in dark glass bottles of 125 ml capacity. In the laboratory, 50 ml of the samples were homogenized and 25 ml sub-sample was taken with a wide-mouthed pipette and placed in a grided Petri dish (15 grids). The Nauplii, copepodites and adult stages of the zooplankton were counted and summed up. The calculation was made using the formula in Wetzel and Likens (2000). Identification of zooplankton taxa was made using key of Fernando (2002).

5.7. Collection of Meteorological data

Data on mean monthly minimum and maximum air temperature and monthly rainfall of the reservoir area were collected from the Debre Zeit (Bishoftu) Meteorological Station.

5.8. Data Analysis

Linear regression and other relevant statistical methods (in Minitab-14 and Sigmaplot 10), were used to examine the causal relationships among the various physico-chemical and biological variables.

5.9. Description of Symbols and abbreviations

Z_{SD}	Secchi depth (m)
Z_e	Depth of euphotic zone (m)
K_d	Mean vertical extinction coefficient (In units m^{-1})
NTU	Nephelometric turbidity unit
DWSM	Dry weight of Suspended Matter (Total Suspended Solids) ($mg L^{-1}$)
TP	Total phosphorus ($(\mu g L^{-1})$)
TA	Total alkalinity ($meq L^{-1}$)
<i>Chl a</i>	Chlorophyll a ($\mu g L^{-1}$)
B	Phytoplankton biomass ($\mu g Chl a L^{-1}$ or $mg Chl a m^{-3}$)
GP	Gross photosynthesis ($mg O_2 m^{-3} h^{-1}$)
NP	Net photosynthesis per unit water volume, ($mg O_2 m^{-3} h^{-1}$)
A	Gross photosynthesis per unit water volume, ($mg O_2 m^{-3} h^{-1}$)
A_{max}	Light- saturated rate of gross photosynthesis per unit water volume

	(mg O ₂ m ⁻³ h ⁻¹)
P _{max} (A _{max} /B)	Biomass-specific rate of gross photosynthesis at light-saturation, mg O ₂ (mg Chl a) ⁻¹ h ⁻¹
ΣA	Hourly areal rate of gross photosynthesis, mg O ₂ m ⁻² h ⁻¹
Σ ΣA	.Daily areal rate of gross photosynthesis, mg O ₂ m ⁻² h ⁻¹

6. Meteorological data of the study area

Variations over the study period of total monthly rainfall and mean minimum and maximum air temperature are shown in Figure 2. The mean monthly minimum air temperature varied from 6.1 °C in December, 2010 to 13.8 °C in August, 2010 with a mean of 10.49 °C, while the mean monthly maximum air temperature ranged from 24.0 °C in July, 2010 to 28.8 in February, 2011 °C with a mean of 26.84 °C. The temporal variation in the mean monthly maximum air temperature was less marked than that of the mean monthly minimum air temperature. Relatively low maximum and minimum air temperature levels occurred during the major rainy period and dry period respectively indicating that the period November - January is not only the driest but also the coldest period of the year in this reservoir region.

Monthly rainfall of the study period ranged from 37.2 mm in May, 2010 to 189 mm in August, 2011. Monthly rainfall declined consistently from its highest peak in July to a relatively low value in September, 2010. The period extending from October, 2010 to February, 2011 was totally without any rainfall.

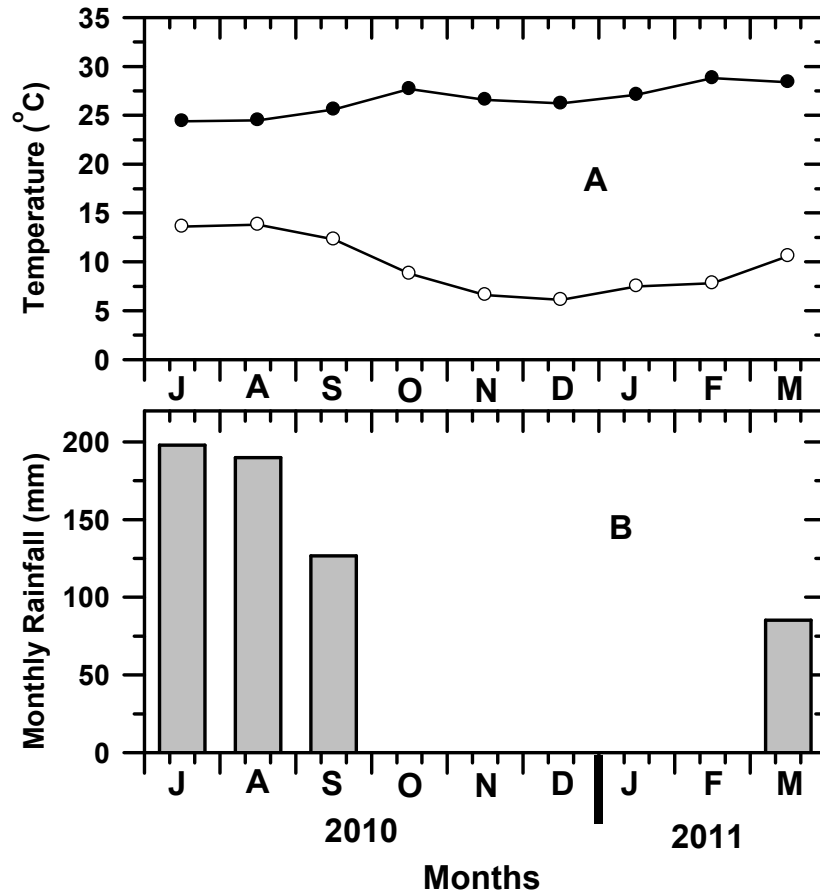


Fig. 2. Meteorological data of Belbela Reservoir: mean maximum (closed circle) and minimum (open circle) air temperature and monthly rainfall(bar graph). (Source: National Meteorological Agency)

Although rainfall data were not obtainable for April and May, there was significant precipitation in both March and April, 2011 (Fishermen, Pers, comm.). The Belbela reservoir region seems to have an extended period of wet season (March-September) with monthly rainfall varying from 50 to 223 mm (Wakena Totoba, 2006). Most of annual precipitation of the reservoir region occurs between June and September, accounting for about 74% of the mean annual precipitation in the catchment areas of the reservoir (Wakena Totoba, 2006). The mean annual rainfall obtained by averaging records of 53 years documented by Debre Zeit Air Force Meteorological Station is about 866 mm (Wakena Totoba, 2006). Yemenu (2009) also reported that the study area received an annual mean rainfall of around 789mm with moderate seasonal variability and bimodal pattern. The climate of the study area is wet to sub-humid according to the Thornthwaite's system of defining climate or moisture regions (NMSA, 1996, cited in Wakena Totoba, 2006).

7. Results and Discussion

7.1. Physico- chemical features of the reservoir

7.1.1. Physical features

Optical features and related variables

Temporal variations in the optical characteristics of the reservoir in relation to euphotic zone *Chl a* (ΣB) and dry weight of suspended matter (DWSM) are shown in Fig. 3. Secchi depth (water transparency) (Z_{SD}) of the reservoir varied from 0.16 m in February, 2011 to 0.26m in November, 2010, with the seasonal minimum Secchi depth value coinciding with peaks of suspended particulate matter and turbidity. The correlation between Secchi depth was strong and significant with DWSM ($r = -0.75$, $r^2 = 0.56$ at $p = 0.0200$) and moderate but insignificant with turbidity ($r = -0.65$, $r^2 = 0.43$ at $p = 0.0568$) although its correlation with both volumetric ($r = 0.15$, $r^2 = 0.02$ at $p = 0.6991$) and areal ($r = -0.31$, $r^2 = 0.1$ at $p = 0.3967$) phytoplankton biomass was poor and statistically insignificant. These results seem to suggest that Secchi depth is primarily a function of abiotic turbidity although, as Håkanson and Boulion (2000) noted, many factors are known to influence Secchi depth including phytoplankton biomass, amount of resuspended material and quantity of coloured matter in the water body. The yellow coloring matter,

presumably humic substances, observed particularly during periods of precipitation is believed to have made considerable contribution to the observed low Secchi disk readings of Belbela reservoir.

The present high Secchi-disk readings of Belebela reservoir are broadly similar to those reported for the shallow entirely exposed rift valley lake of Ethiopia, Lake Ziway (0.233; Wood and Talling, 1988) and Koka reservoir (0.28; Hadgembes Tesfay, 2007) though they are much lower than the values reported for Gefersa reservoir (0.20-0.66m; Nigatu Ebisa, 2010) and many other tropical reservoirs including Botafogo reservoir in Brazil (0.84 to 1.3m; Moura *et al.*, 2009).

The minimum turbidity (67 NTU) in Belbela reservoir was recorded in December, 2010, while the maximum (103 NTU) was observed in February, 2011. The turbidity values of Belbel reservoir are within the range of turbidity values that raw water can have (<1 to 1000 NTU; WHO, 1996). Turbidity decreased consistently from September, 2010 to its seasonal minimum in December, 2010 before it started increasing consistently to its highest seasonal peak in late February, 2011. The high turbidity values observed during the major and minor rainy periods seem to be associated with particulate and coloring materials imported into the reservoir through runoff during precipitation from the surrounding extensive farm plots and pasture land. Meesukko *et al.* (2007) have also reported that suspended solids varied in relation to seasonal changes in precipitation. Concentration of suspended materials in the reservoir varied from a minimum of 46.3 in December, 2010 to a maximum of 85 mg L⁻¹ in September, 2010. The highly significant positive and strong correlation of turbidity with suspended particulate matter ($r=0.95$, $r^2=0.90$ at $p<0.001$) was much higher than that with euphotic zone *Chl a* ($r=0.65$, $r^2=0.43$ at $p=0.0634$) probably suggesting the greater importance of abiogenic turbidity to the underwater climate of the study reservoir. Packman *et al.* (1999) have also reported strong positive correlation between total suspended solids and turbidity in a similar water body in the lowland of Puget. Since Belbela reservoir is shallow and completely exposed to wind action, frequent resuspension of sediments is a likely major source of water column turbidity in Belbela reservoir. Talling (1992) also argued that generation of turbidity by the wind-disturbance of bottom sediments is widespread in waters less than 5 m deep, and is accentuated when the wind-fetch is long.

The turbidity values of Belbela reservoir were generally comparable to those reported by Nigata Ebisa (2010) for Geffersa reservoir (7.2 to 189 NTU) although they are still much lower than those recorded for Legedadi (42 -452 NTU) by Adane Sirage (2006). Turbidity levels incredibly lower than the lower limits of the ranges of turbidity values recorded for Ethiopian reservoirs have also been reported from such other tropical reservoirs as Armando Ribeiro Goncaves reservoir in Northern Brazil (8.9NTU, Chellappa *et al.*, 2009).

The lowest K_d value (5.39) of the reservoir was recorded in February, 2011 while the highest (13.54) was observed in October, 2010 corresponding to the dry and major rainy periods respectively. The observed K_d values of the reservoir are clearly indicative of its water column of generally high turbidity. Lee *et al.* (2005) have indicated that light attenuation coefficient is directly related to the presence of scattering particles in the water column, either organic or inorganic, and thus is an indication of water clarity. K_d was negatively but poorly correlated with turbidity ($r=0.24$, $r^2=0.05$ at $p=0.5728$) and suspended particulate matter ($r=0.22$ $r^2=0.045$ at $p=0.6027$) although its statistically insignificant negative correlation with euphotic zone *Chl a* biomass of phytoplankton ($r=0.5$, $r^2=0.25$ at $p=0.2053$) was moderate. K_d values in excess of 10 are not unusual in water bodies with extremely high turbidity such as reservoirs (Allahsoh *et al.*, 1990) and brown water lakes (Chambers and Prepas, 1988). Such high K_d levels have been reported for Legedadi reservoir (Adane Sirage, 2006) and Lakes Ziway (Giram Tilahun, 2006) and Chamo (Eyasu Shumbulo, 2004) in which high concentrations of silt and clay particles are suspected to impart inorganic (abiogenic) turbidity. The calculated euphotic depths (Z_{eu}) varied temporally from a minimum of 0.34 of October, 2010 to a maximum of 0.82 m of February, 2011, both occurring during the dry period. Shallow euphotic depths are common in lakes and reservoirs with high biogenic or abiogenic turbidity and have been reported for Legedadi (Adane, 2006), Koka (Hadjembes, 2007) and Geffersa (Nigatu Ebisa, 2010) reservoirs .

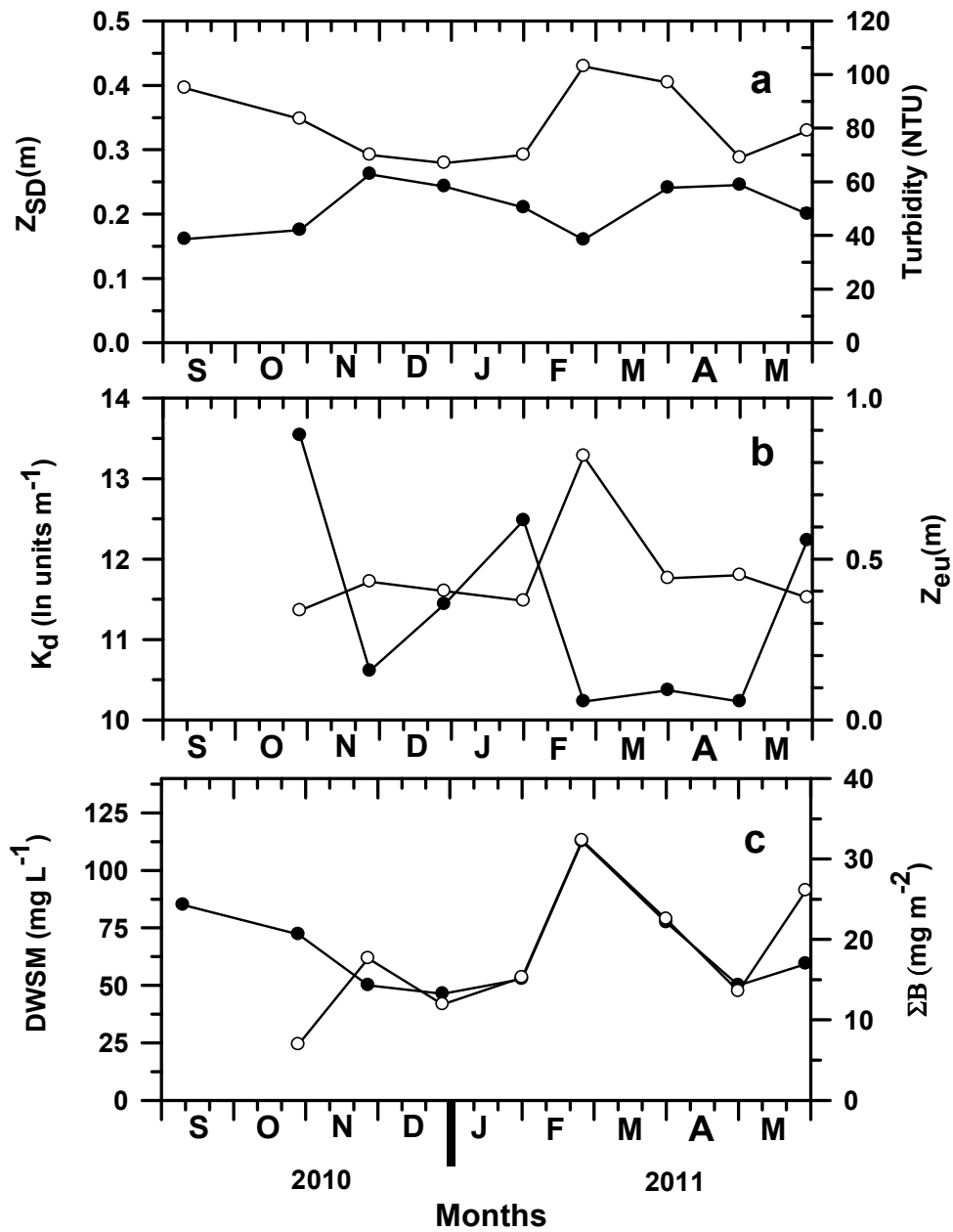


Fig. 3. Temporal variations in secchi depth (Z_{SD} , a, closed circle), turbidity (NTU, a, open circle, euphotic depth (Z_{eu} , b, open circle) and mean vertical extinction coefficient (K_d , b, closed circle) in relation to euphotic zone *Chl a* (ΣB , c, open circle) and total dry weight of suspended matter (DWSM, c closed circle) in Belbela Reservoir during the study period.

Temperature and Oxygen

Figure 4 shows the variation in surface water temperature in relation to air temperature while Fig. 5 illustrates depth profiles of temperature and dissolved oxygen. Surface water temperature varied significantly ($t= 4.9$, $p= 0.002$) from 18.50°C in December, 2010 to 24.1°C in April, 2011, with a mean of 21.3°C . Surface water temperature, which varied in a seasonal pattern roughly similar to that of air temperature, was positively and strongly correlated with air temperature ($r=0.78$ at $p= 0.064$) although the correlation was statistically insignificant. However, it is not always possible to relate surface water temperature with air temperature since a water body absorbs direct and diffuse shortwave and longwave radiation from the sun and the atmosphere while emitting long wave radiation from its surface and because of the convective exchange of sensible heat between lake surface and atmosphere (Livingstone and Imboden, 1989). Different morphometric properties of water bodies and degradation of their water quality due to eutrophication are also known to weaken the correlation of water temperature and air temperature (Skowron *et al.*, 2004). Thus, the surface water temperature change from month to month was smaller as compared to a similar change in air temperature, which is obviously related to the property of water. The surface water temperatures recorded for Belbela reservoir are broadly similar to those reported by Melaku Mesfin *et al.*, (1988) for Koka reservoir, although they are still lower than those recorded by Hadgembes Tesfay (2007) for Koka reservoir.

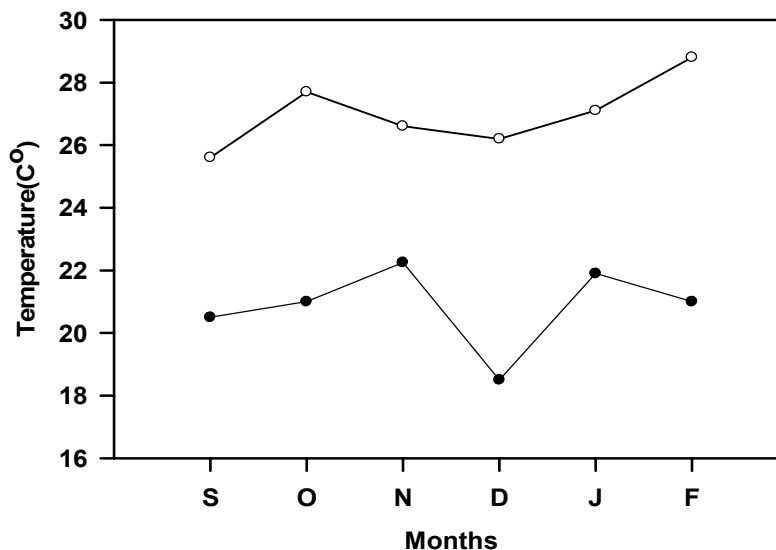


Fig 4. Variations in surface water temperature (closed circle) in relation to air temperature (open circle).

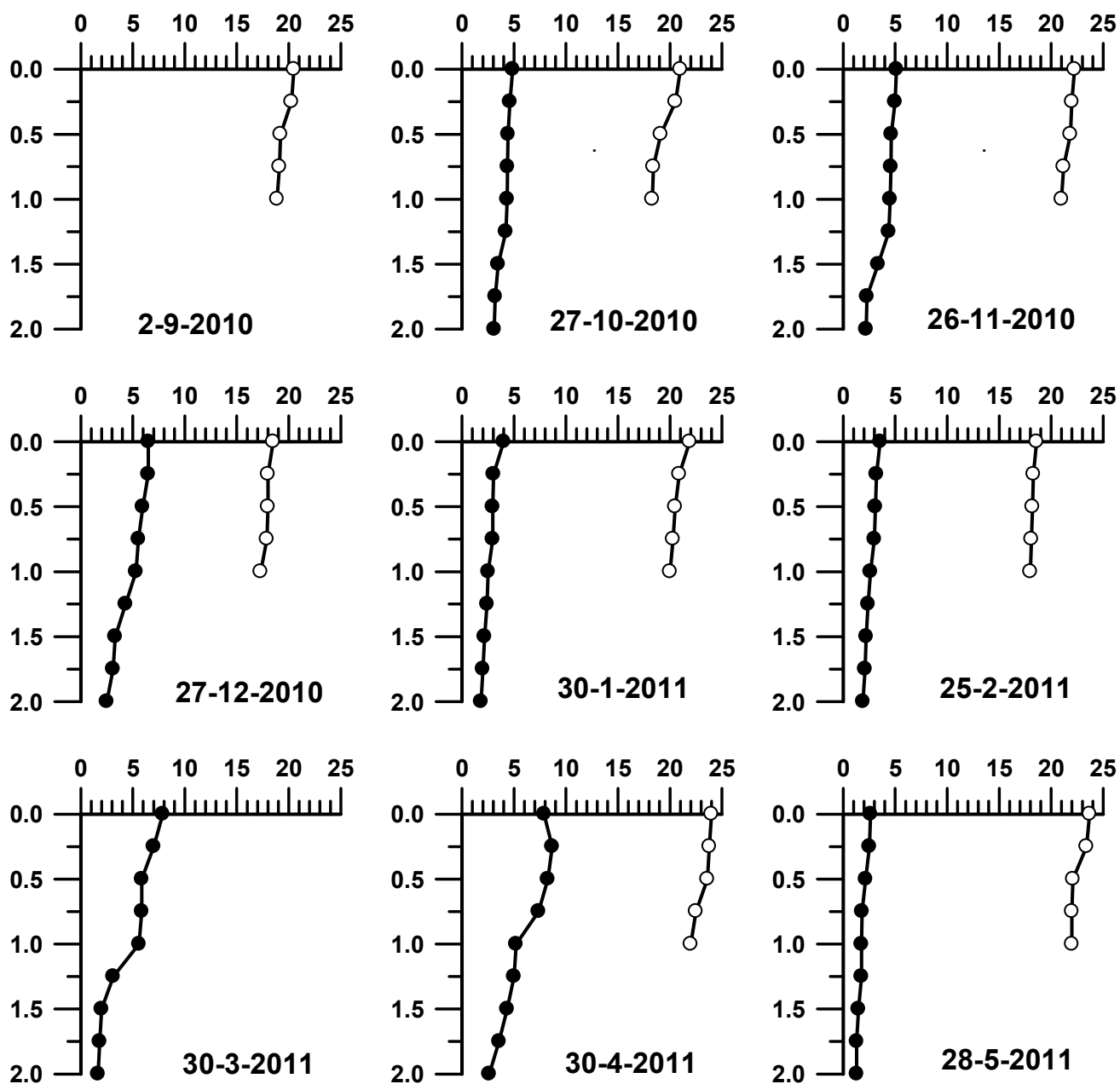


Fig. 5 . Depth profiles of dissolved oxygen (mg L⁻¹, closed circle) and temperature (°C, open circle) at a central station in Belbela Reservoir.

Although determination of complete depth profiles of temperature was not possible owing to the short cable used, well-marked and persistent thermal stratification is not likely to have occurred in the reservoir in light of its shallow depth and complete exposure to wind action. On some sampling dates, there was superficial type of stratification, which is generated by solar heating during the day and destroyed by nocturnal cooling at night. Wood *et al.*, (1992) have also stated that persistent temperature stratification is absent in shallow tropical water bodies apart from the diurnally developed superficial stratification resulting from the absorption of much daytime solar energy as observed in phytoplankton-rich Ethiopian lakes. Many reservoirs are polymictic due to their relatively shallow depths or the effects of enhanced flow-induced turbulence and thermal stratification is determined by the geography of their basins and vegetation cover of the catchment areas (Chapman, 1996).

Oxygen concentration at the surface of the reservoir varied with a minimum of 2.5 mg L⁻¹ in May, 2011 and a maximum of 7.9 mg L⁻¹ in April, 2011. Oxygen maxima at the surface were recorded throughout the study period except in April, 2011 when oxygen maximum was observed at a depth of 0.25m. Kauppasamy and Sivakumar (2008) described distribution of dissolved oxygen in a reservoir water, which was governed by a balance between inputs from the atmosphere, photosynthesis and loss by chemical and biotic oxidation. Lim *et al.* (2003) also argued that high organic matter and low redox potential favour increased sediment oxygen demand, causing the depletion of dissolved oxygen near the sediment surface. The levels and vertical distributions of oxygen in Belbela reservoir seem to be related to the vertical extent and rates of photosynthetic evolution of the gas, oxidative consumption of oxygen and the thermal regime of the water column.

7.1.2. Chemical features

Aggregate chemical parameters and free CO₂ concentration

The temporal variations in some aggregate chemical parameters and free CO₂ in relation to biological parameters are illustrated in Fig. 6. The electrical conductivity of Belebela reservoir ranged from a minimum value of 195.3 μS cm⁻¹ in November, 2010 to 285 μS cm⁻¹ in March, 2011. Higher conductivity values were generally observed during the major and minor rainy periods, which probably suggests the importance of inputs of solutes from the catchments of the

reservoir through runoff during precipitation. The maximum conductivity value of Belbela Reservoir is outside the range that drinking water should have (25-250 μScm^{-1}) (WHO, 1996). Conductivity was only moderately correlated ($r=0.5884$, $r^2=0.3462$ at $p=0.0956$) with total alkalinity although much higher correlations are expected in such a bicarbonate type water body. Seasonal variability in conductivity may be related to evaporation and dilution (Suttar, 2005; Story, 2008). Electrical conductivity can vary with temperature, mainly due to the effect of temperature on the viscosity of water related to ionic mobility (Wetzel, 2001).

The seasonal maximum conductivity value recorded for the reservoir is comparable to that recorded by Wood and Talling (1988) for Koka reservoir (286 $\mu\text{S cm}^{-1}$) although it is considerably higher than the conductivity reported by Melaku Mesfin *et al.* (1988) for Koka reservoir (200 μScm^{-1}), Adane Sirage (2006) for Legedadi reservoir (65-163) and Nigatu Ebisa (2010) for Geffersa reservoir (72.4-136.56 μscm^{-1}). Low conductivity values similar to those of other Ethiopian reservoirs mentioned earlier do not seem uncommon as many reports including those of Mwaura (2006) for the shallow Gathambra reservoir in Kenya (127 to 228 μscm^{-1}), Mustapha(2010) for Oyun reservoir in Nigeria (80.4 to 178.8 μscm^{-1}) and Kow and Wu (2010) for Feisuil reservoir in Taiwan suggest.

Alkalinity ranged from 1.9 in August, 2010 to to 3.6 meq L^{-1} in March, 2011 with its highest peak coinciding with peaks of light-saturated rate of gross photosynthesis (A_{max}) and *Chl a* biomass of phytoplankton. In comparison with the TA of most acceptable water supply sources, 0-8 meq L^{-1} (Bailey and Bilderback, 1998) and to the total alkalinity levels found in nature, 0.4-40 meq L^{-1} (Lind, 1979), the total alkalinities of Belbela reservoir are low and closer to the lower boundary values of both. Water bodies with such low level of alkalinity have low buffering capacity and hence a large change in pH or an increased risk of acidification can be caused by even a small amount of acid (Addy *et al.*, 2004).

The highest TA of Belbela is considerably higher than those reported for such water bodies as Koka reservoir and Lakes Zuguala crater and Afrera, 2.6, 2.1 and 2.2 meq L^{-1} (Elizabeth Kebede *et al.*, 1994) and Legedadi reservoir (0.3 -1.2 meq L^{-1} , Adane Sirage, 2006). pH varied between 7.84 and 8.22.

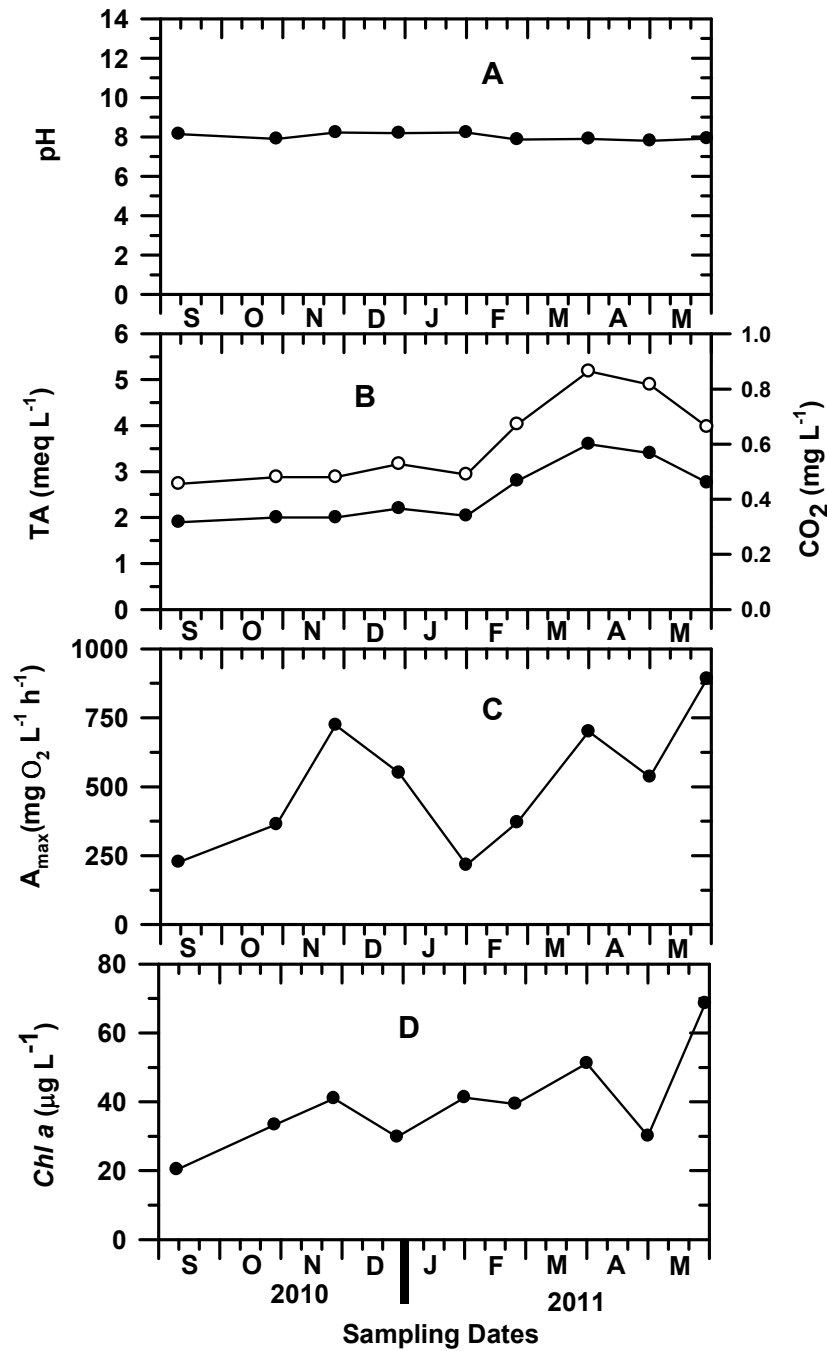


Fig. 6. Temporal variations in the concentrations of free CO₂ (open circle), total alkalinity (TA, closed circle) and pH in relation to *Chl a* biomass and light-saturated rate of gross photosynthesis (A_{max}) of phytoplankton in Belbela Reservoir.

The pH values of Belbela reservoir are in the middle of the range of the highest desirable pH levels for drinking water supply source (6.5-9.5; WHO, 1996). The correlation between total alkalinity and pH was positive and strong ($r=0.7421$, $r^2=0.5507$ at $p=0.0221$). Much stronger and positive correlations between pH and alkalinity data was, however, reported for Legedadi Reservoir (Adane Sirage, 2006), similar to those reported for Ethiopian Lakes (Wood and Talling, 1988) and saline lakes worldwide (Hammer, 1981).

The level of free carbon dioxide was calculated from the relationship between pH and alkalinity at a specific temperature as outlined in Lind (1979). The level of free carbon dioxide ranged from 0.45 in October, 2010 to 0.86 mg L⁻¹ in March, 2011, with the seasonal maximum value unexpectedly coinciding with peaks of *Chl a* biomass and A_{max} of phytoplankton. In light of the low light-saturated rates of gross photosynthesis of phytoplankton in this reservoir, the variations in free CO₂ levels seem to be linked more to the physical regime of the water body and oxidative metabolism of its biota. Free CO₂ levels recorded in the present study are considerably lower than those of Legedadi reservoir reported by Melaku Mesfin and Amha Belay (1989) and Adane Sirage (2006).

Inorganic Nutrients

The temporal variations of inorganic nutrients in relation to phytoplankton biomass measured as *Chl a* are shown in Fig. 7. Nitrate + Nitrite-N (NO₃ + NO₂-N) (hereafter referred to as nitrate) concentration (in µg L⁻¹) varied with a minimum value of about 27 in April and a maximum of 182 in March, 2011, with no apparent association with phytoplankton biomass. The correlation between nitrate levels and *Chl a* biomass of phytoplankton was poor ($r = 0.1336$, $r^2 = 0.0178$ at $p = 0.7318$) corroborating the lack of correspondence observed in the temporal trends of the two parameters.

Although the nitrate concentrations in the study reservoir are comparable to those reported for an offshore station in Koka reservoir (Hadgembes Tesfay, 2007), they are much lower than the values recorded for Geffersa (10 to 300 µg L⁻¹, Nigatu Ebisa, 2010) and Legedadi (240 to 1850 µg L⁻¹, Adane Sirage, 2006) reservoirs probably reflecting the differences among the reservoirs in the extent of application of fertilizers and external loading of the nutrient through runoff.

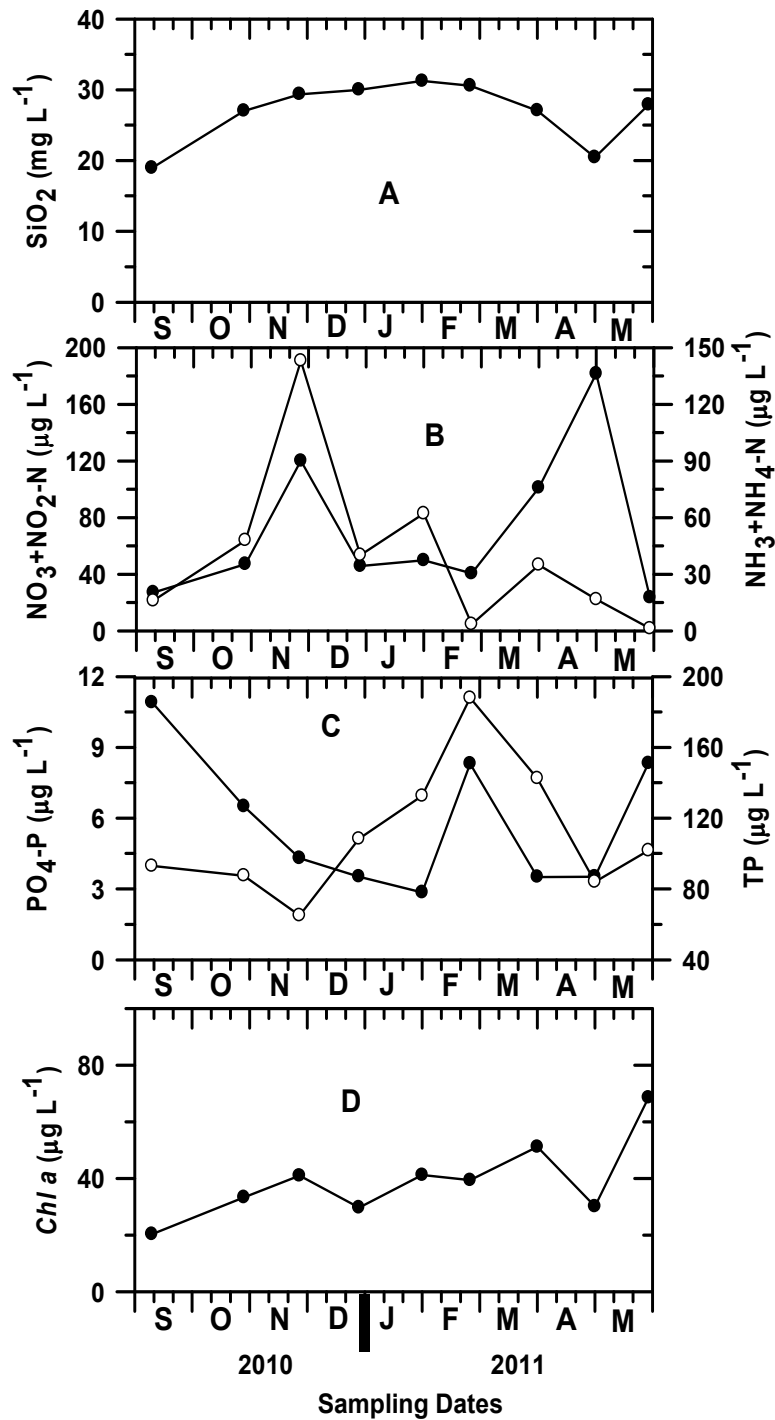


Fig. 7. Temporal variations in the concentrations of silica(A), nitrate + nitrite-nitrogen ($\text{NO}_3 + \text{NO}_2\text{-N}$, closed circle,B), ammonia + ammonium-nitrogen ($\text{NH}_3 + \text{NH}_4^+\text{-N}$ open circle,B) and Soluble Reactive Phosphorus (SRP, closed circle,C) and Total Phosphorus (TP, open circle, C) in relation to Chl a biomass of phytoplankton (D) at a central station in Belbela Reservoir.

Moreover, unlike the other reservoirs in Ethiopia, Belbela is seldom recharged with the water from Belbela river. The present results as well as those reported by Walmsley and Butty (1980), Gunatilake (1983), Thornton (1984), Walmsley and Thornton (1984) and Aweto (2003) on the levels of nitrate in the reservoirs of Africa and Sri Lanka constructed to serve as drinking water supply sources are indicative of the fact that nitrate levels are generally remarkably high in reservoirs, in contrast to what is observed in natural lakes of the tropical region (see Talling and Talling, 1965; Talling and Wood, 1988; Elizabeth Kebede, 1996).

$\text{NH}_3 + \text{NH}_4\text{-N}$ (ammonia + ammonium-nitrogen) (hereafter referred to as ammonium) concentration (in $\mu\text{g L}^{-1}$) fluctuated from 1.25 of May, 2011 to about 143 of November, 2010, with one of its seasonal peaks coinciding with those of nitrate and *Chl a* biomass of phytoplankton. Ammonium was poorly correlated with phytoplankton biomass ($r = 0.0740$, $r^2 = 0.0055$ at $p = 0.8500$). Ammonium concentrations were often lower than nitrate concentration in Belbela reservoir, as they usually are in productive lakes and in lakes after periods of circulation. Due to its rapid oxidation in well-oxygenated waters and preferential uptake by phytoplankton, the concentration of ammonium-nitrogen is usually low relative to other nutrients (Prochazkova *et al.*, 1970; Conway, 1977; McCarthy, 1980).

The concentrations of soluble reactive phosphate-phosphorus (SRP-P) ranged from a seasonal minimum of $2.85 \mu\text{g L}^{-1}$ in January, 2011 to its maximum concentration of $10.9 \mu\text{g L}^{-1}$ in September, 2011, with one of its seasonal peaks coinciding with the highest *Chl a* biomass peak of phytoplankton in May, 2011. Despite the fact that phosphorous is regarded as an extremely important element in controlling the trophic status of some tropical lakes (Kalff, 1983) and chlorophyll a was found to be strongly coupled to measured concentrations of phosphorus (Schindler, 1977; Prairie *et al.*, 1989), the correlation between SRP-P and *Chl a* biomass of phytoplankton was poor ($r = 0.0606$, $r^2 = 0.0037$ at $p = 0.8768$) in Belbela reservoir.

Total phosphorus varied temporally with minimum and maximum concentrations of $65 \mu\text{g L}^{-1}$ in November, 2010 and about $188 \mu\text{g L}^{-1}$ in February, 2011, respectively. Total phosphorous concentration was relatively low during September-November, 2009, which was probably associated with high phytoplankton abundance, particularly those of *Microcystis spp.* (Bicudo and Ramírez, 2003). The molar ratio of TN: TP ranged from 0.5 in April to 5.21 in March, 2011.

Redfield (1958) indicated that a departure from C:N:P molar ratio of 106:16:1 (50:7:1 by weight) imply nutrient deficiency and sub-optimal growth of phytoplankton. Ulen (1978), Forsberg and Ryding (1980) and Hellstrom (1996) have also shown that N:P ratio above 17 indicates P limitation, a ratio below 10 indicates N limitation and values between 10 and 17 indicate that either of the nutrients may be limiting. Hecky and Kilham (1988) also reported that phytoplankton are relieved from phosphorus limitation only at N: P molar ratios < 10:1. Thus, one may be tempted to assign the role of a limiting nutrient to nitrogen in light of the observed range of molar ratios of total nitrogen to total phosphorus. The role of a limiting nutrient is generally given to nitrogen in tropical lakes owing to its generally low ambient concentrations in tropical waters (Talling and Lemoalle, 1998; Sterner, 2009). Its low concentration in tropical lakes was explained by its greater internal loss through denitrification, which is favored by the high temperatures of lakes' bottoms in tropical regions (Lewis, 1996; Flecker *et al.*, 2009). Although data in support of the high frequency of nitrogen-limitation in tropical lakes are increasingly available (see, for example, Talling and Talling, 1965; Lewis, 1996; Elizabeth Kebede and Willen, 1998; Talling and Lemoalle, 1998; Eyasu Shumbulo, 2004; Sterner, 2009), there are some studies including that of Kalff (1983) which show the likelihood of phosphorus-limitation in East African lakes. Considering the fact that nutrient requirement of phytoplankton varies according to species, season and environmental conditions (Vrede and Tranvik, 2006), limitation of phytoplankton growth linked to one particular macronutrient may have not occurred in Belbela reservoir. From their nutrient enrichment experiments, Smith *et al.*, (2007) have also reported the colimitation of phosphorus, nitrogen and Iron during summer thermal stratification in Lake Erie.

Soluble molybdate reactive silica (SiO_2) concentration varied from about 19 mg L⁻¹ in September, 2010 to 31.2 mg L⁻¹ in January, 2011. Unlike the other nutrients, molybdate reactive silica was found at concentrations far from the ambient level assumed to be limiting to the growth of diatoms (< 0.5 mg L⁻¹; Reynolds, 2006) throughout the study period as it usually is in East African lakes including those in Ethiopia (Talling and Talling, 1965; Wood and Talling, 1988; Talling and Lemoalle, 1998). Silica increased continuously from its lowest value in September to the seasonal peak in January before it declined to another low level in April, 2011. Silica concentration showed moderate correlation with phytoplankton biomass ($r = 0.4525$, $r^2 =$

0.2047 at $p = 0.2213$) although its correlation with diatom abundance was very poor ($r = 0.0451$, $r^2 = 0.0020$ at $p = 0.9084$). The silica maximum observed in the present study is much lower than that reported for Legedadi reservoir (77 mg L^{-1} , Adane Sirage, 2006) although it is still comparable to that reported by Wood and Talling (1988) and is more than twice the level recorded for Gefferesa reservoir (13.4, Nigatu Ebisa, 2010). The variations in the levels of this nutrient among reservoirs is obviously related to the geology of their basins, land use practices in their catchment areas and the extent of the silica contribution of the rivers that feed them.

7.2. Biological Parameters

7.2.1. Phytoplankton species composition and abundance

Changes in both species composition and abundance of algal groups were observed in the temporal dynamics of the phytoplankton community in Belbela reservoir. The phytoplankton community was constituted by 47 species (Table 2) belonging to 30 - genera of six algal classes namely Cyanobacteria or cyanophyceae (blue-green algae), Chlorophyceae (green algae), Bacillariophyceae (diatoms), Euglenophyceae (Euglenoids), Cryptophyceae (cryptomonads) and Dinophyceae (dinoflagelates). Among the phytoplankton taxa, blue-green algae with 20 species were the most species-rich, followed by green algae and diatoms, with comparable number of species but with considerably higher abundance of the green algae. Other taxonomic groups of phytoplankton were relatively poorly represented in the phytoplankton community of the study reservoir. Although species diversity index was not calculated in this study, the reservoir can be regarded as having low species diversity in light of the number of species identified in comparison with those of Ethiopian Rift Valley lakes (Elizabeth Kebede and Amha Belay, 1994; Elizabeth Kebede and Willen, 1998). Temporal variations in the abundance and percentage contribution of different algal groups to the total abundance of phytoplankton in relation to *Chl a* biomass of phytoplankton in Belbela Reservoir are shown in Fig. 8. Total phytoplankton abundance varied temporally with seasonal peaks in November, 2010 and May, 2011, coincident with peaks of abundance of blue-greens and *Chl a* biomass of phytoplankton. The first peak of phytoplankton abundance also coincided with peaks of concentrations of nitrate and ammonia and peak of abundance of rotifers. The variations in time in the abundance of major taxonomic groups and *Chl a* biomass of phytoplankton in relation to the abundance of zooplankton groups are illustrated in Fig. 9. The phytoplankton community was characterized by the overwhelming

dominance of blue-greens, whose percentage contribution varied from about 39 % in February to 98% in May, 2011, with most contributions over 80 %. The quantitative importance of green algae was significant with their contributions to total phytoplankton abundance varying from < 2% in May, 2011 to about 56% in February. The dominance of green algae coincided with the seasonal minimum abundance of blue-green algae and considerably low abundance of zooplankton, particularly of rotifers and cladocerans. The contributions from other algal groups combined were always less than 1% throughout the study period.

Table 2. List of phytoplankton taxa identified in samples from Belbela reservoir

<i>Chlorophyceae (green algae)</i>	<i>Cyanophyceae (blue-green algae)</i>	<i>Bacillariophyceae (diatoms)</i>
<i>Actinastrum hantzschii</i> Lagerh.	<i>Anabena cirrinalis</i> Rab.	<i>Cymbella</i> sp.
<i>Carteria</i> sp. Cohn,	<i>A. flos-aque</i> Lyngb.	<i>Nitzschia closterium</i> Ahmad
<i>Closteridium tumidum</i> Dunder	<i>Aphanocapsa delicatissima</i> West & G.S West	<i>N. dissipata</i> (Kutz.) Grun.
<i>Elakatothrix gelatinosa</i> Wille	<i>Aphanothece microspora</i> Thomas	<i>N. navicula</i> Rachlia
<i>Pediastrum clatbratum</i> Jones	<i>Arthrospira</i> cf. <i>khanna</i> Ali	<i>Opephora martyi</i> Pring.
<i>P. duplex</i> Meyen	<i>Chroococcus minutus</i> Vincent	<i>Pleurosigma strigosum</i>
<i>P. simplex</i> Meyen	<i>C. multicoloratus</i> (Woloszyn.) Geit.	<i>Pseudonitzschia</i> sp. Rhodes
<i>Scendesmus dimorphus</i> (Turpin) Kutz.	<i>C. turgidus</i> Rao	<i>Synedra ulna</i> Kutz..
<i>Tetrastrum hentraconthum</i>	<i>Coelosphaerium kuetzingianum</i> Näg.	<i>Cryptophyceae (Cryptomonads)</i>
<i>Euglenophyceae (euglenoids)</i>	<i>Cyldropermopsis curvispora</i> Watan.	<i>Cryptomonas marsonii</i> Skuja
<i>Euglena acus</i> Ehr.	<i>C. africana</i> Kom. et Kling	<i>C. ovata</i> Ehr.
<i>E. limnophila</i> Lemm.	<i>Dactylococcopsis musicola</i>	<i>Dinophyceae (dinoflagellates)</i>
<i>E. spirogyra</i> Lemm.	<i>Microcystis aeruginosa</i> (Kutz.) Kutz.	<i>Peridinium cinctum</i> (O.F. Mill) Ehr.
<i>Trachelomonas caudata</i> (Ehr.) Stein	<i>M. flos-aque</i> (Wittr.)Kirchen	
<i>T. eucholera</i> .	<i>M. cf, bengalensis</i> Kutz.	
<i>T. hispida</i> Deflnder	<i>M. novacekii</i> Kom.	
	<i>Planktothrix rubescens</i> Kom.	
	<i>Planktolyngbya</i> sp.	
	<i>Pseudoanabaena limnetica</i> Nixodorf	
	<i>Raphidiopsis curvata</i> Rich	

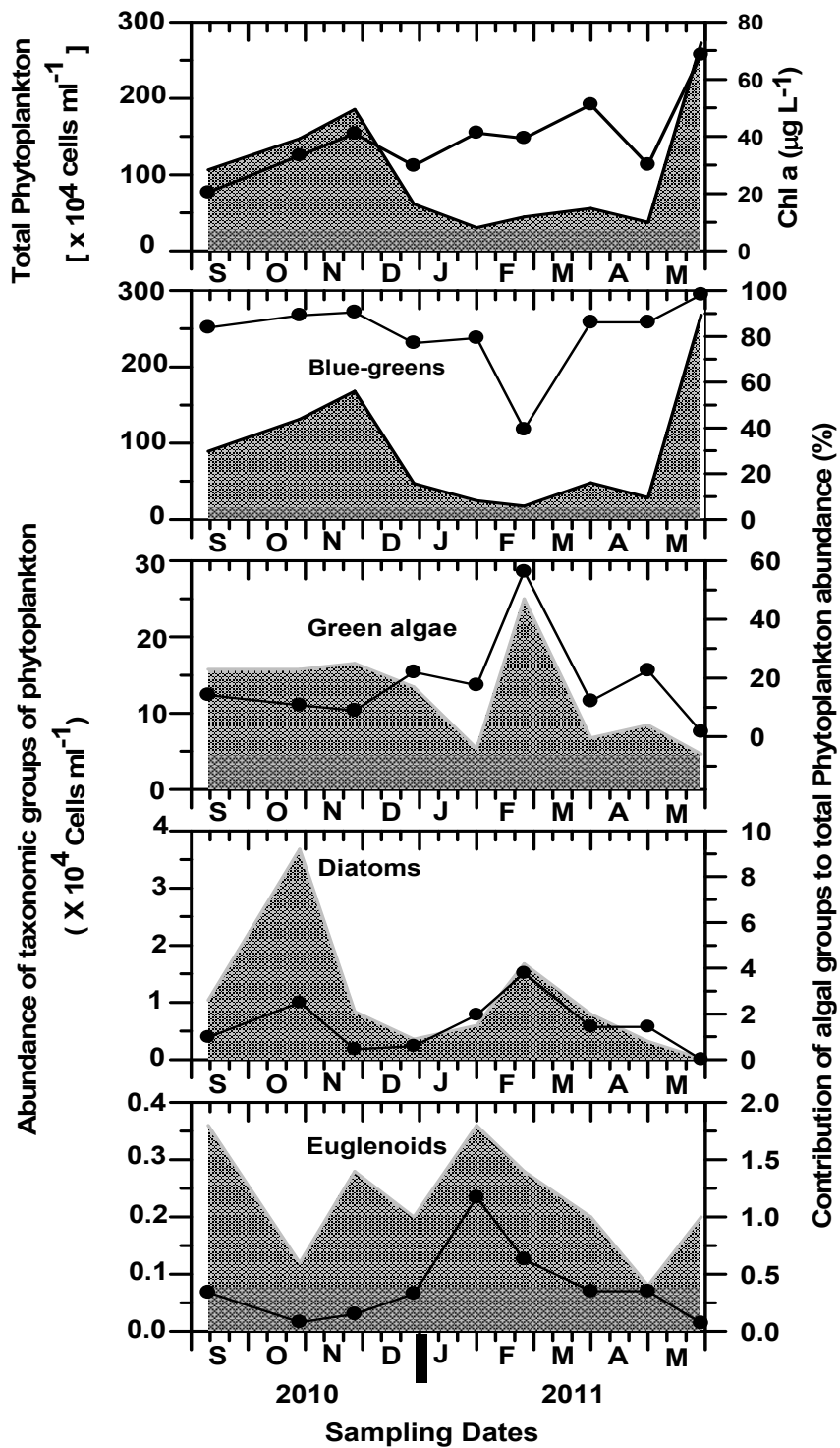


Fig. 8. Temporal variations in the abundance (Area plot) and percentage contribution of different algal groups (Line plot) to the total abundance of phytoplankton in relation to *Chl a* biomass (Line plot) of phytoplankton in Belbela Reservoir.

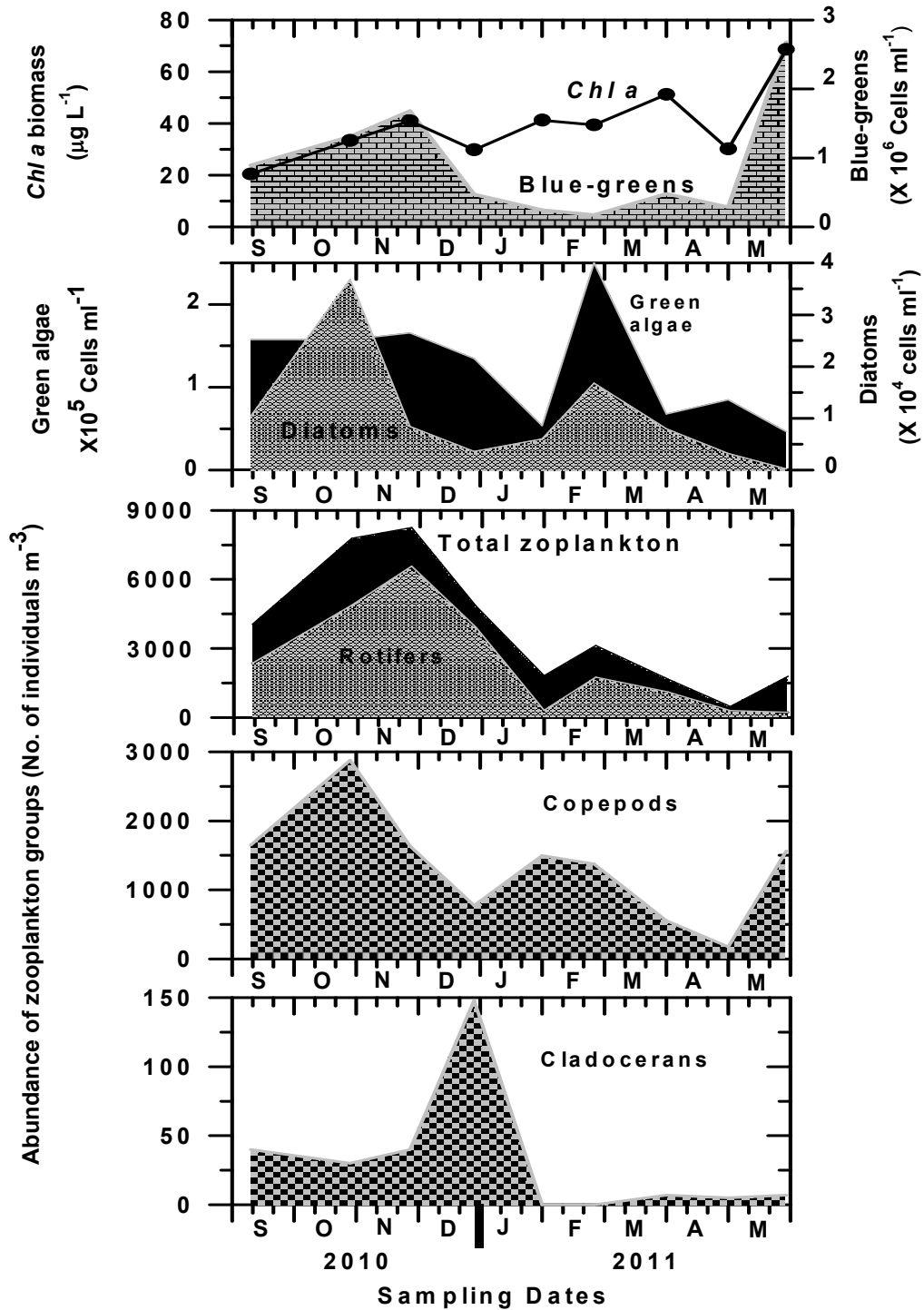


Fig.9. Temporal variations in the abundance of major taxonomic groups and *Chl a* biomass of phytoplankton in relation to the abundance of zooplankton groups in Belbela Reservoir

Figs. 10 and 11 show the temporal variations in the abundance of major phytoplankton taxa in relation to total abundance and *Chl a* biomass of phytoplankton in Belbela Reservoir. Of the blue-greens, species of the colonial genera *Aphanothece*, *Coelosphaerium* and *Microcystis* and the filamentous forms *Cylindrospermopsis* and *Planktothrix* were the most abundant of all phytoplankton taxa.

The major peaks of the colonial blue-greens occurred in November, 2010 coincident with peaks of abundance of total phytoplankton and blue-green algae and the first peak of phytoplankton biomass. The filamentous blue-greens attained their major peaks of abundance in March and May, 2011, the latter corresponding to the largest seasonal peak of total abundance and *Chl a* biomass of phytoplankton. The major peaks of the filamentous forms of blue-green algae also coincided with the period of notably low quantitative importance of all zooplankton groups. Among the green algae and diatoms, species of *Pediastrum* and *Nitzschia* respectively were the most quantitatively important taxa, with their seasonal maximum abundance occurring in November, 2010 coincident with the peaks of abundance of total phytoplankton and blue-greens and one of the major peaks of *Chl a* biomass of phytoplankton.

The overwhelming dominance of blue-greens constituted primarily by *Cylindrospermopsis*, *Microcystis* or *Planktothrix* in tropical lakes and reservoirs is not uncommon and was reported for Geffersa (Nigatu Ebisa, 2010), Koka (Hadgembes Tesfay, 2007) and Legedadi (Adane Sirage, 2006) reservoirs, the crater lakes Bishoftu (Tadesse Ogato, 2007), Hora (Tigsit Woubshet, 2010), Hora-Kilole (Rediat Abate, 2008) and Koriftu (Nebiyu Mohammed, 2010) and rift valley lakes of Ethiopia (Elizabeth Kebede, 1996; Giram Tilahun, 2006). The dominance of algal groups is initiated and enhanced by one or several environmental conditions such as light, temperature, water column mixing and availability of nutrients in the open water depending on the particular algal groups (Reynolds, 2006). High turbidity (Smith, 1986) and temperature (Shapiro, 1990), low level of carbon dioxide (Caraco and Muler, 1998) and high total phosphate (Watson *et al.*, 1997) are among the physico-chemical factors known to initiate and enhance cyanobacterial dominance.

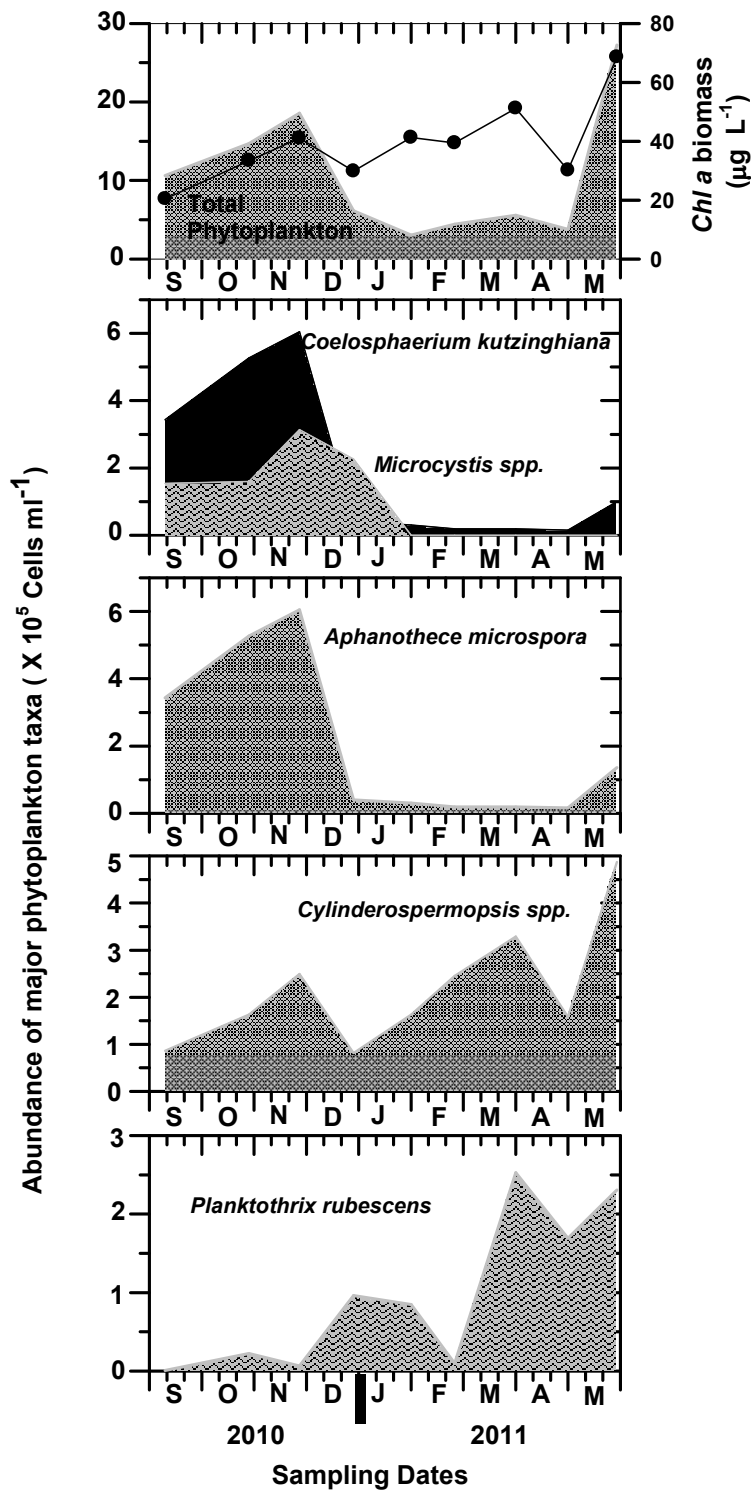


Fig. 10. Temporal variations in the abundance major phytoplankton taxa in relation to total abundance and Chl a biomass of phytoplankton in Belbela Reservoir.

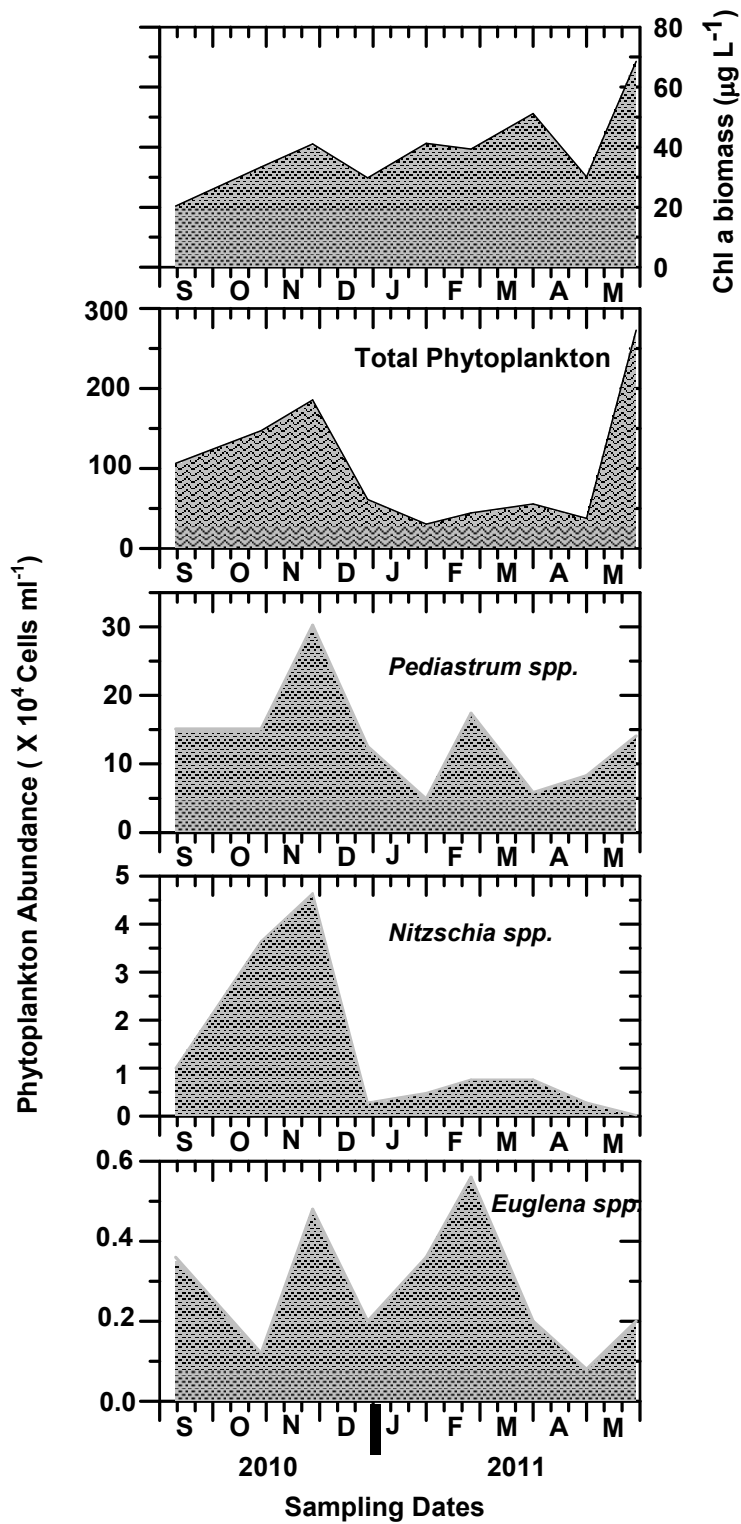


Fig. 11. Temporal variations in the abundance major phytoplankton taxa in relation to total abundance and *Chl a* biomass of phytoplankton in Belbela Reservoir.

Cyanobacteria are also endowed with the ability to store excess phosphorus (Peterrson, *et al.*, 1993), minimize grazing by zooplankton (Hanley, 1987), regulate their vertical position in the water column (Reynolds, 1987) and produce and release toxic extracellular products (Reynolds, 1987), which are structural and physiological adaptations that give them competitive advantage over other algal groups.

Nitrogen-limitation was reported to coincide with dominance by cyanobacteria (Wang *et al.*, 1999). In addition, Hyenstrand *et al.*, (1998) noted that nutrient-limiting conditions may interact with a number of environmental variables including high pH, zooplankton grazing pressure, elevated water temperature and fluctuations in light intensity to impact phytoplankton species composition and dominance by cyanobacteria. The dominance of blue-green algae in a large number of reservoirs in South Africa as a consequence of high nutrient loading and low nitrogen to phosphorus ratio was reported by Thornton (1987). The dominance of blue-green algae in eight Kenyan highland reservoirs due to high nutrient loading after long periods of rain, high temperature, their ability to fix nitrogen during dry periods and regulate their vertical position in the water column was also documented by Mwaura *et al.* (2002).

In Belbela reservoir, the dominance of blue-green algae was probably favoured by turbid, turbulent and nutrient-rich water column. Blue-green algae can regain their vertical position quickly owing to their effective buoyancy mechanism associated with gas vacuoles (Reynolds, 1987). As a consequence, mixing of the water column in this reservoir can affect the blue-green algae only temporarily. Owing to the same adaptive feature of blue-green algae, the poor underwater climate of the reservoir would not be a factor of overriding importance. In fact, positive buoyancy gives blue-green algae a competitive advantage over other algal groups as other algae like diatoms lack an effective mechanism of maintaining their vertical position in the water column.

Temporal dynamics of phytoplankton abundance and biomass are known to be controlled by loss processes including grazing by zooplankton. The dominant taxa of blue-green algae had, however, population peaks in November, 2010 despite the presence of abundant zooplankton, particularly rotifers and copepods. The colonial and filamentous cyanobacteria of the present study reservoir were probably less susceptible to grazing due to their large-sized algal units and

potential toxicity of species of *Cylindrospermopsis*, *Microcystis* and *Planktothrix*, which were most likely not manageable by zooplankton. It was demonstrated by Carney and Elser (1990) that grazers had weak effect on large-sized, cyanobacteria-dominated algal assemblages in eutrophic lakes. Elser and Goldman (1991) have also hypothesized that the coupling between zooplankton and phytoplankton weakens as nutrient availability increases due to the tendency for eutrophic systems to develop blooms of large, grazing-resistant phytoplankton. The impact of zooplankton on the phytoplankton assemblage of Belbela reservoir did not seem to be significant at the time of cyanobacterial dominance although rotifers particularly *Brachionus* species, which were abundant in the reservoir under consideration, are known to have considerable effect on phytoplankton owing to their resistance to the toxicity of blue-greens (Gonzalez, 2000). The results of studies conducted by Girum Tamire (2006) and Rediat Abate (2008) on the eutrophic Bishoftu crater lakes Koriftu and Hora-Kilole respectively, in which colonial and filamentous cyanobacteria had quantitative importance, also seem to indicate the lower contribution of zooplankton grazing to phytoplankton loss. The rare occurrence and sparse population of cladocerans and common occurrence and abundance of large-sized colonial blue-green algae, which are less edible and sometimes toxic to cladocerans (De Bernardi and Guissani, 1990), in Belbela reservoir seems a plausible explanation for the concomitant occurrence of seasonal peaks of abundance of total phytoplankton and blue-greens, *Chl a* biomass, total zooplankton and rotifers in November, 2010.

The diatom *Nitzschia* and the green alga *Pediastrum* were of quantitative importance to the phytoplankton community in Belbela reservoir. Diatoms are among planktonic algae, which can neither swim with flagella nor regulate their buoyancy with gas vacuoles. As a result, they are generally subjected to rapid sinking out of the well-lit zone due to their high density and consequently require more rapid vertical mixing than do cells that sink slowly (Reynolds, 2006). That is why diatoms are quantitatively important in turbulent environments with low irradiance (Graham and Wilcox, 2000), which is known to induce osmotrophy in a number of diatoms found in such habitats (Tuchman, 1996). *Nitzschia* peaked in November, 2010 when nitrate, ammonia and silica were at their peaks although concentration of suspended matter was close to its season minimum while Secchi depth was at its seasonal maximum level. Thus, in light of the optical parameters observed in the month of peak abundance, increased turbulence and the consequent poor underwater climate may not be a plausible explanation. Increased availability of

substrates of allochthonous origin for heterotrophic nutrition may have occurred as the month of peak of abundance is the immediate post-rainy month. Willen (1991) stated that increased frequency of diatom related to the decline of blue green algae is related to the fact that they are good competitors during turbulence in low light and low temperature conditions as long as the water column is not nutrient-deficient. The same diatom genus was found to be among the most abundant phytoplankton taxa in Lakes Hora (Tigist Woubeshet, 2010) and Babogaya (Yeshiemebebet Major, 2006) during the major rainy period.

Like diatoms, desmids gained quantitative importance during periods of vertical mixing (Graham, 2000). This, in part, explains why desmids including *Pediastrum* are found in considerable numbers in Lake Chamo (Girma Tilahun, 2006). Diatoms and green algae are good quality food for zooplankton and are, therefore, effectively grazed upon by zooplankton (Reynolds, 1984). The abundance of *Pediastrum*, however, peaked at the time of rotifer abundance. This is probably due to such structural adaptations as the possession of horn-like projections or chitinous bristles, which are regarded as buoyancy or herbivore deterrence devices (Graham and Wilcox, 2000). Euglenoids had a seasonal peak of abundance in February, 2011, which coincided with that of *Pediastrum* spp. The water column conditions that favored their abundance at this time of the year was not clear. Krysiuk and Messyasz (2006) related euglenoids dominance with huge availability of organic matter in light of their mixotrophic nutrition. The actively motile dinoflagellates, which were reported to be seasonally dominant in the nearby Bishoftu crater lakes including Lakes Babogaya (Yeshiemebebet Major, 2006), Hora (Tigist Woubeshet, 2010) and Hora-Kilole (Rediat Abate, 2008) were rare in Belbela reservoir. Although the ambient levels of nutrients in Belbela reservoir seem to favor the growth of dinoflagellates, species of algae differ in their tolerance to turbulence (Fogg, 1991; Willen, 1991) and these differences may affect competitive interactions. Dinoflagellates are inhibited by turbulence, which impedes cell division and disrupts cells (Pollinger, 1988; Lewis and Hallett, 1997).

7.2.2. *Chl a* Biomass of phytoplankton

The temporal dynamics of phytoplankton biomass measured as *Chl a* concentration are shown in Figs. 7 to 11 in relation to various environmental variables. Phytoplankton biomass measured as *Chl a* concentration ranged from 20.38 $\mu\text{g L}^{-1}$ in September, 2010 to 68.57 $\mu\text{g L}^{-1}$ in April,

2011. Peaks of phytoplankton biomass coincided with peaks of abundance of total phytoplankton and blue-greens. *Chl a* biomass maxima may, however, be associated with changes in cell pigment content, and spatial or successional trends in species dominance (Catalan and Felip, 2000). Tilman *et al.*, (1982) asserted that biomass of phytoplankton communities is primarily a function of the rates of nutrient supply and algal loss. Phytoplankton biomass did not, however, exhibit any clear association with levels of nutrients apart from the concurrence of the first small peak of biomass and seasonal peaks of nitrate and ammonia in November and the largest peak of biomass and increased level of SRP-P, TP and silica in May. Silica is the only nutrient which showed moderate correlation with phytoplankton biomass (see section 7.1.2) although the quantitative importance of diatoms was not comparable to that of blue-green algae and green algae. High phytoplankton biomass values coincided with peaks of abundance of total zooplankton, cladocerans and rotifers although the highest biomass occurred coincident with relatively low abundance levels of all zooplankton groups. Thus, zooplankton grazing does not seem to be the major factor for the observed temporal variations in phytoplankton biomass in Belbela reservoir. According to Talling (1986) water input-output and water-column structures are responsible factors for the seasonal patterns of phytoplankton biomass in African lakes.

The maximum phytoplankton biomass recorded for Belbela Reservoir is higher than those reported by Melaku Mesfin for Koka reservoir ($22.4 \mu\text{g L}^{-1}$) and Elizabeth Kebede *et al.* (1994) for the same reservoir ($16 \mu\text{g L}^{-1}$), Adane Sirage for Legedadi reservoir (22.19 to $39.45 \mu\text{g L}^{-1}$) and Nigatu Ebisa (2010) for Geffersa reservoir (2.29 to $40.67 \mu\text{g L}^{-1}$). The recent phytoplankton biomass values (20.85 to $221.01 \mu\text{g L}^{-1}$) documented by Hadgembes Tesfay (2007) for an inshore station in Koka reservoir, however, included levels which were much higher than the present seasonal maximum biomass in Belbela reservoir.

7.2.3. Photosynthetic Production of Phytoplankton

Depth profiles of gross photosynthesis

Depth profiles of gross photosynthetic rates are illustrated in Fig. 12. The photosynthetic biomass was presumably the same on all sampling dates as composite samples were used for all depths of incubation.

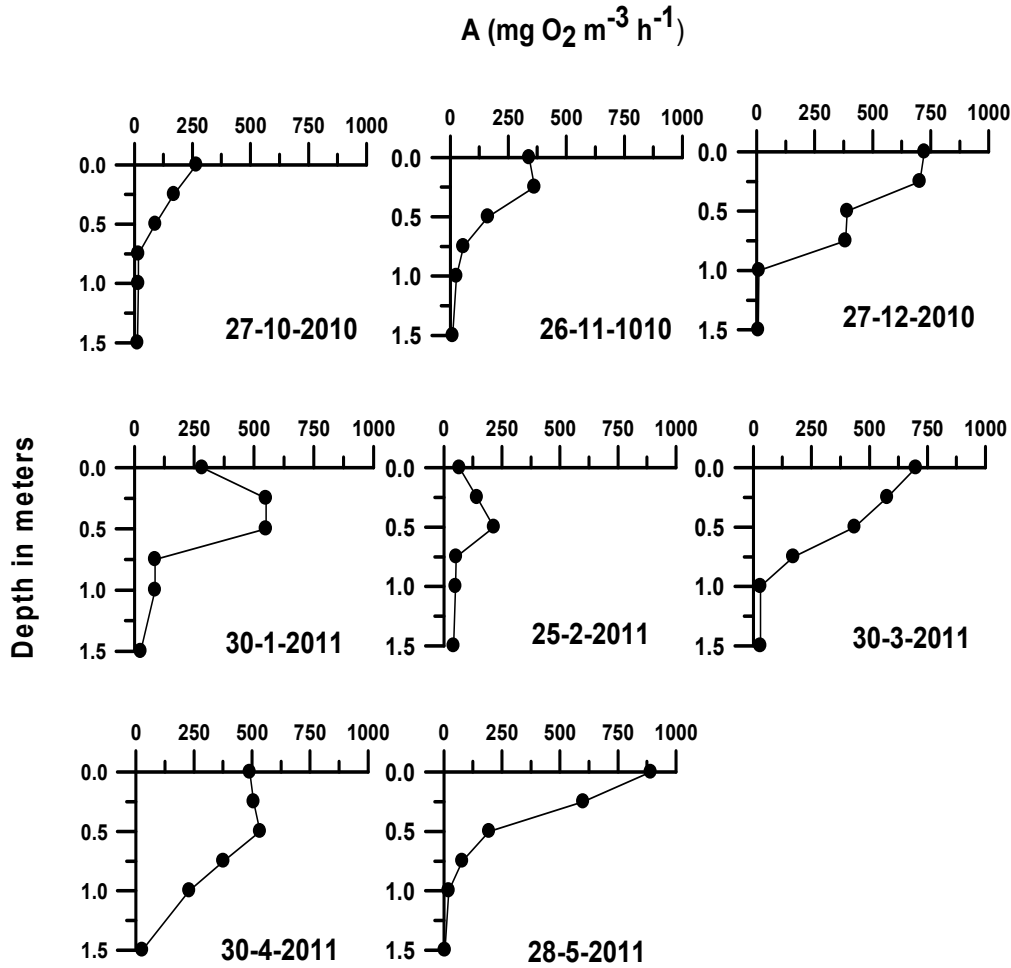


Fig. 12. Depth profiles of gross photosynthesis per unit water volume (A , $\text{mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$) at a central station in Belbela Reservoir.

The observed depth variations in photosynthetic rates were, therefore, responses of similar phytoplankton biomass to differing underwater irradiance. Depth profiles of gross photosynthetic rates (A , $\text{mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$) exhibited typical patterns known for phytoplankton with or without photo-inhibition at the surface. The observed gross photosynthetic rates per unit water volume in the reservoir showed temporal variations in the maximum rates of photosynthesis (A_{max}) and depths of their occurrence. The depth profiles were generally vertically compressed

owing to increased turbidity associated with wind-induced vertical mixing of this shallow reservoir. Such a depth-distribution pattern is typical of turbid systems such as Lake Ziway (Girma Tilahun, 2006) and Legedadi (Adane Sirage, 2006) and Koka (Hadjembes Tesfay, 2007) reservoirs. In such shallow water bodies, resuspension of inorganic particles from the sediment by frequent mixing or loading from the catchments through runoff reduces light penetration (Dokulil, 1994).

The photosynthetic rates at the reservoir's surface were generally depressed during the dry period, which is probably associated with the absence of cloud cover during this period. Surface gross photosynthetic rates ranged from $67 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$ ($17.42 \text{ mg C m}^{-3} \text{ h}^{-1}$) in February to $891 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$ ($231.66 \text{ mg C m}^{-3} \text{ h}^{-1}$) in May, 2011. Depressed rates of gross photosynthesis of phytoplankton at the surface of a water body (photoinhibition) is a common phenomenon and has been reported for lakes (Abebaw Wondie, 2006; Yeshiemebet Major, 2006; Tadesse Ogato, 2007; Zelalem Desalegne, 2007) and reservoirs (Adane Sirage, 2006; Hadjembes Tesfay, 2007) in Ethiopia. Photoinhibition is related to excess photons that become available when ambient levels of photons exceed physiological saturation (Long *et al*, 1994; Falkowski and Raven, 1997). The decrease in photosynthetic rates results from photo-oxidative disruption of pigment systems, inactivation of photosynthetic enzymes or increased photorespiration (Steemann-Nielsen and Jørgensen, 1962; Osmond, 1981; Falkowski and Raven, 1997). Factors that significantly govern the extent of algal photoinhibition include light intensity, cell density, algal strain and time of exposure to the high light intensity (Leon and Galvan, 1999).

Photosynthetic Parameters

Fig. 13 shows the temporal variations in light-saturated rate of gross photosynthesis (A_{\max}), specific rates of gross photosynthesis at light-saturation [P_{\max} , $\text{mg O}_2 (\text{mg Chl } a)^{-1} \text{ h}^{-1}$], and hourly rates of integral photosynthesis (ΣA , $\text{g O}_2 \text{ m}^{-2} \text{ h}^{-1}$) in relation to phytoplankton biomass (B). A_{\max} ranged from about 226 to 891 mg O_2 (≈ 70.5 to 278 mg C) $\text{m}^{-3} \text{ h}^{-1}$, corresponding to the lowest and highest phytoplankton biomass values recorded during the present study period. Its first peak in December coincided with a smaller peak of abundance of green algae, while the second and largest peak in May corresponded to the highest peak of abundance of total phytoplankton and blue-greens and *Chl a* biomass. In tropical lakes, the wide range of this saturation parameter per unit water volume, A_{\max} , is a function of primarily variable biomass

concentration, B (mg m^{-3}) and photosynthetic capacity, the light-saturated biomass-specific rate, P_{max} , $\text{mg O}_2 (\text{mg chl a})^{-1} \text{h}^{-1}$ (Talling and Lemoalle, 1998). A_{max} was moderately correlated with *Chl a* biomass ($r = 0.4785$, $r^2 = 0.2289$ at $p = 0.2304$). Positive and much higher correlations between A_{max} and *Chl a* biomass of phytoplankton are not uncommon and were reported for phytoplankton in Legedadi reservoir (Adane Sirage, 2006) and many tropical reservoirs including those in Sri Lanka (Amarasinghe and Vijverbeg, 2002) and in such nearby crater lakes as Babogaya (Yeshiemebebet Major, 2006), Bishoftu (Tadesse Ogato, 2007), Hora-Kilole (Rediat Abate, 2008) and Koriftu (Zelalem Desalegne, 2007). It is interesting to note that a photosynthetic biomass of about $30 \text{ mg Chl a m}^{-3}$ yielded an A_{max} of $723 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$, while a considerably larger photosynthetic biomass ($\approx 40 \text{ mg Chl a m}^{-3}$) was associated with less than one-third of the former ($216 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$) in Belbela reservoir. Similar lack of congruence between biomass and A_{max} was also recorded for several Ethiopian lakes (Talling *et al.*, 1973; Demeke Kifle and Amha Belay, 1990; Eyasu Shumbulo, 2004; Girma Tilahun, 2005; Yeshiemebebet Major, 2007). High A_{max} values associated with low algal biomass are the result of high specific rates of gross photosynthesis (Talling, 1965; Hammer, 1981). The positive and fairly high correlation between A_{max} and P_{max} ($r=0.6938$, $r^2=0.4813$ at $p=0.0563$) found in this study may in part provide an explanation for the association of high light-saturated rates with low algal biomass observed in Belbela reservoir.

The range of variations in A_{max} of the present study is broadly similar to those reported by Amha Belay and Wood (1984) for the shallow turbid Lake Ziway ($880 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$) and Demeke Kifle and Amha Belay (1990) for Lake Awassa ($217\text{-}425 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$). The temporal variations in A_{max} are often determined by B and P_{max} . However, nutrients may have a regulatory role. In the present study, the correlations of A_{max} with the macronutrients, nitrate ($r=0.3282$, $r^2=0.1077$ at $p=0.4273$) and soluble reactive phosphate ($r= 0.2522$, $r^2=0.0636$ at $p= 0.5468$) were very poor and insignificant. Smith (1979) has, however, shown the presence of strong and positive correlations between A_{max} and inorganic nutrients (N and P). Similarly strong correlations between A_{max} and phosphate, ammonia and nitrate were reported by Rediat Abate (2008) for the phytoplankton in the nearby crater lake, Hora-Kilole.

Consideration of the magnitude of the light-saturate rate of gross photosynthesis per unit of *Chl a* biomass [P_{max} , $\text{mg O}_2 (\text{mg Chl a})^{-1} \text{h}^{-1}$] enables one to compare the photosynthetic capacity of

phytoplankton communities. P_{\max} ranged from ≈ 5.5 to $24.3 \text{ mg O}_2 (\text{mg Chl a})^{-1} \text{ h}^{-1}$ with most values below 15. The observed P_{\max} values are broadly similar to those reported by Amha Belay and Wood (1984) for the shallow lake Ziway ($9.6\text{-}22.5 \text{ mgO}_2\text{m}^{-3}\text{h}^{-1}$) and Demeke Kifle and Amha Belay (1990) for Lake Awassa ($4\text{-}19 \text{ mg O}_2 \text{ m}^{-3} \text{ h}^{-1}$).

The highest P_{\max} value of the present study is higher than the level of photosynthetic capacity (about $20 \text{ mg O}_2 (\text{mg Chl a})^{-1} \text{ h}^{-1}$) considered as representative for many African lakes (Talling, 1965; Talling *et al.*, 1973) and an upper limit for lakes of the temperate regions (Bindloss, 1974). P_{\max} values observed for the phytoplankton community in Belbela reservoir are considerably lower than those reported for several lakes and reservoirs in Ethiopia (see, for example, Talling *et al.*, 1973; Eyasu Shumbulo, 2004; Adane Sirage, 2006; Rediat Abate, 2008 in which maxima of P_{\max} were between 30 and 35).

Several endogenous and exogenous factors are known to affect P_{\max} . P_{\max} is directly affected by temperature (Eppley, 1972), light (Falkowski, 1981), nutrient regimes (Falkowski and Stone, 1975) and cell size (Gilbert *et al.*, 1985). Temperature may not have been a factor of overriding importance to the temporal variation in P_{\max} as water temperature varied temporally within a narrow range. Data on irradiance levels of the study period are not available. The high P_{\max} values recorded during the dry period were probably due to the absence of cloud cover, which would otherwise reduce the quantity of photons reaching the reservoir's surface. Among the nutrients, nitrate was negatively but poorly correlated with P_{\max} ($r=-0.1235$, $r^2=0.0152$ at $p=0.7708$) while phosphate and P_{\max} were positively but moderately correlated ($r=0.6394$, $r^2=0.4088$ at $p=0.0878$) even though statistically insignificant. It was, therefore, possible that nutrients like SRP and probably irradiance and size-structure of phytoplankton were of overriding importance in determining the magnitude of P_{\max} . Tadesse Ogato (2007) also reported a strong and positive correlation between P_{\max} and SRP and a strong but negative correlation between P_{\max} and PAR for phytoplankton of Lake Bishoftu.

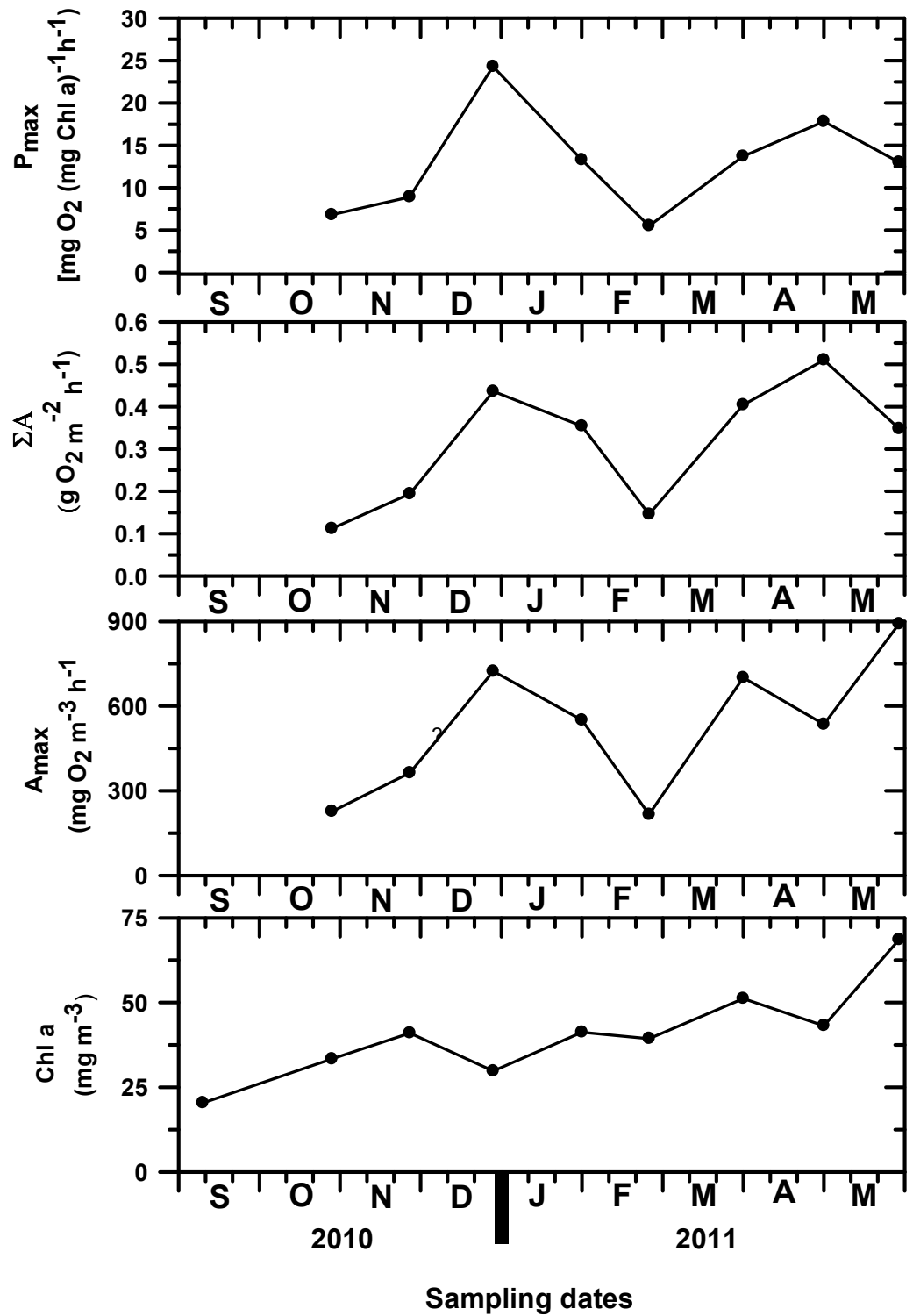


Fig. 13. Temporal variations in photosynthetic parameters in relation to phytoplankton biomass at a central station in Belbela reservoir.

The hourly rate of gross photosynthesis per unit area (ΣA , $\text{mg O}_2 \text{ m}^{-2} \text{ h}^{-1}$) ranged from 0.112 to 0.510 g O_2 ($\approx 0.035 - 0.159 \text{ g C}$) $\text{m}^{-2} \text{ h}^{-1}$. The hourly integrals exhibited a temporal pattern of variation, which was similar to that of A_{max} . The positive but strong correlation between ΣA and A_{max} ($r = 0.7551$, $r^2 = 0.5701$ at $p = 0.0303$) and the negative and weak correlation between ΣA and Z_{eu} ($r = -0.4100$, $r^2 = 0.1681$ at $p = 0.3131$) are consistent with the assertion that gross photosynthesis per unit area is influenced by the light-saturated rate of photosynthesis and the vertical extent of photosynthetic activity. The hourly integral rates of the present study are similar to those of Legedadi reservoir (0.067 to $0.58 \text{ g O}_2 \text{ m}^{-2} \text{ h}^{-1}$, Adane Sirage, 2006) although they are considerably lower than those reported for a reservoir (Hadgenbes Tesfay, 2007) and lakes (Demeke Kifle and Amha Belay, 1990; Abebaw Wondie, 2006; Yeshiemebet Major, 2006; Tadesse Ogato, 2007; Zelelem Desaaegne, 2007) in Ethiopia.

Daily production rates per unit area ($\Sigma \Sigma A$, $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$) were estimated from the hourly integral rates. The empirically derived factor of 0.9 was multiplied by the number of hours of sunlight (10 hr) and then the product was multiplied by the hourly rates per unit area for the estimation of daily rates of gross photosynthesis per unit area (Talling, 1965). The calculated daily rates of photosynthesis ($\Sigma \Sigma A$) ranged from 1.008 to 4.590 g O_2 ($0.226 - 1.193 \text{ g C}$) $\text{m}^{-2} \text{ d}^{-1}$. The observed daily integral rates are broadly similar to those of Legedadi reservoir (0.067 to $5.8 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$, Adane Sirage, 2006) and Lake Babogaya (1.01 to $5.98 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$, Yeshiemebet Major, 2006)

Belbela reservoir exhibited temporal variations in the photosynthetic production of phytoplankton. Owing to the great reduction in variation imposed by marked seasonality in temperature and irradiance, tropical waters are assumed to exhibit limited temporal variability in their primary production (Lewis, 2000). Absolute values of areal production rates do not tell much about the extent of the seasonality in phytoplankton production per unit area in a lake. Thus, Melack (1979a) used coefficient of variation (CV, standard deviation/mean) as an index for determining the extent of temporal variability in the rates of phytoplankton production per unit area. Belbela reservoir, with a CV of 33.3 % and 46.4%, for *Chl a* biomass and hourly areal production rates, falls under Pattern A of Melack (1979a), with most tropical lakes including Lakes Chamo (Eyasu Shumbulo, 2004), the Kenyan freshwater Lakes Naivasha Crater Lake and

Oloiden (Melack, 1879b), Chad (Lemoalle, 1975) and Victoria (Talling, 1965) in which production rates varied in relation to dry-wet seasons or vertical mixing/stratification or increased river discharge or a combination of two or more of these and the associated changes in turbidity and levels of nutrients (Melack, 1979a).

7.2.4. Zooplankton species composition and abundance

Table 3 lists the zooplankton taxa identified in samples collected from Belbela reservoir. Rotifers were the most species-rich zooplankton group with 11 species, 5 of which belonged to the genus *Brachionus*. The second most important zooplankton group was the copepods. The cladocerans were, as usual, poorly represented, with only 2 species, in the zooplankton community of Belbela reservoir.

Table 3. List of zooplankton taxa identified in samples collected from Belbela reservoir.

Rotifers	Cladoceran
<i>Brachionus calyciflorus</i> Pallas	<i>Moina</i>
<i>Brachionuscaudatus</i> Barrois&Daday	<i>Cerodaphnia</i> Sars
<i>Brachionus falcatus</i> Zacharias	Copepods
<i>Brachionus plicatilis</i> Muller	Calanoids
<i>Branchionus angularis</i> Gosse	<i>Metadiaptomus mavretanicus</i>
<i>Keratella tropica</i> Apstein	Cyclopoids
<i>Keratellacochlearis cochlearis</i>	Mesocyclops
<i>Filinia pejleri</i>	Thermocyclops
<i>Filinia pejleri grandis</i>	Copepodites
<i>Hexarthra mira</i> Hudson	Naupli
Trichocerca sps	

The major species of rotifers, which were responsible for the dominance of the group included *Branchionus caudatus*, *B. calyciflorus*, *B. plicatilis*, *Keratella chochlearis*, *K. tropica* and *Filinia peljleri*. Green and Mengestou (1991) also asserted that the zooplankton of Ethiopian lakes exhibit high rotifer diversity with *Branchionus* dominance and a mixture of species found throughout Africa.

Fig. 14 and Table 4 shows the temporal changes in the contribution (%) of the three zooplankton groups to total zooplankton abundance in relation to the abundance of major groups and *Chl a* biomass of phytoplankton. Rotifers, which were largely constituted by *Branchionus spp.*, were the most important zooplankton group with their contributions to total zooplankton abundance varying from about 33 % in May, 2011 to 92% in November, 2010. Fernando (1980) also argued that, numerically, *Branchionus* typically constitutes more than 50% of the total rotifer assemblage in tropical lakes, which may be partially attributed to their wide range of tolerance to alkalinity and salinity. The largest peak of abundance of rotifers coincided with the largest peak of total zooplankton abundance and a smaller peak of green algae, blue green algae and *Chl a* biomass of phytoplankton. The increases in the density of such edible phytoplankton as green algae and diatoms seem to be responsible for the generally high rotifer numbers during the dry period.

The dominance of rotifers is not uncommon. Rotifers were the most abundant zooplankton in Lakes Hora (Tamiru Gebre, 2006), Koriftu (Girum Tamire, 2006), Hora-Kilole (Rediat Abate, 2008) and Geffersa reservoir (Nigatu Ebisa, 2010) and some reservoirs in Brazil (Arcifa, 1984). The abundance of rotifers could be attributed to their parthenogenetic reproductive patterns and short generation time under favorable conditions and to their ability to feed on various food types (Wetzel, 2001). Twombly and Lewis (1987) also attributed zooplankton seasonal dynamics to birth rate and resting egg production, which are determined by physical parameters of the aquatic ecosystem and species specific stimuli. The dominance of rotifers in waters of the tropical regions can be ascribed to their preference for warm waters (Dumont, 1992; Segers, 2003). An abrupt drop in the abundance of rotifers occurred coincident with those of all major phytoplankton groups and one of the major peaks of copepods abundance which may indicate lower food availability for the rotifers and over compitance by copepods.

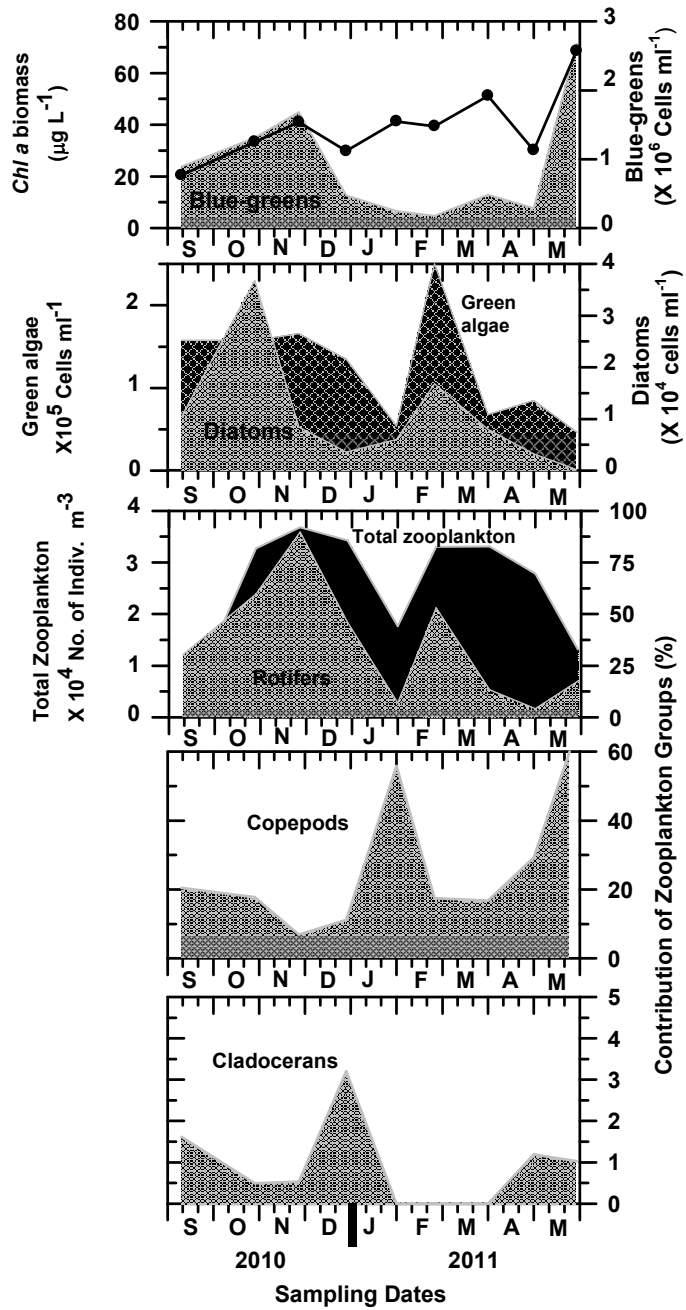


Fig. 14. Temporal variations in the percentage contribution of zooplankton groups to total zooplankton abundance in relation to major phytoplankton groups, *Chl a* biomass and total abundance of zooplankton in Belbela reservoir.

Table 4. Seasonal changes in percentage contributions of major zooplankton groups.

Sampling date		2/9/ 2010	27/10/ 2010	26/11/ 2010	27/12/ 2010	30/1/ 2011	25/2/ 2011	30/3/ 2011	30/4/ 2011	28/5/ 2011	
% Contribution zooplankton	Branchionus	16.2	47.7	1.3	40.9	13.9	31.5	50.7	31.6	23.1	
	Keratella	39.9	17.3	45.5	20.7	10.1	31.8	10.1	37.9	3.5	
	Rotifers	Filinia	21.4	16.7	45.6	13.9	17.4	19.2	21.9	0	2.1
		Hexarthra	0.2	0	0	0.6	1.4	0	0	0	0
		Trichocera	0.2	0	0	0.6	1.4	0	0	0.	4.2
	Cladocera	Moina	0.6	0.16	0.3	2.6	0	0	0.5	0	0.7
		Cerodaphnia	0.97	0.3	0.3	0.6	0	0	0	1.2	0.3
	Copepods	Calanoids	0.3	0.5	0.3	0.2	1	0.3	6.7	5.8	2.1
		Mesocyclops	6.5	4.9	0.1	6.8	1.7	2.1	7.1	18.7	0.3
		Thermocyclops	1.3	0.8	0.9	1	0	2	0	0	1.3
		Copepodites	0.9	1.2	0.1	1.1	1	0.4	0.5	0.6	9.8
		Nauplii	11.4	10.2	5.4	1.9	52	12	8.3	4.11	52.4

Fernando (1994) argued that the low abundance of large-sized zooplankton in many tropical lakes and reservoirs is due to strong mortality associated with fish predation. Wagner *et al.* (1997) also indicated that lakes with moderate zooplanktivory tend to be dominated by smaller-bodied species, such as rotifers and copepods, while lakes with little to no zooplanktivory tend to be dominated by larger-bodied cladocerans such as *Daphnia* due to selective predation by fish on larger zooplankton.

The percentage contribution of copepods ranged from about 7% in November, 2010 to 66% in May, 2011 with the latter corresponding to the highest peak of abundance of total phytoplankton and bluegreens and *Chl a* biomass might indicating high threshold of food availability on which the copepods feed. The low percentage contribution observed in november probably might reflect the increased predation by zooplanktivorous fish. The numerical importance of Copepods in Belbela reservoir is consistent with the results obtained for Lake Hora by Tamiru Gebre (2006) while contrasting with that reported by Rediat Abate (2008) for Lake Hora-Kilole in which copepods were rarely encountered in zooplankton samples. Copepods were the least

abundant zooplankton group in Lake Hora-Kilole (Rediat Abate, 2008). The contribution of Cladocerans never exceeded 5% , with their largest peak of percentage contribution corresponding to a high level of abundance of green algae. Physical conditions such as turbidity affect cladoceran dynamics interfering with their locomotion or food search (Wetzel, 2001).

8. Estimation of Trophic Status of Belbela from Trophic State Index (TSI)

The estimation of trophic state index (TSI) requires six physical, chemical and biological parameters: total phosphorus (TP), total nitrogen (TN), chemical oxygen demand (COD), Secchi depth (SD), chlorophyll-‘a’ (*Chl-a*) concentration and phytoplankton biomass (Sharma *et al.*, 2010). However, Carlson’s Index, which is based on secchi disk, chlorophyll-‘a’ and total phosphorous, has largely been used to assess the trophic status of lakes in almost all countries. Avarage TSI for Belbela reservoir was at 73.73 which places Belbela reservoir under the category of hypereutrophic lakes in both Carlson’s and Indiana trophic classes (see Tables 5 and Table 6 below).

TSI rates individual lakes, ponds and reservoirs based on the amount of biological productivity occurring in the water. Using the index, one can get a quick idea about the extent of productivity of a lake (Hillsborough, 2008). TSI values can be used to rank lakes within a region and between the regions. This ranking enables the water managers to target lakes that may require restoration or conservation activities. An increasing trend in TSI over a period of several years may indicate the degradation of the ecological health of a lake.

Model of Carlson (1977) was used to calculate Trophic States Index (TSI) of Belbela Reservoir.

$TSI - P = 14.42 * \ln [TP] \text{ (in } \mu\text{g/L)} + 4.15$, $TSI - C = 30.6 + 9.81 \ln [Chl-a] \text{ (in } \mu\text{g/L)}$,
 $TSI - S = 60 - 14.41 * \ln [Z_{SD}] \text{ (in meters)}$ and $Average\ TSI = [TSI (P) + TSI (Chl-a) + TSI (Z_{SD})] / 3$ where TP is total phosphorus, Chl-a is chlorophyll a, Z_{SD} is the Sechi depth.

Table 5. Carlson’s Trophic State Index (Carlson, 1977) and trophic status of Belbela reservoir

TSI	<30	30-50	50-70	>70-8	73.73
Status	Oligotrophy	Mesotrophy	Eutrophy	Hypereutrophy	Belbela

Table 6. Indiana Trophic State Index (ITSI) (Source: Sharma, 2010) and Trophic status of Belbela reservoir

TSI Scores	Trophic Class
0-15	Oligotrophic
16-31	Mesotrophic
32-46	Eutrophic
47-75	Hypereutrophic
73.73	Belbela reservoir

According to the calculated scores of trophic state indices, the reservoir is hypereutrophic. When the trophic state index parameters are treated separately, the reservoir can be regarded as being mesotrophic (Z_{SD}), eutrophic (*Chl a*) or hypereutrophic (TP) and in reality the reservoir did not categorized under hypertrophic, the problem of categorizing under hypertrophic can be attributed to the trophic state index used, which was standardized for temperate region but it might be categorized under eutrophic status

9. Conclusions and Recommendations

Belbela reservoir is a very low salinity turbid eutrophic body of water that supports phytoplankton community, which is primarily constituted by cyanobacteria, green algae and diatoms. The phytoplankton community exhibits temporal changes in species composition, abundance, biomass and photosynthetic production in relation to changes in the physicochemical and biological conditions of the water column of the reservoir.

Due to its shallow depth and complete exposure to wind action, the reservoir had consistently high turbidity, which was primarily of abiogenic origin. It is this turbidity, combined with the turbulent and nutrient-rich water column, which favoured the dominance in Belbela reservoir of blue-green algae constituted primarily by such potentially toxic taxa as *Cylindrospermopsis*, *Microcystis* and *Planktothrix*.

The reservoir water is suitable for human and animal consumption in light of the levels of aggregate chemical parameters and such physical variables as turbidity. The acceptability of the water from Belbela reservoir for human consumption is, however, questionable in light of

cyanotoxins and heavy metals as the reservoir harbours potentially toxic cyanobacteria of high levels of abundance and is found in the proximity of floriculture industries that use a variety of chemicals.

Temporal dynamics of phytoplankton seem to be related mainly to physico-chemical variables associated with hydrographic (water mixing) and hydrologic (water input-output) conditions. However, grazing by zooplankton and fish may be a significant factor controlling phytoplankton biomass and composition in the reservoir. Based on the results of the present study, the following recommendations are made:

- Conduction of zooplankton grazing experiments is mandatory to unequivocally resolve the question of phytoplankton control. The impact of fish on the algal flora should also come into picture.
- Before undesirable phenomena similar to that of Koka reservoir take place investigations geared towards the assessment of cyanotoxins and heavy metals in the reservoir water should be carried out.
- There is a need for a closer look at the impact of human activities like irrigation, shore-line modification and removal of plant cover surrounding the reservoir with a view to develop strategies of preventing further degradation of the aquatic ecosystem and conserving its resources.
- Interaction of the sediment and the overlying water with regard to inorganic nutrients should be investigated as the reservoir does not seem to have continuous input from the Belbela river.
- As far as the reservoir is used as the primary source of drinking water by local inhabitants and their cattle, continuous monitoring of the reservoir water quality standards by the concerned authorities is recommended.

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11. Appendices

Appendix 1. Results of Regression analysis among physical, chemical and biological variables at $\alpha=0.05$

Independent variable	Dependent Variable	Sample size (n)	r	r ²	p
DWSM	Z _{SD}	8	-0.75	0.56	0.0200
Turbidity	Z _{SD}	9	-0.65	0.43	0.0568
TSS (DWSM)	ΣB	9	-0.31	0.1	0.3967
<i>Chl a</i>	Turbidity	9	0.161	0.026	0.68
TSS (DWSM)	Turbidity	9	0.95	0.90	0.00
Turbidity	ΣB	9	0.65	0.43	0.9
Turbidity	Kd	8	0.22	0.045	0.5728
DWSM	Kd	8	0.22	0.045	0.6027
Kd	ΣB	8	0.5	0.25	0.2053
AirtT ⁰	Water T ⁰	9	0.78		0.064
Alkalinity	Condac.	9	0.5884	0.3462	0.0956
p ^H	Alkalinity	9	0.7421	0.5507	0.022
Nitrate	Chl a	9	0.1336	0.0178	0.7318
Ammonia	Chla	9	0.0074	0.0055	0.8500
PO ₄ -P	Chla	9	0.606	0.0037	0.8768
Silica	Chla	9	0.04525	0.2047	0.2213
Silica	Diatom	9	0.0451	0.002	0.984
A _{max}	Chla	8	0.4785	0.2289	0.2304
A _{max}	P _{max}	8	0.6938	0.4813	0.0563
Nitrate	A _{max}	8	0.3282	0.1077	0.4273
PO ₄ -P	A _{max}	8	0.2522	0.0636	0.5468
PO ₄	P _{max}	8	0.6394	0.4088	0.0878
A _{max}	ΣA	8	0.7551	0.5701	0.0303

Appendix 2. Formulae used to calculate different parameters

1. Phytoplankton abundance -cells/mL (APHA, 1999; Hotzel and Croome, 1999; Wetzel and

Likens, 2000)

$$\text{Cells /ml} = \frac{N \times 1000 \text{ mm}^2}{A \times D \times F \times CF}$$

Where

N= No. of cells counted

A= 2.604166 mm²

D=Depth of field (1mm)

F= No. of grids counted

CF=Concentration factor

2. Zooplankton abundance was determined according to Wetzel and Likens (2000)

$$V_{\text{net}} (\text{m}^3) = \pi r^2 h$$
$$\text{No. /m}^3 = \frac{N \times GF \times Fs}{V_{\text{net}}}$$

Where

- N= count of zooplankton,
- GF= Grid factor = Total No. of Grids/ No. of Grids counted
- Fs= factor of sub-sample
- V_{net}= Volume of net
- r= radius of the net
- z =depth of the water column hauled

3. Dissolved oxygen(DO) (Lind, 1979; APHA et al.,1999 and Wetzel and Likens, 2000)

$$\text{DO mg/L} = \frac{CF * N * E * 0.698 * V1 * 1000}{Vs}$$

Where CF = $\frac{\text{Total volume of the stoppered bottle}}{\text{Total volume of the stoppered bottle} - 2}$

E=Equivalent weight of oxygen (i.e. 8)

N= Normality of the thiosulphate used (0.01N)

0.698= To convert mg/l value in to ml/L if necessary

1000= to convert value into 1litre

V1=Titre value

Vs=Volume of the sample taken for the titration

$$\text{Gross photosynthesis (GP, mg O}_2\text{m}^{-3}\text{h}^{-1}) = \frac{OLB - ODB}{PQ * T}$$

$$\text{Net photosynthesis (GP, mg O}_2\text{m}^{-3}\text{h}^{-1}) = \frac{OLB - OS}{PQ * T}$$

$$\text{Respiration(GP, mg O}_2 \text{ m}^{-3} \text{ h}^{-1}) = \frac{\text{PQ} \cdot \text{T}}{\text{OS} - \text{ODB}} \cdot \frac{\text{PQ} \cdot \text{T}}{\text{PQ} \cdot \text{T}}$$

Where OLB= Oxygen concentration in the light bottle
 ODB= Oxygen concentration in the dark bottle
 OS = Oxygen concentration in the initial bottle
 PQ= Photosynthetic Qoutient(1.2)
 T = No. of hours of incubation
 To convert all of the above to mgC m⁻³ h⁻¹ = mg O₂ m³ h⁻¹ * 0.312
 assuming a PQ of 1.2.

4. Alkalinity (APHA, et al., 1999, Wetzel and Likens, 2000)

$$\text{Total Alkalinity in meq/l} = \frac{\text{Volume of titrant (HCl) in ml} \cdot \text{Normality of HCl} \cdot 1000}{\text{Volume of sample titrated (ml)}}$$

5. Chl a biomass of phytoplankton

$$\text{Chl a (}\mu\text{g L}^{-1}\text{)} = \frac{13.9 \cdot (\text{E665} - \text{E750}) \cdot \text{V}_e}{\text{V}_s \cdot \text{Z}}$$

Where, E665 = extinction at 665 nm,
 E750 = extinction at 750 nm,
 V_e = volume of extract (in ml),
 V_s = volume of sample filtered (in Litres),
 Z = path length of the cuvette (Spectrophotometric cell) (in cm).

$$\text{6. Total suspended solids (mg/L)} = \frac{\text{TW} - \text{TC} \cdot 10^6}{\text{VS}}$$

Where TW- Weight of the filter paper+ residues in gram
 TC- Weight of the pre heated filter paper at 105 °C in gram
 VS – Total volume of sample filtered in ml

7. Attenuation (Extinction) Coefficient (K_d) (Kirk, 1994)

$$\mathbf{K_d} = \frac{1}{Z} \times \frac{\ln I_0}{\ln I_z}$$

Where I₀ = Surfacc irradiance
 I_z = Irradiance at depth Z

8. Euphotic depth (Z_{eu}) (Kalff, 2002)

$$\mathbf{Z_{eu}} = \frac{4.6}{\mathbf{k_d}}$$