COMMUNITY STRUCTURE AND PRIMARY PRODUCTIVITY OF PHYTOPLANKTON IN CHEBARA RESERVOIR, KENYA

BY

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DECLARATION

Declaration by the student

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Declaration by the supervisors

This thesis has been submitted with our approval as the University supervisors.

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DEDICATION

I dedicate this project to my dear brother and friend, Gerald Henry Ojunga and his wife Esther.

ABSTRACT

River impoundments transform lotic aquatic systems to lentic with changes in physical and chemical properties, biotic assemblage and productivity. Chebara reservoir is located at 36°05′E and 0°22′S and situated within Keiyo-Marakwet County. The reservoir was formed as a result of damming the Moiben River to supply water to Eldoret town. A study was conducted on the physic-chemical properties as well as on community structure and primary productivity of phytoplankton in the reservoir from December 2007 to April 2008. Sampling was done every month at six stations distributed over the reservoir; one station at inlet one station at the outlet, two at minor inlets and one within the reservoir. Secchi depth visibility was measured by vertically immersing a 25cm diameter Secchi disk to disappearance. Phytoplankton were collected using a 28μm diameter plankton net immersed vertically below the photic depth. Phytoplankton were identified and enumerated using a compound. Primary production and biomass were determined by chemical analysis of chlorophyll-*a* concentration and biological oxygen demand (BOD). Phytoplankton diversity indices were determined according to Shannon-Wiener's index. Data on spatial and temporal phytoplankton abundance, temperature, pH, conductivity, dissolved oxygen, BOD and nutrients were analysed using One-Way Analysis of Variance (ANOVA), followed by Duncan's Multiple Range Test (DMRT). All data was analysed at 95% level of confidence. Interrelationships between spatial and temporal physico-chemical factors and phytoplankton abundance were analysed using Pearson's Correlation Matrix and Correspondence Analysis (CA). The relationship between phytoplankton species composition, abundance, sampling stations and physico-chemical was carried out using Canonical Correspondence Analysis (CCA). All statistical analyses were performed with STATIGRAPHIC 2.1 Plus® and STATISTICA 6.0® procedures. There were no significant differences in the spatial or temporal physico-chemical parameters. The reservoir was homogenously oligotrophic and alkaline with only very slight variations. The productivity of Chebara reservoir was low (approximately 0.8 μgmillilitre^{-1}) as estimated by chlorophyll *a*, suggesting oligotrophy. Six phytoplankton classes were identified including Cyanophyceae (22 genera) Bacillariophycae (25 genera), Chlorophyceae (55 genera), Euglenophyceae (3 genera), Pyraphyceae (6 genera) and Crysophyceae (8 genera) and similar to observations made in tropical oligotrophic lakes. The order of abundance was Pyraphyceae > Cyanophyceae > Chlorophyceae> Bacillariophyceae > Crysophyceae > Euglenophyceae> Rhodophyceae. Members of Chlorophyceae showed lower species abundance. There were strong relationships between the various phytoplankton genera and physical and chemical conditions, except for biological oxygen demand which had a weak effect. The study also indicates that phytoplankton growth in the reservoir is more likely to be limited by availability of P than N. The results obtained from this study can be useful for tracking the effects of changing activities in the drainage basin and the tributaries that contribute water directly to the reservoir. Calcium concentrations were consistently low, but the high abundance of pyraphytes in this reservoir could suggest a need to monitor management practices in the reservoir catchment that maintain calcium concentrations and populations of pyraphytes low in order to reduce the water treatment costs. This research further recommends that a research be carried out on macro invertebrates in order to accumulate sufficient knowledge which will be useful for watershed best management practices aimed at ensuring long term protection for water supply.

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CHAPTER ONE

INTRODUCTION

1.1 Background of Study

Aquatic algae are primary producers; hence form the base of the food web in aquatic systems (Agawin *et al*., 2000). Some of the algae are anchored to the substrates and are called benthic, but a majority are free floating and are called the phytoplankton. The phytoplankton occur in streams, ponds, lakes and seas, mainly as single cells, filamentous forms or small aggregates of apparently independent cells.

Cyanobacteria or cyanophytes, commonly called blue-green algae, look similar to real algae but are actually bacteria that can photosynthesize like other algae, and are usually found to dominate nutrient enriched waters (Harper 1992; Annadotter *et al.* 1999). They are usually unicellular but often grow in colonies large enough to be seen by the naked eye (Talling and Talling, 1965). Some cyanophytes can fix nitrogen from the air, and so can become greater in population when nitrogen concentrations are low and competing algae species are limited (Dufour *et al.* 2006). Both phytoplankton and benthic algae exhibit profound responses to pollution in reservoirs (Reynolds, 2006).

The phytoplankton are highly diverse, and because of their small size, responds rapidly to changing environmental factors. Therefore, phytoplankton are recognised as valuable bioindicators of the state of a water body (Agawin *et al*., 2000; Walsh *et al.*, 2003; Callieri, 2008). Moreover, the importance of phytoplankton in tropical reservoir ecosystems include its use in estimating potential fish yield (Hecky and

Kling 1981; Mustapha, 2009), productivity (Park *et al.*, 2003), energy flow (Simciv, 2005), trophic status (Reynolds, 1999) and management (Beyruth, 2000).

The species composition and productivity of phytoplankton is influenced by the quantity and quality of solar radiation, water temperature and chemistry, including the mineral nutrient concentrations, pH and dissolved oxygen (Bell and Kalff, 2001; Pirlot *et al.*, 2005). Other factors such as wind, water inflow or outflow, can have strong, but indirect local influence on algal assemblage and productivity (Chalar, 2006).

River impoundment creates reservoirs of varying sizes that supply water for electric power generation, domestic, agriculture or industry. Although created for one main function, reservoirs can easily serve several other functions, including fisheries (Hecky and Kling, 1981; Mustapha, 2009). The velocity of the water decreases when it enters the reservoir, which leads to deposition of suspended matter (Scheffer, 1998). The water becomes clearer and growth of phytoplankton is enhanced (Bowling and Baker, 1996).

Small Water Bodies (SWBs) such as reservoirs are influenced by the physical, chemical and biological processes within the entire watershed (Huszar and Reynolds, 1997). Changes within the inflowing waters are likely to affect the physico-chemical status of the entire SWBs since the water that gathers in the reservoirs more often depicts the cumulative effects of the water quality changes originating from the catchment areas (Scheffer, 1998; Sterner and Elser, 2002). For example, nutrients from agricultural land within the drainage basin and compounds introduced through

direct precipitation and other human factors such as agriculture can influence the water chemistry and the aquatic biota, thereby affecting species assemblages and aquatic biodiversity (Blomqvist, 1994). Physical and chemical water quality monitoring often provides useful information at the time of the occurrence of an event, such as during the time of impoundment. Biological assessment on the other hand integrates independent and interactive effects of environmental (Sterner and Elser, 2002; A°gren, 2004) and the anthropogenic factors on the abiotic component; thus providing a robust indicator of changes in the characteristics of an aquatic environment

(Li *et al*., 2001; Allan, 2004). Athhough most aquatic biological monitoring systems and programmes have concentrated on the microinvertebrates and macroinvertebrates for assessment of water quality stress (Urbanic, 2004; Bonada *et al.*, 2005), there has also been some increasing interest in the use of phytoplankton as ecological indicators (Harris, 1996; Eloranta and Soininen, 2001).

One of the methods of monitoring water quality changes in reservoirs and other water bodies involves the use of organisms such as algae that respond to varying degrees of physical and chemical perturbations. Extreme environmental conditions render the use of macroinvertebrates inappropriate and limited since they often have shorter lifespan (Carpenter *et al*., 1989; Angeler *et al*., 2000).

The use of biotic communities, such as diatoms (Krammer and Lange-Bertelot, 1986–91; Eloranta, 2001), and fish (Kovacs *et al*., 2002; Bonada *et al*., 2005; Urbanič, 2006) to survey gradients of aquatic environmental variables has several advantages

over physical and chemical monitoring. In running waters for instance, where the water quality changes rapidly, biological monitoring has proved to be a very useful tool due to its integrating nature (Krammer and Lange-Bertelot, 1986–91). Studies that have investigated effects of water quality changes on benthic assemblages have reported an increase in abundance, together with some combination of increases, decreases or no change in taxon richness, depending on the degree of eutrophication (Calow, 1999; Culp *et al*., 2000).Studies showed evaluation of biotic communities offers a comprehensive alternative to assessing changes in the physic-chemical parameters of an aquatic environment (Culp *et al*., 2000).

A variety of factors influence seasonality and succession in plankton populations. Their interactions with other organisms and anthropogenic disturbances are prominent ecological problems for the algae (Annadotter *et al*., 2001).Changes in the water chemistry may lead to pollution (Burgis and Morris, 1987), which often results in marked ecological alterations (Goldman and Horn, 1983; Jones *et al*., 2002). Many researchers have reported mortalities in biotic assemblages resulting from anthropogenic activities and changed water chemistry (Li *et al*., 2001; Ndetei and Muhandiki, 2005). Such changes also lead to significant shifts in phytoplankton composition, structure and abundance. Consequently, phytoplankton composition and production can be used as indicators of physico-chemical characteristics of water in combination with other water quality parameters (Eloranta and Soininen, 2001).

Phytoplankton growth and assemblage in a reservoir is more likely to be limited by availability of phosphorus (P) than nitrogen (N). These two nutrients also significantly influence trophic status of a water body (Culp *et al.*, 2000).

1.2 Statement of the Problem

The Chebara reservoir was created to supply water to Eldoret Municipality. However, damming of rivers creates an aquatic habitat of slow moving water of varying depths and altogether changing the biotic and physico-chemical status of a waterbody. Many phytoplankton are generally beneficial in a water body, but some pose problems of economic importance. For example, cyanophytes cause poisoning of fish, livestock and humans. Pyrrophytes clog pipes making water treatment more expensive. Phytoplankton community composition and biomass of the Chebara reservoir have not been studied before. Therefore, it is not known what physico-chemical properties of the reservoir are pre-eminent in determining the phytoplankton assemblage and production.

1.3 Justification

Man-made lakes (river-impoundments or river–reservoirs, usually referred to as "Dams", are the predominant source of freshwater for various uses such as irrigation, industry, human consumption among other uses. Such impoundments change water from lentic to lotic state with subsequent change in physico-chemical conditions of a water body and composition of phytoplankton. Aquatic environments have high floral taxonomic diversity and rapid response of most of its constituent members to environmental nutrients, light, temperature, turbulence, and anthropogenic inputs. Phytoplankton have been widely recognized as reliable 'bio-indicators' of changes in the physico-chemical conditions of a water body. Chebara reservoir is one of the sources of fresh water for Eldoret Municipality for both domestic and industrial uses. The importance of phytoplankton diversity, periodicity, abundance and biomass

cannot be overemphasised in relation to water quality, usage and treatment. Studies published on this topic for Chebara reservoir are limited, and hence gaps in knowledge. However, such studies can be vital in formulating cheap and sustainable best management practices (BMPs) that can also help reduce, if not prevent, poisoning of water.

1.4 Objectives of the Study

1.4.1 Overall Objective

The overall objective of the study was to obtain information on the limnology and phytoplankton community and productivity of the Chebaara reservoir.

4.1.2 Specific Objectives

The specific objectives were:

i) To determine the composition and diversity of phytoplankton in Chebara reservoir

ii) To determine the abundance and biomass of phytoplankton in Chebara reservoir

iii) To determine spatial and temporal distribution of the phytoplankton in Chebara reservoir

iv) To determine the physico-chemical properties of water in Chebara Reservoir

v) To illustrate the relationship between the phytoplankton abundance and selected physico-chemical properties within the reservoir.

1.5 Research Hypothesis

1. There are significant differences in phytoplankton species composition,

abundance, biomass and distribution in Chebara Reservoir.

2. There is homogeneous phytoplankton distribution in the reservoir. The Chebara reservoir, being slow moving water, is physico-chemically uniform and is relatively nutrient-rich, with high phytoplankton diversity but high productivity. High primary production is brought about by nutrient enrichment.

CHAPTER TWO

LITERATURE REVIEW

2.1 Factors Affecting Phytoplankton Communities in Aquatic Ecosystems

The phytoplankton and benthic algae are an important part of the aquatic environment as they form the base of aquatic food webs (Bell and Kalff, 2001). Their composition, abundance and seasonality are therefore important indicators of reservoir's productivity (Agawin *et al.*, 2000; Adams, 2002).

Phytoplankton are primarily unicellular, though many colonial and filamentous forms occur, especially in fresh waters. They have a wide range of sizes from small flagellates and cocci that are less than 5µm in diameter, to colonial forms such as *Volvox* at 500µm in diameter. Phytoplankton are classified by size into two groups: the net plankton which are retained by nets of approximately 64µm mesh size, and nanoplankton (Zeitzschel, 1978; Manny, 1972). The phytoplankton cell size and surface area to volume ratio have important implications for buoyancy, nutrient absorption (Sterner and Elser, 2002), and resistance to predation (Reynolds, 2006; Sole´ and Bascompte, 2006).

Phytoplankton are widespread in aquatic environments, occurring in virtually all water bodies where they float freely. Whilst most phytoplankton are free floating, and independent of shoreline and substratum and benthos, some shallow-water species have adopted life history strategies involving benthic resting stages (Sole´ and Bascompte, 2006). All algal classes except Rhodophyceae, Charophyceae and Phaeophyceae contribute to the phytoplankton and, in coastal zones, the zoospores

produce from benthic phaeophytes and chlorophytes may temporarily add to phytoplankton assemblage (Reynolds, 2001).

Members of Cyanophyceae (or Cyanobacteria), Chlorophyecae and Chrysophyceae may become dominant in freshwaters, but both Bacillariophytes and Pyraphytes also occur in varying quantities in freshwaters. Desmids predominate in oligotrophic waters. Cyanophytes and Bacillariophytes are common in eutrophic waters. Members of Euglenophyta and Crysophytes may be especially abundant in small nutrient-enriched pools (Ogilvie and Mitchell, 1998).

Large cells such as dinoflagellates and colonial species are capable of minimising grazing in oligotrophic freshwaters, while smaller cells are more susceptible to grazing influences (Reynolds, 2006).

Seasonal variations of phytoplankton composition in natural lakes in cool temperate climates (those with maximum water temperature ($<25^{o}C$) tend to follow patterns which are strongly related to annual cycles of temperature and thermal stratification, though light, nutrients and grazing also have significant influences (Alain and Brenda, 2009). In the tropics, temperature in lake water often varies only a few degrees annually and, apparently, has little influence on phytoplankton composition. However, tropical climates often have distinct wet and dry seasons, and most lakes exhibit corresponding variations in phytoplankton composition (Nwanko, 1996; Iban˜ ez, 1998), but some apparently do not (Huszar *et al*., 2000). A sparse community of cryptophytes, other flagellates or centric diatoms occur during cold seasons. This community gives way to a prominent bloom of diatoms in warm weather,

chrysophytes, cryptophytes or chlorophytes in varying proportions. In warm weather where stratification also occurs, a community of moderate biomass is brought about by grazing and nutrient depletion. Phytoplankton with larger cell size, such as the diatoms, *Ceratium hirundinella* and filamentous or colonial cyanophytes dominate (Sommer *et al*., 1986; Huszar and Reynolds, 1997).

Seasonal variations in hydrology appear relatively unimportant, except for reservoirs (Koma´rkova´ *et al*., 2003) and shallow lakes (No˜ges *et al*., 2003), in which hydrological variations often influence

phytoplankton composition, either seasonally (Fabbro and Duivenvoorden, 2000) or interannually (Zohary *et al*., 1996; Hambright and Zohary, 2000; Kruk *et al*., 2002; Roelke *et al*., 2004).

High late-successional phytoplankton standing crops are associated with positive vertical hydraulic gradient (VHG) (Arfi *et al*., 2001).In addition, hydrological exchange influences assemblage composition. Filamentous chlorophytes, such as *Cladophora spp* commonly occur at upwelling zones, whereas the downwelling zones are frequently dominated by cyanophytes (Roelke *et al*., 2004).

Phytoplankton have different survival strategies in different environmental stresses. Such strategies include possession of gas vesicles in Cyanophyceae and accumulation of lipids in diatoms to regulate buoyancy, a mucilage sheath among many phytoplankton groups to resist predation, ability to fix atmospheric nitrogen in Cyanophyceae and ability to absorb certain materials heterotrophically. Some

phytoplankton, such as dinoflagellates are capable of increasing the concentration of chlorophyll-*a* to absorb more light in low irradiances (Sole´ and Bascompte, 2006; Alain and Brenda, 2009). Most flagellates are able to overcome low nutrient affinity by complementing their diet of organic nutrients (Smayda, 1997; Collos *et al*., 2007). Although inorganic nutrients are regarded as the main source of diatom sustenance, diatoms have also been known to utilize organic substrates as a means of diversifying from conventional trophic pathways (Seitzinger and Sanders, 1999; Berman and Bronk, 2003).

Loss factors such as grazing and sedimentation modify the species composition of phytoplankton and decouple relationships between nutrients and biomass (George and Hewitt, 2006). Losses by grazing are significant in aquatic ecosystems, and may lead to patchiness in distribution of phytoplankton species even when nutrients are sufficient (George and Hewitt, 2006; Tirok and Gaedke, 2006). Resistance to grazing has been exhibited among many algal classes. For example, colonies of fresh water green alga, *Sphaerocystis schroeteri*, may only be reduced by grazing (Reynolds, 2001; George and Hewitt, 2006). Some of the cells pass through the gut of *Daphnia magana* absorb phosphates and emerge both intact and viable. Phosphates so absorbed stimulate photosynthesis and division in the egested cells, and thus compensate for cell losses caused by grazing. Some cyanophytes are less palatable, enhancing their survival and reproduction (Abrusa´n, 2004).

Planktonic members of Cyanophyceae posses gas vacuoles (Roelke and Buyukates, 2002) for floating in the water column. When exposed to sufficient external pressure (brought about by turgor), the vesicles collapse and the cell loses buoyancy. During

photosynthesis in high irradiances, the cell turgor may rise significantly due to the accumulation of photosynthetic products and uptake of K^+ ions (Dinsdale and Walsby 1972; Allison and Waslby 1981). In this manner, planktonic cyanophytes may regulate their position in the water column to maintain favourable growth (Reynolds, 2004). The Cyanophytes therefore have an advantage over other phytoplankton and can form blooms in a stable water column (Reynolds, 2004).

Microscopic size, coupled with mechanisms to reduce density of cells among the majority of algal classes, is the most important adaptation to reduce sinking (Allison and Waslby 1981; Reynolds, 2001). *Anabaena turgidis*, for example if grown in low light produces many gas vacuoles and stays afloat, but vacuolation decreases with the increased irradiance and the cells sink.

2.1.1 Availability of light and mixing

Surface warming of water bodies leads to formation of a layer of warm water overlying a cooler layer; a status termed stratification (Harris *et al*., 1996). Warm water dissolves less gas but more mineral solutes than cold water. However, because most of the nutrients are recycled from sediments the warm upper layer of water contains less nutrients than the lower layer.

Unlike other plants that are fixed in position, phytoplankton is moved vertically or horizontally through water column by mixing processes and their own intrinsic buoyancy mechanisms. As a result, the light that they encounter is not simply a function of the daily incident light, but also the extent to which mixing moves the cells away from the illuminated surface layers (Harris *et al*., 1996). In all aquatic

systems, even those with very clear water, the light intensity diminishes with depth due to absorption by the water itself and scattering by particles suspended within the water column. The depth at which light is at least 1% of its surface value is called the euphotic depth ($\mathbf{Z_{eu}}$), and below this level the algal cells have very little light to support photosynthesis, and must rely on stored reserves for maintenance and growth (Harris *et al*., 1996).

However, stored organic reserves are limited and will only last for a relatively short time. Consequently if the phytoplankton spend too much time below the $\mathbf{Z_{eu}}$, their growth will stop, and will possibly die. If the depth of water mixing ($\mathbf{Z_{mix}}$) is greater than $\mathbf{Z_{eu}}$, then the motion will carry cells in and out of the light zone. The proportion of time that cells spend in the light can then be determined by the ratio of $\mathbf{Z_{eu}}$: $\mathbf{Z_{mix}}$ (Harris and Baxter, 1996).

Many species of phytoplankton are able to modify their photosynthetic responses to light. Thus a cell in the lower photic zone will increase the concentration of photosynthetic pigments; primarily chlorophyll-*a* and the size of its thylakoid apparatus to enhance the efficiency of radiation absorption (Heaney *et al*., 1995). However, shade-adapted phytoplankton may absorb light with increased efficiency but they cannot utilize light with increased efficiency and the yield per photon of light absorbed remains constant. In contrast, in surface or sun-adapted phytoplankton, chlorophyll-a level is reduced and maximum photosynthesis increased (Harris *et al*., 1996; Scheffer, 1998; Roelke *et al*., 2004). The changes in chlorophyll-a concentration may be stimulated by increase in the proportion of blue wavelengths which occurs with increasing water depth, rather than simply due to decrease in total

irradiance (Scheffer, 1998). Many member species of Cyanophyceae form extensive surface blooms and, in *Microcystis*, the carotenoid concentrations increase to enhance light harvesting and to filter out dangerous concentrations of ultraviolet light (Butterwick *et al*., 2005).

In addition to Photosynthetically Active Radiation (PAR), sunlight contains infrared light and other wavelengths that are absorbed by water and cause a rise in water temperature (Kalff, 2002; Reynolds, 2006). However, the magnitude of heating of a water body depends on the size of a water body and its ambient temperature.

Continued surface heating decreases water density resulting in a horizontal stratification to produce two or more layers, a warm surface and a deeper cooler layer. The zone between these layers forms the thermocline, in which temperature decreases abruptly. The density differences prevent exchange of waters between epilimnion (shallow) and hypolimnion layers, but circulation of water occurs within individual layers (Carpenter *et al*., 1992; Kalff, 2002; Downing *et al*., 2006).Stratification of water has ecological implications; as surface waters may become depleted while the deeper waters may remain nutrient-rich, but out of reach of algae occurring above the thermocline (Kalff, 2002).

Shallow water thermoclines develop only under calm conditions, where mixing of the surface by wind action does not occur (Kristensen *et al*., 1992). Once these thermoclines are at a depth beyond the effects of wind action they may remain until the surface waters cool sufficiently before any mixing can occur. Thermoclines persist

throughout the year in the tropics but are seasonal in temperate waters (Heaney and Eppley, 1981; Kalff, 2002).

The optical characteristics of water that determine the extent of light penetration do not change very rapidly, and so light measurements can be made at intervals of one or two weeks and sometimes longer (Kalff, 2002). Mixing in reservoirs is driven largely by metereological conditions such as wind, differences in air and water temperatures and evaporation rates (Humphries and Imberger, 1982; Harris and Baxter, 1996; Kalff, 2002). Changes in mixing are usually tracked from temperature changes in the water column. This requires a series of temperature sensors suspended at different depths, to gather data at frequent intervals, in order to characterise the mixing conditions (Humphries and Imberger, 1982; Angeler *et al*., 2000). Detailed information of water mixing acquired from temperature measurements provides a basis for tracing water movement (Alvarez-Cobelas, *et al*., 1996; Oliver and Ganf, 1999).But if the phytoplankton is not fully captured in the water motion, then they will not be mixed to the extent that the temperature measurements indicate (Kirk, 1994; Phillips, 1996).

The rate of water movement can be estimated from meteorological data in conjunction with temperature data, and if this is compared with the floating or sinking rate of algae, then the extent to which algae are entrained in the movement can be estimated (Reynolds, 2001).This behaviour can be particularly important when bloom forming cyanophytes are a problem, because they are able to float up into well illuminated layers when mixing is relatively weak (Alvarez-Cobelas, *et al*., 1996; Oliver and Ganf, 1999).

Growth of cyanophytes is constrained by low concentrations of light, temperature, and nutrients. In tropical areas, light and temperature are rarely limiting so nutrient availability is usually the key determinant of their proliferation (Dufour *et al.*, 2006). Hydrological disturbances often appear as factors initiating proliferation of cyanophytes (Arfi *et al.* 2001). Anthropogenic factors, particularly the use of fertilizers, have also been known to facilitate growth of cyanobacteria (Ma, 2005; Lürling and Roessink 2006).

Reduced inflow and thermal stratification lead to a shift in dominant phytoplankton to chlorophytes at the beginning of dry periods. Potential nutrient limitations and depressed mixing regimes in later stages of dry periods will shift dominant algae to non-nitrogen-fixing cyanophytes (Angeler *et al.*, 2000). Low NH_3 conditions also favour motile forms of green algae such as *Scenedesmus* (Mustapha, 2009). Nitrogen-fixing cyanophytes such as *Anabaena* become dominant in conditions where nitrogen becomes limiting in dry seasons.

2.1.2 Range of possible nutrient pathways

Nutrient sources of lakes and reservoirs may include riverine inputs, atmospheric deposition, N-fixation and upward flux of nutrients from the bottom (Parra, *et al.*, 1974; Angeler *et al.*, 2000). In intensive agricultural lands, the lake water may be enriched by nitrogen-contaminated groundwater contributing to high nitrate (NO_3^-) concentrations in dry seasons. However, in wet seasons, ammonia (NH_4^+) and phosphates (PO_4^{2-}) may increase **(**Harris, 1996; Manny *et al.*, 1994; Angeler *et al.*, 2000). Significant negative correlation usually occurs between surplus N applications

and dissolved oxygen (DO) while NH_4^+ and NO_3^-concentrations during wet season are positively correlated (Barnese and Schelske, 1994; Chalar, 2006).

Nutrient inputs modify or change the composition of phytoplankton communities and their relative abundance (Mitch and Gosselink, 1993). In many inland waters high nitrate (NO_3^-) concentrations lower the phosphate (PO_4^{2-}) concentrations because of increased acidity of the water body. Low phosphate concentrations lead to high nitrate retention (Kaste *et al*., 1998), thereby leading to possible eutrophication during dry periods. When nutrient balance is shifted towards a high N:P ratio, P may become the limiting nutrient and toxic dinoflagellates become dominant (Skjodal, 1993), especially when silicon dioxide is also depleted. Phosphate addition to acidified lakes reduces acidity by increasing NO_3^- uptake by phytoplankton or plants in a small oligotrophic lake (Davison *et al*., 1995). The increase in pH reduces the probability of occurrence of nuisance phytoplankton (Bowling and Baker, 1996), such as the cyanophytes.

Biochemical cycles show that atmospheric input may account for a significant proportion of nutrient flux in a lake (Downing, *et al.*, 2006) and terrestrial ecosystems (Lindberg *et al*., 1986). A number of studies have found direct atmospheric N and P inputs to be a significant component of total input (Cole *et al*., 1990; Schindler *et al*., 2008).

Internal loading of nutrients usually brought about by nutrient release from sediments and fluctuations in hydrological conditions also contribute to nutrient dynamics of lakes and reservoirs (Reynolds, 2006). High dependence on hydrology is particularly

important in water bodies of Arid and Semi-arid Lands (ASALs) where surface water levels fluctuate seasonally resulting in abiotic-biotic interactions in the aquatic environments (Quintana *et al.*, 1998; Sanchez-Carrilo *et al.*, 2000). External loading may prevent nutrient limitation but flushing acts unpredictably on phytoplankton biomass by washing out phytoplankton cells (Angeler, *et al*. 2000). Although a wide range of nutrients are required by phytoplankton for growth, it is N and P that are commonly limiting. In marine environments N is limiting (Sanchez-Carrilo, *et al*., 2000; Howarth and Marino, 2006), while whole lake enrichment experiments implicate P as limiting for phytoplankton growth (Schindler *et al.*, 1977).

The surface area to volume ratio (S:V) of algal cells determines the rate of nutrient absorption (Sterner and Elser, 2002). For example, studies comparing the effects of cell size on growth rate using diatom *Dictylum brightwelli* and that of nanoplankton *Emiliania huxley* (Parsons and Takahashi, 1973; Sournia, 1981) showed that large cells absorbed more nutrients and grew faster than small cells only at high light intensities. High surface area to volume ratio has also been shown to correspond to increased nutrient uptake in nutrient deficient media (Sterner and Elser, 2002), implying that small sized phytoplankton would be favoured in oligotrophic waters. Large sized phytoplankton, however, have a greater potential for storing up nutrients and may show resilience in the face of periodic nutrient deficiency.

Natural populations of phytoplankton do not show extreme departures from the Redfield ratio C: N: P typical of severe nutrient limitation and may be near to their potential maximum growth (Sterner and Elser, 2002). Thus, species which have the ability to rapidly assimilate nutrients during transient pulses have a competitive

advantage and exhibit patchiness (Maestrini and Bonin, 1981; Sterner and Elser, 2002). Such nutrient rich micro niches may occur due to animal excretion or remineralisation of dead organisms (Ogilvie and Mitchell, 1998).

Many phytoplankton species have the ability to absorb organic compounds in the dark (Vrede *et al.*, 2004; Flynn, 2008). Uptake of particulate organic nutrients by potentially photosynthetic phytoplankton has also been reported and studies have indicated that phytoplankton growth increases in media with low C:N ratios (Vrede *et al*. 2004).Some species of freshwater Crysophytes (*Dinobryon* and *Uroglena*) are capable of digesting bacteria at rates comparable to those recorded for non-photosynthetic flagellates in deeper waters. *Dinobryon* may obtain more than 50% of its carbon requirements from pre-fixed carbon (Droops, 1983).

Reservoir sediments are the major pool of nutrients which largely mediate, through internal loading, the availability of nutrients for phytoplankton growth (Reynolds, 2006). The sediment redox processes are critical in understanding the conditions and rates of release of some nutrients to overlying waters (Scheffer, 1998). The rate of supply and deposition of organic carbon from external load is the major driver of reducing conditions in the sediments (Brown *et al.*, 2009).Thermal stratification significantly increases the reducing potential due to barrier to oxygen transfer from the atmosphere, through the water column to the sediments. Besides nutrients, phytoplankton population growth and biomass are influenced by a complex range of other factors that are cumulative and interactive (Alvarez-Cobelas *et al*., 1996; Sanchez-Carrillo *et al*., 2000).

2.1.3 Eutrophication in aquatic systems

Eutrophication, caused by increased use of agricultural fertilizers in catchments, is one of the most widespread environmental problems of inland waters. Eutrophication alters the chemical and biological characteristics of water bodies (Harper, 1992; Scheffer, 1998) and results in excess production of phytoplankton biomass (Alvarez, 2001; Blomqvist, 2001). Such alterations include de-oxygenation of the sediment and the entire water body with resultant cyanophyte blooms. Eutrophication has been associated with massive fish kills in several naturally productive reservoirs (Kling *et al*., 2001).

Eutrophication may also influence the patterns of succession in a water body. The nutrient source provided by the upwelling water may influence the surface biota and patterns of succession during and after floods (Valett *et al*., 1991; Harris and Baxter, 1996). Such nutrient elevated conditions may also favour high rates of nitrification (Grimm *et al*., 1991).

Some particular phytoplankton, such as cyanophytes, grow in nutrient enriched reservoirs but rarely survive in water bodies with low nutrient loading (Reynolds, 2006).

2.2 Phytoplankton Community Composition and Abundance

2.2.1 Phytoplankton species composition

Phytoplankton species composition, variations and abundance within a water body together with occurrence of certain types of blooms in lakes and reservoirs have been

widely studied (Talling and Talling, 1965; Wilson, 1994). Their relationships with physico-chemical characteristics of water have been used by various researchers to describe the reservoir dynamics (Wetzel, 1999; Reynolds, 2001; Sole´ and Bascompte, 2006).

The patterns of succession of phytoplankton vary in all reservoirs worldwide because their properties are highly variable and each reservoir is unique in this respect (Kalff, 2002). Some phytoplankton species occur more or less all the time but fluctuate in numbers; many other species show clear seasonality and disappear from plankton population for some part of the year (Sarmento *et al*., 2008). For example, in Lake Elementaita the phytoplankton assemblage was reported to show high seasonality and an abrupt switch from one dominant phytoplankton assemblage to another when salinity increases (Phlips *et al*., 1997). Such variation in phytoplankton composition is greatly influenced by a range of physical factors, availability, composition and forms of nutrients, and grazing processes. Phytoplankton community composition would therefore largely reflect interplay of several factors. Other factors that lead to seasonal changes include thermal stratification over the dry periods and turbulent conditions with elevated inflows in early periods of long rains, potential reduction in one or several nutrients to limiting concentrations, potential reduction in available light due to self-shading effect by the phytoplankton cells themselves, and the build-up of grazer zooplankton populations.

In many freshwater ecosystems, the first algal types to increase in concentrations in early rains are the diatoms followed by green algae then cyanophytes and finally the dinoflagellates (Harris, 1996). As temperature stratification breaks down, there can be

a resurgence of some of these populations before the phytoplankton numbers fall away to low levels through cool wet seasons. According to Harris (1996), factors underlying seasonal species succession may make it possible to predict the occurrence of particular species. Droughts and floods can switch the phytoplankton characteristics of water bodies between cyanophytes and diatoms (Heaney *et al*., 1995), thereby making weather variability a major factor on the biota and biochemistry of the ecosystem.

Diatoms are favoured by short residence of water, turbulence, deep clear water columns and strong pulses of silicon dioxide from external source (Harris and Baxter, 1996). They have relatively high sedimentation rates and are physiologically suited to growth under deeply mixed, low light conditions (Harris, 1996). Cyanobacteria are favoured by long residence of the water, quiescent (stratified) states; low dissolved inorganic nitrogen (DIN) in surface waters, and in monomictic systems, strong hypolimnial anoxia, where there is a strong build-up of ammonia, phosphates and sulphides in bottom waters.

Sediment fluxes of nitrogen regulate the form and concentration of N in the surface waters especially NH_3 from anoxic sediments. These are important in determining both algal biomass and species composition (Harris, 1996). Blomqvist, *et al*. (1994) showed that species composition of dominant cyanophytes in freshwaters may be manipulated by changing the dominant form of low dissolved inorganic nitrogen in the system. Small-celled non- nitrogen –fixing cyanophytes, such as *Microcystis*, appear more favoured by the presence of high concentrations of NH_3 but low NO_3^- in

the water, whereas N-fixing forms, for example *Anabaena* and *Aphanizomenon*, are favoured by low NO_3^- conditions (Reynolds, 2001).

Many different methods have been used to estimate phytoplankton abundance, but the most common are chlorophyll-a, primary production and direct enumeration (Prescott, 1954; Margalef, 1976). A close relationship exists between concentration of chlorophyll-*a* in water and total abundance of phytoplankton (Phlips *et al.*, 1997; Reynolds, 2001).

The importance of primary production in the reservoirs and lakes has been described in many studies (Scheffer, 1998; Blomqvist, 2000; Reynolds, 2006). Shallow reservoirs show very high primary production (Scheffer, 1998; Thomas *et al.*, 2000). Such shallow water habitats may have extensive growth of higher aquatic plants, but phytoplankton dominate primary production in most of these aquatic ecosystems (Bell and Kalff, 2001).

Phytoplankton are rarely uniformly distributed in the water column. They also show considerable horizontal and vertical patchiness, with varying scales from less than 1 mm to several kilometres (Platt and Denham, 1980). Both vertical and horizontal distributions may change with time. These temporal changes are also affected by the scale of measurement; they may be diel, seasonal or long-term changes occurring over many years. A phytoplankton assemblage must therefore be thought of as a three-dimensional patch of space, water depth, being influenced by time as well as physical and chemical gradients (Zohary, *et al.*, 1996; Reynolds, 2006).

Diel fluctuations in phytoplankton cell biology are endogenous (Sweeney, 1989; Agawin, *et al*, 2000). Phytoflagellates (*Mallomonas*, *Chroomonas* and *Ochroinonas*) migrate downwards by night but return to the upper water layers during the day (Harpley-Wood, 1976). Under clear sky these phytoplankton occur in the subsurface waters but are found on the surface in cloudy sky. The non-motile green alga *Oocystis* show active movement due to the circulation in water column.

Tropical lakes generally appear to support a higher biomass of phytoplankton, but with less species diversity. Lewis (1986) reported little seasonal change and a dominance of members of Chlorophyceae in lakes in the Philippines. In contrast Lake George, Uganda, is dominated by cyanophytes, making between 70%-88% of total biomass, although the chlorophytes contribute a significant part of the remaining biomass (Lewis, 1986) and diatoms are rare. Such lakes have shown only a twofold annual variation in phytoplankton biomass, with slight peaks occurring during periods of maximum rainfall (Chalar, 2006). Tropical lakes show less successional variation than temperate lakes (Nwanko, 1996; Iban˜ ez, 1998; Pirlot, *et al*., 2005; Chalar, 2006). Many tropical lakes are eutrophic due to rapid remineralization occurring due to higher water temperatures. One of the common features of the phytoplankton of such lakes is the occurrence of pinnate diatoms such as *Nitzchia* (Soininen and Niemela 2001).

2.2.2 Biotic and abiotic changes after river impoundments

Impoundments decrease the velocity of inflowing river water as it enters lake and reservoir basins with subsequent impacts on the physico-chemical conditions of the reservoir or lake (Mustapha, 2009). The decreased water velocity will allow the

sediments and sediment- associated materials in the water column to settle to the bottom of the lake/ reservoir basin. This will also typically result in increased water clarity (Blom *et al.*, 1994; Scheffer, 1998). At the same time, however, decreased water velocity enhances natural processes that flourish in non-flowing or pooled water systems such as the growth of phytoplankton (Blom *et al*., 1994).

Seasonal variation in phytoplankton assemblage in temperate waters is strongly related to cycles of temperature and stratification, but these variations have little effects in the tropical climates (Koma´rkova, *et al*., 2003). Much of the variation within species is averaged away by rescaling space and time (Koma´rkova, *et al*., 2003), and by pooling organisms into collective assemblages with statistical properties that can be more easily tracked. The changes in phytoplankton communities are closely related to temporal and spatial physico-chemical properties of lacustrine environments (Reynolds, 1984; Koma´rkova, *et al*., 2003). Individual phytoplankton cells, for example, experience such extensive small-scale variation, such as turbulence, vertical transport, nutrient encounter and uptake and predator discrimination and viral infection that clear patterns in community organization at this level are not easily discernible (Harris, 1986; Reynolds, 2001).

The composition and diversity of lacustrine biota in tropical reservoirs appear to be similar to those of temperate reservoirs (Roelke *et al*., 2004). Algae are largely cosmopolitan, with very weak endemism, although the genetic differences may not be evident from morphology. Many of the dominant species that occupy tropical reservoirs are indistinguishable from the taxa that are dominant in temperate reservoirs. There are some exceptions. For example, *Cylindrospermopsis stagnale*

(Wolosz.) is common in tropical reservoirs but not in temperate ones; *Asterionella* is a common component of temperate phytoplankton assemblages, but not so in the tropics (Round, 1981). Nevertheless, endemism is generally very weak among phytoplankton and a few endemic taxa have been described from ancient reservoirs (Koma´rkova, *et al.*, 2003; No˜ges, *et al.*, 2003). Algae are broadly distributed in the tropics, just as they are at temperate waters (Fabbro and Duivenvoorden, 2000; Roelke *et al*., 2004). Representation of taxa is also very similar in proportion, although tropical latitudes may show a higher proportion of cyanophytes and a lower proportion of golden brown algae (Nwanko, 1996; Iban˜ ez, 1998; Huszar, *et al*., 2000). Thus, latitudinal differences are subtle, and by no means comparable in scale to those observed in terrestrial communities.

2.2.3 Phytoplankton assemblages in reservoirs

Reservoirs vary in size, morphometry, wind action and seasonal variation in temperature and light regimes. These factors are relevant in differing phytoplankton diversity and abundance. In deep reservoirs, epipelic, epiphytic and epithelic communities develop as a girdle around the periphery, leaving the centre for plankton, whose community composition is fairly stable (Talling and Lemoalle, 1998). In deeper parts of a reservoir, cyanobacteria, favoured by long retention time may increase. In shallow parts of the reservoir, sediment resuspension may periodically modify the underwater light field, limiting phytoplankton development (Reynolds, 2006).

Reservoirs with neutral or alkaline waters, generally rich in nitrogen, phosphorus and organic content, with sediments composed of silt support rich communities of

phytoplankton flagellates such as *Euglena, Phacus, Trachelomonas* and *Leponciclis,* and Chlorococcales such as *Ooocystis, Scenedesmus, Tetraedron, Chlorella* and *Golenkinia*. In oligotrophic reservoirs, poor in calcium support mixed populations of diatoms such as *Melosira, Tabellaria and Rhizosolenia; Cyanophyta* such as *Coelosphaerium and Anabaena;* flagellates including *Mallomonas, Dinobryon, Synura, Uroglena, Gonyostomum, Peridinium, Gymnodinium,, Ceratium, Closterium and Gonium. Chlorococales* less frequent include *Botryococcus braunii* and *Dyctyosphaerium,* but more frequently *Quadrigula, Pediastrum* and *Ankistrodesmus.*

2.2.4 Scope of Study of Chebara Reservoir

Phytoplankton have a large number of genera, and the study of their communities is best achieved by using indices that summarize the community structure (Sommer *et* al., 1993). Diversity estimates, for example, can help to describe ecological systems (Magurran, 1988), community stability and its resistance to disturbances (Barnese and Schelske, 1994) and the patterns of succession (Reynolds, 1988).

This study was attempted to build a body of knowledge concerning the phytoplankton community structure, diversity and periodicity in Chebara reservoir, and to relate the community assemblage to physico-chemical characteristics of the reservoir. The findings from this study will be compared to limnological studies of other tropical reservoirs.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Location of Chebara Reservoir

Chebara Reservoir (Fig.1, Appendix 1) is located at 36°05′E and 0°22′S and situated within Marakwet District lying along Moiben River. The reservoir was created in 1993 to supply potable water for domestic and industrial uses to an estimated population of about one million people.

The reservoir lies at an average altitude of 1,759 metres above sea level. The reservoir consists of shallow pan of water lying on impervious rock surface (Land update, 2006). The surface area is aproximately10 Km^2, but it is subject to marked fluctuation due to seasonal change in the water level of the reservoir. The average depth is 45 meters.

3.2 Climate, Geology and Hydrology

The study area has mean maximum temperature range of 18°C to 28°C, and minimum range of 8°C to 12°C. The mean annual rainfall is 1,000 mm. Long rains occur between March and May, and short rains occur between September and October. Dry months occur between November and March (Land update, 2006).

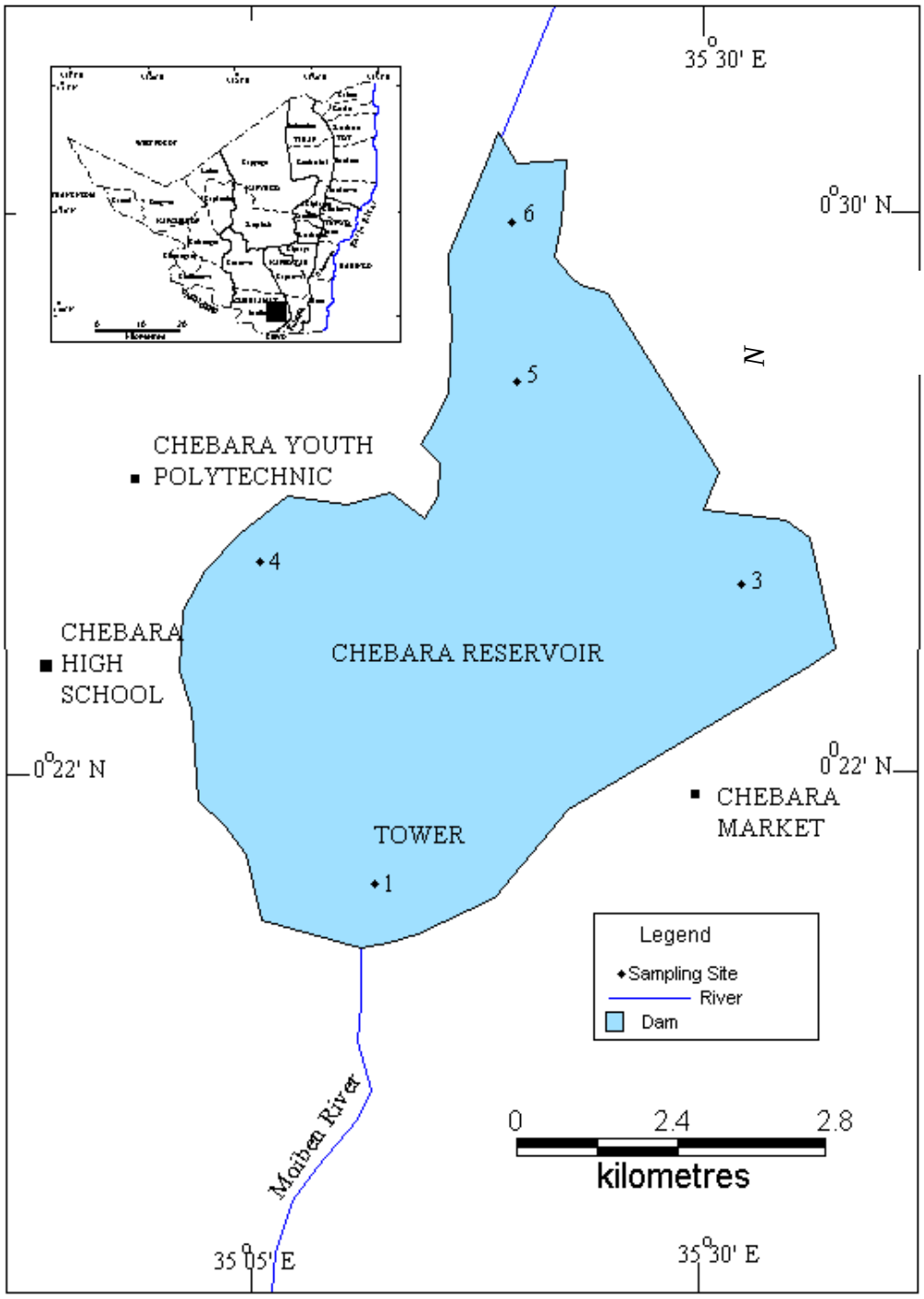


Figure 1: Location of Chebara Reservoir Showing Sampling Stations (Author, 2008)

The rock formation around the reservoir's catchment belongs to the tertiary, quaternary era, volcanic suites or the sediments associated with this suite (65 million to 8000 years ago). The eruptives include pumice turfs, lavas and ignimbrites and all these rocks types appear to have been erupted from both central volcanoes and dispersed fissure source.

The Chebara reservoir bedrock is composed largely of soft volcanic ashes and turfs with only rare outcrops of agglomerates. The rocks on the floor, such as rhyolites, are highly acidic (Land Update, 2006).

Some of the most common animal communities found within Chebara reservoir include pelican (*Pelicanus onocrotalus*), barbus (*Barbus species*) a copepod (*Lovenula africana*), and midge larva (*Leptochironomous deribae*) (Land Update, 2006). The reservoir is nearly devoid of rooted aquatic vegetation except in the North-eastern part, where freshwater rivulets provide suitable conditions for sedges. *Typha species* is primarily dominant, and is found growing at the peripheries of the reservoir.

The main human activities within the study area include crop and dairy farming. The main farming activities within the drainage areas include cultivation of maize, wheat, pyrethrum and a variety of crops. Horticulture, agro-forestry, and livestock rearing are also practiced (Land Update, 2006).

There is scanty human settlement around the reservoir, save for the secondary school situated on the northern side. A brief description of each of the sampling stations is provided below.

3.3 Sampling

Stratified sampling was carried out during the study period at six stations were selected for sampling during this study (Fig 1). Stations 3 and 4 were situated at two minor inlets, and drain water from farmlillitreand and human settlement respectively. These stations were thus chosen on the basis of possible impacts of farming and settlement on the physico-chemical characteristics of water in the reservoir.

A brief description of each of the sampling sites is provided below:

3.3.1 Sampling station 1

This sampling station was situated 10 m away from the reservoir outlet in the open waters and was not frequented by water birds at the time of study. The water in this region is subject to turbulence as it moves out through the outlet. Turbulence and the wind action bring about mixing of the water in this area.

3.3.2 Sampling station 2

Station 2 was situated about 100 m east of the reservoir outlet. The area has dense vegetation in the littoral zone and is protected from wind action. The emergent vegetation was predominantly *Typha* spp. There was also a dense vegetation of both submerged and floating plants including *Elodea* and *Nymphaea* species. The waters here remained relatively calm during all the sampling sessions.

3.3.3 Sampling station 3: Minor stream inlet

This sampling station was selected where one of the inlet streams enters the reservoir. This minor stream lies south of the reservoir and drains from farmillilitreand, as opposed to the main inlet that flows through the forest.

3.3.4 Sampling station 4: Minor inlet stream

This sampling station is also an inlet stream that lies to the south eastern side of the reservoir. The area was near human settlement that included a learning institution. The sampling station has sparse terrestrial and aquatic vegetation, with the presence of waterfowl. This was selected for study to in order to determine if there in significant effects of settlement on the physico-chemical conditions of the reservoir.

3.3.5 Sampling station 5: Open waters

The station was situated 300 m inward away from the main inlet. Water at this sampling station was very clear and devoid of vegetation. The area was open and subject to wind action.

3.3.6 Sampling station 6: Main inlet of Moiben River

The Moiben River drains into the reservoir through a thick protected forest basically free from much human activity. The station had dense vegetation, which gave significant shading. Terrestrial plants include forest species that grew on land overlying the river mouth. The water was turbid and shallow and slow moving.

3.4 Phytoplankton Analysis

3.4.1 Sample collection and preservation

Phytoplankton were collected using a plankton net of 28 μm mesh and 25 cm diameter. The net was immersed vertically below the photic depth of the water as determined using secchi disk (Wetzel 1999). The volume of the water sampled was calculated as follows:

Volume of water sampled = $\pi r^2 d$..... (i)

where:

r = radius of plankton net

d= the photic depth (in meters)

The concentrated samples, measuring 100millilitre each, were then put in plastic bottles, preserved in 0.15 millilitre of Lugol's iodine (APHA, 1998)) and transported to the laboratory for algal species identification and enumeration.

3.4.2 Phytoplankton species identification

Identification of the phytoplankton was carried out at the end of all sampling occasions using an inverted microscope (Olympus®Model CK2) at a magnification of X400. Identification was done to the genus level using different keys (Prescott, 1952; Lund, 1965; Vollenweider, 1969; Nygaard 1977; Cronberg, 1980; APHA, 2003). The various genera and their respective classes were then listed.

3.4.3 Phytoplankton enumeration

A 1 millilitre aliquot of the concentrated sample was pipetted into the Sedgwick-Rafter cell. Counting of the phytoplankton was carried out using a Sedgwick-Rafter cell under an inverted microscope (Olympus® Model CK2) at a magnification of X400 (APHA, 1998). Phytoplankton were counted in at least ten cells of 1 mm x 1mm and numerical estimations of the phytoplankton abundance done using the drop method (Margalef, 1976).

$$\text{Phytoplankton millilitre}^{-1} = (N) \times (50 \times 20 \times 1) \quad \text{..... (ii)}$$

where,

N = number of phytoplankton counted in 1 Sedgwick-Rafter cell

$(50 \times 20)\text{mm}^2$ = total area of the Sedgwick-Rafter chamber

1= 1 millilitre aliquot of the concentrated sample pipetted.

The relative abundance of the various taxa was then calculated according to Margalef (1976) using the formula shown below:

$$\text{Relative Abundance} = \frac{\text{No. of inviduals in a species x 100}}{\text{total number of individuals}} \quad \text{..... (iii)}$$

Phytoplankton diversity indices were determined according to Shannon-Wiener (1949).

$$H' = -\Sigma P_i \text{ Ln } P_i \quad \text{.....(iv)}$$

where,

H' = Shannon-Wiener diversity index

P_i = the relative importance of species *i*, derived from cell numbers (Ni/Nt).

N*i* = number of individuals in a genus in the i^{th} sample

i^{th} = the sample

N = total number of individuals in a sample

3.4.4 Analysis of primary productivity

Primary production (PP) was measured using chlorophyll-*a* analysis**.** One liter of water sample from each station was filtered through Whatman filter paper no.44. A pinch of magnesium carbonate ($MgCO_3$) was added to 10 milliliter of 90% acetone and the filter paper was then immersed it in a 15millilitre centrifuge tube. The filter paper was then immersed in mixture of ($MgCO_3$) and 90% acetone. The mixture was then covered using parafilm, and swirled to mix. The test tube was then placed in a freezer for one hour, and mixed further by swirling. The mixture was then centrifuged and the optical density of the supernatant was read in spectrophotometer Model 80-2088-84 100-120V/200-240V. Readings were made at absorbances of 664nm, 647nm and 630nm for chlorophyll-a. Three readings were obtained for every sample station. Chlorophyll-a concentration (µg chl $millilitre^{-1}$) was calculated according to Strickland and Parsons (1968) as follows:

$$\mu g \text{ chl-a millilitre}^{-1} = 11.85_{664} - 1.54_{647} - 0.08_{630} \quad \text{.....(v)}$$

Concentrations of chlorophyll-*a* in extract was calculated as follows:

$$\mu g \text{ chl in extract} = (\text{Volume of extract}) \times (\mu g \text{ chl millilitre}^{-1}) \quad \text{.....(vi)}$$

3.5 Water Sampling and Analysis

Water sampling was done once a month between December 2007 and May 2008 at each of the six sampling stations for both phytoplankton and physico-chemical properties. Water samples were collected in plastic bottles. All samples for nutrient analysis were collected just below the water surface.

3.5.1 Determination of physico-chemical parameters

Water temperature (°C),, electrical conductivity (EC) and pH were measured *in situ* at each of the sampling stations using JENWAY® 3405 Electrochemical Analyzer with probes for each of these variables. Replicates of three readings were recorded at each station once the probe was calibrated. Measurements were taken at a sensitivity of 0.01 for all parameters. Conductivity was measured to the nearest 1 $\mu S\ m^{-1}$.

Secchi depth visibility was measured using a 25cm diameter Secchi disk. The secchi disk was first lowered and the depth at which the visibility disappeared recorded. The disk was again raised and the depth at which it reappeared was also recorded. Average of the two readings was then determined as the secchi depth to the nearest meters.

3.5.2 Chemical analysis

Water samples were drawn using plastic bottles. Two aliquots of 500 millilitre were collected at each station. The bottles were capped immediately and stored in an ice-packed cool box to arrest any chemical changes. The samples were transported to the laboratory for analyses of chlorophyll-a (phytoplankton biomass), total alkalinity,

nitrate nitrogen (NO_3^--N), ammonia-nitrogen (NH_4^+-N), total nitrogen and soluble reactive phosphorus. Analysis of samples was done three hours after collection.

3.5.2.1 Nitrate-Nitrogen (NO_3^--N)

Nitrate concentrations in the samples were determined using spectrophotometer (APHA, 1998).Nitrate in water was reduced quantitatively reduced to nitrite by running the sample through a column containing amalgamated cadmium filings. The nitrite was determined by diazotizing with sulphamilamide and coupling with N (1 Naphthylethylenediamine) to form a coloured Azo dye. The absorbance of Azo dye was then measured using spectrophotometer Model 80-2088-84 100-120V/200-240V.

The procedure involved adding 2 millilitre of concentrated NH_4Cl solution to 100 millilitres of the water sample. The solution was mixed and poured in to the cadmium reduction column. The first 30 millilitres were discarded and the next 50 millilitres collected. Immediately after reduction, 1.0 millilitre of sulphanilamide was added to 50 millilitres of the reduced sample. The mixture was left standing for 2 minutes. 1 millimetere of N (1 Naphthylethylenediamine) was added and the solution mixed immediately. After 30 minutes, the absorbance of the solution was measured in a 1 cm cell against plain distilled water used as blank at a wavelength of 543 nm. Nitrate present was calculated by deducting the amount of nitrate present in the distilled water from the amount of nitrate present in the sample.

Stock solution of potassium nitrate was prepared by dissolving 0.7223 grams of potassium nitrate, which was dried and cooled in a desiccator, in a volumetric flask. The volumetric flask was then filled with distilled water to 1 litre mark. This formed

1000 µg/ml^{-1} of the stock solution. 25.0 millimetre of stock solution was put in 500 millilitres a volumetric flask and filled with distilled water to the 100 millilitre mark. This formed working standard solutions. 2.0, 4.0, 8.0 and 10.0 millilitres of standard solutions were then put in a set of 100 millilitres volumetric flasks. The diluted standards were treated as for the samples above. The standards were used to verify the reduction column efficiency. The absorbances of the standards were plotted against NO_3^--N concentrations directly from the standard curve. The sample concentration was computed directly from the standard curves and reported as mg /l^{-1}. The nitrate nitrogen was then calculated as follows:

$$NO_3^-\text{-N}\ (\mu g\ L^-) = \frac{(a-b) \times v \times 1000}{w} \quad \text{..... (vii)}$$

where,

a= concentration of NO_3^- in the solution;

b =concentration of NO_3^- in the blank;

v =volume of water sample used

w = total volume of the sample

3.5.2.2 Soluble Reactive Phosphorus

Soluble reactive phosphorus was determined using the Murphy and Riley (1962) method. A composite reagent was prepared containing molybdnoic acid and trivalent antimony. Twenty grammes (20 g) of ammonium polymolybdate were dissolved in 500 millilitre distilled water. The mixture was warmed to 50^{O}C and cooled. Trivalent antimony was prepared by dissolving 0.34g potassium antimony tartrate in 250millilitre of distilled water.

Fourteen millilitres (14 millilitres) of concentrated H_2SO_4 acid were slowly added to 900 millilitre of distilled water. Both trivalent antimony and ammonium polymolybdate solutions were added to the sulphuric acid in a 2 liter volumetric flask, and mixed thoroughly. This formed molybdnoic acid.

Ascorbic acid solution was also prepared by dissolving 27g of ascorbic acid in 500millilitre of distilled water. The composite reagent and the ascorbic acid were then mixed in a 2 litre volumetric flask, mixed thoroughly and the mixture adjusted to 2 litres using distilled water. This formed ascorbic acid reducing agent.

Stock solutions were prepared by dissolving 1.0984 g of KH_2PO_4 in 1000 millilitre of distilled water to make a concentration of 0.250 mgP/ml. 1.0, 2.0, 5.0, 10.0, 20.0 and 25.0 millilitres of the stock standard solution were each transferred into clean 500 millilitre volumetric flasks, and 10 millilitre Olsen's extracting solution and fill to the 500millilitre mark with distilled water. The resultant solutions contained 0.5, 1.0, 2.5, 5.0, 7.5, 10.0 and 12.5 ppm P respectively.

One hundred millilitres (100 millilitres) of the sampled water was put into a conical flask followed by one hundred millilitres (100 millilitres) of the mixed reagent and left for 15 minutes. The optical density was then measured by a spectrophotometer at 885 nm then at 685 nm. Calibration curves were plotted against which phosphate concentrations in water samples were determined then prepared.

$$P\ (\mu gL^{-1}) = \frac{(a-b) \times v \times 1000}{w} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \ldots \text{(viii)}$$

where,

a = concentration of Pin the solution;

b= concentration of P in the blank;

v = volume of water sample used

w = total volume of the water sample

3.5.2.3 Ammonium Nitrogen

Ammonia concentration was measured as ammonium nitrogen (NH_4^+-N) (Boyd, 1990; APHA, 1998). Ammonia concentration was measured as ammonium nitrogen (NH_4^+-N) by colourimetrically according to (Boyd, 1990; APHA ,1998). The principle used here is based on the fact that phenol and hypochlorite react in alkaline solution to form phenylquinone-monoamine. This then reacts with NH_3 in water to form Indophenol.

Phenyl hypochlorite reagent was prepared first by dissolving 34 g sodium salycylate, twenty five grams (25 g) sodium citrate, twenty five grams (25 g) of sodium tartrate and 0.12 g of sodium nitroprusside in seven hundred and fifty millilitres (750 millilitres) of distilled water. The mixture was adjusted to 1litre using distilled water. Sodium hypochlorite solution was prepared by dissolving thirty grams (30 g) of sodium hydroxide in seven hundred and fifty millilitres of distilled water, and adjusted to the one litre mark. The mixture was then allowed to cool.

Standards were prepared by dissolving 4.714 grams of ammonium sulphate ($(NH_4)_2SO_4$) in 1 litre of distilled water. This formed 1000 µg/l stock solution. To prepare standards, 50 millilitre of the stock solution was transferred into 500 millilitre volumetric flask and diluted with distilled water to the 500 millilitre mark. 5.0, 10.0, 15.0 and 20.0 millilitres of the stock solution were each transferred into clean 100 millilitre volumetric flasks, and filled to the mark with 0.5M potassium sulphate. These formed working standard solutions.

Five millilitres of Phenyl hypochlorite reagent was added to 0.2 millilitres of the water samples, standards and blanks followed by 2.5 millilitre of the sodium hypochlorite solution. After 25 minutes the mixture was measured at 665 nm against the blank solutions and the standards. Calibration curves were plotted against which ammonium- nitrogen concentrations in water samples were determined then prepared. The concentration of ammonium- nitrogen was calculated as follows:

$$NH_4^+(\mu gL^-) = \frac{(a-b) \times v \times 1000}{w} \quad \text{.....(ix)}$$

where,

a = concentration of N in the solution;

b= concentration of N in the blank;

v = volume of water sample used

w = total volume of the water sample

3.5.2.4 Total Alkalinity

Total alkalinity was determined by acidimetric method (APHA, 1998). An aliquot of unfiltered water sample was put in a titration flask. Four drops of methyl orange indicator were added then the mixture was titrated with concentrated sulphuric acid as titrant till the end point. Total alkalinity was calculated as follows:

$$\text{Totalalkal inity(mg/L)} = \frac{\text{ml of titrant x N x 50 x1000}}{\text{sample volume (ml)}} \quad \text{..... (x)}$$

where,

N = the normality of the acid.

3.5.3.5 Dissolved Oxygen (DO)

Concentrations of dissolved oxygen were determined by Winkler method (Strickland and Parsons, 1972). In this procedure, water samples were obtained by filling 250miliiliter BOD bottles to 250millilitre mark.

Bottles were each filled by tilting at an angle of 45˚ in order not to include or introduce air bubbles. Fixation was carried out by the addition of manganous sulphate followed by potassium iodide. The mixture was then shaken and then acidified by adding 1millilitre of sulphuric acid along the neck of the bottle. The bottles were then stoppered without displacing the floc. The mixture was further shaken to liberate iodine. 50 millilitres were extracted from each bottle, and mixed with starch indicator, for titration. Titration was done using 0.01N thiosulphate until iodine faded to a pale-straw colour. Calculations for DO concentrations were done as follows:

O_2 (mgL^{-1}) = 0.1006 x FxV.....(xi)

where,

F = the standardisation factor

V = volume titrated

Standardisation of thiosulphate was done by filling 250miliiliter BOD botlles with distilled water, then adding 1.0 millilitre cocentrated sulphuric acid foolowed by 1.0 millilitre. The solution was shaken gently to mix. 50 millilitre aliquot was added to the mixture followed by 0.01N standard iodate solution.Iodine was then allowed to liberate at room temperature for two minutes.The mixture was then titrated using 0.01N thiosulphate solution using a 15 millilitre buret. Standardisation was repeated three times to increase accuracy . Calculation for F was done as follows:

$$F = \frac{5.00 \text{ for } 0.01 \text{ thiosulph ate}}{V} \quad \text{..... (xii)}$$

3.5.2.6 Biological Oxygen Demand (BOD)

BOD was measured by determining the Oxygen concentrations determined following Winkler method described in section 3.4.2.5. Two BOD bottles were filled with water as in sampling for DO and measurement. One bottle was fixed as for DO. The other bottles were wrapped using aluminium foil and transported in dark conditions. The bottles were incubated in the dark for 5 days, and then fixed. Fixation was carried out by the addition of manganous sulphate followed by potassium iodide. The mixture was shaken and then acidified by adding 1millilitre of sulphuric acid along the neck of

the bottle. The bottles were then stoppered without displacing the floc. The mixture was further shaken to liberate iodine. 50 millilitres were extracted from each bottle, and mixed with starch indicator, for titration. Titration was done using 0.01N thiosulphate until iodine faded to a pale-straw colour. Three sub samples were obtained for each station. Calculations for BOD concentrations and for F were done as for DO described in section 3.4.2.5

3.5.2.7 Calcium

Calcium concentration in the water samples was determined by flame photometric method. Stock standard solution was prepared by dissolving 2.4973 grams of dry calcium carbonate ($CaCO_3$) in 200 millilitres of water containing 5 millilitre concentrated hydrochloric acid. The mixture was boiled, to drive off carbon dioxide, and then cooled. It was then diluted with distilled water to one litre mark in a volumetric flask. This stock standard solution contained $1000mg^{-1}$ Ca.

Working calcium solution was prepared by diluting 50 millilitres of the stock calcium solution in 100 millilitre of distilled water. This solution contained 500pp Ca. Lanthanum solution was prepared by dissolving 10.6939 grams of $LaCl_3.7H_2O$ in water and diluted to one litre. One litre of distilled water was carefully added to 28.0 millilitre of concentrated sulphuric acid to form 1,0 N H_2SO_4 acid.0.0, 1.0, 2.0, 3.0, 4.0 and 6.0 millilitres of the calcium solution were each put in 100 millilitre volumetric flasks, followed by 15 millilitre of 1N H_2SO_4 acid. The resultant solution was then diluted to 100 millilitre mark with distilled water. This became the standard solution. 10 millilitre of water sample was pipetted into 50 millilitre volumetric flask. 10 millilitre of lanthanum chloride was added, and then diluted to the mark with

distilled water. The contents were then shaken well. Standard, blank and sample solutions were then sprayed into the flame photometer. Readings were made at wavelength of 422.7nm. Calibration curves were against which calcium concentrations in water samples were determined then prepared

$$Ca\ (\%) = \frac{(a-b)x\ v\ x\ f\ x\ 100}{1000\ x\ w\ x\ 1000} \quad \text{.....(xiii)}$$

where,

a = concentration of Calcium in the sample solution;

b = concentration of in the blank;

v = volume of water sample used

f = dilution factor

w = total volume of the water sample

3.6 Data Analysis

All statistical analyses were performed with STATIGRAPHIC 2.1 Plus® and STATISTICA 6.0® (StaSoft, 2001) softwares. Normality of data distribution was checked by means of the skewness and kurtosis (Zar, 2001).

Data on water temperature, pH, conductivity, dissolved oxygen and the nutrients were calculated as means (± SE) for each station on each sampling occasion. Differences of means for physico-chemical parameters among stations were analyzed using a One-way Analysis of Variance (ANOVA). Differences of means in both spatial and temporal variability were analyzed by Two-way ANOVA. Duncans Multiples Range Test (DMRT) was used to discriminate between the means.

The spatial and temporal variation in phytoplankton abundance was also analyzed using Kruskal Wallis Test (Zar, 2001). Multiple comparisons of means was done using Duncan's Multiple Range test (DMRT). All statistical analyses were done at 95% level of confidence. The independent interrelationships between physico-chemical factors and phytoplankton abundance were analysed using Pearson's Correlation Matrix (Zar, 2001) and Correspondence Analysis (CA) multivariate approach based on sampling station and season.

The relationship between phytoplankton species composition, abundance, sampling stations and physico-chemical parameters was carried out using Canonical Correspondence Analysis (CCA).

CHAPTER FOUR

RESULTS

4.1 Phytoplankton Abundance and Biomass

There was significant difference in both seasonal (Figure 2) but there was no significant difference in spatial (Figure 3) phytoplankton abundance (p= 0.700). Figure 3 indicates mean phytoplankton abundance for all the stations at each sampling date. The study revealed a twofold annual variation in phytoplankton biomass and abundance. The highest abundance was observed in March (Figure 2) and at station 3 (Figure 3), while the lowest abundance in April (Figure 2) and at station 5 (Figure 3).

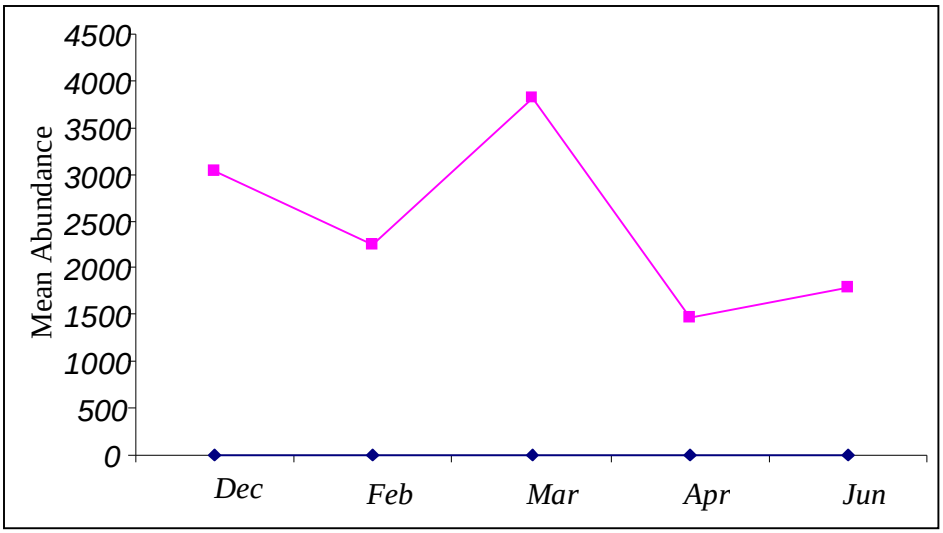


Figure 2: Temporal Variation in Phytoplankton Abundance in the Sampling Period (Author, 2008)

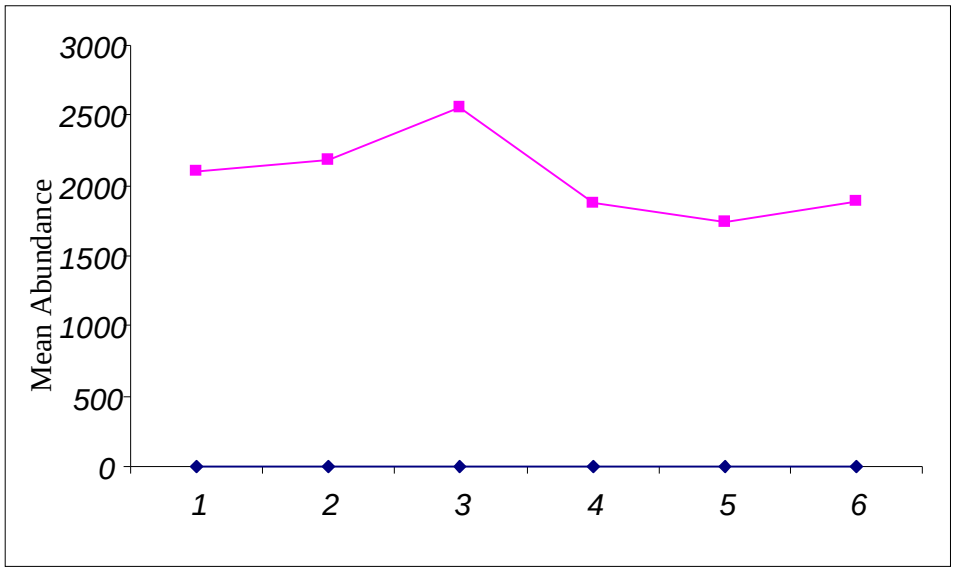


Figure 3: Spatial Variation in Mean Phytoplankton Abundance in Chebara Reservoir during the Study Period (Author, 2008)

Concentration of chlorophyll-*a* was similar for all the stations and dates (p=0.581). The highest concentration was at station 5 and the lowest at station 2 during all the sampling dates (Fig. 4). Low concentrations of chlorophyll-*a* in station 2 corresponded with low Biological Oxygen Demand concentrations. Slight variations in chlorophyll-*a* concentration were also observed at different dates (Table 5) at the same stations.

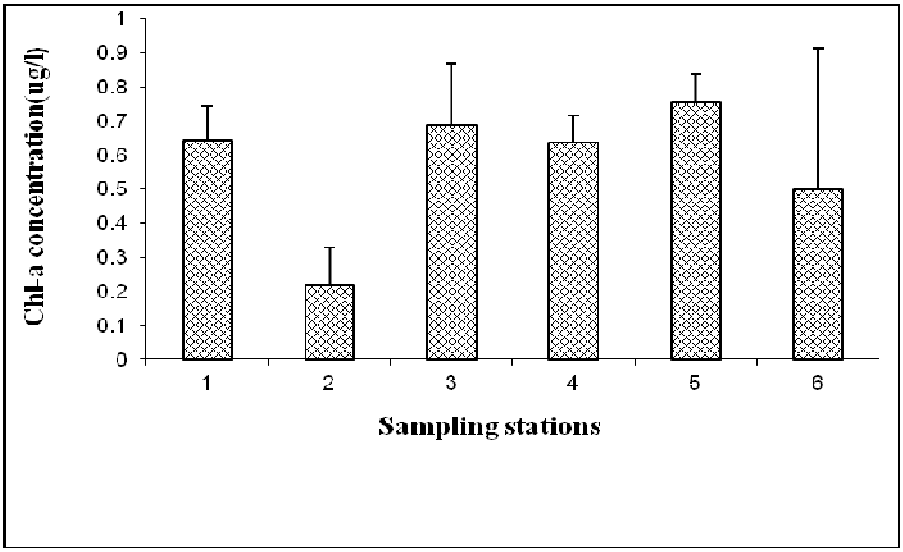


Figure 4: Mean Spatial Chlorophyll-*a* Concentration (µg L^{-1}) in Chebara Reservoir during the Sampling Period (Author, 2008)

4.2 Phytoplankton Relative Abundance and Composition

A summary of phytoplankton abundance and diversity for Chebara reservoir for different stations and dates is presented in Table 1 and 2 below respectively.

Six phytoplankton classes were identified including Cyanophyceae (22 genera) Bacillariophycae (25 genera), Chlorophyceae (55 genera), Euglenophyceae (3 genera), Pyraphyceae (6 genera) and Crysophyceae (8 genera) on the different sampling dates. The order of abundance was Pyraphyceae > Cyanophyceae > Chlorophyceae> Bacillariophyceae > Crysophyceae > Euglenophyceae> Rhodophyceae. The dinoflagellates were the most abundant of the groups observed in the reservoir, with a temporal relative abundance of 151.98 (Table 1) and 181.49 on spatial scale (Table 2). On both temporal and spatial scale *Ceratium* and *Peridinium*

were the most abundant, followed by *Microcystis* (Table 2 and 3). Three phytoplankton classes Chlorophyceae, Bacillariophyceae and Cyanophyceae showed the highest variety of genera, and accounted for over 50% of the total phytoplankton assemblage Members of Chlorophyceae showed lower species abundance. Chlorophyceae showed greater number of genera (55

genera) than the Cyanophyceae (22 genera) (Table 1). Cyanophytes showed the highest density of all the classes, with *Microcystis* showing the greatest abundance of all cyanophytes.

Table 1: Temporal Variations in Relative Abundance of Various Phytoplankton Genera in Chebara Reservoir over the Study Period

Class	Genus	Date 1	Date 2	Date 3	Date 4	Date 5	TOTAL
Cyanophyta	*Anabaena*	0.03	0.089	0.05			**0.175**
	Anabaenopsis	0.43	0.58	0.601	0.27		**1.886**
	Aphanocapsa	0.33	0.28	1.18	1.84	2.07	**5.686**
	Aphanothece			3.35		0.056022	**3.411**
	Chlorococcus	3.46	0.89	4.84			**9.189**
	Chroococcus	2.8	6.15		0.27	0.06	**9.283**
	Coelosphaerum	0.86	0.66	0.45	0.48		**2.441**
	Coenococcus	0.20	0.09	0.11	0.68	1.23	**2.306**
	Cyanercus	0.36			0.07		**0.431**
	Dactylococcopsis	2.0	4.91	11.04	8.77	5.43	**32.149**
	Glaucocystis		1.86		0.07		**1.938**
	Gleocapsa	1.25	1.15	2.93			**5.337**
	Gleothece	0.20					**0.20**
	Holopedium	0.07					**0.074**
	Microcystis	3.72	9.65	7.15	16.0	10.92	**47.412**
	Merismopedia		2.12	1.83	0.48	0.23	**4.671**

Class	Genus	Date 1	Date 2	Date 3	Date 4	Date 5	TOTAL
	Nostoc	0.03					**0.037**
	Oscillatoria	2.54	0.18	0.818	3.0	2.017	**8.533**
	Phormidium	0.033					**0.0337**
	Schizothrix						**0**
	Synechococcus	0.56	0.22	0.58			**1.3667**
	Synechocystis		0.04				**0.048**
	TOTAL	**18.83**	**28.85**	**34.91**	**31.88**	**22.017**	**136.501**
Chlorophyta	*Actinastrum*		0.04	0.03			**0.079**
	Ankistrodesmus	0.17	0.09	0.05	0.75	0.28	**1.337**
	Asterococcus	0.03	0.04		0.07		**0.156**
	Botryococcus	0.03			0.34	0.39	**0.779**
	Carteria	0.033					**0.0337**
	Cerasterius				0.611829		**0.6118 29**
	Chaetophora	0.13	0.73				**0.876**
	Characium	0.20	0.04				**0.240**
	Chlorella	1.0	0.04	0.03	0.34	0.28	**0.798**
	Cladophora			0.261712			**0.2617 12**
	Closteriopsis	0.04		0.074	0.056022		**0.107**
	Coelastrum	1.0	0.04	0.13		1.12	2.302
	Cosmarium		1.02	0.58	0.618	0.73	**2.935**
	Cylindrocystis	0.07	0.49	0.21			**0.769**
	Dactylococcus	0.23		0.03			**0.302**
	Docidium		0.040	0.03		0.06	**0.131**
	Elakatothrix		0.09	0.13			**0.221**
	Geminella			0.03			**0.031**
	Gleocystis	0.03	1.46	0.92	1.09	0.06	**3.553**
	Golenkinia	0.13					**0.139**
	Hyalotheca		0.27	0.03			**0.308**
	Hydrodactyon	0.07					**0.074**
	Kirchneriella	0.037	0.137	0.057	0.61		**0.832**
	Lagerheimia	0.03		0.10			**0.142**
	Mesotaenium				0.06		**0.062**
	Mougeotia			0.03			**0.031**
	Nephrocytium	0.20	0.62	0.89	0.07		**1.773**
	Nitella	0.36				0.06	**0.42**
	Oedegonium	0.07					**0.074**
	Ooocystis			6.83	6.39	7.73	**21.0**
	Palmella	9.30	4.03			0.06	**13.342**
	Palmodactyon	0.4		0.14			**0.539**
	Palmellococcus	0.72	0.181		1.22		**2.138**

Class	Genus	Date 1	Date 2	Date 3	Date 4	Date 5	TOTAL
	Pediastrum	1.09		0.05	0.1		**1.283**
	Protococcus		0.044	0.03			**0.079**
	Pseudoulvella		0.75				**0.752**
	Quadrigula	0.044	0.026				**0.808**
	Scenedesmus	0.26	0.22	0.05		0.06	**5.085**
	Schitococcus	3.56					**3.561**
	Schroederia		0.13				**0.133**
	Selenastrum	0.033		0.65	0.14	0.06	**0.881**
	Sphaeriella	0.033					**0.0337**
	Sphaerocystis	0.033					**0.0337**
	Spirogyra	0.10	1.33			0.11	**1.546**
	Spondilosum					0.06	**0.062**
	Stigeoclonium		0.04	0.86	2.10	2.63	**5.659**
	Tetradesmus		0.04			0.11	**0.163**
	Tetraedron	0.165	0.18	0.05	0.07	0.06	**0.523**
	Tetraspora	0.03	0.4				**0.437**
	Volvox	2.57	0.4	0.4		0.11	**3.56**
	Zoochlorella		0.31	0.34		0.06	**0.712**
	Zygnema	0.33	0.44	0.26	0.07	0.06	**1.125**
	Zygnemopsis	0.10	0.8	0.71	1.70	0.50	**3.819**
	TOTAL	**24.50**	**14.34**	**14.50**	**20.9**	**14.629**	**88.811**
Bacillarophyta	*Acnanthes*	0.033	0.4		3.6709 72	1.794	**5. 907**
	Amphora		0.09	0.036	0.27	0.6162 46	**1.007**
	Bacillaria		0.300	0.030	0.07	0.11	**0.507**
	Cocconeis	0.03	0.66	0.05	0.68	0.17	**1.603**
	Coscinodiscus		0.13	0.08			**0.217**
	Cyclotella	0.53	0.043	0.03	0.07		**0.676**
	Cymbella	0.4	0.09	0.03	0.34	0.17	**1.026**
	Denticula		0.04	0.105	0.68	0.06	**0.884**
	Epithemia		0.13				**0.133**
	Eunotia		1.0	0.37	1.70	0.95	**3.993**
	Frustulia		0.22	0.24		0.11	**0.574**
	Gomphonema			0.03	0.07	0.11	**0.217**
	Gomphocymbella	0.03					**0.037**
	Gyrosigma	0.032			0.61	6.0	**6.644**
	Hanztchia		0.75	0.55			**1.307**
	Melosira	0.95	2.03	0.293	6.12		**9.48**
	Navicula	0.79	0.97	1.39	2.11	1.79	**7.053**
	Niztchia		0.13	0.76	0.48	0.56	**1.938**
	Opephora	0.03	0.09			0.11	**0.238**
	Pinnularia	0.03	0.4	0.08	0.27	0.50	**1.334**
	Rhaphidonema		0.35			0.11	**0.477**
	Rhoicosphenia	0.03	0.09		0.07		**0.194**
	Rhopalodia	0.03	0.04				**0.085**

Class	Genus	Date 1	Date 2	Date 3	Date 4	Date 5	TOTAL
	Strauroneiss	0.07	0.09	0.45	1.97	2.13	**4.79**
	Synedra	0.43	0.04	1.15	1.43	2.19	**5.237**
	TOTAL	**3.42**	**8.10**	**5.63**	**20.60**	**17.50**	**55.238**
Rhodophyta	*Erythrotrichia*		0.04	0.11			**0.152**
	Porphyridium				0.07	0.45	**0.526**
	TOTAL		**0.04**	**0.12**	**0.07**	**0.45**	**0.673**
Pyraphyta	*Ceratium*	19.6	14.11	14.9	9.04	15.35	**73.07**
	Closterium	0.033	0.09	0.08	0.27		**0.476**
	Glenodium	0.07	0.23				**0.331**
	Peridinium	19.10	13.67	15.65	6.59	16.41	**71.423**
	Straustrum	0.79	1.54	2.01	2.2433 72	0.11	**6.712**
	Cystodinium		0.04	0.026			**0.0709**
	TOTAL	**39.58**	**29.74**	**32.64**	**18.158**	**31.88**	**151.98**
Euglenophyta	*Euglena*	0.461	0.31	0.11			**0.884**
	Phacus	0.07	0.13	0.05		0.22	**0.483**
	Trachelomonas	0.96	0.89	1.94	0.20	2.6	**6.565**
	TOTAL	**1.48**	**1.33**	**2.09**	**0.20**	**2.80**	**7.91**
Crysophyta	*Characiopsis*	0.56	2.66	0.03			**3.241**
	Chlosteriopsis		0.03				**0.03**
	Chrysidiastrum	1.35		2.43		1.18	**4.965**
	Dinobryon	0.17	8.98	4.95	0.61	2.97	**17.673**
	Goniochloris	0.10					**0.10**
	Mallomonas	0.07					**0.074**
	Pleurogaster	0.03					**0.10**
	Synura	0.10					**0.10**
	TOTAL	**2.40**	**11.647**	**7.41**	**0.61**	**4.15**	**26.208**

Table 2: Spatial Variations in Relative Abundance of Various Phytoplankton Genera in Chebara Reservoir over the Study Period

Class	Genus	Abundance						Total
		Stn 1	Stn 2	Stn 3	Stn 4	Stn 5	Stn 6	
Cyanophyta	*Anabaena*	0.19					0.05	**0.24**
	Anabaenopsis	0.71		0.78	0.05	0.12	0.9	**2.57**
	Aphanocapsa	0.67	0.32	1.45	0.11	3.67	0.12	**6.31**

	Aphanothece	0.19	2.15		1.60	2.01	1.86	**7.80**
	Chroococcus	9.67	3.11	4.46	3.038	1.67	0.38	**22.30**
	Coelosphaerum	0.33	0.18	0.27	1.12	0.17	1.38	**3.46**
	Coenococcus	0.05	0.32	0.20	0.32	0.17	1.33	**2.38**
	Cyanercus		12.70	0.16			0.27	**13.12**
	Dactylococcopsis		0.18	6.41	7.5	5.4	4.41	**23.95**
	Glaucocystis	0.05		0.39		0.06	1.65	**2.14**
	Gleocapsa	2.24		1.49	1.12	1.95	1.81	**8.59**
	Gleothece	0.29						**0.29**
	Holopedium			0.08				**0.08**
	Microcystis	6.00	6.03	5.71	12.40	14.17	8.81	**53.10**
	Merismopedia	1.10	0.82	0.94	1.44	1.04	2.02	**7.34**
	Nostoc		1.51	0.04			1.27	**2.82**
	Oscillatoria	1.19		1.84	0.48	3.21		**6.71**
	Schizothrix					0.06		**0.06**
	Synechococcus			0.16	1.38		0.16	**1.70**
	Synechocystis	0.43		0.12				**0.55**
	TOTAL	**23.10**	**27.32**	**24.48**	**30.556**	**33.68**	**26.38**	**165.5**
Chlorophyta	*Actinastrum*	0.10		3.29	0.32			**3.70**
	Ankistrodesmus	3.29	0.09	0.12	0.05		0.58	**4.13**
	Asterococcus		0.09	0.04	0.59		0.05	**0.77**
	Botryococcus	0.05						**0.05**
	Carteria		0.05		0.05			**0.10**
	Cerasterius					0.12	0.05	**0.17**
	Chlamydomonas	3.39			0.1			**3.39**
	Chaetophora	0.86	0.41	0.23		0.06	0.11	**1.67**
	Characium		1.55	2.46	0.32			**4.34**
	Chlamydomonas				0.05			**0.05**
	Chlorella	0.33	0.05				0.32	**0.70**
	Cladophora						0.53	**0.53**
	Coelastrum	0.05				0.06	0.64	**0.74**
	Cosmarium		0.09	0.43	0.21	0.17	0.96	**1.86**
	Cylindrocystis	1.10	0.69	0.47	0.69	0.23	0.58	**3.75**
	Dactylococcus		0.05	0.31	0.05	0.06		**0.47**
	Docidium	0.33			0.05		0.05	**0.44**
	Elakatothrix	0.05		0.20	0.05			**0.30**
	Geminella	0.10						**0.10**
	Gleocystis		0.05	0.04		0.29	0.80	**1.17**
	Golenkinia	2.62	0.46					**3.08**
	Hyalotheca		0.18	0.04	0.32		0.27	**0.81**
	Hydrodactyon			0.08				**0.08**
	Kirchneriella					0.06		**0.06**

Table 2 (Contd.): Spatial Variations in Relative Abundance of Various Phytoplankton Genera in Chebara Reservoir

Class	Genus	Abundance						Total
		Stn 1	Stn 2	Stn 3	Stn 4	Stn 5	Stn 6	
	Lagerheimia	0.62	0.05		0.21			**0.88**
	Mougeotia	0.05						**0.05**
	Nephrocytium	0.10		0.27	0.11	0.12	0.64	**1.23**
	Nitella	0.95	0.55	0.08	0.11			**1.69**
	Oedegonium	0.19	0.18					**0.37**
	Ooocystis	0.10		9.27	6.65	0.29	8.39	**24.69**
	Palmella	5.00	10.74			0.52		**16.25**
	Palmellococcus	0.05		0.74	0.05	0.12		**0.96**
	Palmodactyon	0.43	0.59					**1.02**
	Pediastrum	0.19		0.16	0.11		0.11	**0.37**
	Protococcus	1.38						**1.38**
	Pseudoulvella	0.10				0.69	0.27	**1.05**
	Quadragula			0.20	0.59	0.06		**0.84**
	Radiofilum	0.10	0.27					**0.37**
	Scenedesmus	0.05			2.39			**2.44**
	Schitococcus	0.67	0.23	1.84			0.96	**3.69**
	Schroederia		2.79	0.04				**2.83**
	Selenastrum	1.05	0.14		0.16	0.06		**1.40**
	Sphaeriella			0.04			0.11	**0.15**
	Sphaerocystis							
	Spirogyra			0.04	0.11		0.05	**0.20**
	Spondilosum	0.05					1.65	**1.69**
	Stigeoclonium	0.05	0.05	0.94	0.80	1.32		**3.15**
	Tetradesmus	1.43	0.64	0.12			0.32	**2.50**
	Tetraedron			0.04	0.16	0.06		**0.26**
	Tetraspora	0.10	0.27			0.52		**0.88**
	Volvox	0.71	0.46	2.31	0.48	0.06	0.06	**4.07**
	Zoochlorella		0.46	0.16	0.11		0.53	**1.25**
	Zygnema	0.86	0.27	0.12	0.16	0.12	0.27	**1.79**
	Zygnemopsis	0.95	0.96	0.43	0.85	0.57	0.21	**3.98**
	TOTAL	**24.00**	**22.40**	**24.48**	**15.91**	**5.50**	**18.47**	**110.45**
Bacillarophyta	*Acnanthes*	0.57			1.38			**1.96**
	Amphora			0.04		3.26	0.053	**3.36**
	Bacillaria	0.33	0.09	0.23		0.92	0.27	**1.84**
	Cocconeis	0.38		0.20	0.75		0.16	**1.48**
	Coscinodiscus	0.19				0.06	0.11	**0.35**

	Cyclotella		0.37	0.35		0.11	**0.82**
	Cymbella	0.19	0.14	0.35	0.27	0.11	**1.05**
	Denticula	0.05			0.11	0.17	0.53	**0.86**
	Epithemia					0.17		**0.17**
	Eunotia	0.19	0.59	0.70	1.54	0.80	1.06	**4.90**
	Frustulia	0.10	0.14		0.27	0.17		**0.67**
	Gommphocymbella						0.05	**0.05**
	Gomphonema			0.04			0.16	**0.20**
	Gyrosigma	0.05		0.08	0.48	0.23	0.53	**1.36**
	Hantchia	1.81						**1.81**

Table 2 (Contd.): Spatial Variations in Relative Abundance of Various Phytoplankton Genera in Chebara

Class	Genus	Abundance						Total
		Stn 1	Stn 2	Stn 3	Stn 4	Stn 5	Stn 6	
	Melosira	0.86	0.91	2.78	3.51	4.35	1.70	**14.11**
	Navicula	1.76	2.06	0.20	0.11	0.12	2.92	**7.15**
	Niztchia	0.38		0.90	0.21	0.75	0.05	**2.29**
	Opephora	0.05	0.73			0.11	0.11	**1.00**
	Pinnularia	0.10	0.18	0.51			0.43	**1.21**
	Rhaphidonema			0.08	0.27	0.06	0.05	**0.45**
	Rhoicosphenia	0.05		0.16				**0.20**
	Rhopalodia		0.05	0.04				**0.08**
	Strauroneiss	1.14	0.27		0.69	3.78		**5.89**
	Synedra	0.76	1.19	2.82	0.11		1.91	**6.78**
	TOTAL	**8.95**	**6.72**	**9.46**	**9.69**	**14.95**	**10.30**	**60.06**
Rhodophyta	*Erythrotrichia*	0.1		0.12				**0.22**
	Porphyridium	0.05		0.31				**0.36**
	TOTAL	**0.15**		**0.43**				**0.58**
Pyraphyta	*Ceratium*	14.62		11.18	18.52	19.93	14.6	**78.85**
	Closterium	0.14	0.05		0.05	0.06		**0.3**
	Cystodinium	0.1	1.6					**1.7**
	Glenodium		13.43	0.08			0.27	**13.78**
	Peridinium	16.93	0.09	10.05	17.35	18.67	16.93	**79.99**
	Straustrum	2.86		0.31	1.01	1.09	1.6	**6.87**
	TOTAL	**34.62**	**15.17**	**21.62**	**36.93**	**39.75**	**33.4**	**181.49**
Euglenophyt	*Euglena*	0.05		0.23	0.16	0.06	0.74	**1.24**

a								
	Phacus		0.09	0.12	0.11	0.06	0.16	**0.54**
	Trachelomonas	0.95	1.19	0.43	1.06	2.06	3.13	**8.82**
	TOTAL	**1.00**	**1.28**	**0.78**	**1.33**	**2.18**	**4.03**	**10.60**
Crysophyta	*Characiopsis*			0.31	0.05		3.13	**3.49**
	Chlorellidiopsis		0.05	0.31			0.16	**0.52**
	Chrysidiastrum	0.05	9.78					**9.83**
	Dinobryon	3.48		13.26	1.01	1.89	1.91	**21.55**
	Goniochloris		0.14					**0.14**
	Mallomonas						0.11	**0.11**
	Pleurogaster			0.05	1.06			**1.11**
	Synura		0.09	0.04				**0.13**
	TOTAL	**3.53**	**10.06**	**13.97**	**2.12**	**1.89**	**5.31**	**36.88**

Table 3 is a summary of both temporal and spatial variations in phytoplankton abundance, dominanance evenness and diversity indices during the study period. The order of temporal abundance was March > December> February > June > April. More genera occurred in February > December> March > June > April.

The order of spatial abundance *was* station 3 > station 2 > station 1 > station 6 > station 4 > station 5. The order of spatial genera occurrence was station 1 > station 6 > station 3 > station 4 > station 2 > station 5.

Table 3: Temporal and Spatial Variations in Abundance, Dominance, and Evenness and Diversity Indices for Phytoplankton in Chebara Reservoir over the Study Period

Sampling Dates	**December**	**February**	**March**	**April**	**June**
No .of Genera	86	90	78	56	58
Abundance	3037	2260	3821	1471	1785
Dominance	0.096	0.068	0.079	0.068	0.85
Evenness	0.38	0.42	0.37	0.43	0.39
Shannon diversity	3.02	3.27	3.05	3.11	2.91
Stations	**1**	**2**	**3**	**4**	**5**	**6**
No. of Genera	80	63	76	66	58	79
Abundance	2100	2189	2557	1879	1746	1884
Dominance	0.07	0.09	0.07	0.1	0.11	0.08
Evenness	0.42	0.38	0.40	0.39	0.37	0.42
Shannon diversity	3.23	2.93	3.17	2.92	2.77	3.16

Diversity and richness were high on both temporal and spatial scales, although mean spatial abundance was slightly lower than temporal abundance. Species evenness and dominance were low for sampling occasions and stations, indicating heterogeneity and patched distribution of phytoplankton.

4.3 Physico-chemical Characteristics of Water in Chebara Reservoir

The status of selected physico-chemical attributes of Chebara water are summarised in tables 4 and 5 below.

The study revealed that there was significant difference in nitrate concentration among sampling stations (Table 4) and dates (Table 5). Samples from stations 1, 2, 3, 4 and 5 showed similar concentration of nitrate, with concentrations below 0.5 µg millilitre^{-1}, but station 6 had higher concentrations. The mean concentrations of nitrate nitrogen at stations 1 and 6 were higher than the other stations. However all the

stations had nitrate nitrogen concentrations above minimum reporting level (MRL) of 0.05 μg millilitre^{-1}

Table 4: Spatial Variation in Physico-Chemical Attributes in Chebara Reservoir over the Study Period Between December 2007 and June 2008

Parameter	Sampling Stations						Sig.
	1	2	3	4	5	6	
NO_3^- (µg millilitre^{-1})	0.47±0.06	0.41±0.04	0.42±0.07	0.36±0.03	0.42±0.04	0.57±0.13	<0.0001
NH_4^+ (µg millilitre^{-1})	0.56±0.07	0.56±0.06	0.39±0.04	0.49±0.05	0.46±0.06	0.52±0.06	0.467
PO_4^{3-} (µg millilitre^{-1})	0.007±0.001*	0.008±0.001	0.007±0.001*	0.008±0.001	0.007±0.001*	0.01±0.003	<0.0001
pH	7.77±0.07	7.79±0.04	7.86±0.12	7.62±0.09	7.42±0.18	7.64±0.07	0.720
Electrical Conductivity (0.1 µS m^{-1})	1.85±1.71	1.81±1.68	1.75±1.63	1.77±1.65	1.82±1.68	1.77±1.62	0.360
Secchi depth (m)	8.7±0.54	8.9±0.46	8.6±0.6	9±0.45	8.2±0.8	1.43±0.87	<0.0001
Temperature (ºC)	19.4±0.4	19.2±0.3	19.3±0.3	19.2±0.4	19.4±0.3	18.6±0.5	0.644
BOD (mg l^{-1})	4.62±0.32	3.36±0.29	3.81±0.11	5.83±0.24	0.89±0.31	2.51±1.36	0.260
DO (mg l^{-1})	6.42±0.66	4.92±0.86	5.44±1.48	6.11±0.9	6.3±0.73	6.02±0.86	0.905
Chl *a*	0.64±0.14	0.22±0.15	0.69±0.25	0.71±0.12	0.76±0.11	0.50±0.59	0.581
Ca	Trace	Trace	Trace	Trace	Trace	Trace	

Table 5: Temporal Variations in Physico-Chemical Attributes in Chebara Reservoir over the Study Period Between December 2007 and June 2008

Parameter	Sampling Dates					
	December	**February**	**March**	**April**	**June**	**Sig.**
NO_3^- (µg millilitre^{-1})	0.62±0.04	0.28±0.05	0.64±0.07	0.33±0.03	0.32±0.03	0.0001
NH_4^+ (µg millilitre^{-1})	0.42±0.05*	0.42±0.04*	0.56±0.08	0.55±0.04	0.60±0.03	0.0001
PO_4^{3-} (µg millilitre^{-1})	0.007±0.001	0.0040±.004	0.006±0.006	0.008±0.001*	0.008±0.01*	0.0001
pH	7.93±0.2	7. 9±0.15	7.92±0.05	8.15±0.05	7.65±0.02	0.720
EC (0.1 µS m^{-1})	0.14±0.008*	0.12±0.009	0.13±0.015	0.14±0.02*	0.16±0.009	1.000
Secchi depth (m)	8.4±0.I8	8.7±0.18	8.3±18	7.8±0. 5	7.3±0.6	0.0001
Temperature (ºC)	18.5±0.3	19.5±0.2	19.25±0.1	20±0.00*	20±0.00*	0.713
BOD (mg l^{-1})	3.4±0.22	3.68±0.05	3.4±0.33	4.46±0.05	4.58±0.32	0.260
DO (mg l^{-1})	7.53±0.13	4.29±0.093	5.01±0.1	5.82±0.13	5.84±0.1	0.905
Chl *a*	0.80±0.12	0.84±0.05	0.82±0.1	0.68±0.05	0.76±0.03	0.701
Ca	Trace	Trace	Trace	Trace	Trace	

There was significant difference in ammonium nitrogen concentration among sampling dates ($p < 0.001$) (Table 5) but not sampling stations ($p = 0.467$) (Table 4). Highest mean concentrations of ammonium nitrogen were recorded at stations 1 and 2 and the lowest at station 3. There was a wide range of ammonium concentrations at different stations, the least being 0.39 µg millilitre^{-1} and the highest 0.56 µg millilitre^{-1}. There was no significant difference for ammonium nitrogen concentrations in December 2007, February and March 2008. These months were dry seasons, and were different from the values obtained in rainy season (May and April).

The total reactive phosphorus concentration was significantly different for both the stations (Table 4) and dates (Table 5). Mean total reactive phosphorus concentrations for most stations were equal to or below MRL. Stations 1, 2, 3, 4 and 5 did not show any significant difference. However, there were slight variations among stations; with a range of between 0.007 and 0.008 µg millilitre^{-1} (Table 4). Mean total reactive phosphorus concentration was highest at station 6 but lowest at stations 1, 3 and 5. The least phosphorus concentration was recorded in February (Table 5). Mean total reactive phosphorus concentration was generally lower than mean ammonium-nitrogen and nitrate-nitrogen concentrations.

Chebara water was generally alkaline, with the highest spatial pH value being 7.86 and the lowest being 7.42 (Table 4). The highest temporal pH was 9.02 and the lowest 7. 65 (Table 5). There was no significant difference in electrical conductivity (EC) for all the

stations (p= 0.36) and dates (p =1.000). The highest EC value was recorded (1.846 μS m^{-1})) for stations 1, and the lowest (1.754 μS m^{-1}) for station (Table 4).

Secchi depth was significantly different for stations (Table 4), and for dates (Table 5). Stations 1,2,3,4 and 5 were similar (p=1.000), but significantly greater than secchi depth for station 6 (Table 4). Secchi depth values for December, February, March, were similar and significantly greater than secchi depth for April and June.

There was no significant difference in water temperature (p=0.644) for all the stations (Table 4) and dates (Table 5) (p=0.715). The highest water temperature was recorded in April and June, and the lowest in December. The lowest temperature was 18.6 ^{o}C at the station 6 and the highest was19.4ºC for stations 1 and 5.

There was no significant difference in biological oxygen demand (BOD) concentrations among the stations (p=0.260) and dates (p=0.260). The highest BOD concentrations were recorded at station 4 and the lowest at station 5 (Table 4). Station 6 also showed considerably low concentrations. The highest BOD concentrations were recorded in March and the lowest in April (Table 5).

Table 4 and 5 also revealed that Dissolved oxygen (DO) concentrations were also similar among the stations (p=0.898 and dates (p=0.905). The highest DO level was recorded at station 1 and the lowest at station 2.

Calcium was found in traces in Chebara reservoir, thereby indicating very low ion concentrations.

4.4 Relationships between Phytoplankton and Physico-chemical Parameters

To reduce the number of genera used in the ordinations and avoid overcrowding the plots, genera whose contribution to the total abundance was less than 0.4 % within the group were eliminated. This reduced the number of genera from 123 to 32 with a combined abundance of 84.07 % (Table 2). To reduce the numbers further, the remaining 32 genera were subjected to pair-wise multiple correlation to eliminate redundant ones. Genera whose Pearson Product Moment was more than 0.85 were considered redundant. Only one genus from among the redundant groups was selected for ordination analysis with physicochemical parameters.

For the genera with more than 0.4% contribution to the total abundance (Table 7) the following redundant groups were identified: *Microcystis*, *Pinnularia*; *Chroococcus*, *Stigeoclanium*; *Dactylococcopsis*, *Schitococcus* and *Chlamydomonas*; *Merismopedia* and *Coelosphaerum*; *Aphanocapsa* and *Oscillatoria*; *Eunotia* and *Melosira*; *Chlamydomonas*, *Schitococcus*, *Niztchia* and *Dinobryon*; *Volvox*, *Melosira* and *Ankistrodesmus*; *Gleocystis* and *Straustrum*; *Cosmarium*, *Peridinium*, *Ceratium*, *Eunotia*, *Navicula*, *Trachelomonas*, *Synedra* and *Pinnularia*. Genera that were not redundant were included in the ordinations, and included *Aphanothece*, *Anabaenopsis*, *Scenedesmus*, *Zygenemopsis*, *Trachaelomonas* and *Nephrocytium*. From among the redundant groups, those included in

the ordinations were *Microcystis*, *Chroococcus*, *Dactylococcopsis*, *Merismopedia*, *Oscillatoria*, *Dinobryon*, *Melosira*, *Straustrum*, *Ceratium*, *Navicula* and *Synedra*. Consideration was given to those with higher relative abundance. In total 18 genera representing 58.5% of total phytoplankton abundance was used in the ordinations.

Table 6: Taxa with Relative Abundance > 0.4%

Division	Genera	Within Group Relative Abundance
Cyanophyceae	*Microcystis*	8.60
	Dactylococcopsis	6.71
	Chroococcus	3.92
	Oscillatoria	1.59
	Gleocapsa	1.46
	Aphanothece	1.24
	Merismopedia	1.21
	Aphanocapsa	1.03
	Coelosphaerum	0.55
	Anabaenopsis	0.45
Bacillariophyceae	*Synedra*	2.39
	Navicula	1.81
	Melosira	1.51
	Pinnularia	0.90
	Eunotia	0.57
	Niztchia	0.46
Chlorophyceae	*Ankistrodesmus*	1.41
	Stigeoclonium	0.92
	Schitococcus	0.88
	Volvox	0.85
	Chlamydomonas	0.84
	Gleocystis	0.70
	Cosmarium	0.69
	Scenedesmus	0.67
	Zygnemopsis	0.67
	Nephrocytium	0.45
Crysophyceae	*Dinobryon*	5.85
Euglenophyceae	*Trachelomonas*	1.41
Pyraphyceae	*Ceratium*	15.32
	Peridinium	14.39
	Straustrum	1.33

Ordination results (Fig.5) indicated that increased PO_4^{2-}, $NO_3^-$and NH_4^+ but reduced BOD concentration and secchi depth led to marked increase in *Ceratium* species, *Scenedesmus* species, *Mensmopedia*, *Nephrocytium*, *Anabenopsis* at station 6 and increase in *Straustrum* at station 4, but led to reduced abundance of *Chroococcus*, *Zygnemopsis*, *Dactylococcopsis* species, Dinobryon at stations 2 and 3. Increased PO_4^{2-}, NO_3^-, NH_4^+, pH, DO, and led to marked decrease in abundance of *Synedra*, and *Dinobryon*, but increased abundance of *Trachelomonas*, *Navicula*, *Oscillatoria*.

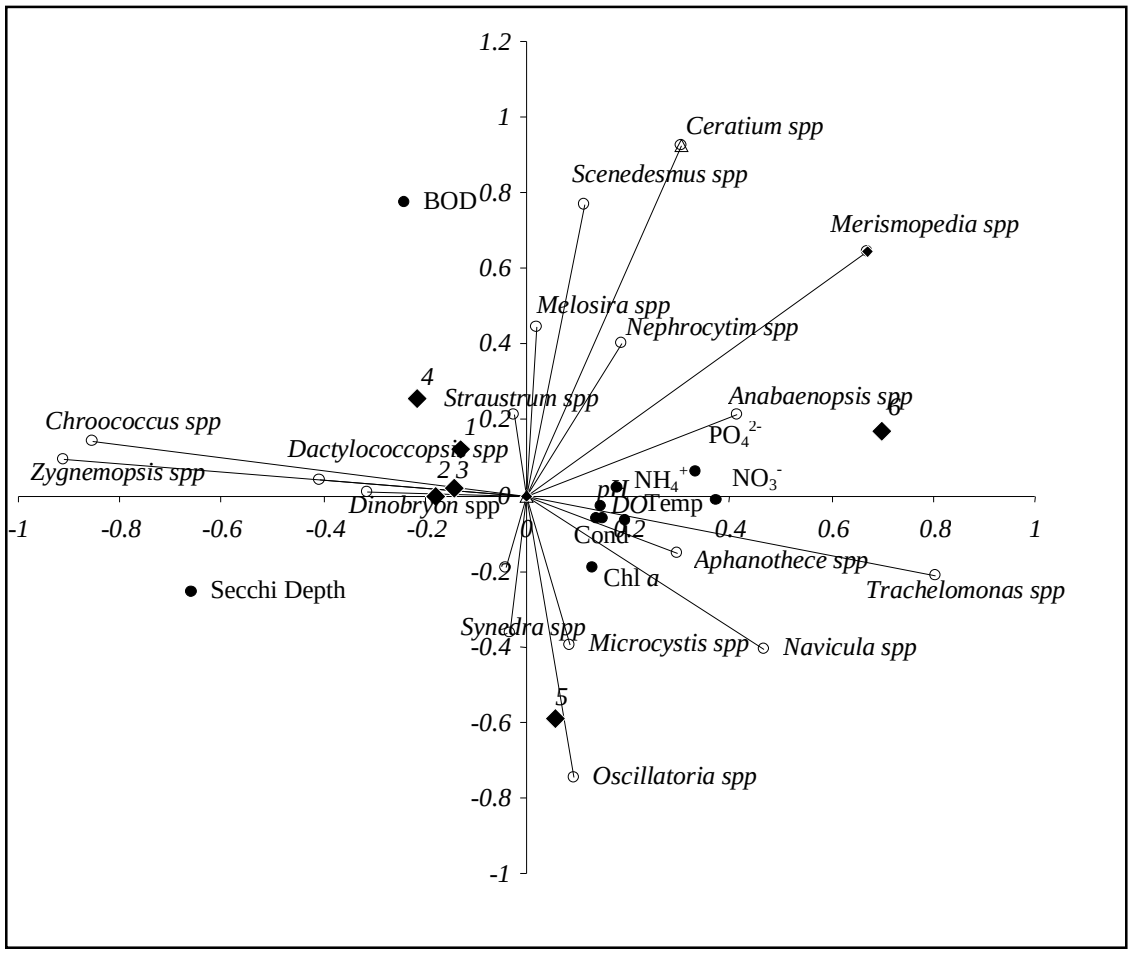


Figure 6: Biplot ordination for phytoplankton and physico-chemical factors at in Chebara Reservoir (Chl-a= Chlorophyll *a*; DO = dissolved oxygen; H_4^+ = Ammonium; Temp = temperature; NO_3^-= Nitrates and BOD = biological oxygen demand) (Author, 2008)

Increased PO_4^{2-}, $NO_3^-$and NH_4^+ but reduced BOD concentrations and secchi depth led to marked decrease in *Chroococcus*, *Zygnemopsis*, *Dactylococcopsis*, *Dinobryon* and *Synedra*. Increase in BOD favoured *Straurostrum*, *Dactylococcopsis*, *Chroococcus* and

Zygnemopsis, but decrease in *Aphanothece, Trachelomonas, Microcystis, Navicula* and *Oscillatoria*.

Increased PO_4^{2-}, NO_3^-, NH_4^+, pH, DO, temperature and conductivity led to marked decrease in *Aphanothece, Trachelomonas, Navicula* and, *Oscillatoria*, but decrease in population of *Chroococcus, Zygnemopsis, Dactylococcopsis*. High turbidity favoured growth of *Dinobryon* and *Synedra*, but led to decreased populations of *Ceratium, Scenedesmus, Merismopedia*, *Nephrocytium* and *Anabaenopsis*.

The correlation of various phytoplankton genera with various physico-chemical parameters of the reservoir is shown in Table 8. Most phytoplankton including *Microcystis Dactylococcopsis, Chroococcus, Oscillatoria, Aphanothece, Merismopedia, Anabaenopsis, Synedra, Navicula, Melosira, Scenedesmus, Zygnemopsis, Nephrocytium, Dinobryon, Trachelomonas* and *Ceratium* showed weak negative correlation with nitrate, ammonium, soluble reactive phosphorus, pH, conductivity, secchi depth, temperature BOD and DO. Only *Straustrum* showed positive correlation with all the physico-chemical factors. *Microcystis* exhibited negative correlation with nitrate, ammonium, total phosphorus, pH, conductivity, secchi depth, temperature BOD and DO. However, a generally weak negative correlation was observed.

Table 7: Spearman's Rank Correlations between Phytoplankton and Various Physico-Chemical Parameters Investigated in the Sampling Period

Taxa	NO_3^-N	NH_4^+N	PO_4^{2-}	pH	Conduct	Secchi depth	Temperature	BOD	DO
Anabaenopsis	0.7	-0.5	-0.3	0.3	-0.4	-0.7	0.1	-0.09	0.1
Aphanothece	-0.1	0.4	0.6	-0.3	0.3	-0.1	-0.4	-0.6	-0.3
Ceratium	0.1	0.5	0.9	-0.03	-0.3	0.14	-0.9	0.3	-0.3
Chroococcus	-0.1	0.2	-0.6	0.7	0.2	0.5	0.5	0.6	0.1
Dactylococcopsis	-0.7	-0.2	0.1	0.4	-0.5	0.4	-0.3	0.03	-0.8
Dinobryon	0.2	0.1	-0.3	0.9	-0.2	-0.1	0.1	-0.03	-0.5
Melosira	0.06	-0.4	0.4	0.5	-0.9	-0.1	-0.6	0.3	-0.6
Merismopedia	0.3	-0.1	0.6	-0.03	-0.6	-0.2	-0.6	0.3	-0.03
Microcystis	-0.2	-0.6	0.1	-0.8	-0.2	-0.3	-0.09	-0.4	0.1
Navicula	0.4	0.2	0.2	-0.6	0.5	-0.5	0.00	-0.8	0.2
Nephrocytium	0.6	0.8	0.1	0.5	0.4	-0.1	-0.1	0.1	0.00
Oscillatoria	0.1	-0.4	-0.7	0.1	0.2	-0.4	0.6	-0.6	-0.1
Scenedesmus	-0.06	0.44	0.7	-0.3	-0.1	0.4	-0.6	0.5	0.2
Straustrum	0.2	1.0	0.3	0.1	0.6	0.3	-0.1	0.2	0.2
Synedra	-0.2	-0.9	-0.3	-0.4	-0.4	-0.3	0.2	-0.4	-0.03
Trachelomonas	0.4	0.3	0.6	-0.6	0.3	-0.6	-0.4	-0.8	0.03
Zygnemopsis	-0.5	0.6	-0.2	0.4	0.4	0.8	0.2	0.5	-0.1

CHAPTER FIVE

DISCUSSION

5.1 Phytoplankton Community, Composition and Abundance

The results from the study show that phytoplankton assemblage in Chebara reservoir was heterogeneous on both temporal and spatial scale. The spatial distribution and seasonal replacements of phytoplankton assemblages are affected by such factors as the shape and size of the lake basin, latitudinal position of the water body, wind action, seasonal changes in temperature, external hydraulic loads, light availability and nutrient dynamics (Gasse, *et al*., 1983; Nwanko, 1996; Huszar *et al*., 2000). Other processes act on time periods of days to weeks, like meteorological (wind, rain and cloudiness) and hydrological events (water and inputs, hydrological withdrawal and water level fluctuations). Depending on their intensity and on their frequency, these may drive non-equilibrium dynamics and enhance the species diversity of the ecosystem (Margalef, 1958).

In very large lakes and reservoirs, the phytoplankton community is never constant because communities develop in water masses of different chemical status and move with the circulation of waters (Dufour *et al*., 2006; Reynolds, 2006). In small water bodies the horizontal variation in phytoplankton community composition is slight with the greatest difference in the shallow littoral region where there is the possibility of input from benthic flora, and in the inflow region where the new water may contain species from the inflow (Scheffer, 1998).

Phytoplankton diversity and abundance were higher in December, March and February in that order. This corresponded with dry seasons during which light intensity and transparency are high, and possibly with high flushing rates (Floder and Burns, 2005). During the dry period, the water level of the reservoir seems to be a key factor controlling the access of the phytoplankton to the nutrients stored in the sediments (Chalar, 2006).

Phytoplankton diversity and abundance was higher at stations 3, 2 and 1 in that order. At station 3, nutrient input from farm and could have been a factor in stimulating phytoplankton development. Agricultural activities around a catchment can contribute significantly to phosphorus load via erosion (Chalar, 2006). However, at station 1, high transparency, and possibly high flushing rates could contribute to high phytoplankton diversity and abundance (Floder and Burns, 2005).

Phytoplankton biomass was low at stations 2 and 6 (Figure 3). Both stations were shaded by dense vegetation in the littoral zone and protected from wind action. In water columns with low transparency, light limitation forces competition and maintains phytoplankton diversity under natural regimes of light flactuations (Huisman *et al,* 1999a, Floder and Burns, 2005). In stable aquatic ecosystem, species with the lowest critical light intensity will exclude all others (Floder and Burns, 2005). Under high flushing rates as in station 6 (Figure 3)., in-lake processes are weak, and the biomass is maintained low but dominated by species adapted to permanent water mixing, high turbidity and low retention time (Reynolds, 1993).

Species evenness was low for both sampling stations and dates. Different phytoplankton genera only showed patchiness on spatial scales, with no significant difference in spatial phytoplankton abundance. In stable ecosystems, phytoplankton densities are low, and diversity values moderate and corresponding to species limited to light and retention time (Roelke and Buyukates, 2002). However, as conditions become favorable, community abundance increases and diversity reaches maximum, until where resource competition sets in.

Seasons are also a major agent of change in the structure of phytoplankton communities. Seasons bring about fluctuations in various environmental factors including temperature, salinity, conductivity, pH and available nutrients that determine phytoplankton growth patterns (Harris, 1996). The weather variability showed a major impact on biota and biochemistry of Chebara reservoir. There was significant difference in seasonal phytoplankton abundance, with a peak observed in December, March, and February. These were dry months and probably reflected the effect of high light intensities on algal photosynthesis and growth (Heaney *et al.*, 1995). The month of March also marked transition between dry and wet seasons.

Different phytoplankton classes attained peaks abundance at different times. Cyanophytes were highest in number in December 2007 but their abundance decreased in the wet seasons. Chlorophytes reached their peak in February and April 2008, and diatoms and euglenophytes reached their peak in April 2008. Crysophytes and dinoflagellates showed

seasonal succession such that while crysophytes were abundant in March, dinoflagellates showed higher populations in wet seasons. Droughts and floods tend to switch fresh water bodies between cyanophytes which either regulate their buoyancy or float to diatoms-which sink rapidly (Harris, 1995). The first three months of sampling corresponded with the dry season when discharge into the reservoir was lowest. There is little or no water input; therefore, nutrients concentrations are also low and the reservoir remained generally clear. Dinoflagellates such as *Ceratium* and *Peridinium*, and chloropytes such as *Scenedesmus*, *Ankistrodesmus*, *Pediastrum*, *Cosmarium*, *Selenastrum*, *Zygnema* and *Chlorella* which are adapted to low nutrient conditions flourish in dry season (Huisman , 1999a, Roelke and Buyukates, 2002).

In dry season light intensity was high. High irradiance leads to a growth surge, which eventually depletes nutrients. Depleted phosphorus and nitrogen could possibly lead to reduced phytoplankton growth (Wetzel, 1983; Kalff, 2002). The high growth continues until other factors such as predation nutrient depletion and death /sinking play against the high growth. In wet seasons, due to low irradiance and low temperature, growth rate decreases as was evidenced in the wet months (April, May and June). Diatom communities of unpolluted streams and rivers were most closely related to stream order and altitude (that is, mainly temperature, discharge regime and geology (Phillips, 1996 ; Quintana, *et al*., 1998), but ambient light, chemical compounds other than carbonates, and competition and predation are important as well (Chessman, 1986; Ward, 1986).

Based on results of correlations, nitrate-nitrogen, soluble reactive phosphorus, ammonium-nitrogen, pH, dissolved, BOD and conductivity all played a role in determining the community structure and composition of phytoplankton community in Chebara reservoir. High diversity of phytoplankton in Chebara reservoir positively correlates with availability of nutrients, high secchi depth and open nature of the reservoir which exposes it high irradiances. Phytoplankton abundance was greatest in March which was the beginning of long rains. The highest values of biomass were recorded at the end of the dry season and during the transition period which corresponds to the period when mineral nutrients are flashed from the catchment into the reservoir. Thomas, *et al.*, (2000) reported that phytoplankton abundance and productivity in tropical reservoirs is usually rain induced .The rain water drains from land deposits nutrients, suspended solids and algal cells (or resting/reproductive stages) into the water body, or in some cases may brings nutrient dilution effect (Phlips, *et al.*, 1997; Kalff, 2002).

The order of assemblage of phytoplankton in Chebara reservoir was Pyraphyceae > Cyanophyceae>Chlorophyceae>Bacillariophyceae>Crysophyceae>Euglenophyceae>Rho dophyceae. This was similar to observations made in Lakes Tanganyika and Victoria (Lung'ahyia *et al*., 2000), and Lake Tanganyika (Kimerei *et al*., 2005). The abundance of Chlorophyceae and Desmidiaceae in stations 1, 2 and 3 was due to the stations having high transparency and water resident time. Occurence of *Anabaena*, *Pediastrum*, and *Closterium* in their respective classes may have been caused by their response to availability and utilization of nitrate and phosphate ions (Opute, 2000).

Station 2 showed the highest phytoplankton abundance. This is probably because the station is sheltered thereby protecting the phytoplankton from drifting and disturbance under the influence of wind. In both stations 2 and 6 shade adapted genera such as *Ceratium* and *Peridinium* were the most abundant. Dinoflagellates, especially *Ceratium* and *Peridinium*, showed the highest spatial and temporal abundance of the groups followed by *Microcystis*. *Peridinium* and *Ceratium* which occur in warm oligotrophic waters (Davison *et al*., 1995; Tirok and Gaedke, 2006). *Ceratium* particularly have shade tolerant forms and show increased frequencies up to a depth of 100 metres (Graham *et al*., 1944). They can take more phosphorus and nitrogen in the deeper waters and can still remain photosynthetically active at that depth.

Many genera of cyanophytes and chlorophytes occurred at low densities and which showed patchiness. Hutchinson (1961) noted the paradox of phytoplankton in which 10-50 species appear to coexist within apparently uniform water body. Hutchinson *et al*. (1970) and Platt and Denham (1980) suggested that several species can coexist in equilibrium (even with very slight spatial and temporal nutrient variations), provided that each species is limited by a different factor or resource.

The dominance of diatoms is typical of tropical rivers (Wood and Talling, 1988; Alfred-Ockiya and Otobo, 1990). Diatoms were most abundant in wet season and at station 6 which was close to the inlet of the main river. During wet season mixing of water and wash outs may occur during which time diatoms receive shortly pulsed external sources of silica. These nutrients have been known to be limiting to phytoplankton growth (Harris

and Baxter, 1996). Station six also showed high turbidity, resulting in low light intensities in water. Diatoms have high sedimentation rates and are physiologically situated to grow under deep and mixed waters which also have low light conditions (Harris and Baxter, 1996) where silica is available. Therefore greater frequencies of diatoms respond to increased flows, as in the inlet and outlet regions (Krammer and Lange-Bertelot, 1991). Significant abundance of diatoms was also observed at station 1 in which the water column was deep and clear (Harris and Baxter, 1996). Station 6, being located in riverine zone of the reservoir, was showed higher inputs of nutrients coming from rocks and sediments. The dominance of *Melosira* and *Synedra* was probably due to high pulses of silica and sulphate especially abundant during the rains. Other diatoms such as *Navicula*, *Synedra*, Nitzchia, *Eunotia* and *Pinnularia* were observed to occur at low densities. Of the diatoms listed, *Pinnularia* and *Eunotia* showed the least abundance are associated with low water pH, and their limited occurrence may be linked to high pH (Krammer and Lange-Bertelot, 1991). Both *Synedra* and *Melosira* species are associated with waters containing high concentrations of DO, low concentrations of BOD and low concentrations NH_4^+, PO_4^{2-} and nitrates (Krammer and Lange-Bertelot, 1991; Alfred-Ockiya and Otobo, 1990). *Navicula* and *Niztchia species* flourish in alkaline waters having low DO, and high BOD but are also tolerant to eutrophication (Steinberg and Schieffele, 1988). However, since diatoms are basically benthic, they showed generally low abundance in the plankton.

The occurrence of nitrates and phosphates may have caused the abundance of chlorophyceae and cyanophyceae. These nutrients have been known to be limiting to

phytoplankton growth (Talling and Lemoalle, 1998). In waters with high nutrient status, cyanophytes can proliferate and form blooms (Reynolds, 1984). But low populations of cyanophytes were observed due to low nutrient status of the reservoir. Only few desmids were identified in Chebara reservoir. Desmids prefer brackish and highly saline conditions (Opute, 2000) as opposed to Chebara reservoir whose water was fresh and oligotrophic. *Microcystis* was the most abundant of all cyanophytes, being favoured by relatively strong build up of ammonium and phosphates but low concentrations of NO_3^-. Other cyanophytes that occurred in low concentrations of NH_4^+ include *Chroococus* and *Aphanothece* (Blomquist *et al*, 1994; Reynolds, 2006).

A high abundance of cyanophytes was observed during the dry season. However, chlorophytes showed higher species diversity and higher spatial distribution. Dry seasons were also characterised by relatively high concentrations of NH_4^+ but low concentrations of phosphates. Studies have revealed that in reservoir discharges with low nitrogen relative to phosphate concentrations, nitrogen becomes limiting, providing a competitive advantage for N-fixing forms of cyanophytes over chlorophytes (Webster *et al*., 1996). *Microcystis* can also flourish in low nitrogen supply (Reynolds, 1999). This was demonstrated by the high relative abundance of members of cyanophytes over the chlorophytes at Chebara.

Chebara reservoir has a mean water depth of 45 meters and clear, and may have favoured cyanophytes at stations 1, 2 and 3.Depth of water determines the extent of light penetration and mixing of water mass and consequently survival and growth of

phytoplankton. Many cyanophytes posses buoyancy mechanisms and are able to control their vertical position in the water column (Harper, 1992; Vanni, 1999**).**

The surrounding vegetation, rocks and soils greatly influence the type of phytoplankton in oligotrophic lakes and reservoirs. The host plants such bryophytes and pteridophytes influence the epiphytic phytoplankton communities, while soils and rocks determine the chemistry of a water body. When there are high concentrations of nutrients in the water, the transfer of nutrients from terrestrial plants becomes insignificant and the number and type of epiphytes will reflect the conditions of the water (Harper, 1992; Sceffer, 1998; Simciv, 2005). For example, *Scenedesmus* and *Ankistrodesmus* multiply in high concentrations of organic nitrogen (NH_4^+), but low concentrations of NO_3^- and are capable of converting nitrate into NH_4^+ (Opute, 2000). Highest mean concentrations of ammonium nitrogen were recorded in stations 1 and 2 and the lowest at station 3. Relatively high mean concentrations of ammonium nitrogen were also recorded at stations 4 and 5. *Ankistrodesmus* showed highest abundance at stations 1 and 6 while *Scenedensmus* showed highest abundance at station 4, and patchiness in station 1.

Station 6 the highest PO_4^{2-} level. Both station 2 and 6 had a lot of vegetation. Chrolococcales such as *Scenedesmus*, *Chlorella*, *Oocystis* and *Tetraedron* were in greatest abundance in these stations. These phytoplankton may be epipelic or epiphytic and flourish in neutral waters that are rich in organic phosphorus and nitrogen (Scheffer, 1998; Simciv, 2005). The same environments also favour the growth of cyanophytes such as *Microsystis*, *Aphanizomenon*, *Anabaena* and *Gleotrichia*, *Chroococcus* and

Coelosphaerum which are known to pref were er anoxic conditions with high NH_4^+ concentration. Pyraphytes such as *Straustrum* were also observed at these stations.

Spatial variability is a structural character of an ecosystem and allows for complex population interactions involving energy transfer, competition, predation, nutrient depletion, death /sinking and niche formation (Reynolds, 2006). It is therefore expected that less disturbed stations exhibit higher species diversity, with evenly distributed taxa (Kling *et al*., 2001; Jones *et al*., 2001, Death, 2004).

Although there was observed seasonal variation, it is noteworthy that most phytoplankton genera showed similar growth patterns throughout the study period. In general diatoms dominated the colder nutrient- rich waters such as at station 6, chlorophytes were more abundant in warmer oligotrophic waters and dinoflagellates appeared to grow in intermediate environmental conditions. Perodicity was clear, with peaks followed by lows. Many algal cells can sink, permanently from the water column. At the bottom they utilize all oxygen and since they do not receive sufficient light for photosynthesis, the cells die. Death and decay cause oxygen depletion bringing about anaerobic conditions in the hypolimnion. Detritus in the hypolimnion is dense with rich supply of nutrients but beyond reach by phytoplankton. The phytoplankton then receives limited nutrients and their growth rate is reduced, till mixing occurs, or water levels reduce as in dry seasons (Chalar, 2002).

The productivity of Chebara reservoir was low as estimated by chlorophyll-*a*, suggesting oligotrophy. Low concentrations of chlorophyll-*a* at station 2 corresponded with low BOD concentrations, indicating relatively low primary productivity at this station. This could be attributed to the shading effect of terrestrial and some aquatic vegetation that was in both stations. Depth of water is an important diving force in nutrient dynamics of a reservoir. Shallow lakes and reservoirs are generally more productive (Thomas, 2000). In some shallow lakes, resuspension events are correlated with increases in phytoplankton biomass as estimated by chlorophyll-*a* (Schelske *et al*., 1995; Hamilton and Mitchell, 1997; Ogilvie and Mitchell, 1998). Mixing can result from the entrainment of meroplankton into the water column (Schelske *et a*l., 1995), and nutrients deposited in the lower regions of the lake are brought up to the photic zone, being available for phytoplankton growth On the contrary, deep lakes tend to be oligotrophic (Kotut *et al*., 1998). The water volumes are high, mixing is limited if not rare and thermal stratification normally occurs. This leads to algal growth confined only within the epilimnion. The hypolimnion remains rich in nutrients, but beyond reach of phytoplankton. Chebara reservoir is also characterised by narrow littoral and sublitoral zones, but an extensive profundal zone. This reservoir generally supports a lot of green algae and diatoms, supports a lot of green algae and diatoms, but with marked patchiness.

There was negative correlation between transparency and most of phytoplankton. Increased transparencies favoured *Dactylococcopsis, Chroococcus, Scenedesmus, Zygnemopsis, Ceratium* and *Straustrum*. Higher transparency could have allowed

grazing by zooplankton, and depletion of nutrients leading to reduced densities at different stations. (Harris, 1996)

Trophic state indices based on phytoplankton chlorophyll-*a* concentrations, secchi depth and phosphorus concentrations indicate that the reservoir is oligotrophic. It also indicates that phytoplankton growth in the reservoir is more likely to be limited by availability of P than N. Thus, small increases of phosphorus in the reservoir could stimulate phytoplankton to produce even blooms, whereas increases in N may not. Nuisance conditions associated with phytoplankton diversity and productivity were absent probably due to the reservoir's oligotrophic status.

The results indicated low BOD concentrations, suggesting oligotrophic fresh water bodies that support many species of chlorophyceae and fresh water diatoms (Krienitz *et al.*, 1998; Kimirei *et al.*, 2005). The results also indicated that P is probably the limiting factor in phytoplankton growth in the reservoir. The results of this study correspond with results obtained earlier in fresh waters by Ryther and Danstan, 1971. In their study, they concluded that lower concentrations of P in fresh waters inhibited growth of phytoplankton.

Mean concentration of phosphorus, an indicator mainly of living and non-living P, decreased from a high of 0.01 μg millilitre^{-1} (station 6) to a low of 0.007 μg millilitre^{-1}(stations 1,3 and 5). This pattern is probably due to sedimentation of organic and inorganic particles in the three stations. These stations are sheltered from strong winds.

Spearman's rank correlations results suggested such factors as temperature, nutrient dynamics, sechi depth, pH, conductivity BOD and DO as some of the factors possibly contributing to phytoplankton depletion in dry seasons. Different correlations existed between the phytoplankton and the physico-chemical parameters of the reservoir. Temporal negative correlations with temperature and transparency for most genera were an indication that the phytoplankton and leading to their reduced occurrence. Higher transparency may have allowed for grazing (Reynolds, 2006). In dry season, phytoplankton proliferate and competition for nutrients may ensue (Grover and Chrzanowski, 2004) Genera with low adaptability to utilise nutrients even at low concentrations become reduced.

5.2 Physico-chemical Parameters in Chebara Reservoir

The physical and chemical conditions of the Chebara reservoir show homogeneity on spatial and temporal scales. According to Scheffer (1998) these conditions often interact to determine the productivity nature and make up the assemblage of the autotrophic organisms. Chebara water showed oligotrophy in relation to phytoplankton assemblage and abundance.

Temperature of a water body influences the occurrences, and intensities of occurrence, of other parameters such as DO, conductivity and total alkalinity (Kalf, 2002). Temperatures were generally medium, a high of 20°C and low of 18 ºC and similar for all the stations and times of sampling. Temperatures of this reservoir are falls within the same range as

those recorded in many East African lakes that include Lake Victoria (Lunganyia *et al*, 2000), Lake Tanganyika (Kimereri *et al*., 2005), Lake Turkana (Burgis and Morris, 1987) and Lake Naivasha (Kitaka,1991), Turkwell Gorge (Kotut, 1998). The low temperatures of 18 ºC recorded are due to the location of the reservoir in a forest causing low insulation, its situation in high altitude where atmospheric temperatures low. Presence of outlet leads to water losing heat as it flows out of the reservoir. Temperature of a water body also influences the occurrences, and intensities of occurrence, of other parameters such as DO, conductivity and total alkalinity (Kalff, 2002).

The pH of the reservoir is neutral to slightly alkaline. This value is falls within the same range as the pH of deep tropical lakes such as Lake Victoria (Lunganyia *et al*., 2000) and Lake Tanganyika (Kimirei *et al*., 2005). According to Kalff (2000), spatial variations of pH within the same water body can be attributed to the combined effects of both phytoplankton population and nutrient condition. In shallow alkaline nutrient-rich water bodies, high phytoplankton population with subsequent high biomass allows the removal of large quantities of CO_2 and HCO_3^- in the daytime (Lewis, 2006). Significant shifts in pH could be caused by variations in water volumes that influence the extent of dilution of dissolved chemicals (Reynolds, 1998; Kalff, 2002; Lewis, 2006). Most lakes are basic (alkaline) when they are first formed but become more acidic with time due to the build-up of organic materials (Menden-Deuer and Lessard, 2000). As organic substances decay, carbon dioxide (CO_2) forms and combines with water to produce a weak carbonic acid. The minor inlet river at station 3, which drains through open agricultural and settlement land, showed the highest pH. This variation could be attributed to changes in the

chemical composition of water within the catchment area. Forests acts as filters and purifiers and as such could alter or correct the pH of waters that drain through them (Angeler *et al*., 2000). This could explain the relatively low pH values recorded at stations 5 and 6. Station six also recorded the lowest abundance of cyanophytes. According to Harris and Baxter (1996), the abundance of cyanophytes becomes reduced at low pH levels. This could explain the observation made in Chebara reservoir.

Conductivity was low and fairly uniform for all the sampling stations, suggesting that the reservoir draws from smaller catchment and short water retention time. This finding concurs with findings from African lakes (Burgis and Morris, 1987). Variations in conductivity may be brought about by precipitation and discharge of water into a reservoir (Kalff, 2002). This is because the water discharged from a drainage basin often reflects the activities being carried out in the basin and its intrinsic characteristics. Harris (1986) also observed that larger catchments usually yield larger conductivity and alkalinity values as compared to smaller catchments. This is mainly because large catchments have many different land features, are exposed to evapo-transpiration over a long time, often drain across intensely disturbed land surfaces and posses greater contact between run-off and rock surfaces (George and Hewitt, 2006).

The spatial and temporal variation in nutrient was minimal within the reservoir. Soluble reactive phosphorus status of phosphorous was heterogeneous, concentrations being similar and slightly higher for April and June, but lower for February and March. Mean concentration of soluble reactive phosphorus an indicator mainly of living and non-living

P, decreased from a high of 0.01 µg millilitre^{-1}(station 6) to a low of 0.007 µg millilitre^{-1}(station 1, 3 and 5). These results strongly suggest that external loading by incoming water as observed at station 6 also contributed to phosphorus of this reservoir (Sanchez-Carrilo *et al*., 2000a). High soluble reactive phosphorus concentrations at station 6 with a corresponding small secchi depth suggested that there was greater contact, and therefore a frequent exchange of phosphorus between water and sediment. Nitrate- nitrogen and ammonia- nitrogen concentrations were greater than concentrations of soluble reactive phosphorus, indicating that phosphorus was limiting in the entire reservoir. This study confirms previous studies that P is limiting in tropical waters (Reynolds, 1999; Sanchez-Carrilo *et al*., 2000; Reynolds, 2001). Phosphorus in deep aquatic ecosystems is adsorbed on sediments, on solids suspended in water or in insoluble salts. Thus in deep waters reservoir, phosphorus can only be available if mixing occurs. However, it has been established that a great proportion of phosphorus in East African fresh waters comes from atmosphere rather than from sediments (Cole et al., 1990; Schindler et al. 2008).

Higher DO concentrations at station 1 compared to stations 3, 4 and 6 (main inlet) is likely to result from the effect of mixing at stations 1, 3 and 6, and possibly from atmospheric inputs. High DO at station 4 corresponds to high productivity at station 4 (Table 2) (Death, 2004). Station 1 is open waters at the reservoir's outlet and higher DO concentrations at this station could possibly be from atmospheric inputs into the turbulent water as it flows through the outlet, and the high primary productivity that leads to liberation of oxygen. The highest BOD concentrations were recorded at station 4 and the lowest at station 5. This may be explained by biochemical utilization of DO along the river channels at sample stations 3, and 4. Station 3, which drains open agricultural and

settlement lands, had low DO. Death (2004) reported that catchments with high human activity tend to increase BOD caused by respiring soil flora and fauna and by marked inputs of phosphates and nitrates in run-off from farm fields (George and Hewitt, 2006).

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6. 1 Conclusions

Chebara reservoir is a moderately warm, oligotrophic water body; low in N and P.

Nitrates, soluble reactive phosphorus, ammonium, pH, dissolved oxygen, BOD and conductivity played a role in determining the composition of phytoplankton community in Chebara reservoir. The study also indicates that phytoplankton growth in the reservoir is more likely to be limited by availability of P than N. Thus, small increases of P in the reservoir could stimulate phytoplankton to produce even blooms, whereas increases in N may not.

Phytoplankton assemblage at Chebara reservoir was heterogeneous throughout the study period, although there were distinct temporal and spatial surges and disappearances. Whereas diatoms dominated the colder nutrient rich waters (as at station 6), the green algae were more abundant in warmer oligotrophic and open waters and dinoflagellates more uniformillilitrey distributed in all stations.

The productivity of the reservoir was low (approx. 0.8 μg $millilitre^{-1}$).

6.2 Recommendations

On all occasions, survey of the spatial variability of DO, pH, temperature and electrical conductivity were observed, no consistent spatial trends were observed. This and indicates that a single depth profile of measurements could be considered indicative of condition throughout the reservoir.

Pyraphytes require calcium for their growth, and their occurrence in a water body implies expensive treatment costs and procedures. Although calcium concentrations were consistently low, the high abundance of pyraphytes in this reservoir could suggest a need to monitor management practices in the reservoir catchment that maintain calcium concentrations and populations of pyraphytes low in order to reduce the water treatment costs.

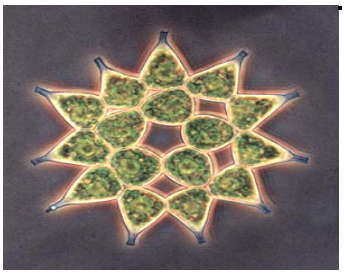
Waterfowls were more in site 2, 4 and 6.Their feeding and roosting (as was evident in the faecal dropping at site 4) could have possibly increased phosphate concentrations at these stations mentioned. Due to the potential sensitivity of the reservoir to P loading, it may be necessary to keep track of water fowl populations on the reservoir as a way of checking eutrophication.

It is important to note that for effective water quality monitoring of Chebara reservoir, the tributaries that contribute water directly to the reservoir should be selected as monitoring stations. For this reason stations 3 and 4 were selected. This site is at the points where the tributaries empty their water into the reservoir.

Although water treatment can remove many contaminants, it is better and more cost-effective to prevent contamination at the source of water supply. There is growing recognition of using phytoplankton as a means of monitoring physico-chemical conditions as a means of reducing the cost of treatment systems required for safe drinking water. Other ecological studies involving invertebrates, fish, zooplankton and benthic genera should be carried out to generate a complete ecological map in the reservoir.

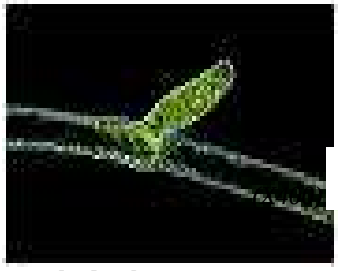
The data results obtained from this study are useful for tracking the effects of changing activities in the drainage basin, and for evaluating the effectiveness of watershed Best Management Practices (BMPs) aimed at ensuring long term protection for water supply.

a: *Closterium*



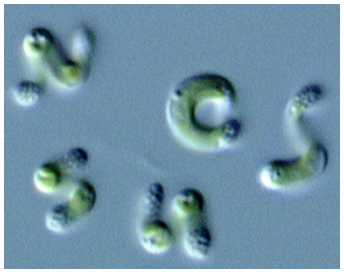
b: *Pediastrum*

c: *Navicula*

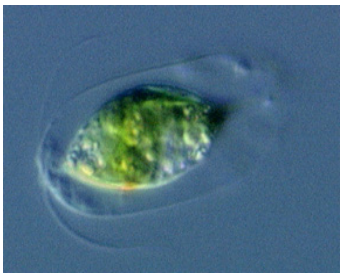


d: *Cladophora*

e: *Scenedesmus*

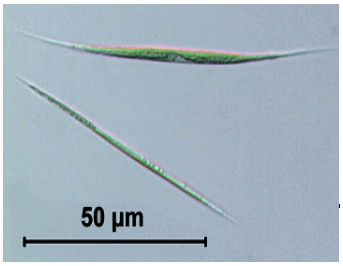


f: *Selenastrum*

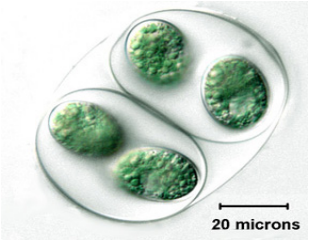


g: *Haematococcus*

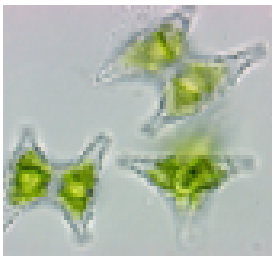
h: *Trachelomonas*



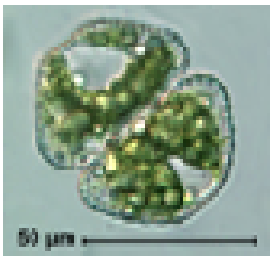
i: *Ankistrodesmus*



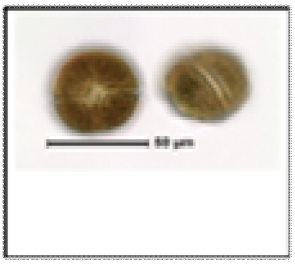
j: *Oocystis*



k: *Straustrum*



l:*Cosmarium*



m: *Peridinium*

Figure 7: Photomicrographs Showing Various Phytoplankton Cells Identified in Chebara Reservoir During the Study Period (Author, 2008)

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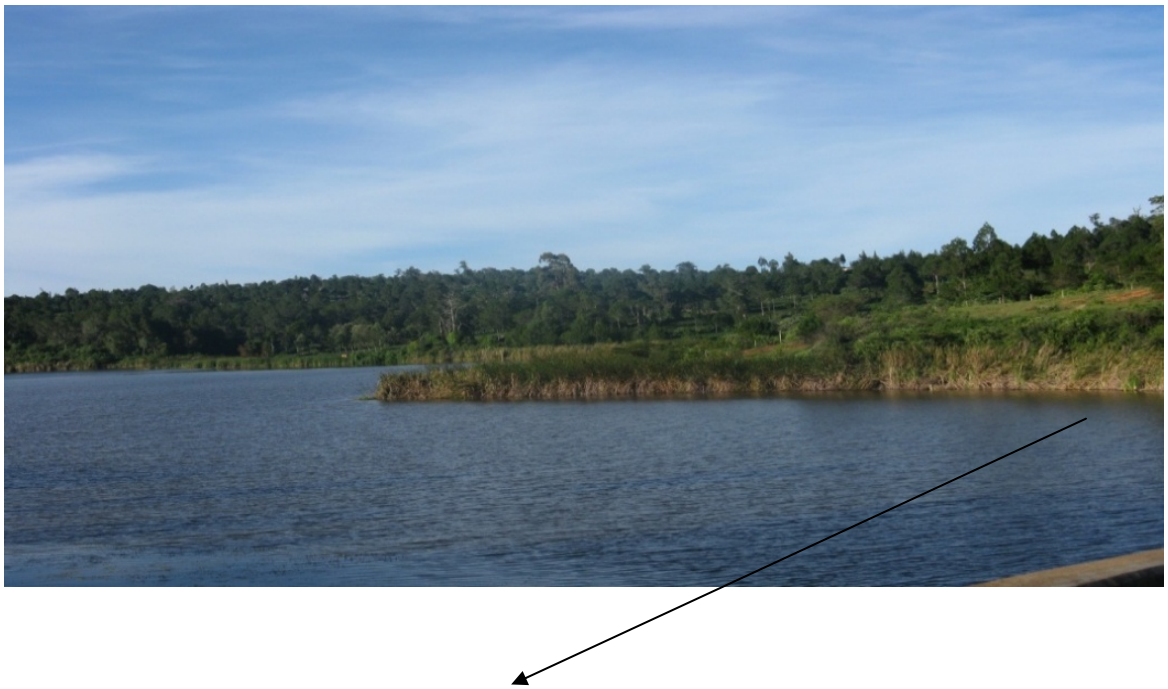
APPENDICES

Appendix 1: Photographs Showing the Various Sampling Stations of Chebara Reservoir (Author, 2008)

Station 1

Station 2



Station 3

Station 4

Station 5

Station 6

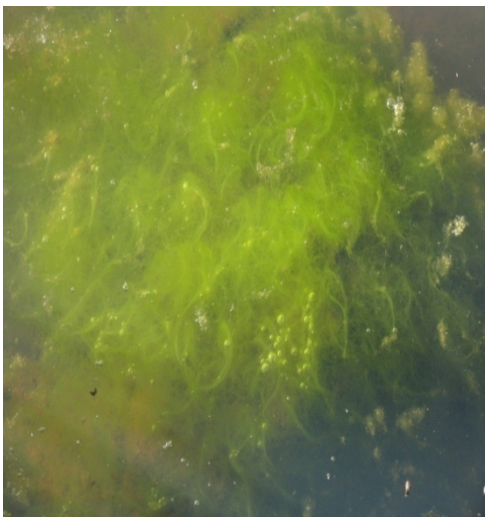
Appendix 2: Photographs Showing the Various Aquatic and Terrestrial Vegetation Within and Around Chebara Reservoir (Author, 2008)

2a. *Crassura granvikii*

2b. *Crassocephalum picridifolium*

2c. Nymphaea *lotus*

2d. *Nymphaea lotus*



2e. *Spyrogyra sp*

2f. *Spyrogyra sp*

2g. *Orobanche minor*

2i. *Peucendanum sp*

2j. *Polygonum salicifolia*

2k. *Periploca linearifolia*