A STUDY OF THE LIMNOLOGY AND ECOLOGY OF MAZVIKADEI RESERVOIR

By

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Abstract

An assessment of the limnology of Mazvikadei reservoir was carried out from May to October 2015. The main objective of the study was to determine whether limnology of the reservoir has changed since the filling phase. The reservoir has matured into an oligotrophic status characterized by low algal biomass, low concentrations of P and N, high Secchi disk clarity and low chlorophyll *a* concentration. Mazvikadei is well oxygenated with dissolved oxygen concentration of up to 7.9 mgl^{-1} and a slightly alkaline pH (8.54). During the entire sampling period the reservoir was not strongly stratified except in July when there was stratification in the first 5 m. The average conductivity in the reservoir has increased to 263.2 μScm^{-1} and the transparency has increased to 4.6 m. The improvement in transparency is linked to change in the trophic status of the reservoir. The phytoplankton community in the reservoir comprised of six main groups namely Dinophyta, Euglenophyta, Chlorophyta, Bacillariophyta, Desmids and Cyanophyta. Bacillariophyta and Dinophyta dominated in the cool dry season and Chlorophyta in the hot dry season. Cyanophyta had less representation although it occurred in small numbers right through the sampling period with the highest numbers being observed in May. Euglenophyta and Desmids were most abundant in May. Fifty four species were recorded.Species richness was highest in May at the onset of the cool dry season in response to high nutrient concentrations. Evenness decreased during the cool dry season. Dominance with respect to species representation was Chlorophyta (21 species), Bacillariophyta (14 species) and Cyanophyta (7 species). Phytoplankton abundance and composition were significantly correlated with temperature, nitrates and total nitrogen. The major groups of zooplankton recorded were the Cladocera, Copepoda and Rotifera. Cladocera were the most dominant although rotifers and copepods were well represented. Cladocerans became most abundant during the cold dry season (June and July) when they assumed 75 % dominance. Rotifers and copepods dominated during the hot dry season. Ostracods were only observed in September in the hot dry season. Species richness has increased to 19, being represented by 6 rotifers, 6 cladocerans, 6 copepods and 1 ostracod. The zooplankton community was highly influenced by the amount of reactive phosphorus and phytoplankton abundance. The relationship between reflectance and ammonia, nitrates and reactive phosphorus was not stable over time. The best time to use remote sensing for water quality monitoring was in the cool dry season where processes in the reservoir are relatively stable. Ammonia and nitrates could best be approximated by the near infra red and red band respectively. No relationship was found between reactive phosphorus and reflectance. The study showed that Mazvikadei reservoir has matured and assumed the physico-chemical characteristics and plankton community typical of an oligotrophic lake. Due to its oligotrophic status, remote sensing will not be effective in water quality monitoring.

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Table of Contents

CHAPTER ONE

1 INTRODUCTION

1.1 General introduction

A large number of man-made reservoirs have been constructed throughout Southern Africa in order to cope with seasonal rainfall. Although primarily designed for specific water supply purposes, most of these reservoirs now fulfill a multipurpose role. The need for such a large number of reservoirs is related to the unpredictability in rainfall and run-off within the region (Thornton, 1987). The total global reservoir area is unknown, but in 2000 the World Commission on Dams counted about 48 000 large dams, 46 percent of them in China, 19 percent in the rest of Asia, 3 percent in Africa (60 percent of which are in South Africa and Zimbabwe), and 2 percent in South America (Kolding and van Zwieten, 2006).

Zimbabwe is a land-locked semi-arid country located in Southern Africa. The country is characterized by large variations in annual rainfall with very few perennial rivers and lacks natural lakes. This has necessitated the construction of dams across the main river systems in order to secure adequate water supply (Rusere, 2005). The government and non-governmental organizations (NGOs) recently re-embarked in dam construction projects particularly in drought prone areas of Zimbabwe (Dube and Kamusoko, 2013). These range in size from small to large reservoirs and are used to supply water for urban use, mining, industries and irrigation purposes (Rusere, 2005). Most of these reservoirs also play an important role in fish production and contribute significantly to the livelihoods of the communities residing along their shores (Petr, 1994). In Zimbabwe five reservoirs namely Kariba, Chivero, Manyame, Mutirikwi and Mazvikadei hold the most important commercial fish stocks. Mazvikadei dam being one of the large dams holds large quantities of water that is used to supplement irrigation (Rusere, 2005).

Generally the limnology of large and medium sized African reservoirs has been studied extensively and therefore information about factors that are likely to affect the small reservoirs exists. These factors include the changing flows of tributary streams and water fluctuations (Nhiwatiwa and Marshall, 2007). These changing patterns in seasonal flow in turn affect aquatic species migration and the water level fluctuations influence the physico-chemical characteristics of reservoirs and lakes (McLachlan, 1969). Water quality deterioration in reservoirs and lakes usually emanates from excessive nutrient inputs resulting in eutrophication (Magadza, 2003). Changes in the physico-chemical characteristics can provide information on the current state of water quality in these reservoirs. The changes can in turn affect biodiversity and functioning of the reservoir (Mustapha, 2008).

Due to agricultural activities within its catchment, productivity within Mazvikadei reservoir could have increased over the years. There is at present no information on its trophic status so as to institute appropriate management strategies. This will require spatial and timely monitoring. GIS and Earth Observation are useful tools that can be used to collect water quality data in space and time. The integration of remote sensing and *in situ* monitoring will lead to increased knowledge on the water quality in the reservoir. Using remote sensing is less labour intensive, less time consuming and more cost effective. The applicability of this will be explored during this study.

Dams are structures designed to divert river water with the intention to alter the natural distribution and timing of the river flow in order to meet human needs. This can alter the natural ecosystem processes and functions of rivers (Bergkamp *et al.*, 2000). Pre-impoundment studies on the biotic and abiotic environment of the river provide a basis to understand the impact of damming on the riverine ecology (Argawal and Thapliyal, 2005). Evidence shows that altering the limnological characteristics of water bodies affects the biological productivity of any water body including fish populations. It is therefore imperative to assess changes within dammed reservoirs with time.

1.2 Justification of study

The establishment of artificial impoundments in Zimbabwe provided opportunities to undertake limnological studies as far back as the late 1950s (Sanyanga and Mhlanga, 2004). These studies however, focused primarily on Lake Kariba and Lake Chivero which have been extensively studied compared to other reservoirs. Considerable information is available on Lake Kariba and Lake Chivero, from their formation to their current state (Marshall, 1995; Magadza, 2003; Marshall, 2005; Mhlanga and Siziba, 2006). For example, there is relatively sufficient knowledge on fish ecology, plankton dynamics and physico-chemical aspects of these reservoirs but a glaring paucity of information on the limnology of Mazvikadei Dam (in southern Africa, the word Dam is used as the colloquial for the word reservoir). Agricultural expansion within the catchment of Mazvikadei Dam and invasion by exotic species such as *Oreochromis niloticus* could have altered the limnology of this reservoir since the last studies by Masundire in 1992. During the first year of filling, Masundire (1992) undertook a basic limnological study to assessed physico-chemical parameters and the plankton communities that had established in the reservoir. A more detailed study after about two decades will provide indications on the changes that have taken place in the reservoir. At present, knowledge on the status of the limnology of Mazvikadei reservoir is patchy despite its widespread recognition as a resort and irrigation source. Knowledge on the changes in the water quality, stratification regimes and changes in the biota of the reservoir can be used as an important tool to institute appropriate management strategies for the reservoir.

1.3 Research Aims and Objectives

1.3.1 Research Hypothesis

The physical and chemical limnology and the biological characteristics of Mazvikadei dam have changed from the pre-impoundment phase, filling phase to the present as the lake has aged in response to changes associated with human activities.

1.3.2 General objective

The main aim of this study was to investigate the physical and chemical limnology and aspects of the ecology of Mazvikadei reservoir in order to inform management and conservation decisions.

1.3.3 Thesis structure

1.3.3.1 Thermal and oxygen stratification regimes of Mazvikadei, an African tropical reservoir.

The aim of this chapter was to investigate the thermal and dissolved oxygen regimes of the reservoir associated with the stratification patterns.

1.3.3.2 Physico-chemical limnology and plankton dynamics of Mazvikadei, an African tropical reservoir.

The aims of this chapter were to study the physico-chemical limnology of the reservoir and the composition, abundance and diversity of plankton communities in the dam in relation to water quality and seasonal processes.

1.3.3.3 Exploring the relationship between reflectance and nutrient concentration over time in Mazvikadei Reservoir.

The aims of this chapter were to determine if the relationship between the physico-chemical variables and reflectance is stable over time and also to determine which bands are best approximators of water quality variables for the reservoir.

1.3.3.4 General discussion

This chapter gives a general discussion of the findings and points to some research gaps and recommendations for management.

1.4 Study Area

The study was carried out at Mazvikadei reservoir (17°13'14"S and 30°23'30"E) which is located on the Mukwadzi River in Banket, northwest of Harare (Fig. 1.1). Construction of the dam started in 1985. The wall, which is an earth fill embankment, is 63.5 metres in height making it the second highest dam wall in Zimbabwe. The construction of the dam was completed in 1988 and the reservoir filled for the first time in 1990. The reservoir has a storage capacity of 360 million cubic metres with a surface area of 2300 hectares at full capacity. Mazvikadei has a mean depth of 16 m and a maximum width of 2 km. Table 1.2 shows the morphometric features of Mazvikadei reservoir at full capacity. The reservoir is surrounded by commercial agricultural land and it provides water for farm irrigation. It is also a popular weekend resort due to its proximity to Harare, Zimbabwe's capital.

Most of the reservoir's catchment lies in granitic rock and soil with metamorphomised sediments in some parts. The catchment also includes part of the Great Dyke, characterized by serpentine soils with high levels of magnesium, nickel and chromium content (Masundire, 1992).

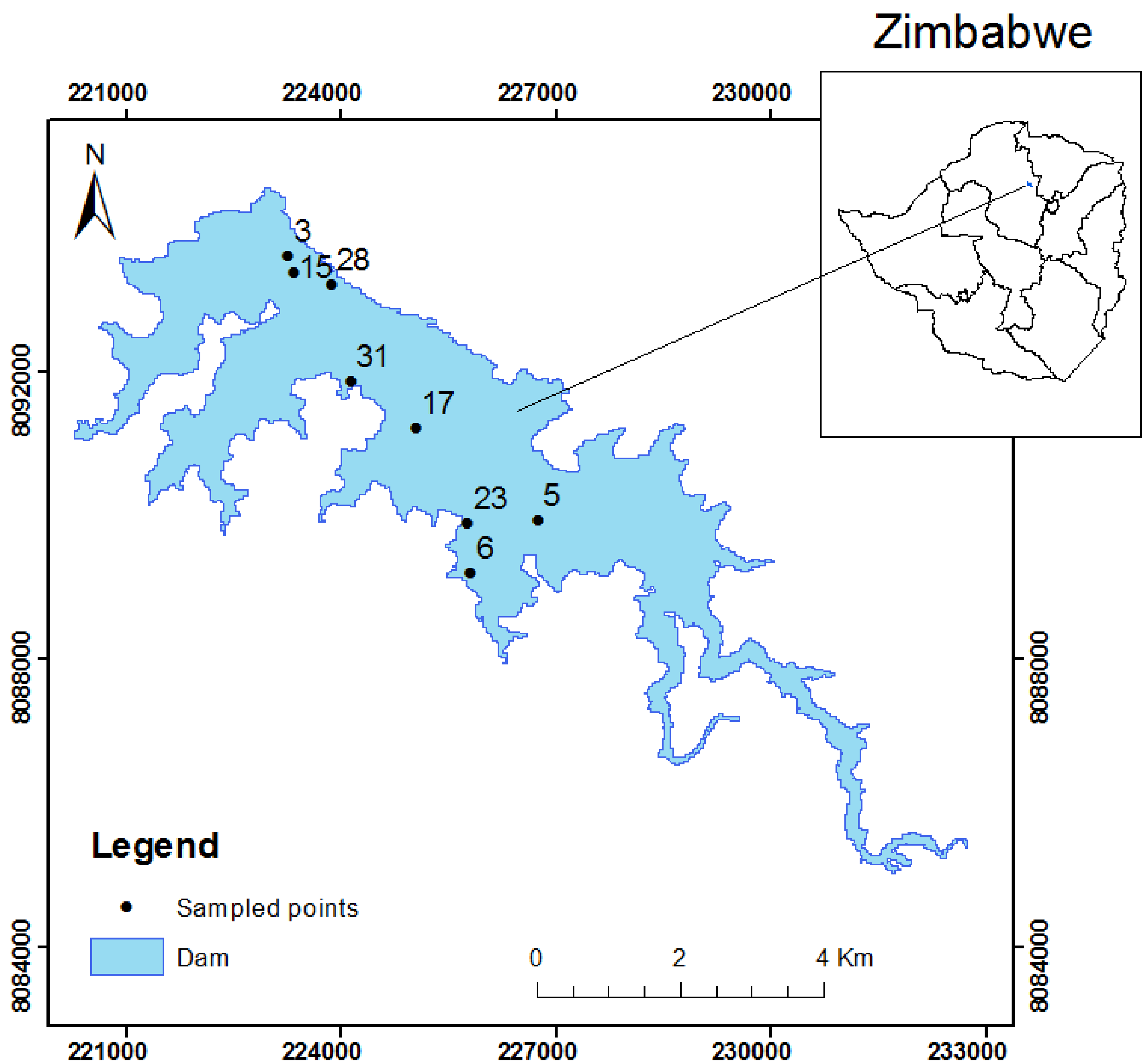


Figure 1.1 Mazvikadei reservoir in the north eastern region of Zimbabwe showing distribution of sampled points from May-October 2015.

Macrophytes such as *Nymphaea sp*. and *Phragmites sp.* were common along the banks of the reservoir. Submerged macrophytes such as *Lagarosiphon major* and *Potamogen crispus* were abundant in the reservoir throughout the sampling period from May-October 2015.

Table 1.1 Morphometric features of Mazvikadei Reservoir (when full) (Source: Masundire, 1992)

Height of wall	63.4 m
Length of main embarkment	320.0 m
Length of saddle dam	810.0 m
Length	11.5 km
Mean breadth	2.0 km
Maximum depth (z_m)	58 m
Mean depth (z)	15.9 m
Volume (V)	365.0 10^6 m^3
Volume development (D_V)	0.82
Surface area	2300.0 ha
Shoreline (L)	116.0 km
Shoreline development (D_L)	6.8
Catchment area	1120.0 km^2
Mean annual runoff from catchment	162.0 10^6 m^3
Renewal time	2.0 yrs

1.5 Sampling sites

Since characteristics of a reservoir exhibit distinct longitudinal gradients linked to the reservoir's river-lake hybrid nature (Kimmel *et al*. 1990) during this study these gradients were characterized by selecting eight sampling sites based mainly on depth and location. The sampling stations were randomly distributed along the reservoir. These were along the Mazvikadei dam from the inflow of the Mukwadzi River to the dam wall. Sampling was undertaken at each of these sites. Eight sites (3, 5, 6, 15, 17, 23, 28 and 31) were chosen for the monthly profile sampling and are shown in Fig 1.1. The co-ordinates for the different study site locations in Mazvikadei reservoir are shown in Table 1.2.

Table 1.2 Co-ordinates for the different study site locations in Mazvikadei reservoir.

SITE	S	E
3	17.225662	30.397744
5	17.259238	30.429994
6	17.265883	30.421122
15	17.227710	30.398419
17	17.247400	30.414133
23	17.259585	30.42082
28	17.229262	30.403367
31	17.241533	30.405747

CHAPTER TWO

2 THERMAL AND OXYGEN STRATIFICATION REGIMES OF MAZVIKADEI, AN ARFICAN TROPICAL RESEVOIR.

2.1 Introduction

The stratification of a lentic water system such as a lake is the single most important factor that regulates its biotic processes, creating density differentials that influence vertical mixing, and the distribution of chemical ions and particles with respect to water depth (Nhiwatiwa and Marshall 2006). Stratification causes depth related variations in the distribution of temperature, dissolved solids and suspended particulate matter. The latter two components (chemical and suspended particulate stratification) are generally negligible in the bulk fluid of inland lakes and reservoirs. Thus, stratification is dominantly dependent on temperature variation, which in turn is a function of the overall energy balance and the internal mixing processes of a lake (Fischer *et al*., 1979). Inland lakes are primarily made up of fresh water and therefore the dominant stratification component is thermal stratification. The seasonal cycle of lake stratification results from the seasonal variation in solar radiation and controls the long-term temperature and distribution of chemicals (in particular oxygen) in the lake (Fischer *et al*., 1979).

In summer, the high solar radiation input and warm air temperatures contribute to a strong thermal stratification of the lake. Surface water is warmer than bottom water. Winds tend to keep the surface water well-mixed, and this upper well-mixed region of the lake is called the epilimnion. Below the mixing action of the wind and the penetration depth of the solar radiation, a strong temperature and accompanying density gradient develops. This region of strong gradients is called the thermocline, or sometimes picnocline. Below the thermocline a weaker temperature gradient is observed and the water is cool and comparatively quiescent. The bottom region of the lake is the hypolimnion. As the air temperature gets cooler and the solar radiation input decreases in the winter, the surface water begins to cool. Eventually, the surface water and thermocline cool down to the temperature of the hypolimnion and the lake is no-longer stratified. In this state the lake can easily be mixed, even by a light wind. The lake is therefore expected to completely mix and experience a turnover event.

Lakes may be classified according to the frequency of overturn. Lakes that overturn twice a year are called dimictic while those that overturn once are called monomictic. Lakes that overturn several times a year are polymictic and this is usually the case in shallow tropical lakes which are easily influenced by wind action. In amictic lakes there is no mixing and for some deep tropical lakes there may be poor mixing and in this case they are called oligomictic. Meromictic lakes experience incomplete mixing. These classifications are important to limnologists because they determine the species of plants and aquatic life that will populate the lake (Imboden and Wuest, 1995; Rutherford, 1994).

The majority of reservoirs in tropical Africa are shallow, polymictic and found in regions where evaporation approaches or exceeds precipitation, with water typically being derived from rivers (Mustapha, 2008). These reservoirs are usually filled up with water during the rainy season, but their volume become greatly reduced during the dry season as a result of evaporation, water withdrawal and municipal water usage. Many tropical African reservoirs have short water residence times, small sizes, large watersheds, high shoreline development ratios and large water level fluctuations due to seasonal influences (Dalu *et al*., 2013). Despite the advances that have been made in understanding the structure and energetics of mixed layers, it is surprising that in tropical lakes even the most fundamental information, such as persistent thermocline depth and stability of stratification, is limited and inadequate (Nhiwatiwa and Marshall, 2006).

Shallow lakes and ponds can stratify and destratify (with regard to both density and chemistry) on a daily basis, and may even remain persistently stratified over multiple days (Branco and Togersen, 2009). Thus, the entire water column of a shallow lake or pond may behave as a diurnal mixed layer (diurnal stratification). In small water bodies diurnal temperature ranges may be greater than seasonal ones. Lakes George, Uganda and Nakuru in Kenya experience a diurnal rather than a seasonal cycle of stratification (Ganf and Viner, 1973; Melack and Kilham, 1974). However the pattern of change is seasonal rather than diurnal for deeper lakes. Some of the water bodies do not stratify and never develop an anaerobic hypolimnion for example Lake Manyame (Coterill and Thornton, 1985).

Masundire (1992) observed that Mazvikadei reservoir showed no apparent thermal stratification during its filling phase which is a common feature of tropical African reservoirs. After 17 years since his study and with the on-going climatic changes, there could be changes in the stratification regimes of the reservoir. Hence the objective of this aspect of the study was to study thermal and oxygen stratification regimes of Mazvikadei Reservoir.

2.2 Materials and Methods

Sampling was carried out in the dam monthly from May to October 2015 during the last week of each month. Measurements were done from two sampling points, site 28 which was in the deep area and had a depth of 32 m and a shallow site 28 with a depth of 6 m. The seasonal pattern of stratification was determined by measuring temperature and dissolved oxygen at 1 m intervals at the two sites in the dam using a water quality meter probe (WTW Oxi 330) that measures both parameters.

2.3 Results

Thermal stratification was established weakly in the reservoir in July but broke down completely during the other months of sampling (Figures 2.1-2.7).The shallow water site did not show any apparent thermal stratification throughout the sampling period as compared to the deeper site. Imboden and Wuest (1995) define the thermocline as the water layer in which water temperature decreases rapidly with increasing depth. For the purposes of this study the thermocline was defined by a drop of 2 ^{o}C in the first 5 m. In line with this definition it can be noted from Fig 2.3 for the thermal profile of site 28 in July that there was the highest drop in temperature of 2.9 ^{o}C in the first 5 m. This was then followed by the months of August and September which had a drop of 1.8 and 1.7 ^{o}C respectively. The lowest drops in temperature in the first 5 m were observed for the months of May and June in the cool dry season. The temperatures dropped to 1.4 and 1.2 ^{o}C in May and June respectively. The mean profile temperature for the deep water site was highest in May and low from June- August during the cool dry season and rising again at the beginning of the hot dry season in September. The mean profile temperatures for the shallower site were quite high for all the months except in July where it dropped to 19.8 ^{o}C (Table 2.1). The mean surface temperatures were high in May ranging from 21-24.6 ^{o}C. Temperatures decreased from July-August within a range of 18.8-22.7 ^{o}C. The temperatures increased again from September-October to a range of 19.4-26.3 ^{o}C.

The oxygen stratification patterns during the period of study were generally similar to that of thermal stratification (Figures 2.1-2.7). Oxygen concentrations at the beginning of the cool-dry season were as low as 2 mgl^{-1} at the bottom in May but from June to September at 5 m depth the oxygen concentrations reached up to >7 mgl^{-1} at the surface and <5 mg^{-1} at the bottom. The mean profile oxygen concentrations for the deep site and the shallow site followed the same pattern with high concentrations from July-September and low in May-June (Table 2.2).

Table 2.1 Mean profile temperatures (± SD) of site 28 (deep water) and site 31 (shallow water).

Month	Mean profile temperature (^{o}C)	
	Site 28 (deep water)	Site 31 (shallow water)
May	23.05± 0.68	23.13± 0.64
June	19.98± 0.59	20.10± 0.12
July	19.16± 0.66	19.80± 0.73
August	19.93± 1.13	21.62± 0.90
September	21.32± 1.34	23.08± 0.66

Table 2.2 Mean profile dissolved oxygen concentrations (± SD) of site 28 (deep water) and site 31 (shallow water)

Month	Mean profile dissolved oxygen concentrations(mgl^{-1})	
	Site 28 (deep water)	Site 31 (shallow water)
May	3.99±0.39	4.92±0.08
June	5.21±1.01	5.98±0.39
July	5.68±0.23	7.65±0.37
August	5.91±0.69	7.30±0.91
September	5.53±1.11	6.76±0.16

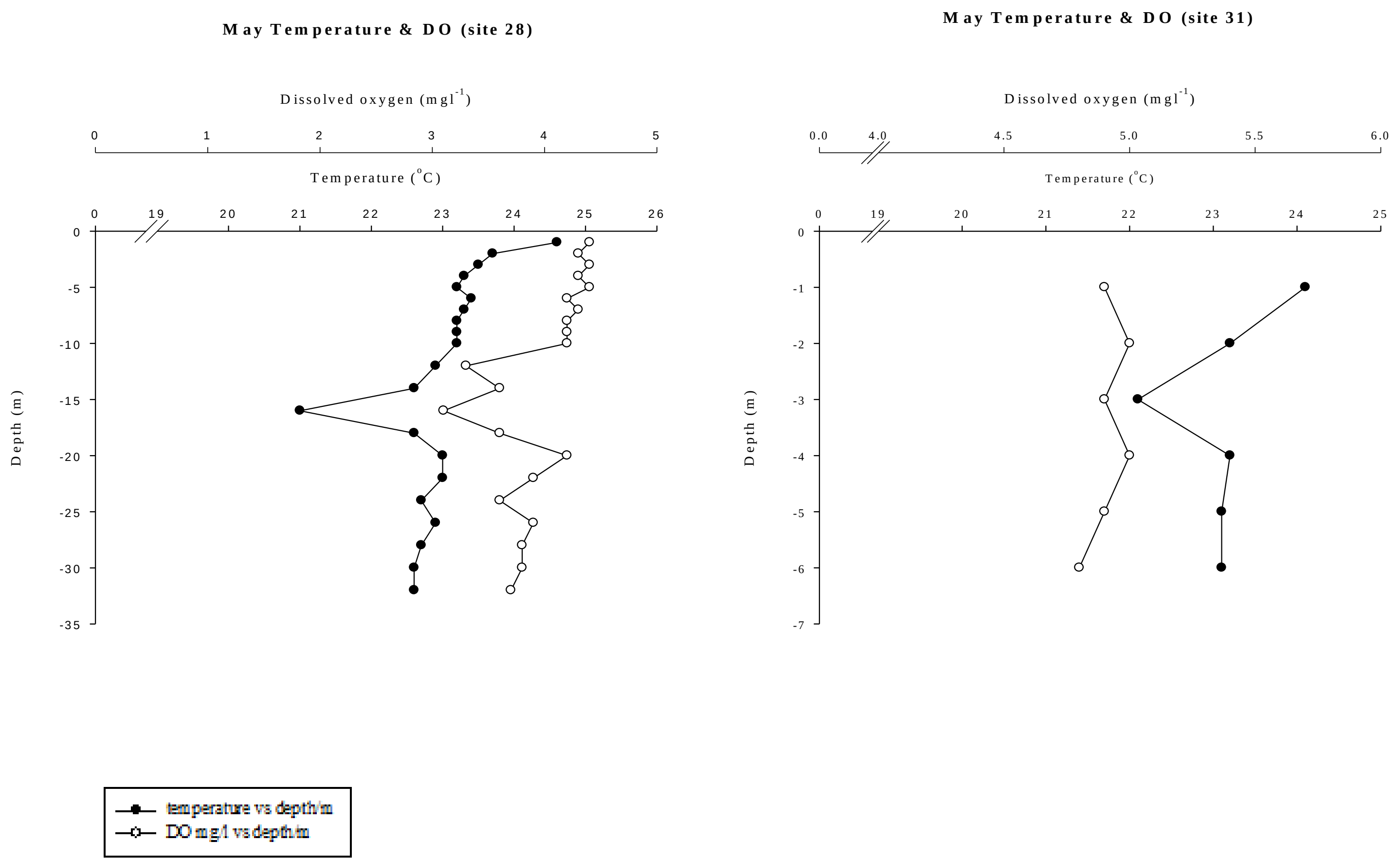


Figure 2.1 Dissolved oxygen and temperature profiles for Sites 28 and 31 (May 2015)

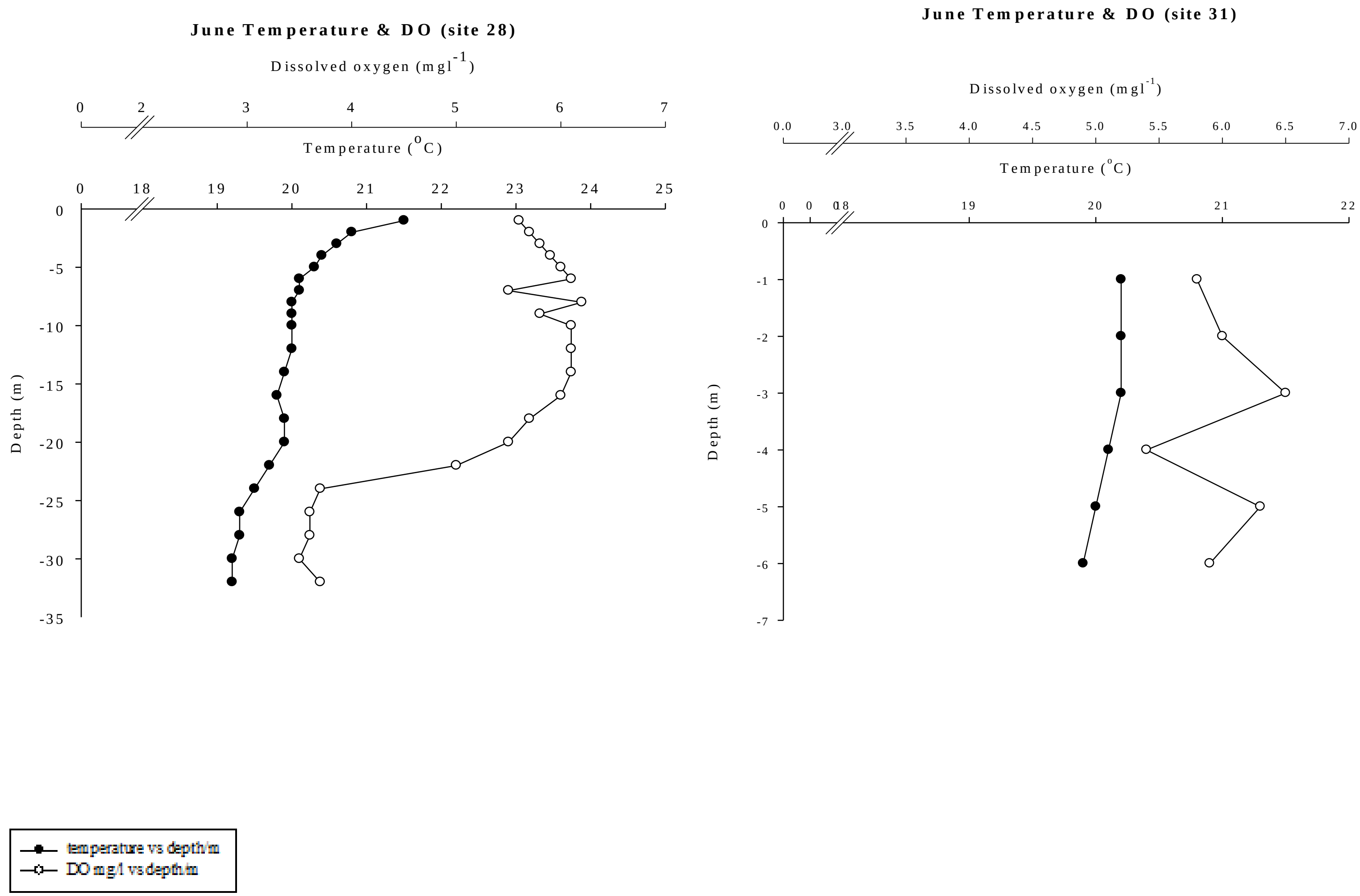


Figure 2.2Dissolved oxygen and temperature profiles for Sites 28 and 31 (June 2015)

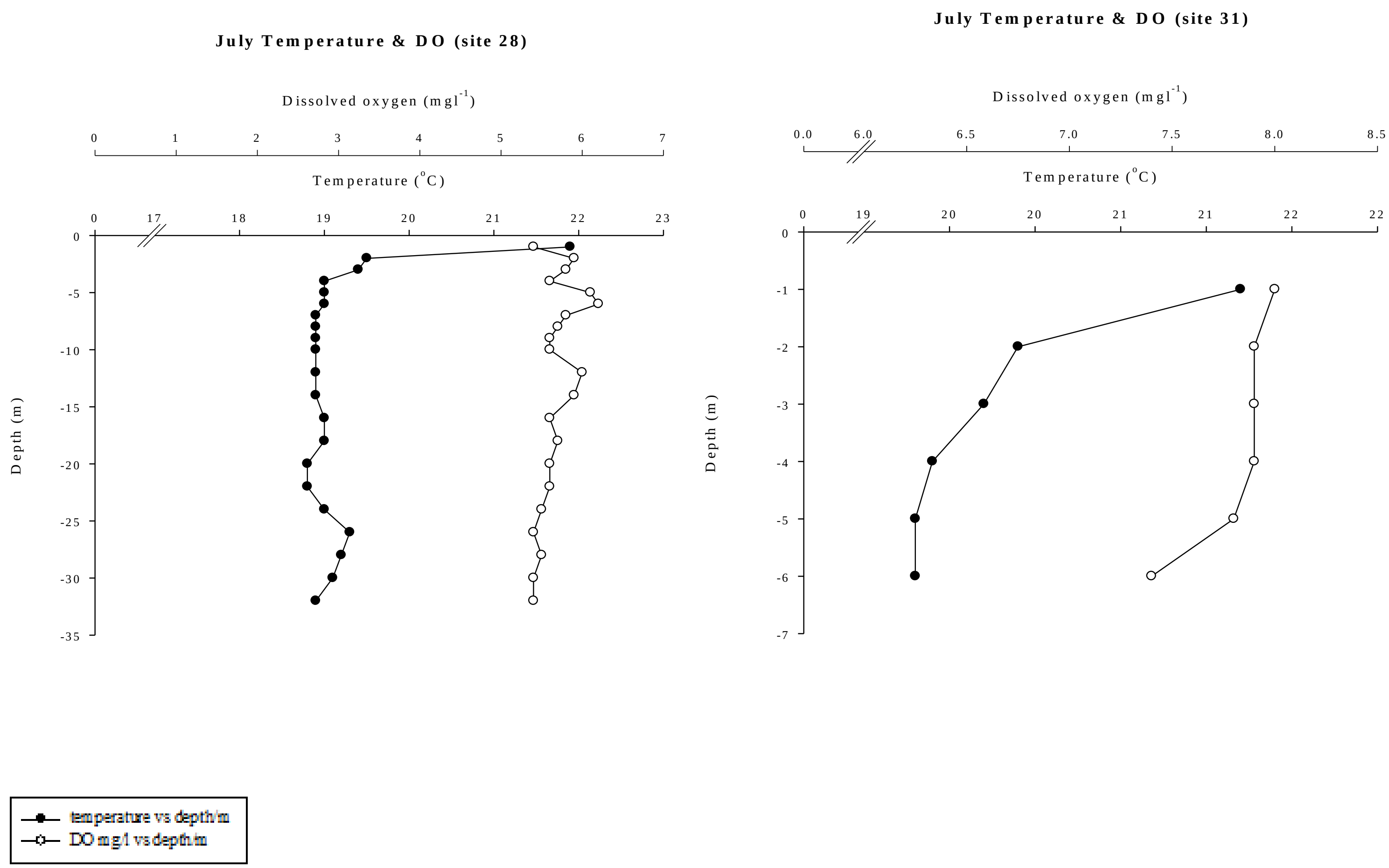


Figure 2.3Dissolved oxygen and temperature profiles for Sites 28 and 31 (July 2015)

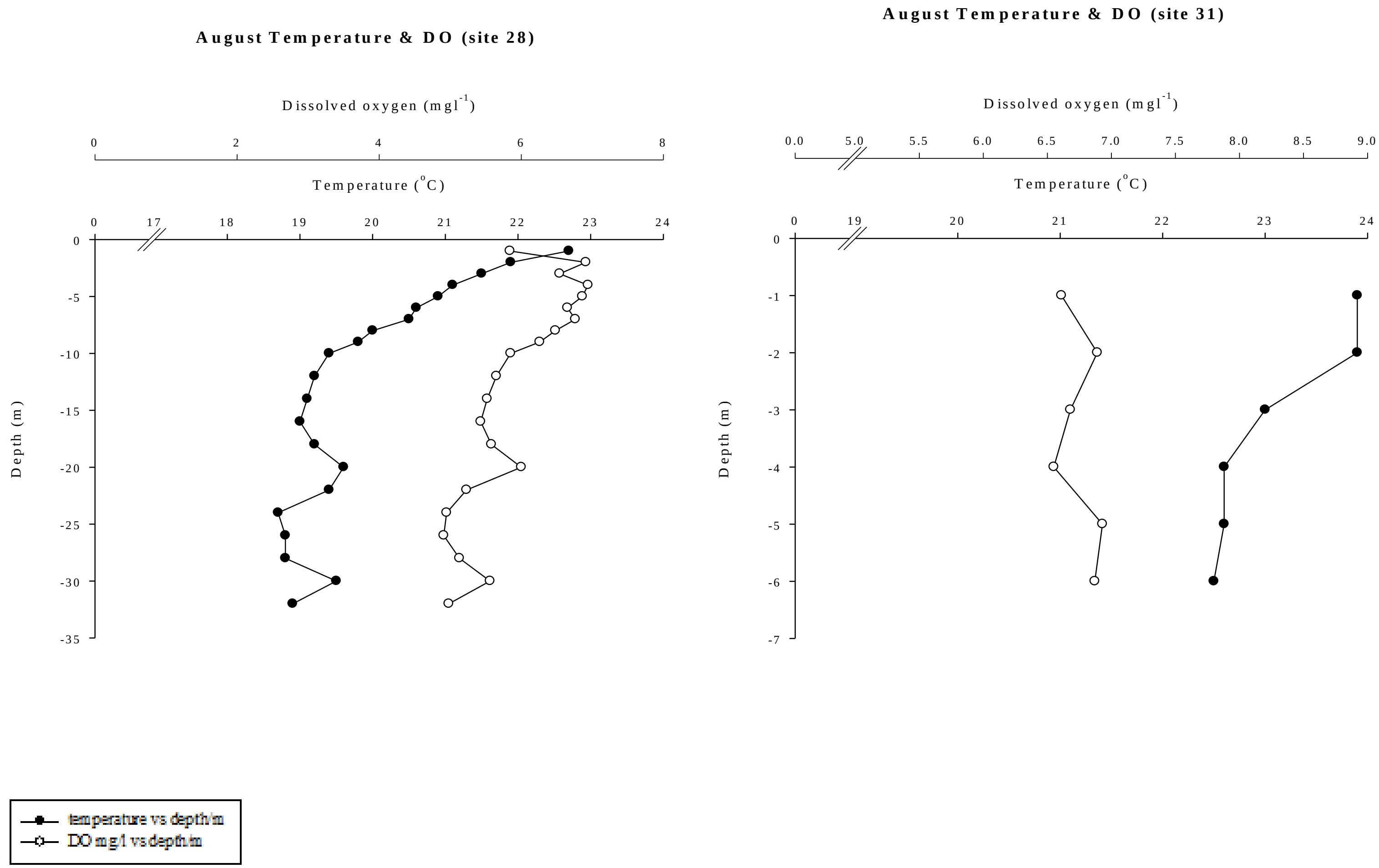


Figure 2.4Dissolved oxygen and temperature profiles for Sites 28 and 31 (August 2015)

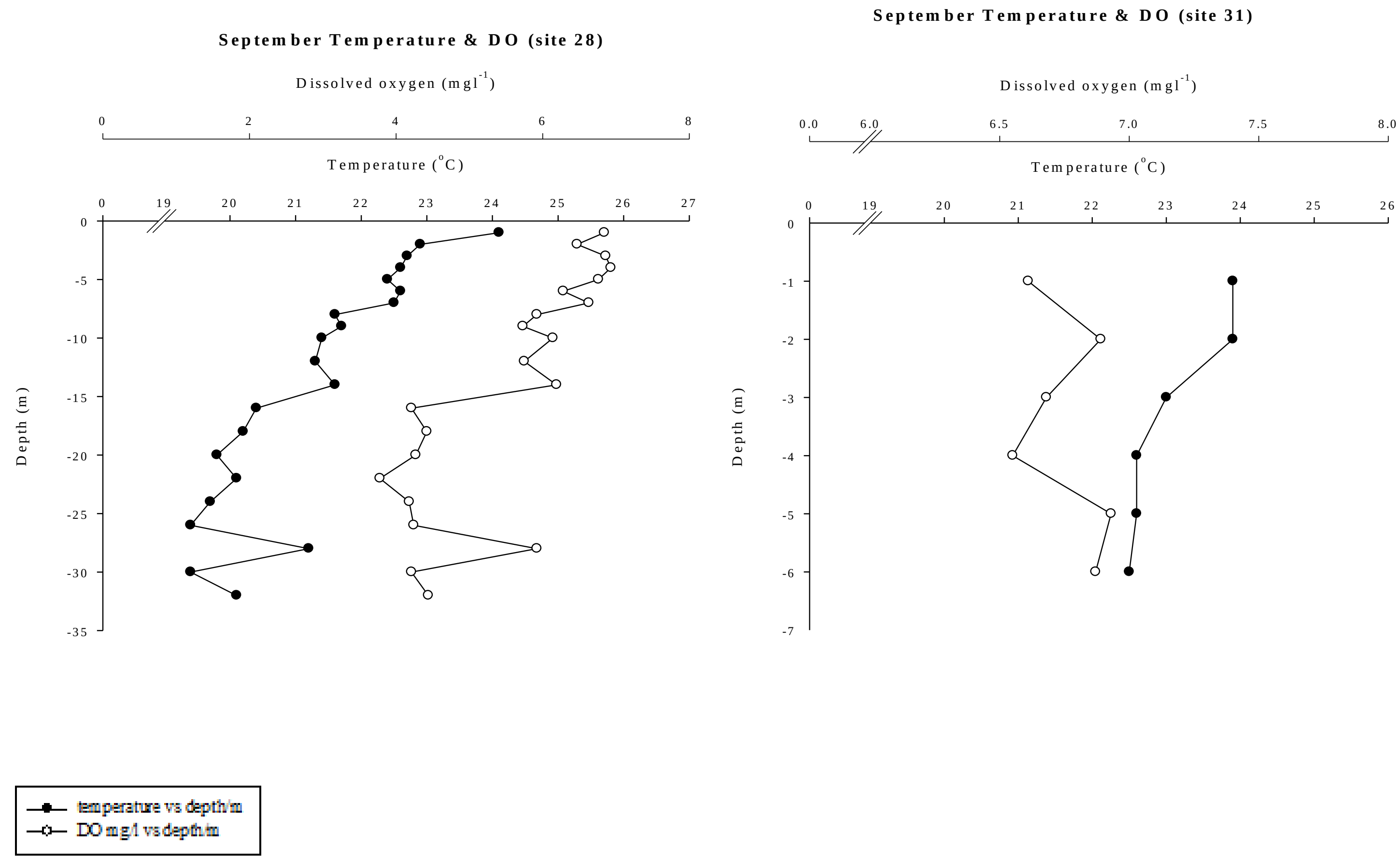


Figure 2.5Dissolved oxygen and temperature profiles for Sites 28 and 31 (September 2015)

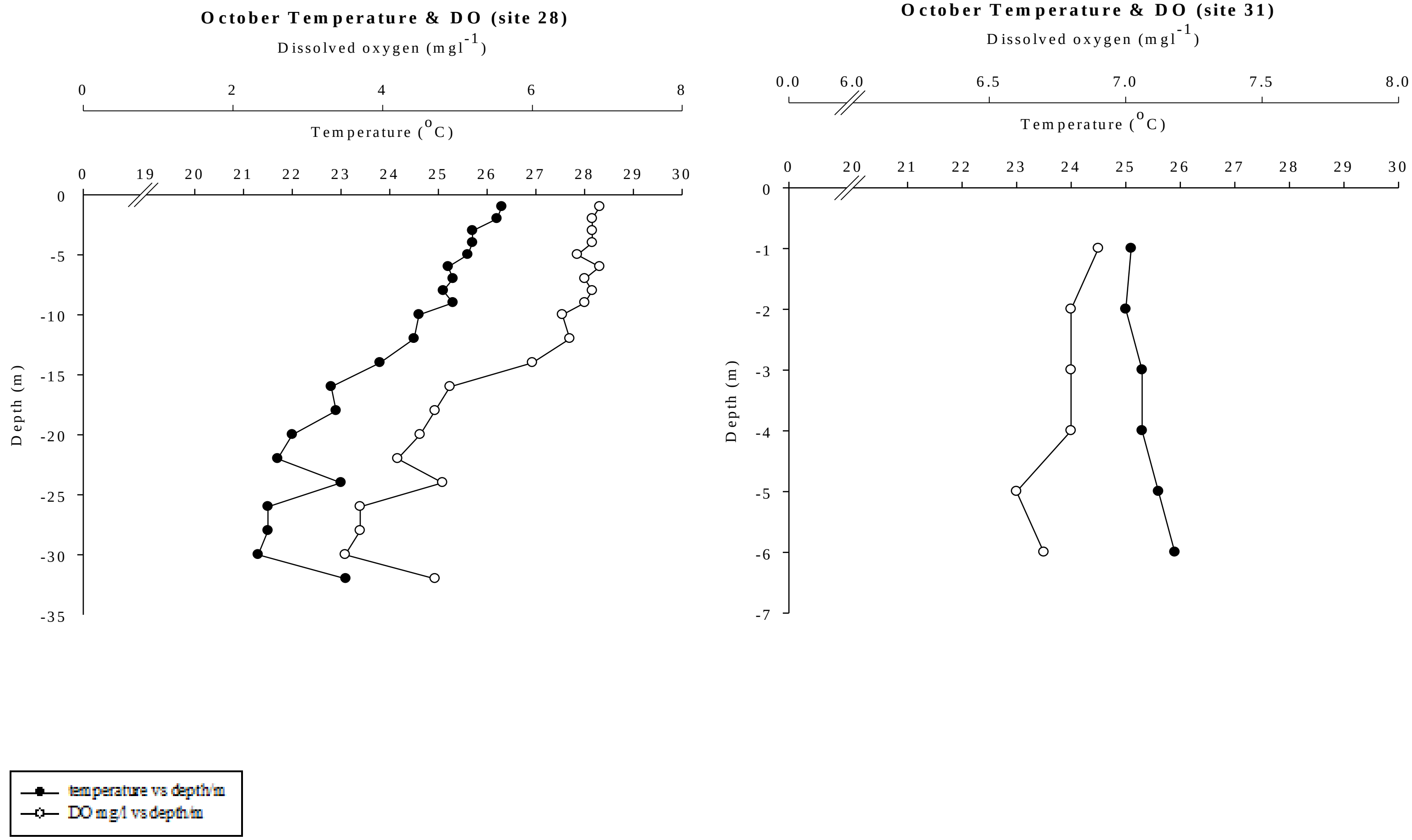
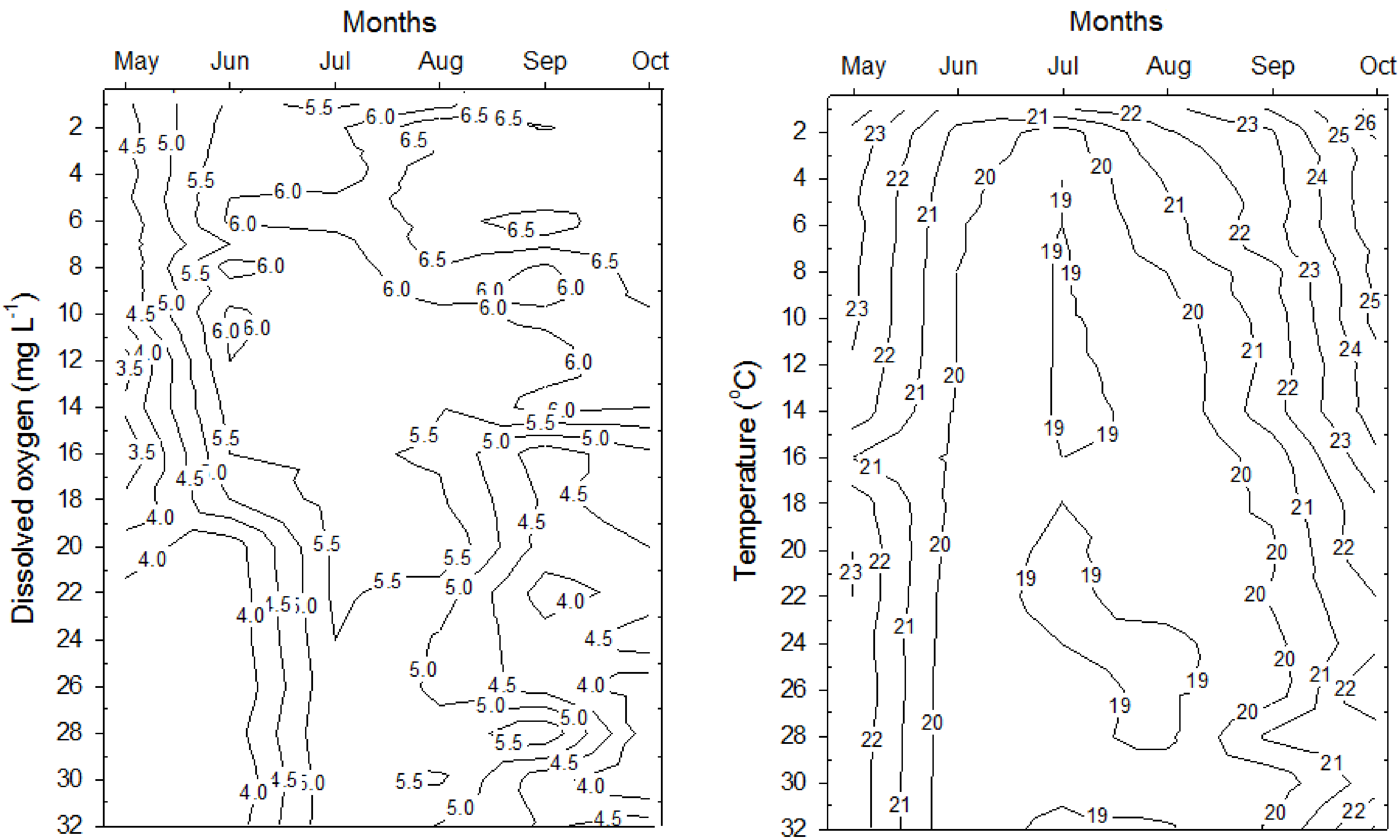


Figure 2.6 Dissolved oxygen and temperature profiles for Sites 28 and 31 (October 2015)



a) b)

Figure 2.7 Monthly variations in a) dissolved oxygen (mgl^{-1}) and b) water temperature (°C) in Mazvikadei reservoir (May – Oct 2015)

2.4 Discussion

Mazvikadei dam did not show seasonal thermal stratification patterns similar to other small and medium-sized dams (Nhiwatiwa and Marshall, 2006; Dalu *et al*., 2013), although it is in the sub-tropical region. These reservoirs stratify during the hot dry season but the stratification breaks down completely in the cool dry season during the turnover period. In July when the reservoir did show weak stratification it was only established in the first 5 m after that the stratification was not so distinct. This is the same pattern that was observed by Masundire (1992). It was observed that the reservoir did not show any apparent stratification but showed an unstable thermocline in August which is was the same period that a thermocline was observed for this study. The thermocline observed in July may have been due to intense surface heating as surface temperatures were as high as 24.5^{o}C at some sites. This lack of stratification observed in Mazvikadei reservoir depends on various other factors that may be at play. Medium-sized water bodies such as Mazvikadei are more likely to be affected by wind and other external factors that would not have such a great effect on smaller water bodies (Nhiwatiwa and Marshall, 2006).

The short-term fluctuations observed in the thermal profiles may have been due to partial mixing in the water column induced by the wind which is more frequent in tropical lakes (Wetzel, 2001). The thermal stratification of a reservoir also depends on the maximum depth and surface area of the reservoir, where it is likely to be more pronounced in large, deep reservoirs (Gorham and Boyce, 1989). Mean depth relates the volume and surface area of a reservoir. Tropical reservoirs in Zimbabwe which have shown seasonal thermal stratification patterns were much smaller than Mazvikadei (2300 ha surface area, 58 m maximum depth) and also had a lower maximum depth. These include Malilangwe reservoir (211 ha surface area, 14 m maximum depth) (2300 ha surface area, 58 m maximum depth) and the smaller Munhwahuku dam (5.7 ha surface area, 5 m maximum depth) (Nhiwatiwa and Marshall, 2006; Dalu *et al*., 2013). Although Mazvikadei had almost the same surface area as Lake Chivero (2632 ha) (Mhlanga *et al*., 2006), Mazvikadei had a higher maximum depth of 58 m about twice that of Lake Chivero (27 m). The high maximum depth is common to large reservoirs such as Lake Kariba (Balon and Coche, 1974) the interaction of both the surface area and high maximum depth could have affected the stratification regime observed in the reservoir. The shallower areas did not stratify at all as they are more prone to wind induced mixing. Moderate summer or winter breezes can keep the shallower areas completely mixed throughout the year.

It is expected that once tropical water bodies stratify deoxygenation of the hypolimnion occurs rapidly as organic matter is broken down (Dalu *et al*. 2013). However unlike most tropical reservoirs that exhibit pronounced deoxygenation during the stratification period, Mazvikadei reservoir was well oxygenated throughout the water column during the entire sampling period except in May when the reservoir showed anoxic conditions only at the bottom. This may be attributed to the deeper waters receiving less light during that period because of the high phytoplankton biomass caused by the inflow and therefore reducing phytoplankton productivity at the bottom. Organic matter carried by the Mukwadzi river with the inflow may have been deposited at the bottom of the reservoir and this provided energy for bacteria. The bacteria consumed oxygen during decomposition. This coupled with the weak stratification observed that would not allow the water to mix led to the gradual depletion of oxygen in the deeper waters and deoxygenation at the bottom. The inflow may also have boosted productivity within the reservoir and the reservoir underwent a temporary deoxygenation phase.

During the filling phase of Mazvikadei dam, Masundire (1992) observed that during May the whole water column was anoxic and the smell of hydrogen sulphide was quite distinct. This could be attributed to the decomposition of large quantities of organic matter that were recently flooded by the rising water during the filling phase of the reservoir. Bacterial decomposition resulted in the accumulation of hydrogen sulphide in the bottom waters then. In contrast, in this study the smell of hydrogen sulphide was not detected. This suggests that the reservoir has matured since the filing phase and the extent of deoxygenation are now less severe. The rate of decomposition processes have also decreased with time as the reservoir has aged.

A sharp decline in the temperature was observed at a depth of about 15 m in May. This could have been caused by an inflow of a cold water current from Mukwadzi river emanating from late season rains. During the wet season, the river inflow is usually colder than the reservoir waters (Wetzel, 2001). This inflow varies with the flow rate and when it is high, the thermal profile may be altered (Lindim *et al*., 2010), as was observed in Mazvikadei.

During this study the maximum temperature recorded was 26.3 ^{o}C, which was higher than the 25 ^{o}C observed by Masundire (1992). This suggests a warmer year in Zimbabwe but without enough data it is difficult to accurately determine by what degree the reservoir has warmed up. Warming is an emerging issue on which further information should be generated. Warming in southern Africa has only been recorded in Hartbeespoort and Roodeplaat dams in South Africa (van Ginkel and Silberbauer, 2007) and in Lake Kariba (Mahere *et.al*., 2014, Ndebele-Murisa *et al*., 2014). Warming in large African lakes and reservoirs has been reported in Lakes Tanganyika (O'Reilly *et al*. 2003, Verburg *et al*. 2003), Victoria (Sitoki *et al*., 2010, Marshall *et al*., 2013), Albert (Lehman *et al*., 1998), Malawi (Vollmer *et al*., 2005), and Kivu (Lorke *et al*., 2004). The Great Lakes such as Tanganyika and Victoria have also had studies that have shown that indeed global warming is a reality for lake ecosystems. It is however important to note that in this study, the temperature ranges recorded from May-October in Mazvikadei reservoir, were similar to those measured in a much warmer region of the country (south east Lowveld) by Dalu *et al*. (2013). It could be that the dry season was exceptionally warm and hence the higher water temperatures, but yet again it could be an indication of an irreversible trend towards lake warming. This is a subject area which can be explored in future studies. Sampling was only done for six months from May-October 2015, this was a limitation in that the information obtained was not sufficient to fully understand the patterns of the reservoir.

CHAPTER THREE

3 PHYSICO-CHEMICAL LIMNOLOGY AND PLANKTON DYNAMICS OF MAZVIKADEI, AN AFRICAN TROPICAL RESERVOIR.

3.1 Introduction

3.1.1 Factors affecting physicochemical characteristics of reservoirs

Understanding the limnology of reservoirs is necessary in order to predict the effects of human activities on water quality, biodiversity, and ecological responses to human activities and fish production. The information is also useful in understanding the patterns observed in both tropical and temperate systems as a function of size and climatic gradients (Bootsma and Hecky, 1993). Water quality refers to the chemical, physical, biological and radiological characteristics of water. It is a measure of the condition of water relative to the requirements of one or more biotic species and to any human need or purpose (Johnson *et al*., 1997). The physico-chemical composition of surface and underground waters is dependent on natural factors (geological, topographical, meteorological, hydrological and biological) in the drainage basin and varies in response to seasonal differences in runoff volumes, weather conditions and water levels (Awoyemi *et al*., 2014). The use of the physico-chemical properties of water to assess water quality provides an indication of the status, productivity and sustainability of a water body. The changes in physical characteristics like temperature, transparency and chemical elements of water such as dissolved oxygen, chemical oxygen demand, nitrate and phosphate provide valuable information on the quality of the water, the sources of the variations and their impacts on the functions and biodiversity of the reservoir (Mustapha, 2008).

An important factor that influences water quality in relatively still, deep waters, such as lakes and reservoirs, is stratification. Stratification occurs when the water in a lake or reservoir is separated into two layers with different densities. It is most commonly caused by temperature differences, leading to differences in density but occasionally by differences in solute concentrations. Water quality in the two layers of a stratified reservoir is subject to different influences (Elci, 2008). The surface layer receives more sunlight while the lower layer is physically separated from the atmosphere (which is a source of oxygen) and may be in contact with decomposing sediments which exert an oxygen demand. As a result of these influences it is common for the lower layer to have a significantly decreased oxygen concentration compared with the upper layer. When anoxic conditions occur in bottom sediments, various compounds may increase in interstitial waters through dissolution or reduction and diffusion from the sediments into the lower water layer. Substances produced in this way include ammonia, nitrate, phosphate, sulphide, silicate, iron and manganese compounds (Elci, 2008; Imberger, 1998).

Human intervention in aquatic ecosystems also has significant effects on water quality. Some of these effects are the result of hydrological changes, such as the building of dams, draining of wetlands and diversion of flow. The most obvious is pollution through the discharge of domestic, industrial, urban and other wastewaters into the water-course and the contamination from chemicals emanating from agricultural land within the drainage basin (Ibe and Agbamu, 1999). The degree to which the impact of human activities on water quality disrupt the ecosystem and restrict water use are both widespread and varied (Peters and Meybeck, 2000). In developing countries eutrophication is a major threat to aquatic ecosystems into which the sewage effluent is discharged. Eutrophication results from point sources, such as wastewater discharges with high nutrient loads (principally nitrogen and phosphorus), and from diffuse sources such as run-off from livestock feedlots or agricultural land fertilized with organic and inorganic fertilizers (Adekunle, 2009). During its filling phase, Masundire (1992) postulated an increase in productivity in

Mazvikadei reservoir in response to an increase in agricultural activities in particular crop production and the use of artificial fertilizers within the catchment. This study assessed the physico-chemical parameters of the reservoir in order to determine whether the trophic status of the reservoir had changed since its filling phase.

3.1.2 Use of plankton dynamics in assessing water quality

Trophic changes in aquatic ecosystems can be monitored using water quality as an indicator. The indicators that can be used affect the structural and functional properties of the ecosystem (Gulati, 1983; Beak, 1965). Phytoplankton have been used as biological classification tools for lakes and slow flowing waters since the 19th century (Teiling, 1955). They are fundamental components of the aquatic food web since they act as primary producers of organic matter, oxygen producers, and food resources for grazers and compartments for the microbial loop. They consequently respond first to water quality changes thereby initiating a chain reaction which is successively reflected within other functional groups of the food web (zooplankton, benthic fauna, fish and birds) (Whitton and Patts, 2000).

Zooplankton structure and grazing activity can be used as an indicator of water quality. Zooplankton form an important link in the food web of aquatic systems because of their central position between autotrophs (algae and macrophytes) and other heterotrophs (fish and other carnivores) (Gulati, 1983). Their strategic position in the aquatic food web both in terms of feeding and energy flow, as well as their sensitivity to both man- made and natural changes renders zooplankton suitable for the biological monitoring of water quality. High growth and reproduction rates of zooplankton indicates the good quality of food (Porter *et al*., 1982). Zooplankton have lower densities, larger size and lesser number of species. Their intermediate position between phytoplankton and fish enables zooplankton to respond to changes in food and predation. This can be a useful indicator of the trophic status of conditions in a reservoir (Gulati, 1982). Generally zooplankton community grazing is high and variable in lakes of low trophy but it is low and relatively constant in lakes of high trophy (Porter *et al*., 1982). Filteration rates of *Daphnia* spp may provide information on both the trophic status as well as dissolved substances in lake waters (Gulati, 1982).

The use of phytoplankton for water quality assessment does not, however, represent community structure (Washington, 1984), since the occurrence of an indicator species can reflect either clean or polluted conditions. Other biotic indices are likely to be specific for one particular type of pollution, as indicator organisms cannot be equally sensitive to all types of pollution (Beak, 1965).

3.1.3 Plankton seasonal development

Seasonal succession in plankton is highly known to be predictable and directional but in some instances it may be disturbed by irregular physical events (Sommer *et al*., 1986). It is a process that is repeated annually and changes occur in the community assembly under the influence of various external factors and internal interactions which shape the communities (Sommer *et al*., 1986). Phytoplankton have very short generation times and they also react rapidly to shifts in the environment (Willen, 2000). Changes in the physical or chemical status of the water can cause an alteration of species and their abundances. Long lasting water quality changes may then be reflected among the phytoplankton from one growth season to another (Willen, 2000).

The seasonality and abundance of phytoplankton in tropical African lakes is affected by the stratification of the lake. The general annual cycle of thermal stratification in these lakes includes a short phase of partial mixing which is immediately followed by a peak of algal abundance in response to nutrient release. Diatoms make the major contribution of phytoplankton after mixing.

Chlorophycae (mostly desmids) are most abundant during the rainy season followed in decreasing order of abundance by Cyanophycae, Bacillariophycae, Euglenophycae, Dinophycae, Cryptophycae, Chrysophycae and Xanthophycae. This order changes slightly during the dry season and the abundance of diatoms is greater than that of cyanophytes and dinophytes are more abundant than euglenophytes. This seasonal pattern is found in tropical African lakes such as Kariba, Malawi, Tanganyika and Victoria with the blue-greens dominating in summer and diatoms in winter (Ndebele-Murisa *et al*., 2010).

Some tropical African lakes however have a stable phytoplankton seasonality where there is remarkable stability in algal species composition and abundance in summer and winter in the lake (Zohary, 2004).

3.2 Materials and Methods

3.2.1 Basic water quality measurements

Samples and measurements were done for 8 months (May 2015 to October 2015) in the reservoir from the 8 sites along the reservoir length. A 5 litre Ruttner sampler was used to collect water from the water column at 1 m intervals from the bottom to the surface of the dam. All the samples collected at each depth were then integrated and subsamples were taken for laboratory analysis. At each depth pH and temperature, conductivity and Total Dissolved Solids, dissolved oxygen and percentage of oxygen saturation were measured using a pH meter (WTW PH 340i), conductivity meter (WTW Cond 330i) and an oxygen meter (WTW Oxi 330i) respectively. Water transparency was measured using a 20 cm Secchi disk. The collected water samples were placed in 500 ml polythene bottles and stored in a cooler box before being analyzed within 24 hrs in the laboratory.

Biological Oxygen Demand was measured by incubating the water samples which had been artificially saturated with oxygen in dark bottles for 5 days at 25 ^{o}C. Dissolved oxygen was measured with an electric probe (WTW Oxi 330i) and BOD was calculated as the difference between the initial oxygen level when the water was saturated and the oxygen level after 5 days of incubation. Other chemical variables (nitrates, ammonia and reactive phosphorus) were measured with a spectrophotometer using filtered water samples (Whatman GF 47 mm filters) except for total nitrogen, total phosphorus and chemical oxygen demand. The summary of each method is given below.

a) Chemical oxygen demand

Chemical Oxygen Demand was determined by the reactor digestion method (EPA Method 410.4). Concentrated sulphuric acid provided the primary digestion catalyst with mercury ions in the digestion solution being used to complex chloride interferences. The secondary catalyst, silver sulphate, assists in the oxidation of straight chain hydrocarbons. The samples were heated for 2 hrs with a strong oxidizing agent, potassium dichromate with the heat from the digestion block (150 ^{o}C) also acting as a catalyst. The dichromate ion (Cr^2O7^{2-}) was reduced to Cr^{3+} and the amount of Cr^{6+} was determined by measuring at a wavelength of 420 nm. Precision was at the standard deviation of ± 2.7 mgl^{-1} COD.

b) Nitrate-nitrogen

Nitrate-nitrogen was determined by the cadmium reduction method, EPA method 353.2. Nitrate in the sample is reduced to nitrite in the column containing cadmium which has been treated with copper. The Cd^{2-} ions released in this way are simultaneously bonded to a complex with the help of ethanolamine in order to prevent the formation of $Cd(OH)_2$ and possibly also $CdCO_3$ which interferes with the efficiency of the reductive column. The nitrite is determined by diazotizing with sulphanilamide and coupling with N-1-Napthylethylenediamine to form a pink colored azo dye. Absorbance was measured at 545 nm and precision was at the standard deviation of 0.01mgl^{-1} NO^{3-}.

c) Ammonia

Ammonia was determined by the EPA method 350.1. Monochloramine (NH_2Cl) and free ammonia (NH_3 and NH_{4+}) can exist in the same water sample. Hypochlorite was added which combines with free ammonia to form more monochloramine. In the presence of a catalyst (sodium nitroprusside) monochloramine in the sample reacts with a substituted phenol to form an intermediate monoimine compound. The intermediate then couples with excess substituted phenol to form green indophenols which is proportional to the amount of monochloramine present in the sample. The measurement wavelength used was 635 nm. Precision was at a standard deviation 0.01 mgl^{-1} NH_3-N.

d) Total nitrogen

Total nitrogen was determined by the cadmium reduction method. An alkaline persulphate digestion converts all forms of nitrogen to nitrate. The nitrate was determined by reduction with copperised cadmium. Measurements were taken at wavelength 545 nm and precision was at a standard deviation of 0.01 mgl^{-1} NO^{3-}.

e) Reactive phosphorus

Reactive phosphorus was determined by the USEPA PhosVer 3 method 8048. Orthophosphate reacts with molybate in an acid medium to produce a mixed phosphate/molybate complex.

Ascorbic acid then reduces the complex giving an intense molybdenum blue color. Absorbance was measured at 882 nm and the precision was at a standard deviation of ± 0.02 mgl^{-1} PO_4^{3-}.

f) Total phosphorus

Total phosphorus was determined by the USEPA PhosVer 3 with acid persulphate digestion method 8190 using unfiltered water samples. Phosphates present in organic and condensed inorganic forms (meta-, pyro-, or other polyphosphates) had to be converted to reactive orthophosphate before analysis. Pretreatment of the sample with acid and heat provided the conditions for hydrolysis of the condensed inorganic forms. Organic phosphates are converted to orthophosphates by heating with acid and persulphate. Orthophosphate reacts with molybdate in an acid medium to produce a mixed phosphate/ molybdate complex. Ascorbic acid then reduces the complex giving an intense molybdenum blue color. The measurement wavelength was 882 nm and the precision was at a standard deviation of ±0.06 mgl^{-1} PO_4^{3-}.

e) Chlorophyll a (Acetone extraction)

Subsamples of 200 ml were filtered using Whatman GF 47 mm filter paper. Pieces of the filter were placed into a 10 ml centrifuge tube and 10 ml of acetone was added. The samples were placed in darkness for 24 hrs for extraction. After incubation the centrifuge tube was shaken vigorously to ensure homogenous distribution and then centrifuged for 10 minutes at approximately 2000 ms^{-2}. The absorbance of the final extract was measured at a chlorophyll *a* absorption maximum (665 nm) and correction for turbidity was made by measuring the absorption at 750 nm, where chlorophyll *a* is non-absorbing. Calculations were done using the formula:

Chlorophyll *a* concentration (mgl^{-1}) = $\frac{(\text{Abs } 665 - \text{Abs } 750) \times e \times 10\,000}{87.67 \times V \times l}$

e =volume of the extract

l =the length of the cuvette (100 mm)

V =filtered volume (0.2 litres)

87.67=the absorption coefficient for (expressed in litres $g^{-1}cm^{-1}$)

3.2.2 Plankton sampling

Sampling was carried out monthly in the dam from May- October 2015 from the 8 sampling sites along the reservoir length. Water samples were collected at 1 m depth intervals from the surface to the bottom and all the samples collected were then integrated in a container and then subsamples of zooplankton and phytoplankton were taken using 64 µm and 20 µm mesh nets respectively of diameter 40 cm. The concentrated water samples were then collected in small 250 ml bottles that were labeled. A preservation solution of 70% alcohol was added to the sample bottles with zooplankton and Lugol's iodine solution was added to the bottles with phytoplankton for fixing purposes. The samples were taken to the laboratory for identification and abundance estimation (counting) under an inverted microscope (OLYMPUS CKX41) (Figure 3.1).

The density of phytoplankton and zooplankton was determined by counting the numbers present in 25 ml subsamples from each site and then the numbers per litre were estimated. Identification of taxa was done using dichotomic identification keys presented by Fernando (2002) and Lund and Lund (1995). Indices of diversity were calculated for phytoplankton and zooplankton using PAST version 2.0.

Figure 3.1 Inverted microscope used in plankton identification.

3.2.3 Data analysis

Kruskal-Wallis tests ($p<0.05$) were done to test for differences in physicochemical characteristics a) between sites (H_o = there is no difference between the 8 sites) and b) between months (H_o = there is no difference between the six months of sampling from May-October 2015) for each of the physicochemical variable. The computer software STATISTICA version 7 was used.

3.2.3.1 Principal Component Analysis

Prior to the multivariate analysis, all physicochemical variables except pH (already a logarithm) were log-transformed to reduce skewness in the data. The datasets were consistently linear in nature (Detrended Canonical Analysis). Therefore, Principal Component Analysis (PCA) was done in order to get an overall assessment of the possible relations among the limnological variables measured at different sites and times in Mazvikadei reservoir.

3.2.3.2 Redundancy Analysis

Prior to all multivariate tests, the data was tested for linearity with Detrended Correspondence Analysis (DCA) (Lepš and Šmilauer, 2003). Zooplankton and phytoplankton community datasets were again consistently linear in nature (Detrended Canonical Analysis). The first step in the Redundancy Analysis (RDA) was to first identify significant explanatory variables (environmental factors) using Forward Selection procedures (999 Monte Carlo permutations). Once these variables were identified for both the phytoplankton and zooplankton communities, the selected variables were again applied to the RDA procedure and it was determined with Monte Carlo permutations whether they had a significant effect or not. 54 phytoplankton, 19 zooplankton and 15 environmental variables were used in the analysis. The statistical program Canoco ver. 4.5 was used.

3.3 Results

3.3.1 Water chemistry

Table 3.1 and Table 3.2 below summarise the mean values of the environmental variables measured in Mazvikadei reservoir from May-October 2015. Temperature showed a similar trend for all the sites during the study period (Figure 3.2 a). Temperatures were high in May with ranges of 22.8-23.6^{o}C. Temperatures decreased in June and July during the cool dry season with the lowest temperatures being observed in July with values ranging from 18.78-19.95^{o}C and site 3 having the lowest temperature of 18.78 ^{o}C. Temperatures then increased from August to October. The highest temperatures were recorded in October with a range of 23.6-26.1 ^{o}C.

The pH trend was similar at all the sites during the 6 months of sampling (Figure 3.2 b). There was a slight decrease of ph with depth at all the sites. The average pH in the lake was 8.63. The highest pH values were observed at the start of the sampling period in May ranging from 10.3-12.3. There was a sudden decrease in the pH levels in June to as low as 7.48 at site 28. From July-October there was a slight increase in pH but the water was alkaline with no values of less than 7 being recorded throughout the sampling period.

The average conductivity in the reservoir was 262.6 μScm^{-1} .Conductivity increased from May-August with values ranging from 251.4-267.9 μScm^{-1} but conductivity dropped in September to range between 253.2 μScm^{-1} (site 28)-261.2 μScm^{-1} (site 31) (Figure 3.2 c). This was then followed by a significant increase in October. The highest values were recorded in October when all the sites recorded above 270 μScm^{-1} except for site 31 which had a value of 260.2 μScm^{-1}. There was a general decrease in conductivity with depth.

TDS concentrations followed the same trend as conductivity (Figure 3.2 d). The average TDS concentration in the reservoir was 165.2 mgl^{-1}. It steadily increased from May to July with values ranging from 156.3-165.3 mgl^{-1}. TDS decreased in August to as low as 160.25 mgl^{-1} at site 5. The TDS concentrations increased in September to ranges of 179.94-180.64 mgl^{-1} but decreased again in October to ranges of 162.4-180 mgl^{-1} for 7 of the 8 sites. TDS concentrations for site 31 however remained high in October.

The average Secchi disk transparency in the reservoir was 4.36m. Secchi disk transparency followed a similar trend at all the sites increasing from May-June within a range of 3-4.5m (Figure 3.2 e). The highest transparency values were observed in July when all the sites recorded above

6m. After this the transparency decreased from August-October with the lowest values being observed in October which ranged from 2.9-4m.

Chemical Oxygen Demand (COD) concentrations were low in May for 5 of the 8 sites ranging from 8-31 mgl^{-1}(Figure 3.2 f). The other 3 sites had high COD concentrations in the same month within a range of 58-119mgl^{-1} but these concentrations decreased in June. Overally the COD concentrations in June were low ranging from 3-58 mgl^{-1}. There was an increase in COD concentrations in July with high values of 134 mgl^{-1} (site 3) and 174 mgl^{-1} (site 5) being recorded but most of the sites had low COD concentrations. There was a general decrease in COD concentrations from August-September but at 3 of the sites high COD concentrations of 26-41mgl^{-1} were observed in September. The concentrations then increased for all the sites in October with a range of 9-61mgl^{-1}.

The Biological Oxygen Demand followed almost a similar trend at all the sites fluctuating during the entire sampling period. The average BOD for the entire sampling period was 1.98mgl^{-1}. BOD values were high in May ranging from 2-7 mgl^{-1}(Figure 3.3 a) The BOD then decreased in June to a range of 0.84-5.2 mgl^{-1}. The BOD increased again in July to a range of 3.96-4.93 mgl^{-1}. BOD decreased in August to a range of 0.15-1.14 mgl^{-1}. From September- October the BOD values were almost the same changing slightly within the range of 0.5-2.1 mgl^{-1}.

Dissolved oxygen (DO) and percentage oxygen saturation followed exactly the same trend for all the sites throughout the sampling period as these two variables are correlated (Figure 3.3 b&c). The average dissolved oxygen concentration and percentage oxygen saturation for Mazvikadei reservoir was 5.95mgl^{-1} and 76.93% respectively. The DO concentrations and percentage saturation increased from May-June with ranges of 3.97-6.1 mgl^{-1} and 52.3-76.6% respectively.

The highest DO concentrations were observed in July with some sites showing concentrations of greater than 7 mgl^{-1}. From August-October the concentrations and saturation increased for most of the sites within ranges of 5.4-7.6 mgl^{-1} and 66.7-102.5% respectively. During the same period site 31 showed a decrease in both dissolved oxygen and percentage oxygen saturation. There was gradual decrease in both variables with depth for all the sites.

Table 3.1Changes in environmental variables with site for Mazvikadei reservoir

	site 3		site 5		site 6		site 15		site 17		site 23		site 28		site 31	
Variable	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
AMM(mgl^{-1})	0.0146	0.0099	0.0116	0.0026	0.0107	0.0016	0.0105	0.0007	0.0103	0.0004	0.0104	0.0009	0.0107	0.0004	0.0106	0.0004
RP(mgl^{-1})	0.0207	0.0220	0.0147	0.0220	0.0187	0.0228	0.0314	0.0437	0.0175	0.0225	0.0116	0.0174	0.0199	0.0293	0.0173	0.0269
NITR(mgl^{-1})	0.0210	0.0340	0.0119	0.0095	0.0064	0.0040	0.0092	0.0052	0.0062	0.0041	0.0053	0.0035	0.0115	0.0083	0.0088	0.0070
TP(mgl^{-1})	0.02	0.01	0.06	0.08	0.02	0.01	0.03	0.02	0.09	0.20	0.03	0.02	0.03	0.02	0.24	0.43
TN(mgl^{-1})	0.08	0.07	0.10	0.10	0.08	0.04	0.08	0.04	0.09	0.07	0.09	0.07	0.08	0.04	0.08	0.05
TRANSP(m)	4.27	1.73	4.43	1.25	4.53	1.22	4.22	1.66	4.30	1.49	4.52	1.37	4.62	1.19	4.02	1.16
BOD(mgl^{-1})	1.81	1.53	2.62	1.82	1.81	1.62	1.98	2.09	1.87	1.20	1.66	1.52	2.49	2.51	1.58	1.39
COD(mgl^{-1})	42.17	49.53	50.67	63.13	20.50	19.95	26.50	17.46	28.17	45.49	35.83	41.86	28.33	22.07	19.33	5.50
Chl a (μgl^{-1})	0.0016	0.0026	0.0019	0.0031	0.0027	0.0033	0.0020	0.0033	0.0014	0.0016	0.0014	0.0019	0.0026	0.0043	0.0035	0.0045
pH	8.51	1.34	8.73	1.62	8.89	1.70	8.53	1.32	8.54	1.22	8.80	1.43	8.54	1.41	8.45	1.01
COND(μScm^{-1})	263.04	6.93	262.13	7.45	262.95	6.62	262.74	6.94	262.92	6.11	262.33	7.20	263.21	7.52	261.17	5.55
TDS(mgl^{-1})	165.86	7.32	164.57	8.26	164.45	8.11	165.10	7.85	164.76	8.03	164.12	8.28	165.13	7.67	167.19	10.35
TEMP(	21.37	2.08	21.45	1.86	22.50	2.26	21.34	1.91	21.77	2.05	22.29	2.09	21.20	1.85	21.89	1.65
DO(mgl^{-1})	5.69	0.86	5.61	0.69	6.55	1.08	5.44	0.52	6.09	1.03	6.46	0.99	5.30	0.68	6.48	0.98
% SAT	73.33	10.96	72.47	8.00	84.71	14.68	69.68	6.20	78.44	13.71	84.69	14.43	67.58	8.55	84.52	11.52

AMM = Ammonia, RP = Reactive Phosphorus, NITR = Nitrate, TP = Total Phosphorus, TN = Total Nitrogen, TRANS = Transparency, BOB = Biological Oxygen Demand, COD = Chemical Oxygen Demand, Chl *a* = Chlorophyll *a*, COND = Conductivity, TDS = Total Dissolved Solids, TEMP = Temperature, DO = Dissolved Oxygen, % SAT = Percentage Oxygen Saturation.

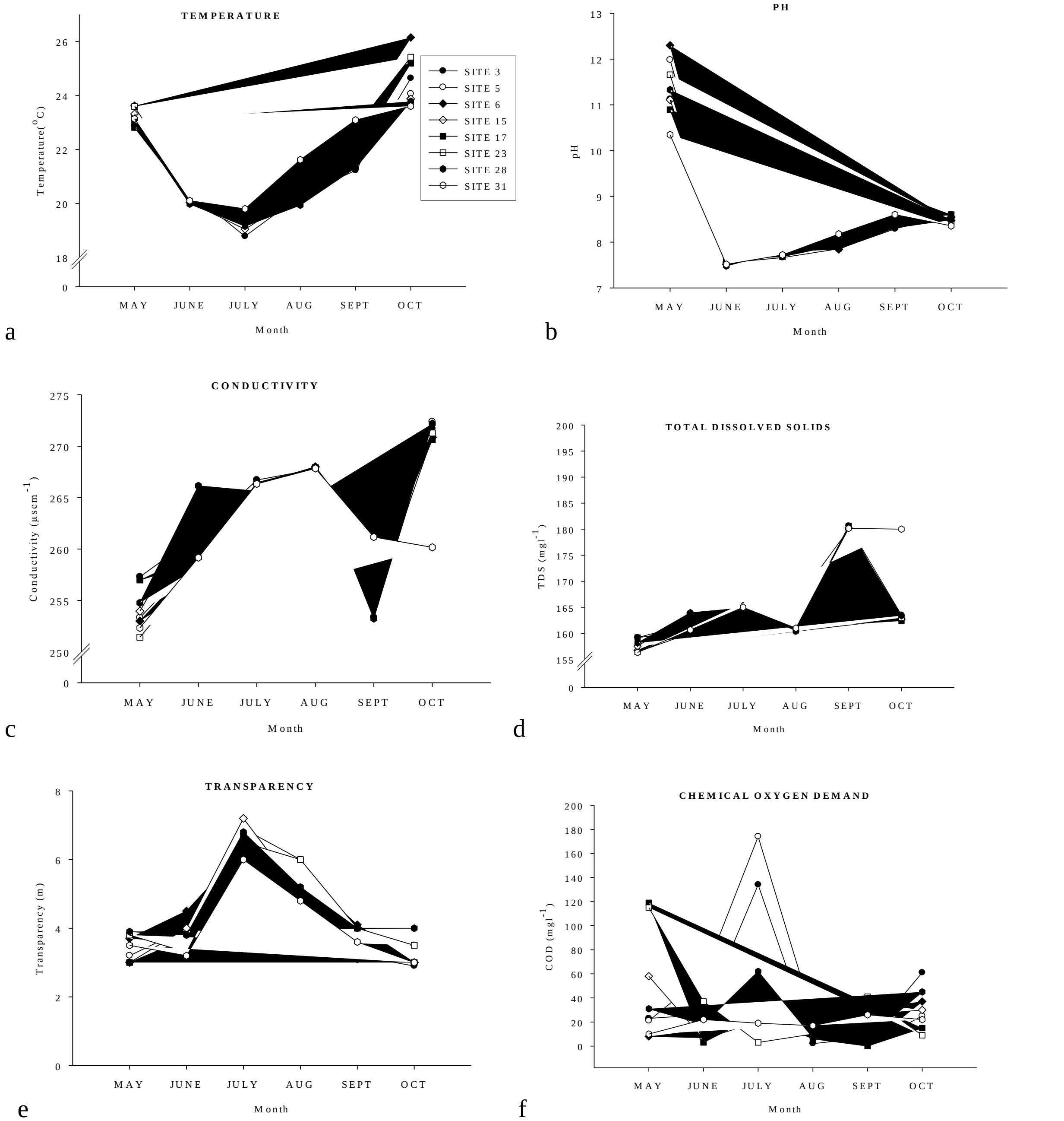


Figure 3.2 Changes in a) temperature, b) pH , c)conductivity, d)total dissolved solids, e)transparency and f)chemical oxygen demand in Mazvikadei reservoir from May-October 2015

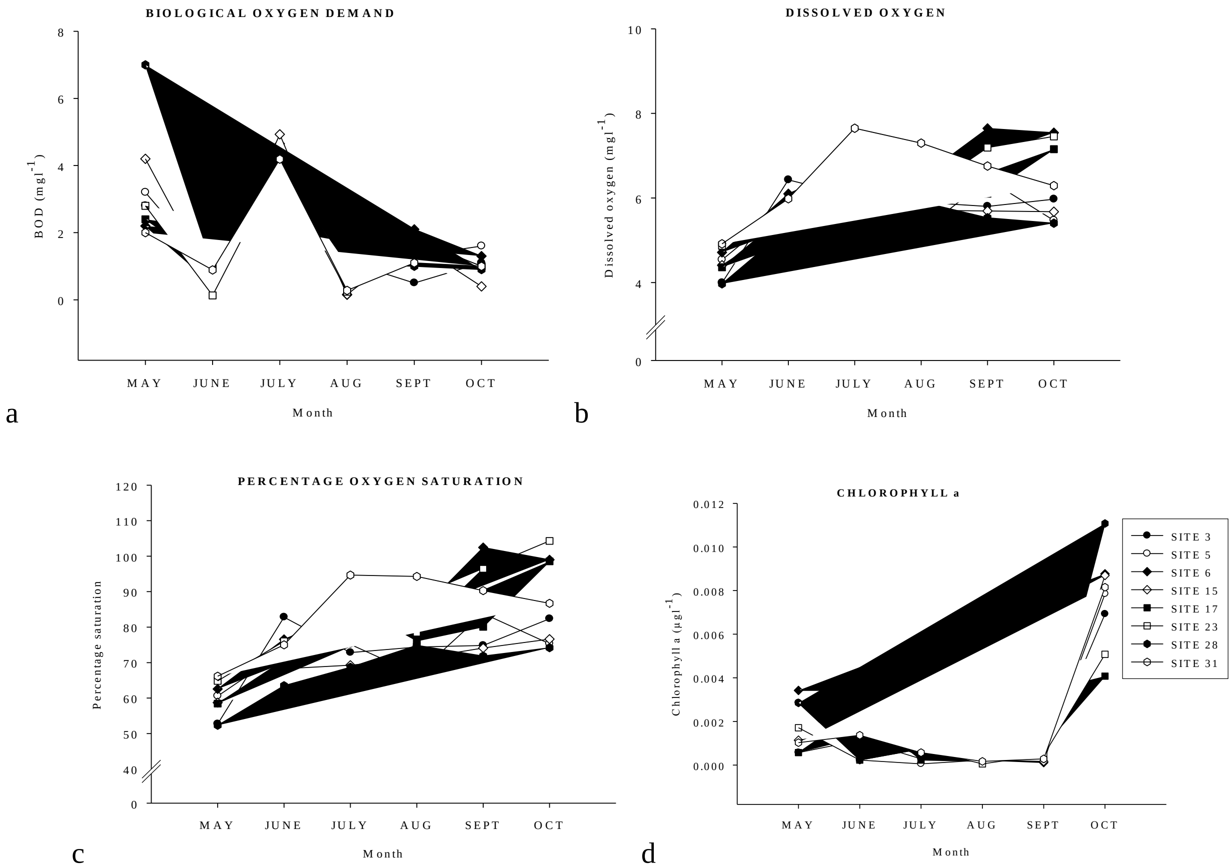


Figure 3.3Changes in a) biological oxygen demand, b)dissolved oxygen , c)percentage oxygen saturation and d) chlorophyll *a* in Mazvikadei reservoir from May-October 2015.

The average chlorophyll concentration in the reservoir was 0.0021 μgl^{-1} Chlorophyll *a* concentrations were high in May with a range of 0.0057-0.01μgl^{-1} (Figure 3.3 d) The concentrations remained almost constant in June except for site 17 which showed an increase and site 28 which showed a decrease. In July the concentrations decreased at all the sites. The concentrations remained low from August-September within a range of 0.000057-0.000855μgl^{-1}. The highest concentrations throughout the sampling period were recorded in October with values of greater than 0.004μgl^{-1}.

3.3.2 Nutrients

The average ammonia concentration in the reservoir was 0.0112 mgl^{-1} Ammonia concentrations were low and almost constant from May-October ranging from 0.009-0.034mgl^{-1}(Figure 3.4 a). However in July there was a sharp increase at site 3 and slight increase at sites 5 and 6.

Reactive phosphorus (RP) was highest in May for all the sites ranging from 0.045-0.12 mgl^{-1}(Figure 3.4 b) The average concentration in the lake was 0.019 mgl^{-1}. The concentrations were almost constant from June to September varying slightly within a range of 0.0005-0.022 mgl^{-1}. In July at site 3 the RP concentration rose unexpectedly to 0.022 mgl^{-1}. The concentrations increased slightly in October to a range of 0.0095-0.023 mgl^{-1}.

The average nitrate concentration in the reservoir was 0.01 mgl^{-1}.Nitrate concentrations rose steadily from May-August within a range of 0.0006-0.028 mgl^{-1}(Figure 3.4 c) The concentrations decreased for most of the sites from September-October to concentrations of 0.0026-0.09 mgl^{-1}. In September nitrate concentration at site 3increased to 0.09 mgl^{-1}.

Total phosphorus (TP) concentrations were low throughout the sampling period but were distinctly high at site 17 in May (0.496 mgl^{-1}) and at site 31 in July (1.112 mgl^{-1}) (Figure 3.4 d). The average TP concentration in the reservoir was0.065 mgl^{-1}.

Total nitrogen (TN) concentrations showed a similar trend at all the sites from May-October (Figure 3.4 e). The average TN concentration in the reservoir was 0.085mgl^{-1}. The highest concentrations were observed in May with a range of 0.152-0.31 mgl^{-1}. These concentrations dropped significantly in June and July to as low as 0.0267 mgl^{-1} at site 28. The concentrations increased slightly from August-October with values ranging from 0.056-0.084 mgl^{-1}.

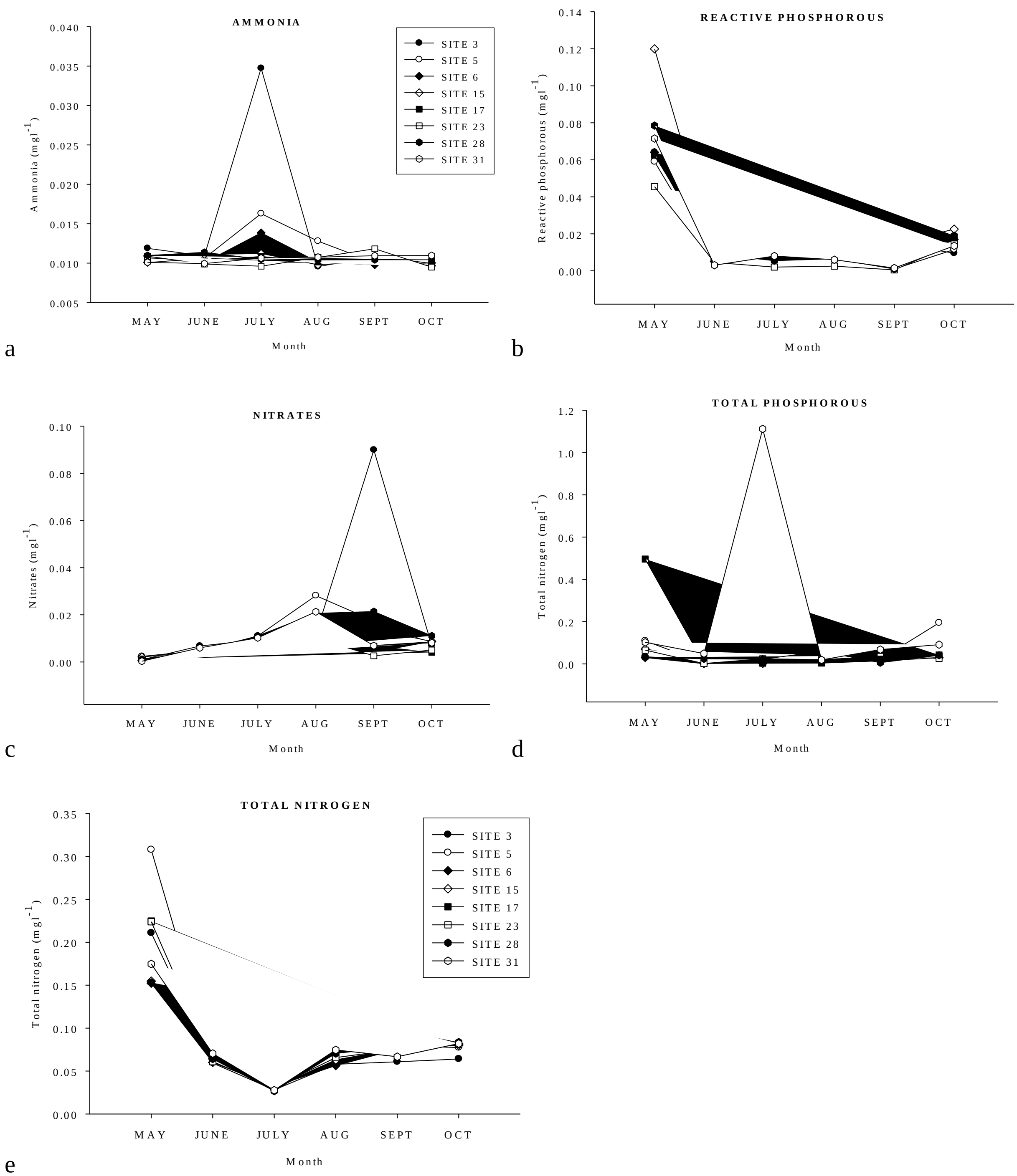


Figure 3.4 Changes in a)ammonia b)reactive phosphorus c)nitrates d)total phosphorus and e)total nitrogen in Mazvikadei reservoir from May-October 2015

Table 3.2 Kruskal-Wallis ANOVA test for significant differences in physicochemical characteristics between the sampled sites (3, 5, 6, 15, 17, 23, 28, 31) and between months (May-October 2015). Significance was tested at $p < 0.05$

	SITE		**MONTH**	
Variable	p value	H value	p value	H value
Ammonia(mgl^{-1})	0.86	3.25	0.67	10.31
Reactive phosphorus(mgl^{-1})	0.41	7.16	<0.001	29.03
Nitrates(mgl^{-1})	0.73	4.43	<0.001	33.74
Total phosphorus(mgl^{-1})	0.29	8.39	0.0009	20.84
Total nitrogen(mgl^{-1})	0.93	2.39	<0.001	40.65
Transparency(m)	0.86	3.24	<0.001	36.86
BOD(mgl^{-1})	0.96	1.97	<0.001	32.49
COD(mgl^{-1})	0.89	2.94	0.04	11.8
Chlorophyll a (μgl^{-1})	0.98	1.6	<0.001	32.44
pH	0.98	1.49	<0.001	42.56
Conductivity(μScm^{-1})	0.99	0.48	<0.001	40.05
TDS(mgl^{-1})	0.99	1.18	<0.001	39.59
Temperature(^oC)	0.94	2.29	<0.001	42.86
Dissolved oxygen(mgl^{-1})	0.03	14.94	<0.001	22.47
% saturation	0.05	14.29	<0.001	24.74

BOD = Biological Oxygen Demand, COD = Chemical Oxygen Demand, TDS = Total Dissolved Solids

There were significant differences ($p < 0.05$) among the sites for dissolved oxygen and percentage oxygen saturation. All the other variables were not significant among the sites. There were significant differences between the sampling months for all the variables ($p<0.05$) except for ammonia (Table 3.3).

3.3.3 Principal Component Analysis

The first four ordination axes accounted for 73.8% of the cumulative variance of the environmental data (Table 3.2) . The first two axes (depicted in Figure 3.6) accounted for 53.0% of the total variation, with axes one and two each explaining 35.3% and 17.8%, respectively. There was a clustering of sites sampled in the same month which suggests changes in water quality among the different sampling occasions although the extent of the differences varied. The months of May, July and October were distinct while the other sites were highly similar but were nevertheless clustered separately. In May, there was increasing levels of reactive phosphorus, total nitrogen and pH while the month of July was characterized by increasing water transparency (Secchi disc) and ammonia. The sampling sites in October were characterized by high levels of chlorophyll *a* and water temperature.

Table 3.3Summary of PCA analysis results for the overall assessment of possible relations between physico-chemical variables.

Axes	**1**	**2**	**3**	**4**	**Total variance**
Eigenvalues	0.353	0.178	0.124	0.083	1.000
Cumulative percentage variance of species data	35.3	53.0	65.5	73.8	
Sum of all eigenvalues					1.000

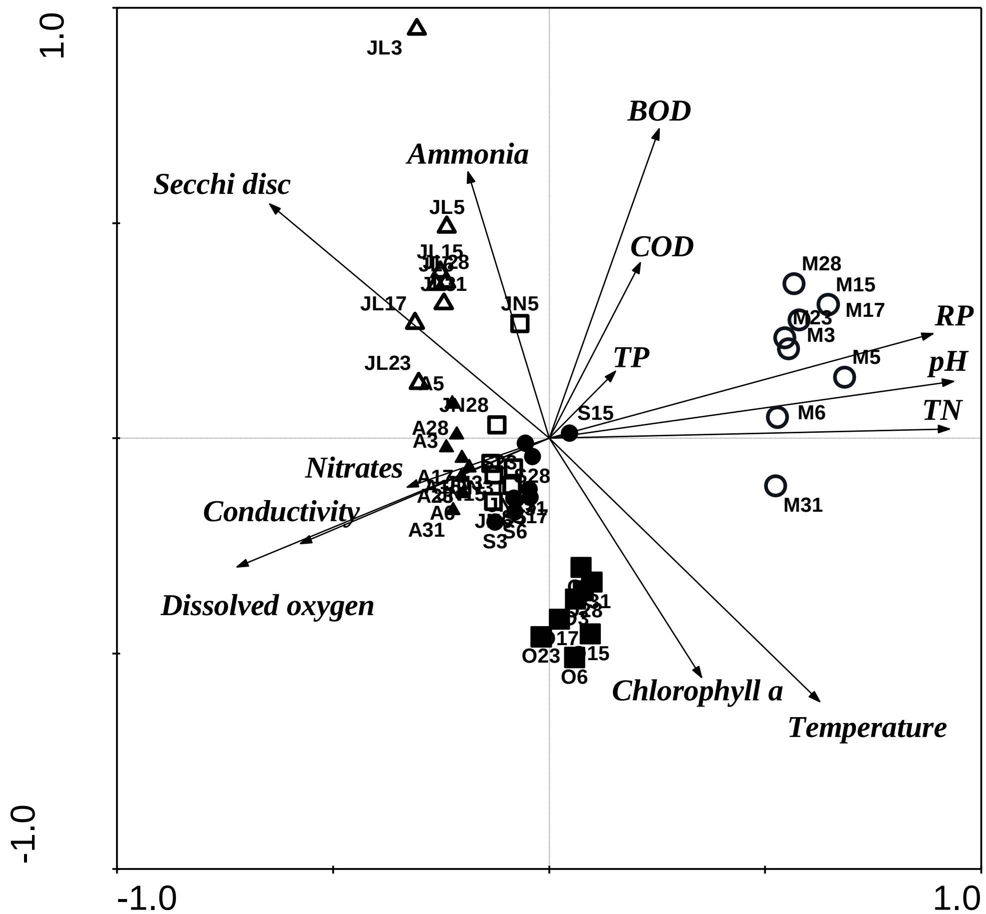


Figure 3.5 PCA plot showing the relations among the limnological variables measured at the different sites in Mazvikadei dam from May-October 2015

3.3.4 Phytoplankton

3.3.4.1 Species composition and seasonal variation

The relative abundance of the phytoplankton groups is shown in Table 3.5 below. The highest composition of Chlorophyta was in September and October in the hot dry season with a mean composition of 63.29 % as shown in Fig 3.7 a below. The composition of Bacillariophyta was highest from May-June with a mean composition of 44.01%. A large proportion of Dinophyta was observed in the cool dry season from July-August with a mean composition of 46.88%. The highest composition of Cyanophyta was observed in May with a mean composition of 20.08%. In May desmids had a percentage composition of 18.19%. The Euglenophyta averaged 1.68% with the greatest composition being recorded in May. The percentage composition of Bacillariophyta, Desmids and Cyanophyta decreased gradually from May-September and a significant increase in all the three groups was observed in October. The Chlorophyta however increased gradually from July-October.

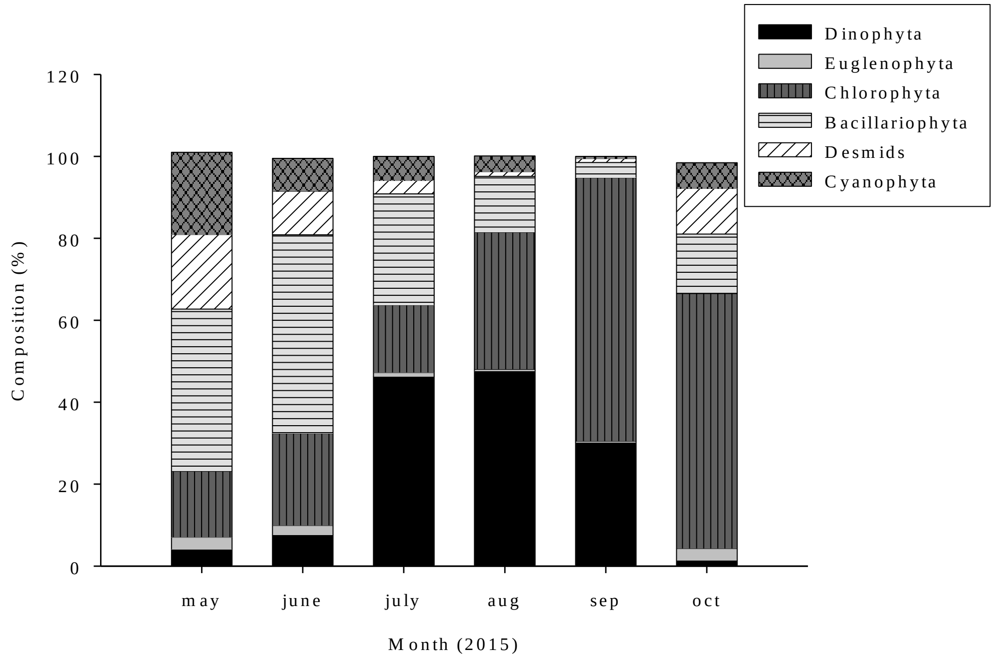
Species richness was greatest in May (36 species). Species richness was low during the cool dry season in June, July and August with a mean of 27 species (Table 3.2). The Shannon Wiener diversity index and the Simpson's index followed the same trend. Both were high in May at the onset of the cool dry season (H'=2.928; Simpson=0.921) and decreased during the cool dry season dropping in July (H'-1.828; Simpson=0.697). The Shannon Wiener index of diversity then increased in the hot dry season (H'=2.672; Simpson=0.8831). The phytoplankton communities were more even in May, June and October. From July-September the evenness was low decreasing to 0.1847 in September and two or three groups of phytoplankton became dominant.

Table 3.4Relative abundance (individual per litre) of phytoplankton of Mazvikadei reservoir

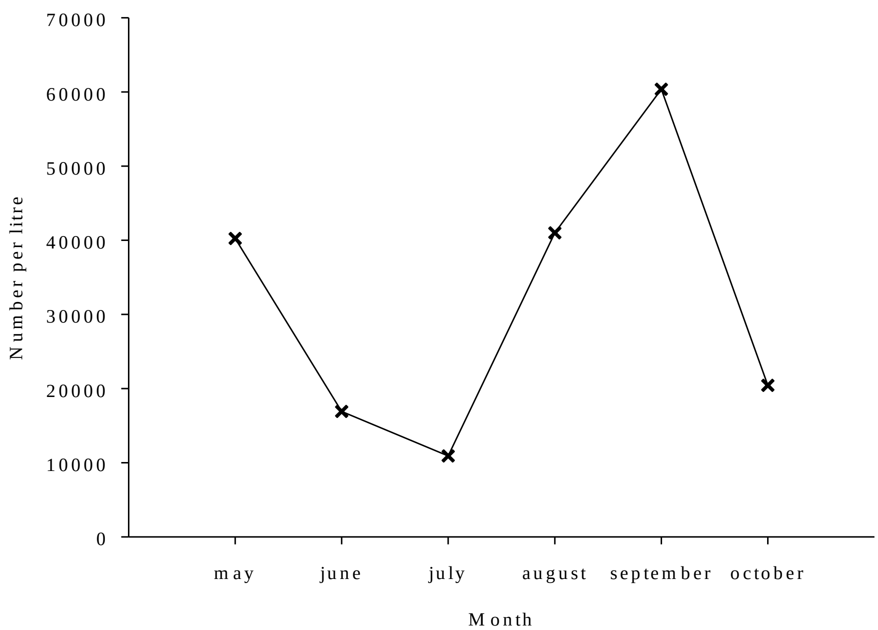
Division	May	June	July	August	September	October
Dinophyta						
Ceratium	1400	1280	5040	19480	18160	280
Peridinium	240	0	0	40	40	0
Euglenophyta						
Euglena	280	320	120	160	200	600
Tetraedron trigonum	920	80	0	0	0	0
Trachelomonas sp.	0	0	0	0	0	0
Chlorophyta						
Amphora sp.	280	120	0	160	40	280
Ankistrodesmus	80	0	0	0	0	0
Chlorella sp.	0	40	0	240	80	760
Chroococcus sp.	960	0	0	0	0	0
Coelastrum sp.	440	80	40	400	1000	400
Dictyosperarium	80	0	0	40	0	0
Eudorina elegans	240	0	0	160	1120	280
Eutetramorus planctonicus	0	0	40	0	240	1200
Eutetramorus fotti	40	0	80	40	1120	400
Gloeocystis sp.	80	120	480	880	680	0
Heleochloris mucosa	3440	240	120	1200	4960	1760
Heleochloris sp.	40	360	560	9440	24120	6040
Korschpalmella miniata	160	0	0	0	320	0

Division	May	June	July	August	September	October
Monoraphidium arcuatum	520	2680	80	200	280	320
Oocystis sp.	40	0	120	360	3520	0
Pediastrum boryanum	0	160	40	120	80	200
Pediastrum duplex	0	0	120	40	40	160
Pediastrum simplex	80	0	0	40	0	0
Radiococcus nimbatus	0	0	0	80	1160	440
Scenedesmus sp.	0	0	80	320	80	480
Schroederia sp.	0	0	40	0	0	0
Bacillariophyta						
Achnanthes	1160	280	240	80	0	0
Cocconeis	400	0	0	0	0	0
Cymatopleura	640	320	40	200	160	280
Cymbella sp.	1120	120	400	0	120	80
Diatoma vulgaris	280	320	80	960	240	80
Eunotia	160	40	0	120	80	0
Fragilaria sp.	880	1360	1080	1200	520	800
Frustulia	720	1560	0	40	120	440
Gyrosigma	1800	1000	520	840	360	1120
Navicula	160	40	80	400	40	0
Nitchzia linearis	840	240	80	280	120	80
Pinnularia sp.	6760	2520	280	760	360	0
Synedra sp.	920	400	160	720	80	80
Surirella	80	0	0	0	0	0
Desmids						
Arthrodesmus sp.	80	0	0	0	0	0
Closterium setaceum	160	920	320	400	320	1560
Cosmarium	1720	40	0	0	0	0
Spirogyra sp.	40	80	0	0	0	0
Staurastrum gracile	0	0	0	0	0	80
Staurastrum tetracerum	5320	760	40	80	280	640
Cyanophyta						
Anabeana sp.	0	1280	520	280	120	640
Anabaena spiroides	320	0	0	40	0	0
Chroococcus sp.	2360	0	0	0	0	0
Gomphosphaeria sp.	960	0	0	120	0	0
Merismopedia sp.	40	80	0	280	0	200
Microcystis aeruginosa	800	0	0	0	40	40
Oscillatoria sp.	3280	0	120	840	160	400
Unidentified filamentous	320	0	0	0	0	0
Total number of taxa (N)	36	29	26	31	34	31
Ind l^{-1}	40240	16920	10920	41000	60360	20440
Simpson's index	0.921	0.907	0.749	0.697	0.7351	0.8831
Shannon Wiener index	2.928	2.685	2.118	1.828	1.837	2.762
Eveness	0.519	0.505	0.32	0.201	0.1847	0.5106

The total individual numbers per litre was high in May but decreased in June and July during the cool dry season and increased again in August (Figure 3.7 b). The highest number of individuals per litre for the whole sampling period was observed in September when there was a peak of >60 000 indl^{-1} at the beginning of the hot dry season. The phytoplankton numbers decreased in October. Individual species within the phytoplankton groups that contributed significantly in terms of abundance were *Ceratium sp., Heleochloris sp., Heleochloris mucosa, Coelastrum sp., Cymatopleura sp., Gyrosigma sp., Fragillaria sp., Pinnularia sp.* and *Straurastrum tetracerum.*



a)



b)

Figure 3.6Temporal changes in a) percentage composition and b) total abundance (individuals per litre) of phytoplankton taxa

3.3.4.2 Relations between environmental variables and the phytoplankton community composition and abundance.

Table 3.3 summarises the results of RDA analysis of the relations between measured environmental factors and the phytoplankton community of Mazvikadei reservoir Phytoplankton abundance and composition were significantly correlated (F-ratio = 2.903, p = 0.002) with temperature, nitrates and total nitrogen. Together these variables accounted for 21.3% of the variation in the phytoplankton community in Mazvikadei reservoir. Axes one and two accounted for 15.7% of the percentage variance of the phytoplankton species community. Similarly, the first two axes accounted for 73.9% of the species-environment variance. It was interesting to note that nitrates and total nitrogen concentrations had a strong clustering effect on the phytoplankton community but similar trends were not apparent for dissolved oxygen (Figure 3.8). In relation to the sites, all the other sites were clustered around the origin which suggests a lack of major differences in the phytoplankton community. However, the sampling sites of May 2015 were distinctly clustered together.

Table 3.5Summary results of RDA analysis of the relations between measured environmental factors and the phytoplankton community of Mazvikadei Reservoir (May-October 2015).

Axes	**1**	**2**	**3**	**4**	**Total variance**
Eigenvalues	0.093	0.064	0.040	0.015	1.000
Species-environment correlations	0.799	0.749	0.831	0.580	
Cumulative percentage variance of species data	9.3	15.7	19.7	21.3	
Cumulative percentage variance of species-environment relation	43.9	73.9	92.7	100.0	
Sum of all eigenvalues					1.000
Sum of all canonical eigenvalues					0.213
Summary of Monte Carlo test					
Test of significance of all canonical axes					
Trace					0.213
F-ratio					2.903
P-value					0.002

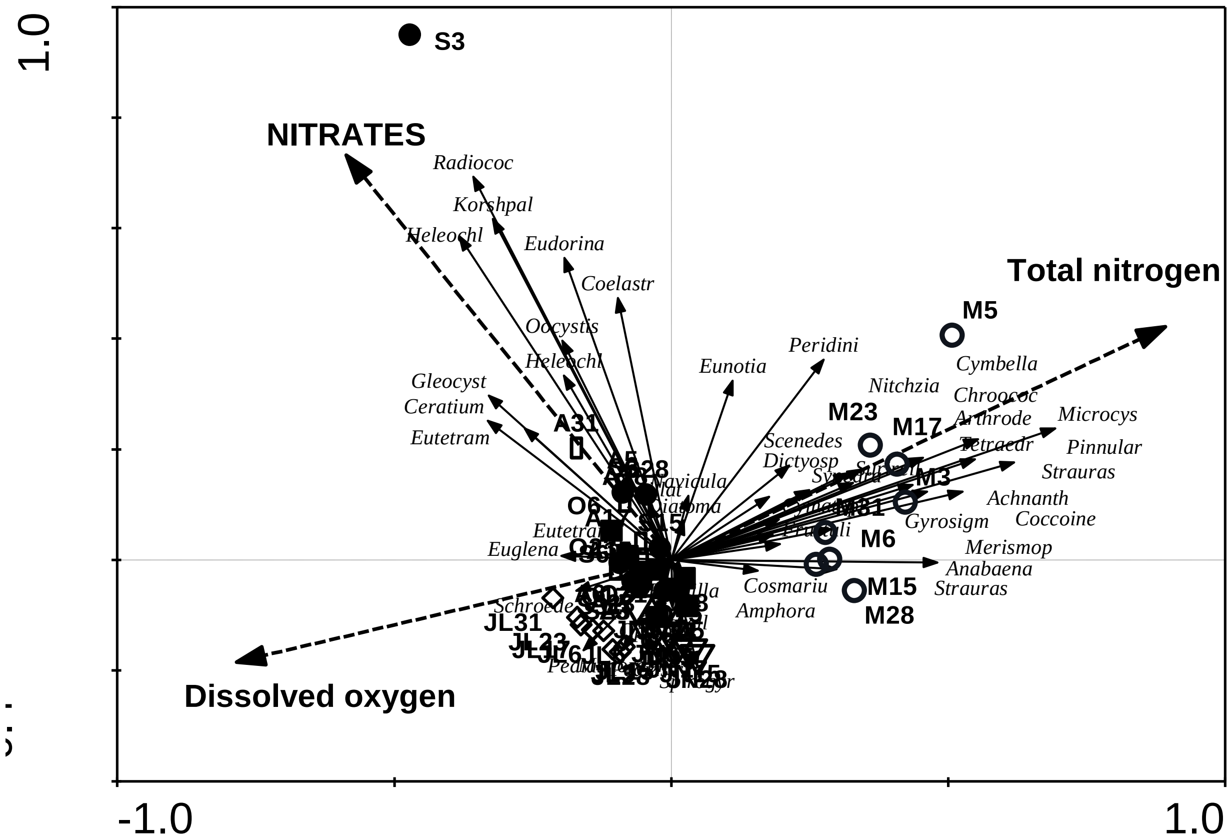


Figure 3.7 RDA ordination plot showing the relationship between significant environmental variables and the phytoplankton communities of Mazvikadei reservoir from May-October 2015. (Site codes: M – May, JN – June; JL – July; A – August; S – September; O – October)

3.3.5 Zooplankton

3.3.5.1 3.4.1 Species composition and seasonal variation

The relative abundance of zooplankton in Mazvikadei reservoir during the sampling period is summarized in Table 3.8 below The highest percentage composition of Rotifera was observed in September (29.6%). There was a high percentage composition of Rotifera in May and June with a mean composition of 13.53%. The percentage composition of the copepods was highest in October in the hot dry season (35.13%). From May-July there were less copepods with a mean composition of 18.27%. During the entire sampling period the cladocerans had the highest composition each month and the composition increased sharply in the cool dry season in the months June and July with a mean composition of 74.97%. Interestingly, the Ostracoda were only observed in the hot dry season in October although they had a low percentage composition of 5.4%.

The Shannon Wiener and Simpson's indices of diversity followed a similar trend (Table 3.3). They were both highest in May (H'=2.141; Simpson=0.8482). The lowest diversity was observed in July during the cool dry season (H'=1.607; Simpson=0.7264). The zooplankton community structure was almost even throughout the sampling period from May-October with evenness ranging from 0.4639-0.6623.

Table 3.6 Zooplankton species composition and occurrence in 1988 (Source: Masundire, 1992)

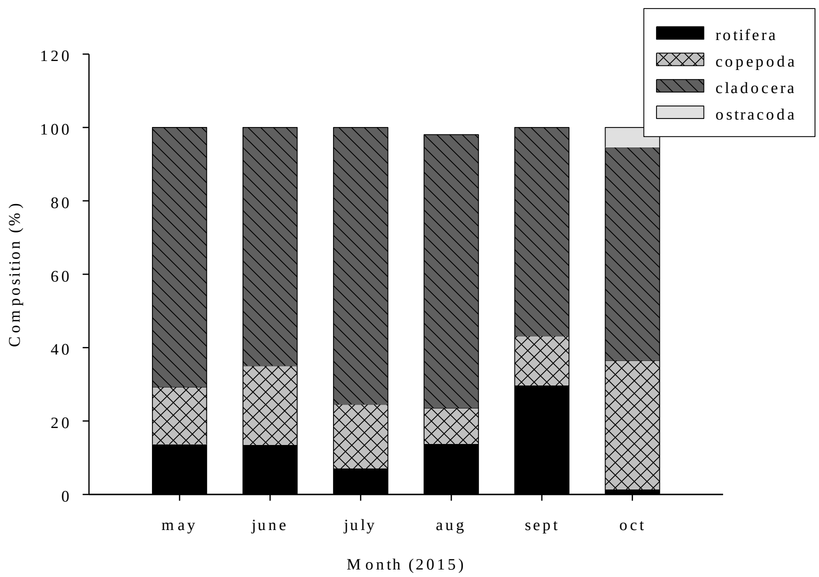
Division	April	May	August
Rotifers			
Filinia opoliensis	+	-	+
Trichocerca cylindrica	+	-	-
Trichocera sp.	+	-	-
Polyarthra sp.	+	-	-
Keratella tropica	+	-	-
Brachionus falcatus	+	-	-
Platyias quadriconis	+	-	-
Ascormorpha sp.	+	-	-
Cladocera			
Daphnia longispina	-	-	+++
Bosmina longirostris	++	-	++
Ceriodaphnia cornuta	++	-	+
Diaphanosoma excisum	+++	-	+
Copepoda	-		
Thermocyclops neglectus	+++	-	+++
Thermodiaptomus syngenes	++	-	++

- absent + present ++ common +++ abundant

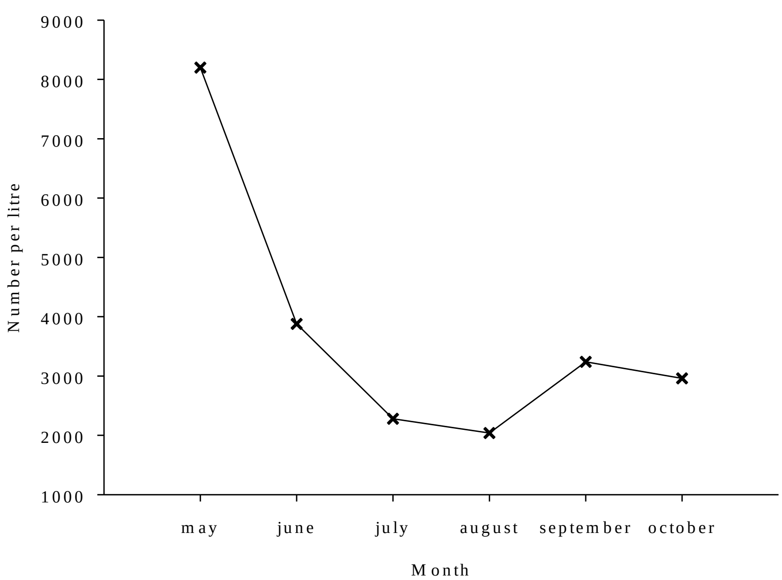
Table 3.7Relative abundance (individual per litre) of zooplankton in Mazvikadei reservoir (May-October 2015)

Division	May	June	July	August	September	October
Rotifera						
Asplancha sp.	0	80	0	0	0	0
Branchionus caudatus	80	160	40	80	0	0
Branchionus fortificula	720	200	120	40	40	0
Keratella cohlearis	280	40	0	120	880	0
Polyarthra vulgaris	40	0	0	0	40	40
Trichocera sp.	0	40	0	40	0	0
Copepoda						
Ectocyclops sp.	40	0	0	80	40	0
Eucyclops neumanii	200	40	0	0	0	0
Eucyclops eucanthus	80	80	0	0	0	160
Thermocyclops eminii	40	240	120	120	200	320
Thermocyclops incicus	240	120	80	0	0	80
Mastigodiaptomus sp.	0	0	40	0	40	80
Nauplii	680	360	160	0	160	400
Cladocera						
Bosmina longirostris	2200	1760	760	800	1320	1040
Ceriodaphnia sp.	160	0	0	0	0	0
Daphnia longispina	1520	40	0	0	0	0
Daphnia coronata	0	160	80	120	120	80
Daphnia pulex	1040	520	880	560	400	440
Diaphanosoma excisum	880	40	0	40	0	160
Ostracoda						
Oncocypris sp.	0	0	0	0	0	160
Total number of taxa(N)	15	15	9	11	10	11
Ind l^{-1}	8200	3880	2280	2040	3240	2960
Simpson's index	0.8482	0.7554	0.7264	0.7559	0.7368	0.8134
Shannon Wiener index	2.141	1.94	1.607	1.784	1.638	1.986
Eveness	0.5673	0.4639	0.5542	0.5414	0.5143	0.6623

The total abundances of zooplankton was low being <4000 indl^{-1} from June-August but rose to a maximum of 8200 indl^{-1} in May (Figure 3.9 a). The numbers increased again from September-October in the hot dry season. The individual species in the zooplankton groups that contributed significantly in terms of abundance were *Bosmina longirostris, Branchionus fortificula, Keratella cochlearis, Thermocyclops eminii, Daphnia pulex, Daphnia longispina* and *Diaphanosoma excisum*.



a)



b)

Figure 3.8 Temporal changes in a) percentage composition and b) total abundance (individual per litre) of zooplankton taxa in Mazvikadei reservoir

3.3.5.2 Relations between environmental variables and zooplankton community composition and abundance.

Table 3.4 summarises the results of RDA analysis of the relations between measured environmental variables and the zooplankton community in Mazvikadei reservoir. Reactive phosphorus and phytoplankton abundance were identified in the forward selection statistical procedure as significant variables (F-ratio 4.230, p = 0.002) explaining zooplankton community dynamics in Mazvikadei dam. The first 4 axes accounted for 35.3 % of the variance in the zooplankton species data, with axes one and two explaining 12.1% and 3.7 % respectively. Of the variance in the species-environment relationship, axis one accounted for 76 % and axis two 24.3 %. The species-environment analysis depicted in Figure 3.10 does not go beyond the first two axes, so it is a two dimensional relationship.

Table 3.8 Summary results of RDA analysis of the relations between measured environmental variables and the zooplankton community of Mazvikadei reservoir (May-October 2015)

Axes	**1**	**2**	**2**	**2**	**Total variance**
Eigenvalues	0.121	0.037	0.109	0.087	1.000
Species-environment correlations	0.892	0.662	0.000	0.000	
Cumulative percentage variance of species data	12.1	15.8	26.7	35.3	
Cumulative percentage variance of species-environment relation	76.7	100.0	0.0	0.0	
Sum of all eigenvalues					1.000
Sum of all canonical eigenvalues					0.158
Summary of Monte Carlo test					
Test of significance of all canonical axes					
Trace					0.158
F-ratio					4.230
P-value					0.0020

An analysis of Figure 3.10 reveals that samples collected in May 2015 were associated in higher concentrations of certain species which include *Ceriodaphnia* sp., *Daphnia pulex, Bosmina longirostris*, *Polyathra* sp., *Branchionus fortificula, Diaphanosoma excisum* and *Eucyclops* sp.. In the same quadrant there is also the influence of phytoplankton abundance as a significant factor on the zooplankton community. August, October and several September sites were clustered together. The effect of phytoplankton abundance is also evident on these sites. Species such *Ectocyclops* sp., *Keratella cochlearis, Thermocyclops eminii*, *Daphnia pulex* and *Oncocypris* sp. were dominant in August and October. The June and July sites were not strongly associated with the dominance of many species except a small increase in *Trichocera* sp. and *Mastigodiaptomus* sp.

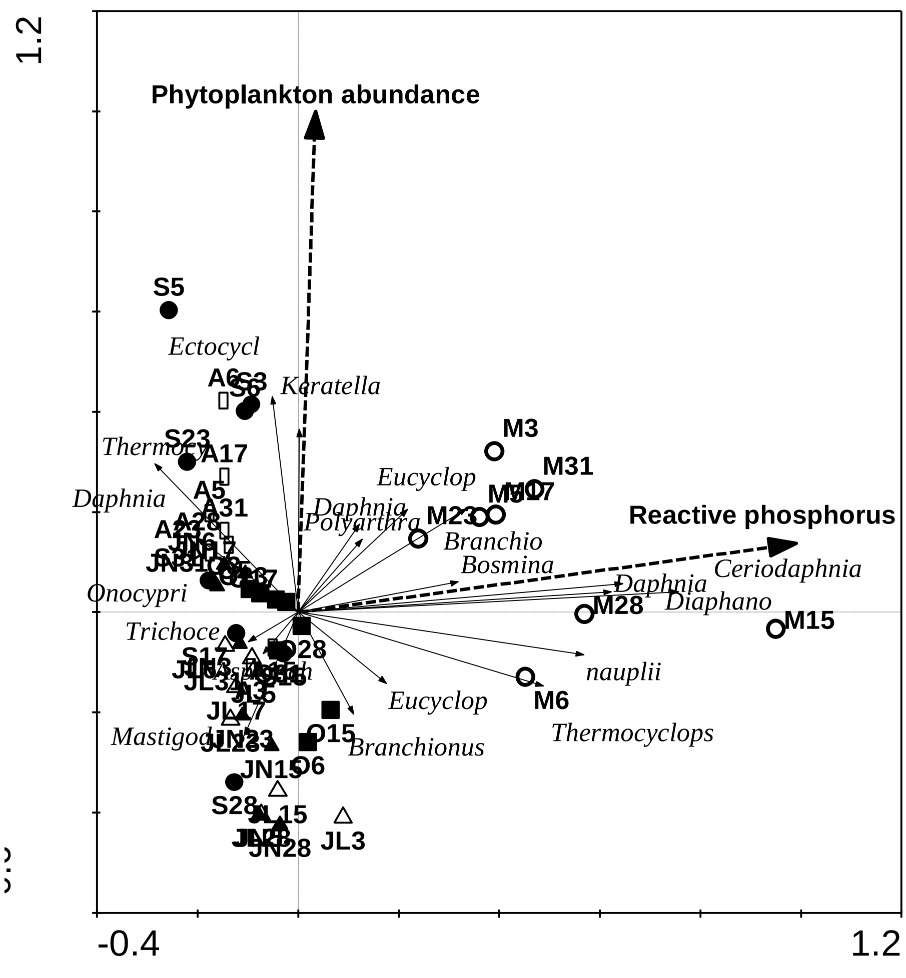


Figure 3.9RDA ordination plot showing the relationship between significant environmental

variables and the zooplankton communities of Mazvikadei reservoir from May-October 2015. (Site codes: M – May, JN – June; JL – July; A – August; S – September; O – October)

3.4 Discussion

All indices employed during this study showed that Mazvikadei reservoir can be classified as an oligotrophic reservoir characterized by the low algal biomass, low concentrations of P and N, high Secchi disk clarity and low chlorophyll *a* concentration.

Mazvikadei is well oxygenated. This contrasts with anoxic conditions detected in the water column during the filling phase (Masundire 1992). Dissolved oxygen in Mazvikadei is within the range recorded in Cleveland dam (7.7 mgl^{-1}) (Ndebele, 2009) and in the epilimnion of Lake Kariba (7.6 – 7.8 mgl^{-1}) (ILEC, 1998), two other oligotrophic lakes. In 1988 the Mazvikadei had a neutral pH ranging from 7.19 in May and 7.80 in August (Masundire, 1992). The pH is now slightly alkaline (8.54) and is comparable to that for Lake Kariba (Balon and Coche 1974) and Cleveland Dam (Ndebele, 2009). The average conductivity in the reservoir has increased to 263.2 μScm^{-1} in comparison to 160 μScm^{-1} (range 146 - 158 μScm^{-1}) recorded during the filling phase. The conductivity levels in Mazvikadei are higher than in Cleveland dam (114 μScm^{-1}) (Ndebele, 2009). It also falls out of the range reported in Lake Kariba (95 $\mu S\ cm^{-1}$ in Basin 1 to 100 $\mu S\ cm^{-1}$ in Basin 5 (Balon and Coche, 1974). The reservoir however, falls in Class 1 which according to Talling and Talling (1965) comprises of soft water lakes and reservoirs with conductivities < 600 μScm^{-1}. Such lakes are fed by rivers of low salt content which regionally include lakes Victoria and George (Talling and Talling, 1965), and locally Kariba (Coche, 1974; Magadza *et al*., 1987). The transparency of Mazvikadei has increased to 4.62 m from a transparency of 0.75 m and 3.30 m recorded in May and August 1998 respectively during the filling phase (Masundire, 1992). The improvement is linked to change in the trophic status of the reservoir. Both N and P were higher during the filling phase. The reservoir experienced a post-impoundment eutrophic or nutrient-rich phase but has now assumed and an oligotrophic or nutrient-poor phase.

The phytoplankton community in Mazvikadei reservoir comprised of six main groups namely Dinophyta, Euglenophyta, Chlorophyta, Bacillariophyta, Desmids and Cyanophyta. These groups are typical of those recorded in Cleveland dam (Ndebele, 2009) and Lake Kariba during the early years (Ramberg, 1987; Cronberg, 1997), which are two systems that are also oligotrophic. Individual species within the phytoplankton groups that contributed significantly in terms of abundance in Mazvikadei reservoir were *Ceratium sp., Heleochloris sp., Heleochloris mucosa, Coelastrum sp., Cymatopleura sp., Gyrosigma sp., Fragillaria sp., Pinnularia sp.* and

Straurastrum tetracerum. Fifty four species were recorded in Mazvikadei reservoir. This is about twice the number of species recorded in Cleveland dam (29 species) (Ndebele, 2009) but is lower than an estimated 150 in Lake Kariba (Cronberg, 1997). Kariba is much larger in size than Mazvikadei reservoir and therefore it has higher niche availability leading to higher species richness.

Species richness was highest in May at the onset of the cool dry season. This is directly related to the high nutrient concentrations that were also observed during that particular period. The high nutrient concentrations observed in May can be attributed to the import of dissolved ions via the river inflow that occurred as a result of the late rains. An increased amount of suspended solids carried into the reservoir led to high concentrations of reactive phosphorous in the reservoir. When sediments were disturbed in this case by the late rains and river inflow the rate of phosphorous release may have increased (Wetzel, 2001). Under the influence of wind generated mixing there could also have been a partial mixing of the riverine layer and the epilimnion providing a mechanism by which nutrients were redistributed vertically within the reservoir. Additional nutrients were released contributing to increased productivity in the reservoir during that period. As the nutrients were assimilated with time this led to a decrease in the species richness in the subsequent months after May.

Evenness decreased during the cool dry season with the Bacillariophyta and the Dinophyta becoming dominant during that time. This is a similar trend with that observed in Lake Kariba, Malawi, Tanganyika and Victoria (Ndebele-Murisa *et al*., 2010, Salonen *et al*. 1999, Heck and Kling 1981, Cocquyt and Vyverman 1994, Descy *et al*. 2005). A comparison can also be made with the phytoplankton community in Lake Kariba which is now stable and exhibits a regular seasonal pattern linked to the nutrient dynamics in the lake (Ramberg, 1987; Cronberg, 1997). In Lake Kariba diatoms (*Aulacoseira*, *Cyclotella*, *and Synedra*) attain a maximum biomass at turn-over (June-July). This was also observed in Mazvikadei reservoir where diatoms (Bacillariophyta) were dominant in the cold dry season. Diatoms in freshwater exhibit a "*bloom and bust*" lifestyle in which when there is high nutrient and light availability during the turnover period their competitive edge allows them to quickly dominate the phytoplankton communities (Furnas, 1990). Phytoplankton growth is dependant on temperature. Although the Dinophyta can adapt to a wide range of temperatures and light irradiance they do thrive at lower temperatures (Baek *et al*., 2008), hence the group dominated during the cool dry season.

Due to the limited sampling period (six months) it is difficult to clearly depict phytoplankton seasonal succession in Mazvikadei reservoir. The sampling period, from May-October 2015 only reflects the results of the cool dry season and the hot dry season. There was dominance of the Bacillariophyta and Dinophyta in the cool dry season and Chlorophyta dominating in the hot dry season. The group Cyanophyta had less representation although it occurred in small numbers right through the sampling period with the highest numbers being observed in May. Other groups mainly Eugleophyta and Desmids were most abundant in May as well. These phytoplankton species may be more competitive at higher temperatures recorded in May. Generally the phytoplankton community in Mazvikadei reservoir is dominated by 3 groups; Chlorophyta, Dinophyta and Bacillariophyta. This contrasts with the observation by Ndebele (2009) in Cleveland dam where Cyanophyta *(Microcystis aeruginosa*) was observed to be dominant with respect to abundance. Ndebele (2009) observed that in terms of species representation the dominant groups in Cleveland dam were Chlorophyta (13 species), Bacillariophyta (8 species) and Cyanophyta (3 species). In Mazvikadei reservoir dominance with respect to species representation was Chlorophyta (21 species), Bacillariophyta (14 species) and Cyanophyta (7 species). The differences in the community structures may be attributed to time of sampling since Ndebele (2009) sampled during the hot dry and hot wet season (September- January) and this study was carried out in the cool dry and hot dry season (May –October).

The succession of phytoplankton in a reservoir reflects the change in the lake's trophic status. Although there is no documented information of the phytoplankton community composition at the filling phase of Mazvikadei reservoir for comparison, the current community that has been established in Mazvikadei reservoir is characteristic of an oligotrophic system. Diatom species such as *Nitchzia* and *Fragillaria* which were also observed in Lake Kivu are normally associated with oligotrophic, phosphorous deficient African tropical reservoirs (Sarmento *et al*., 2006). The low abundance of Euglenophytes and Cyanophytes and the high densities of desmids was also an indication of oligotrophic conditions within the reservoir. It is also comparable to that of Lake Kariba where the most common phytoplankton species in the lake are the two diatoms *Aulacoseira* and *Synedra* and blue-green algae *Cylindrospermopsis raciborskii* and (Cronberg, 1997). During the filling phase in Lake Kariba desmids and benthic diatoms, relics of the riverine community were dominant. This could have been the case also in Mazvikadei reservoir during its eutrophic phase but it has now matured and assumed an oligotrophic state.

Phytoplankton abundance and composition were significantly correlated with temperature, nitrates and total nitrogen. Nitrates and total nitrogen had a strong clustering effect on the phytoplankton community because a large proportion of total nitrogen contains dissolved organic nitrogen which can readily be converted into ammonia which is the best form of nitrogen for assimilation by plants and algae since they cannot use nitrogen in its elemental form (Ghaly and Ramakrishnan, 2015). Species that were highly correlated by the two nutrients were chlorophytes namely *Radiococcus* sp., *Heleochloris* sp., *Coelastrum* sp., *Ceratium* sp., *Gleocystis* sp., *Eutetramorus sp., Eudorina elegans, Korshpalmella miniata* and *Oocystis* sp. Most of the sites from June –July were clustered around the origin suggesting a lack of major differences in the phytoplankton community during that period. This may be because there were no significant differences between the sites in terms of environmental variables so since the conditions were almost the same across all the sites the community composition would also be almost similar.
The sites for May were however distinctly clustered together. This may as a result of the nutrient release that occurred because of the inflow caused by late rains in May. This event only occurred in May throughout the sampling period and may have led to this distinct clustering.

In 1992 Masundire recorded 8 rotifers in Mazvikadei reservoir which occurred only in April except for *Filinia opoliensis* which was also recorded in August. During this study 6 rotifer species were recorded. Except for *Branchionus fortificula* and *Trichocera* sp. all the other species recorded in 2015 were new species.

Masundire (1992) recorded four cladocerans comprising of *Daphnia longispina*, *Bosmina longirostris*, *Ceriodaphnia cornuta* and *Diaphanosoma excisum*. All species occurred in August and except for *D. longispina* the rest occurred in April. None of these species were recorded in May. The most abundant species in April during Masundire’s study was *D. excisum* while *B. longirostris* and *C. cornuta* were common. During this study six cladocerans were recorded in May. The cladoceran species that were recorded in both studies are *B. longirostris*, *D. longispina* and *D. excisum*. New species recorded during this study are *Daphnia coronata* and *Daphnia pulex*.

. A *Ceriodaphnia* sp. was also recorded in 2015. All the cladocera except for *D. coronata* occurred in May 2015 which contrasts with Masundire's study where he did not record any cladocerans in May. This is because in May when Masundire did his study anoxic conditions prevailed in the reservoir which could have killed all forms of life that were in the reservoir at that time. The most dominant cladocerans in May 2015 in order of dominance were *B. longirostris*, *D. longispina* and *D. pulex*. In August 2015 all other cladocerans except *Ceriodaphinia* sp. and *D. longispina* were recorded. The dominant species in August 2015 in order of importance were *B. longirostris*, *D. pulex* and *D. coronata*.

Masundire (1992) recorded 2 copepods in Mazvikadei in 1988, a cyclopoda *Thermocyclops neglectus* and a calanoida *Thermodiaptomus syngenes*. The two species occurred in April and August but were absent in May. In 1988 the most abundant species was *T. neglectus* followed by *T. syngenes*. Six new species of copepods were recorded in 2015 which included none of those recorded in 1988. These included *Ectocyclops* sp., *Eucyclops neumanii*, *Eucyclops eucanthus*, *Thermocyclops eminii*, *Thermocyclops incicus* and *Mastigodiaptomus* sp. The community of copepods in Mazvikadei has changed. There are obvious differences in zooplankton community composition observed in August 1988 to the current study. In the current study the most dominant species in order of abundance were *T. incicus*, *E. neumanii* and *E. eucanthes*. They were all recorded in May 2015 except *Mastigodiaptomus* sp. which also did not occur in August 2015. Copepods can survive in a wide range of conditions. They have been recorded to survive even in low oxygen waters (Strickle *et.al.*, 1989). Calanoids however are more responsive to shifts in the external environment and so when conditions were not favorable in the cool dry season the calanoid *Matigodiaptomus sp*. was not observed. The high number of nauplii recorded in May 2015 can best be explained by the high food availability which led to higher reproduction rates for the copepods.

The zooplankton community structure and species composition in Mazvikadei reservoirs has changed from that observed during its filling phase. There are now 19 species compared to 9 recorded during the filling phase showing an increase in species richness over time. Since the filling phase the phytoplankton community has also stabilised as the reservoir matures, the zooplankton community has also responded to this since the two are highly correlated. The absence of some species in 2015 may be attributed to the sampling frequency. Samples were only taken once a month so some species could have been missed during the other periods. The new species may also point to zooplankton succession in reservoirs as they age with the pioneer species such as *Ascormorpha* being gradually replaced by the *Daphnia sp.*

However as observed by Masundire (1992) Cladocera are still the dominant group although rotifers and copepods are also well represented. Lake Kivu an African oligotrophic lake also had all the different groups but was dominated by copepods. It would be expected that since the reservoir is oligotrophic it would be dominated by calanoids which can graze at low phytoplankton concentrations but this was not the case (Isumbisho *et al*., 2006). Cladocerans became most abundant during the cold dry season (June and July) in Mazvikadei reservoir when they assumed 75 % dominance. Cladoceran abundance and composition is dependent on temperature and predation rates. In cold oligotrophic lakes with low predation rates the cladocerans have a longer life span (Pietrzak *et al*., 2013). This may also be the reason why the cladocerans were dominant during the cool dry season in Mazvikadei an oligotrophic reservoir. Rotifers and copepods dominated during the hot dry season due to high food availability as they are grazers. Ostracods were only observed in September in the hot dry season because the ostracod eggs have a protective shell which allows them to go into a dormant state. Once favorable conditions such as warmer temperatures and food availability are prevalent the eggs hatch (Fernando, 2002).

The highest number of zooplankton in May highly correlated with the high phytoplankton abundance at that time that was caused by the nutrient releases in the reservoir. The zooplankton community is highly influenced by the amount of reactive phosphorous. This is a cascading effect where reactive phosphorous affects phytoplankton growth and this in turn affects the amount of food available for the zooplankton.

The study showed that Mazvikadei reservoir has matured and assumed the physico-chemical characteristics and plankton community typical of an oligotrophic lake.

CHAPTER 4

4 EXPLORING THE RELATIONSHIP BETWEEN REFLECTANCE AND NUTRIENT CONCENTRATIONS OVER TIME IN MAZVIKADEI, AN AFRICAN TROPICAL RESERVOIR.

4.1 Introduction

Remote sensing and GIS are effective tools for water quality mapping and land cover mapping. They are essential tools for monitoring, modeling and environmental change detection. GIS can be a powerful tool that can be used for developing solutions to water resources problems. It can be used for assessing water quality, determining water availability, forecasting flooding, understanding the natural environment, and managing water resources on a local or regional scale (Skidmore *et al*., 1997). Remote sensing can also be used to provide spatial information of a limited number of parameters in water quality assessment. On the other hand, *in situ* sampling provides information on many parameters at a single point. Remote sensing and *in situ* sampling can both be used to provide information on water quality at one moment in time (Dekker *et al*., 1996). *In situ* measurements and collection of water samples for subsequent laboratory analyses are currently the main method used to evaluate water quality. These measurements are accurate for a point in time and space but do not provide a spatial or temporal view of water quality changes in a large water body. Thus, the technologies such as remote sensing and GIS can be very useful as a tool in generating temporal and spatial information in water quality monitoring (Usali and Ismail, 2010).

Turbidity and chlorophyll *a* are usually the parameters used for physico-chemical characterisation of small reservoirs. These two parameters have optical properties that make them measurable by remote sensing, and can accurately indicate the environmental status of the reservoirs (Moore, 1980). Turbidity is the measure of the light scattering in water due to suspended matter (Rasmussen *et al*., 2009). Chlorophyll *a* is the substance responsible for the pigment of algal plants, which gives a good indication of algal biomass in water (Duan *et al*., 2007). High turbidity of a reservoir is generally caused by inflows that are heavily loaded by suspended matter (Valentin *et al*., 2008). As the amount of chlorophyll *a* increases so does the turbidity measurements.

A lot of research has been done on remotely sensed retrieval of chlorophyll *a* from satellite images in marine systems using the Sea-viewing Wide Field of-view Sensor (SeaWiFS) (O'Reilly *et al*. 1998), Landsat TM (Ekstrand 1992; Dalu *et al*., 2015), Medium Resolution Imaging Spectrometer (MERIS) (Gordoa *et al*. 2008), and Moderate Resolution Imaging Spectroradiometer (MODIS) (Binding *et al*., 2012). MODIS-Aqua data has been used in the retrieval of chlorophyll *a* as a measure of the phytoplankton biomass in Lake Naivasha (Ndungu *et al*., 2013). This is because the imagery is readily available free of charge and has daily temporal resolution allowing frequent monitoring of water quality. This is useful especially for water quality studies in developing countries where continuous data collection is a challenge due to limited research and monitoring funds. Furthermore, successful retrieval of satellite data is fundamental because it allows retrospective analysis of the chlorophyll *a* in aquatic systems and therefore provides data at times when ground measurements do not exist. The spatio-temporal dynamics of turbidity has also been evaluated using chlorophyll *a* concentration retrieved from MODIS-Aqua 500 m resolution images (Ndungu *et al*., 2013).

One major advantage of remote sensing observations over traditional measurements for water quality monitoring is that it provides both spatial and temporal information of surface water characteristics (Lindell *et al*., 1999). With present advanced satellite sensors, a large number of water quality information e.g. on chlorophyll *a*, suspended sediment, yellow substances, turbidity, Secchi disk depth, wave height, color index and surface water temperature can be observed on a regular basis (Zhang *et al*., 2002). Remote sensing it is also a cheaper and repetitive quantitative technique for measuring water quality that will allow adequate management of a water body than routine *in situ* monitoring. Remote sensing has been successfully used for water quality monitoring (Baban, 1993).

In situ measurements are often restricted to selected sampling points while remote sensing data provides the synoptic view of the water body and provides measurements of the characteristics of an area rather than a point. Remote sensed data can integrate several characteristics compared to one composite *in situ* measurement. Due to the continuous or frequent feedback from satellite measurements remote sensing has the ability to improve models due to the large quantities of data generated. However an intensive *in situ* sampling programme is useful in order to validate the remotely sensed data and to provide measurements of the factors that can not be observed from space such as changing biota, substrate and the chemical characteristics.

An improved understanding of how *in situ* generated data relate to remotely sensed data over time will enable the development of remotely sensed environmental indicators. These indicators will then be monitored using space based measurements and be easily related to characteristics of the environment not directly observable from space (Koponen *et al.*, 2002). Such information is at present lacking for reservoirs in Zimbabwe. Most remote sensing data has focused on modeling chlorophyll *a* in water systems but none of them have focused on the physico-chemical variables that are essential for phytoplankton growth namely phosphates, nitrates and ammonia. The detection of high levels in the system of these nutrients using remote sensing can be valuable for future monitoring of water systems. The study had two main objectives: (i) to determine if relationships exist between nutrient parameters and reflectance; and (ii) assess the stability of such relationships over time as a means of validation as would be required if remote sensing is to be used as an early warning system.

4.2 Materials and Methods

4.2.1 In situ water quality measurements

Water was collected at each of the 8 sites from May-October 2015 in 500 ml polythene bottles and were taken to the laboratory for analysis. Water transparency was measured using a 20 cm diameter, black and white quadrated disk. Nitrates, reactive phosphorous and ammonia were determined using standard methods from EPA and Hach.

4.2.2 Remote sensing

Mazvikadei Dam is enclosed in the satellite scene of row 72 and path 170. Satellite images were downloaded from the United States Geological Survey (USGS) Earth Explorer website (www.earthexplorer.usgs.gov.). In this study, Landsat OLI (Operational Land Imager)/TIRS (Thermal Infrared Sensor) data for the months May-October 2015 was used. Using Qantam GIS version 2.10 Pisa semi automatic classification plug in Top of Atmosphere (TOA) radiances were atmospherically corrected using the Dark Object Subtraction method. The output of the atmospheric correction was reflectance data sets from DN values. The sampling points were used to extract reflectance values of each of the bands at these points. The reflectance values for each band were then combined with the physico-chemical variables at each of the sampling points. The following linear regression model was fitted to the reflectance values of the bands blue (450-510 nm), green (530-590 nm), red (640-670 nm), near infra-red (850-880 nm) and the Short Wave Infra red 1 (1570-1650 nm) and the three physico-chemical variables ammonia, nitrates and reactive phosphorus. Different band ratios were also computed and fitted into the regression model. The single band and the band ratios were the independent variables. R^2 values showing the goodness of fit of the linear regression model were computed and evaluated.

$$\mathbf{Y = mx+b}$$

where:

y = concentration of physicochemical variable

m = slope of the line

x = reflectance for each individual band or band ratio

b = y intercept

4.3 Results

4.3.1 Single bands

For both the single bands and the band ratios the relationship with the physico-chemical variable was not stable over time as the R^2 values fluctuated from month to month throughout the sampling period. The association between the dependent and independent variables did become stronger when the use of band ratios was employed for some of the physico-chemical variables at different times. A relationship between ammonia and all the bands was observed (Table 4.1) although the highest correlation was observed for the near infra-red band in June which had the highest R^2 value of 0,799 (Figure 4.1). The highest R^2 value 0f 0.4701 for reactive phosphorus was observed in June for the near infra-red band (Table 4.2). This was less than 0.5 meaning the model could not explain more than 50 % of the relationship. There was high correlation between nitrates and the blue band in July with an R^2 value of 0.6715 (Figure 4.2).

Table 4.1 R^2 values for the single bands and ammonia

		AMMONIA			
MONTH	Red Band	Blue Band	Green Band	NIR Band	SWIR 1 Band
May	0.038	0.04	0.07	3.00E-05	0.014
June	0.507	0.41	0.718	0.799	0.49
July	0.0029	0.0455	0.098	0.0577	0.0088
August	0.0077	0.0153	0.0502	0.0415	0.0068
September	7.00E-05	0.0003	0.0022	0.0009	0.0066
October	0.0776	0.072	0.0761	0.0578	0.0707

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

Table 4.2R^2 values for the single bands and ammonia

		REACTIVE PHOSPHORUS			
MONTH	Red Band	Blue Band	Green Band	NIR Band	SWIR Band
May	0.2147	0.2708	0.2804	0.067	0.0727
June	0.1478	0.3141	0.3659	0.4701	0.4379
July	0.015	0.2076	0.2824	0.0647	0.0017
August	0.0809	0.008	3.00E-06	0.0047	0.0755
September	0.0037	4.00E-07	0.0095	0.0069	0.0007
October	0.0477	0.0462	0.0591	0.0495	0.0529

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

Table 4.3 R^2 values for the single bands and nitrates

		NITRATES			
MONTH	Red Band	Blue Band	Green Band	NIR Band	SWIR Band
May	0.3963	0.326	0.3164	0.4824	0.2814
June	0.0013	0.0045	0.1915	0.0439	0.021
July	0.0707	0.6715	0.618	0.2464	0.134
August	0.1885	0.0831	0.1053	0.0955	0.0189
September	0.1765	0.2073	0.2781	0.1777	0.1338
October	0.4716	0.4012	0.41	0.4012	0.4691

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

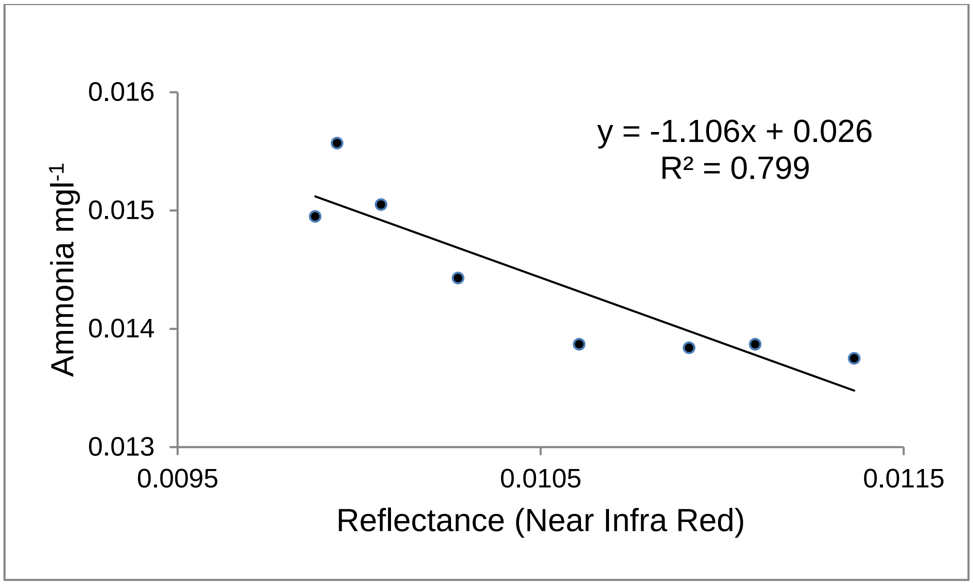


Figure 4.1Linear regression model using ammonia and the near infra red band in June 2015

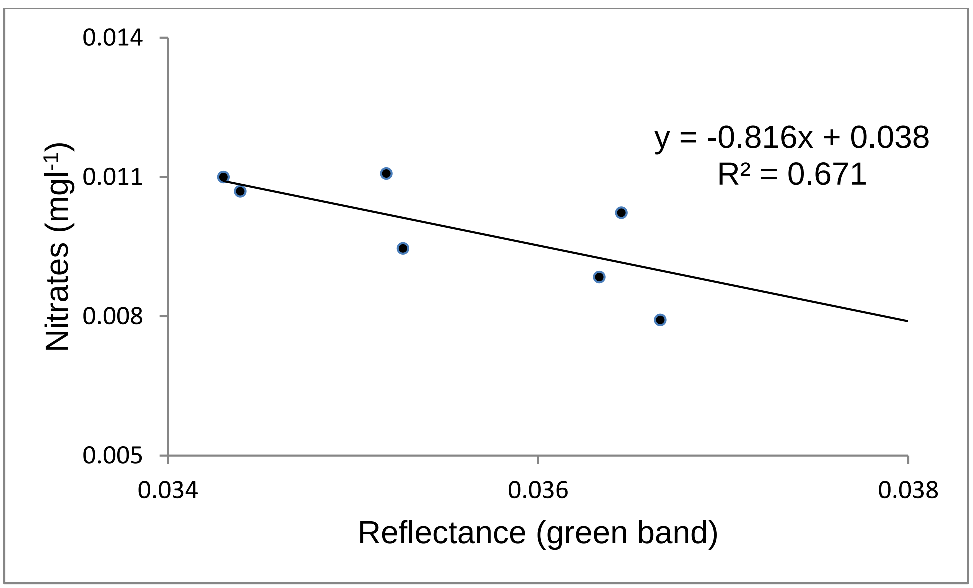


Figure 4.2Linear regression model using nitrates and the green band in July 2015

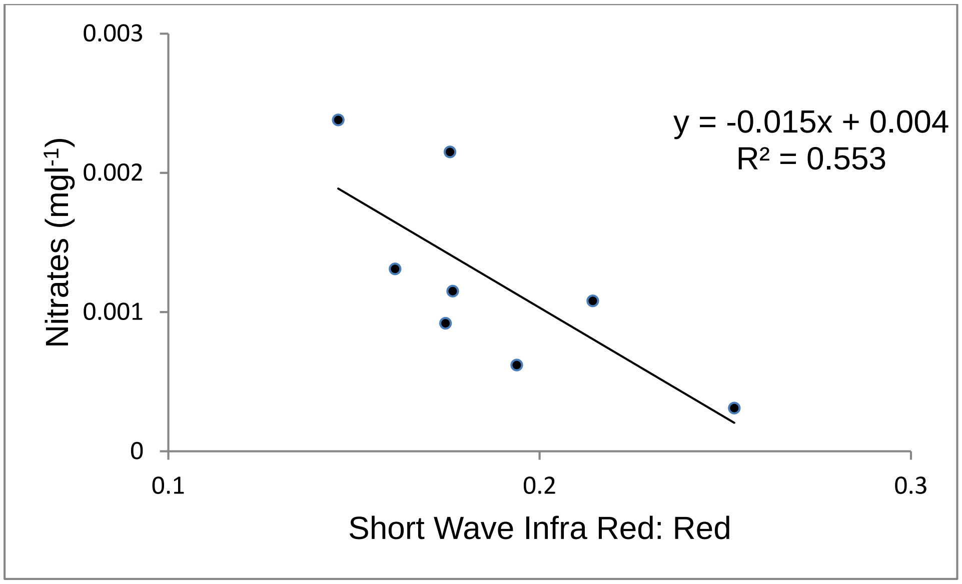


Figure 4.3 Linear regression model using the band ratio short wave infra red: red and nitrates for May 2015

4.3.2 Band ratios

The highest R^2 value for ammonia was obtained for the band ratio Red: Blue in June. High values were also obtained for the NIR: Red, Red: Green band ratios in June. There was no significant relationship between reactive phosphorus and the different band ratio combinations throughout the sampling period from May-October 2015 (Table 4.5). A significant relationship was observed for nitrates and the band ratio Short wave infra red: Red with the highest R^2 value of 0.5538. Generally the R^2 values for the nitrates and all the band ratios was high in May and October (Table 4.6). An observation that can be made from Table 4.4 and Table 4.6 is that the ratios Red: Green and SWIR: NIR seemed best for nitrates and Red: Green and Red: Blue for ammonia for all the band ratios.

Table 4.4 R^2 values for different band ratios and ammonia

			AMMONIA		
MONTH	NIR: Red	Red: Green	Red: Blue	SWIR 1:NIR	SWIR 1: Red
May	0.0531	0.0003	0.0385	0.03	0.0473
June	0.4085	0.4734	0.5081	0.3425	0.0181
July	0.0155	0.0104	0.0092	0.0006	0.2499
August	0.0655	0.019	0.0198	0.0969	0.0199
September	0.0115	3.00E-07	0.0002	0.0343	0.0149
October	0.0373	0.0584	0.0513	0.0487	0.0826

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

Table 4.5 R^2 values for different band ratios and reactive phosphorus

	REACTIVE PHOSPHORUS				
MONTH	NIR: Red	Red: Green	Red: Blue	SWIR 1:NIR	SWIR 1: Red
May	6.00E-05	0.0541	0.1863	0.0986	0.0355
June	0.0497	0.1275	0.1317	0.3615	0.1926
July	0.0473	0.0412	0.0448	0.0127	0.3713
August	0.1262	0.1081	0.1107	0.1821	0.1163
September	0.0459	0.0049	0.0079	0.0173	0.0357
October	0.0329	0.0454	0.0479	0.061	0.0056

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

Table 4.6 R^2 values for different band ratios and nitrates

	NITRATES				
MONTH	NIR: Red	Red: Green	Red: Blue	SWIR 1:NIR	SWIR 1: Red
May	0.4285	0.4991	0.4305	0.5328	0.5538
June	0.0422	0.0013	0.0024	0.0115	0.1321
July	0.057	0.031	0.0197	0.0727	0.0124
August	0.0038	0.0129	0.0116	9.00E-05	0.0236
September	0.1513	0.1606	0.162	0.0994	0.2008
October	0.3981	0.4633	0.4635	0.4968	0.2573

SWIR 1=Short Wave Infra Red, NIR=Near Infra Red

4.4 Discussion

The relationship between the physico-chemical variables and reflectance was not stable over time. This was expected as various processes are occurring in the reservoir at any time due to various external factors that influence these processes. During the hot wet season, the concentrations of nitrogen increase in the reservoir because of storm drains and various forms of run off that wash the nitrates from the catchment that is used for agricultural activities. Nitrogen may also be released into the air because of ammonia volatisation. Atmospheric deposition in the form of acid rain can also affect nutrient concentration in water (Paerl, 1997). Sediments are also important sources of nutrients for aquatic organisms. The sediment particles are resuspended during the rainy season and increases the concentration of nutrients in the reservoir. There are various sources of phosphate into reservoirs from firm rock deposits, run off from surface catchments and interaction between water and sediment from dead plant and animal remains at the bottom of reservoirs. The instability of this relationship over time therefore shows that remote sensing is limiting when it comes to using reflectance to predict the concentrations of these three nutrients in a water body. The good fit that was obtained in the months of June and July shows that the use of the remote sensing to predict nutrient concentrations in the reservoir gives the best results in the cool dry season. This may be because at this particular time the nutrients will be almost evenly distributed throughout the water column soon after turnover occurs.

Water constituents such as the nutrient variables affect the absorption and scattering properties of incoming light (Dekker *et al*., 1996) and therefore affecting the reflectance. This is the first time the short wave infra red band has been used to estimate nutrient concentrations in a reservoir in Zimbabwe. The band showed a high correlation with the nitrates. This relationship needs to be explored further. Most studies have implored the use of the red, green and near infra red bands (Cannizaro and Carder, 2006; Han and Jordan, 2005; Dalu *et al.*, 2015). Remote sensing however was not useful for predicting reactive phosphorus. This may be because Mazvikadei is an oligotrophic reservoir defined by its low N and P levels. The use of remote sensing in determining nutrient status of reservoirs may therefore be more useful for mesotrophic and eutrophic reservoirs.

Since the relationship between the nutrient variables and reflectance is not stable over time the use of remote sensing for operational water quality monitoring over time in Mazvikadei reservoir cannot be employed. The use of remote sensing for predicting nutrient variables is a much better way of monitoring water quality than measuring the amount of chlorophyll *a* when the aquatic system has already been degraded. Measuring the nutrient levels in reservoirs allows accurate monitoring of the changes in reservoirs which enables resource managers to employ remedial action. The effect of suspended sediments or colored dissolved organic matter on the remotely sensed data was not employed. Further errors may also have originated from haze and shadow covering some of the sites in the reservoir area which affected the reflectance values. In order to fully understand the reservoir pattern more data should be generated during the rainy season.

5 GENERAL DISCUSSION

This study has provided insights into how the physico-chemical variables and fauna and flora of medium sized reservoirs change over time after the filling phase. The reservoir did not show any apparent stratification soon after the filling phase and currently it only exhibits a weak stratification in the first 5 m in the cool dry season. Nutrient concentrations varied throughout the sampling period under the influence of seasonal factors. The physico-chemical variables have changed since the filling phase of the reservoir.

The study provides information on the phytoplankton community which can be used for future comparison studies in Mazvikadei reservoir. Due to the limited sampling period, a typical seasonal plankton successional pattern was not clearly shown however the chlorophytes and the bacillariophytes were the most dominant in the reservoir. The increase in the species richness in the zooplankton community from the filling phase to the present suggests that the zooplankton community has stabilized since the filling phase.

The study also explored if any relationship exists between nutrient variables and reflectance and if it exists whether it is stable over time. Previous studies in remote sensing have employed the use of chlorophyll *a* and turbidity to measure water quality in reservoirs. This however is the end point of aquatic degradation. It would therefore be better to use nutrient variables to assess water quality as an early warning of pollution in reservoirs. It was noted that the best time to use this approach for this reservoir was in the cool dry season. There is still much to be done in demonstrating the suitability of remote sensing in monitoring nutrient levels for long term water quality monitoring for mesotrophic and eutrophic reservoirs. It is recommended that farming activities around Mazvikadei reservoir be monitored to reduce the problems of nutrient loads from ammonia fertilizers from the catchment into the reservoir. Whilst there remains some gaps in the knowledge with regards to the specific driving mechanisms of processes in oligotrophic tropical reservoirs the study has made a contribution to the knowledge of the limnology and aspects of plankton ecology in oligotrophic reservoirs.

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