CHAPTER ONE

1.1 Introduction

Southern Africa is characterized by large variations in annual rainfall, very few perennial rivers and lacks natural lakes with the exception of a few temporary pans and coastal lakes. This has necessitated the construction of dams across many rivers to ensure adequate water supply (Thornton, 1987). These dams have led to the creation of reservoirs of varying sizes and the largest Lake Kariba was built for hydroelectric power generation. Dam building in Zimbabwe started in 1902 with the construction of Matopos dam and in the 1970's the process gathered momentum with widespread development of commercial agriculture. Locally and elsewhere the term dam has become synonymous with reservoir and will be used as such throughout this dissertation.

The International Commission on Large dams (ICOLD, 1998) estimates that there are approximately 800 000 small dams worldwide. About 50% of reservoirs are used for irrigation, 20% for hydroelectric power generation and the rest for flood control as well as domestic and industrial water supply and recreation. Classification of dams has been attempted on the basis of factors such as morphometry, hydrology, geomorphology (Marshall, 1994) but it has been difficult to come up with a common definition. For this study a small dam was any water body less than 10 ha in surface area.

Zimbabwe has a particularly large number of small dams estimated at 14 000 covering an area of about 126 089 ha. This is 86% of the number of small dams in Southern Africa excluding South Africa and about 93% of the total surface area in the region covered by small dams yet Zimbabwe contributes only 6.8% of the geographic area of Southern Africa (Ersal, 1994). Hence density (number per unit area) is high compared to other countries. About 80% of the 10 747 listed dams in Zimbabwe are 5 ha or less in surface area and constitute 10% of the total area covered by water bodies in the country (Marshall and Maes, 1994) (Figure 1.1). Most of these small dams are in commercial farming areas (61%), with 25% in communal areas and 14% in resettlement areas. This distribution reflects the high demand for water where crop farming and cattle ranching are carried out on a large scale (Figure 1.1).

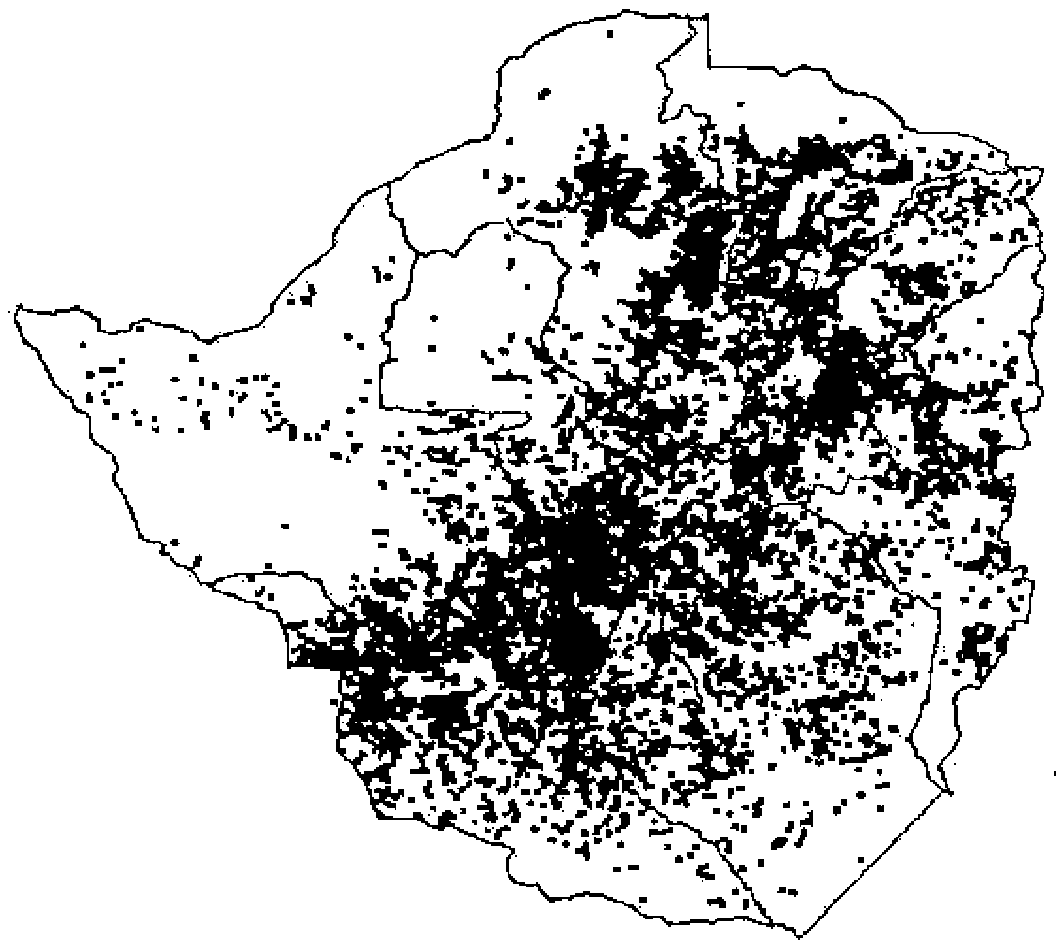


Figure 1.1. Distribution of dams in Zimbabwe (Ersal, 1994).

Most of the small dams are built across seasonal rivers (intermittent streams) which dry up during the dry season and flood during the wet season. When this is integrated with losses of water due to abstraction, evaporation and groundwater seepage, pronounced fluctuations in water level and capacity sometimes up to 95% reduction of volume occur which has an effect on the limnology. The reduction in surface area in such water bodies will depend on their surface-area-to-volume ratio but generally most of the shallow zones are desiccated completely during this period (Thornton, 1987).

A historical bias in limnological investigations towards larger water bodies like Lake Kariba or Chivero has resulted in small dams not being studied in Zimbabwe. Fluctuations in water level can cause wide variations in physical and chemical variables in larger water bodies such as Lake Chilwa (Moss and Moss, 1968; McLachlan, 1969; Morgan and Kalk, 1970; McLachlan *et al*., 1972), Lake Kariba (McLachlan, 1970), Lake Kainji (Bidwell, 1976), Lake Chivero (Marshall, 1978a,b; Marshall and Falconer, 1973), Lake Murray, Papua New Guinea (Osborne *et al*., 1987) and these are almost inevitably more extreme in small dams. Zooplankton has been studied extensively in larger dams in Zimbabwe such as the work done on Cleveland dam (Elenbaas and Grundel, 1994), Lake Chivero (Munro, 1966; Thornton and Taussig, 1982; Magadza, 1994) and Lake Kariba (Begg, 1974a, 1976; Mills, 1977; Magadza, 1980; Masundire, 1991, 1992, 1994; Green, 1985; Marshall, 1991, 1997). A few studies have been done on phytoplankton in large dams such as Lakes Chivero (Munro, 1966; Robarts *et al*., 1982) and Kariba (Thomasson, 1965; Hancock, 1979; Ramberg, 1985, 1987; Cronberg, 1997).

Much less work has been done on plankton in small dams in the country, apart from a list of phytoplankton and zooplankton species occurring in five small impoundments in the Eastern Highlands (Thornton and Cotterill, 1978) and a list of zooplankton in eighteen small dams in Marondera and Nyanga (Green, 1990). The biology of fish in small dams has been little studied although McGown (1969), Kenmuir (1981, 1982), Maar (1956), Marshall and Maes (1994) and Sugunan (2000) made some observations mostly on fish production and the potential for aquaculture. This is in contrast to what is known about various aspects of fish biology and production in larger dams such as Chivero (Marshall and Lockett, 1975; Marshall, 1979 a & b, 1981, 1982 a, b & c; Cochrane, 1982; Mutsekwa, 1989; Moyo, 1997), Lake Manyame (Bowmaker, 1976; Mutsekwa, 1989) and Lake Kariba (Badenhuizen, 1967; Bowmaker, 1969, 1973, 1975; Coke, 1969; Balon, 1972, 1974; Bell-Cross and Bell-Cross, 1971; Begg, 1970, 1974 ,1976; Mitchell, 1976, 1978; Marshall, 1979 a & b, 1981, 1982 a & b, 1987; Marshall *et al*., 1982; Mtada, 1987; Hustler and Marshall, 1990; Moyo, 1990; Sanyanga *et al*., 1995; Chifamba, 1998), which support important fisheries.

The main aim of the study was to understand how small reservoir ecosystems function coupled with hydrological processes that are unique to them. This topic is often ignored in lake studies and large dams because of their more stable hydrological regimes but it is important in environments subject to pronounced and short-term changes in water level.

Objectives

The objectives of this study on small dams therefore were to first determine physico-chemical characteristics and the effects of fluctuations in water level over time. Changes in water and sediment chemistry determine water quality and will have immediate effects on the general productivity of the dams and their suitability as habitats for plankton, macrophytes and fish. Therefore the second objective was to study the phytoplankton and zooplankton communities of the dams, their diversity, succession, abundance and relationship to the water chemistry and hydrology of the dams. A third objective was to study the structure of the fish populations to determine which species were present, their abundance and how they cope with seasonal changes in potential food sources as well as water quality.

1.2 Study site

The work was done in two dams located on Ingwerati farm, about 20 km West of Harare, Zimbabwe, on the Mumwahuku River (17 ^{o}51'S: 30^{o}52'E). They were on the same stream thus draining the same catchment and having the same hydrology (Figure 1.2). The top dam was breached during the 1997-1998 rainy season and it was dry for a year before the commencement of this study. Two sampling stations, one located in the deepest part and another in the shallower water at the top end of each dam were selected (Figure 1.2).

The dams have a catchment area of 18 km^2 collectively and drain an area of mainly granitic soils although the dams themselves are built on a banded ironstone ridge (Baldock, 1991). Most of the catchment is uncultivated and consists of dambos, i.e. seasonally waterlogged valley grasslands dominated by grasses and sedges with very few trees (Whitlow, 1985). The water is used to irrigate crops and the Upper dam is drained into the Lower one as the water is used and so its level fluctuates more extensively. The Upper dam is about twice the surface area of the Lower one (Table 1.1). The mean depth was almost similar between the two dams. Volume development values suggest that the Upper dam has a more conical shape ($D_v < 1$) compared to the Lower one ($D_v > 1$) (Table 1.1).

Macrophytes grew rapidly when the reservoirs filled after the dry season and *Polygonum senegalense*, *Cyperus* sp., *Nymphaea* sp. and *Ludwigia adscendens* were the most important species. *Azolla filiculoides* appeared in the Lower dam during the 2000 rainy season while *Juncus* sp. and *Cyperus* sp. were found only in the shallows of the Upper dam. Submerged macrophytes such as *Vallisneria spiralis* were present only in the Upper dam. *Polygonum senegalense*, *Juncus* sp. and *Cyperus* sp. were able to survive on the exposed shores after the water level dropped during the dry season (Personal observations).

Table 1.1. Some morphometric characteristics of the two Mumwahuku dams at full capacity (D. Stewart, personal communication)

	Upper dam	Lower dam
Volume ($m^3 \times 10^4$)	7.33	2.93
Surface area (ha)	5.7	2.5
Maximum depth (z_m)	5.0	3.5
Mean depth (z)	1.3	1.2
Volume development (D_v)	0.78	1.03
Maximum length (m)	650	350

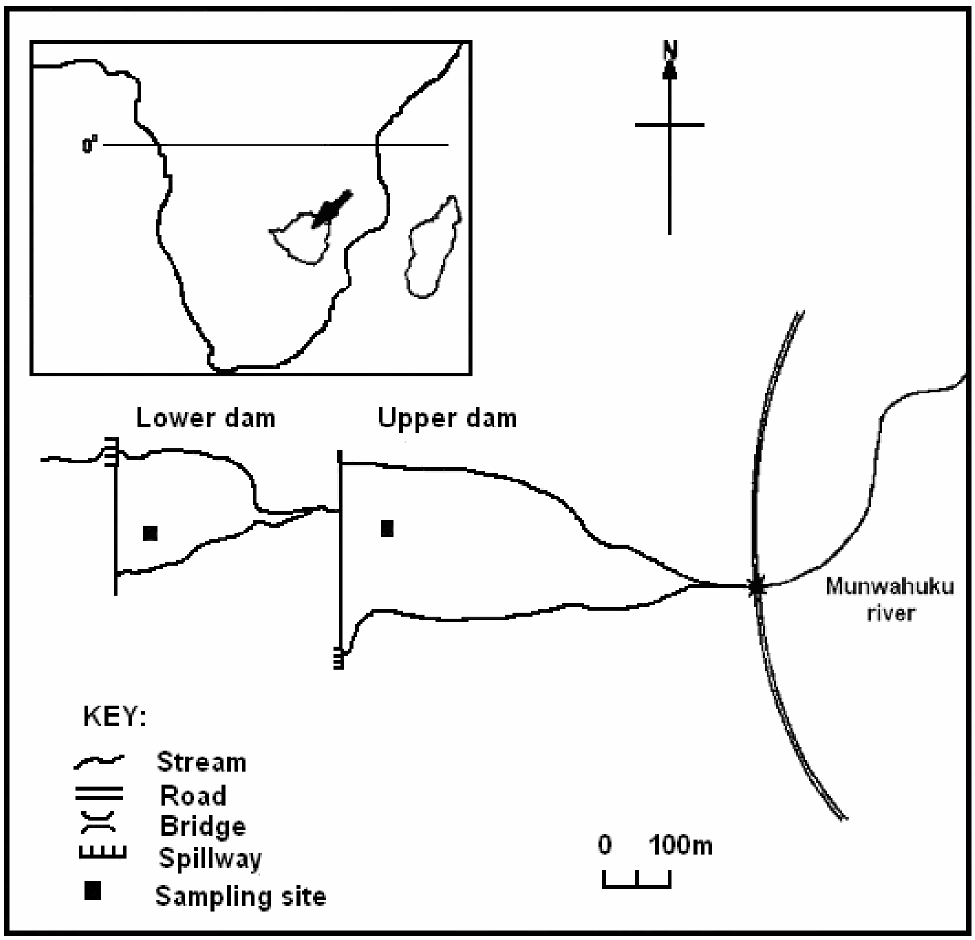


Figure1.2. Location of the Mumwahuku dams in Zimbabwe (insert) and the sampling stations in each dam

1.3. Climate and hydrology

The water level in the two dams was low, around 4% and 56% of the maximum depth in the upper and lower dams respectively at the beginning of the study in November 1999 but they filled to capacity in January 2001 following heavy rains that lasted until May (Figure 1.3). Both dams were full until August when their levels began to fall reaching their lowest level (10% and 77% of maximum depth in the upper and lower dams respectively) in December 2000 (Figure 1.3). The drop in water level in the lower dam was less severe because of the water discharged into it from the upper one.

The rainfall pattern of the 2000-01 season differed from the previous one with the rains ending abruptly in March. The dams filled by February but the levels of both dropped rapidly from May onwards because water was abstracted for winter crops and both dams were almost completely drained by September 2001. The mean monthly air temperatures were highest from October to February (22 °C) and lowest in June and July (13 °C) (Figure 1.3). Seasonal changes in the mean air temperature closely followed changes in solar radiation but there was an inverse relationship with sunshine hours (Figure 1.3) because of cloud cover during the wet season and its absence in winter. The prevailing winds were NEE for most of the year although NNE, SEE, SSE and E winds were recorded.

River flow like rainfall was highly seasonal such that by the end of June 2000 and April 2001 there were no more river inflows into the two dams. Munwahuku River in the two years of study only started flowing after heavy rains in January and normally both dams spilled within 24hrs of such an event. Direct inflows into the two small dams could not be measured because of some logistical problems. Estimates of runoff into the dams have been derived from runoff rates for the Muzururu catchment which was nearest to Munwahuku and was not affected by sewage effluent. It was assumed that rainfall patterns were uniform throughout the Lake Manyame catchment and in the Munwahuku subcatchment. Vegetation

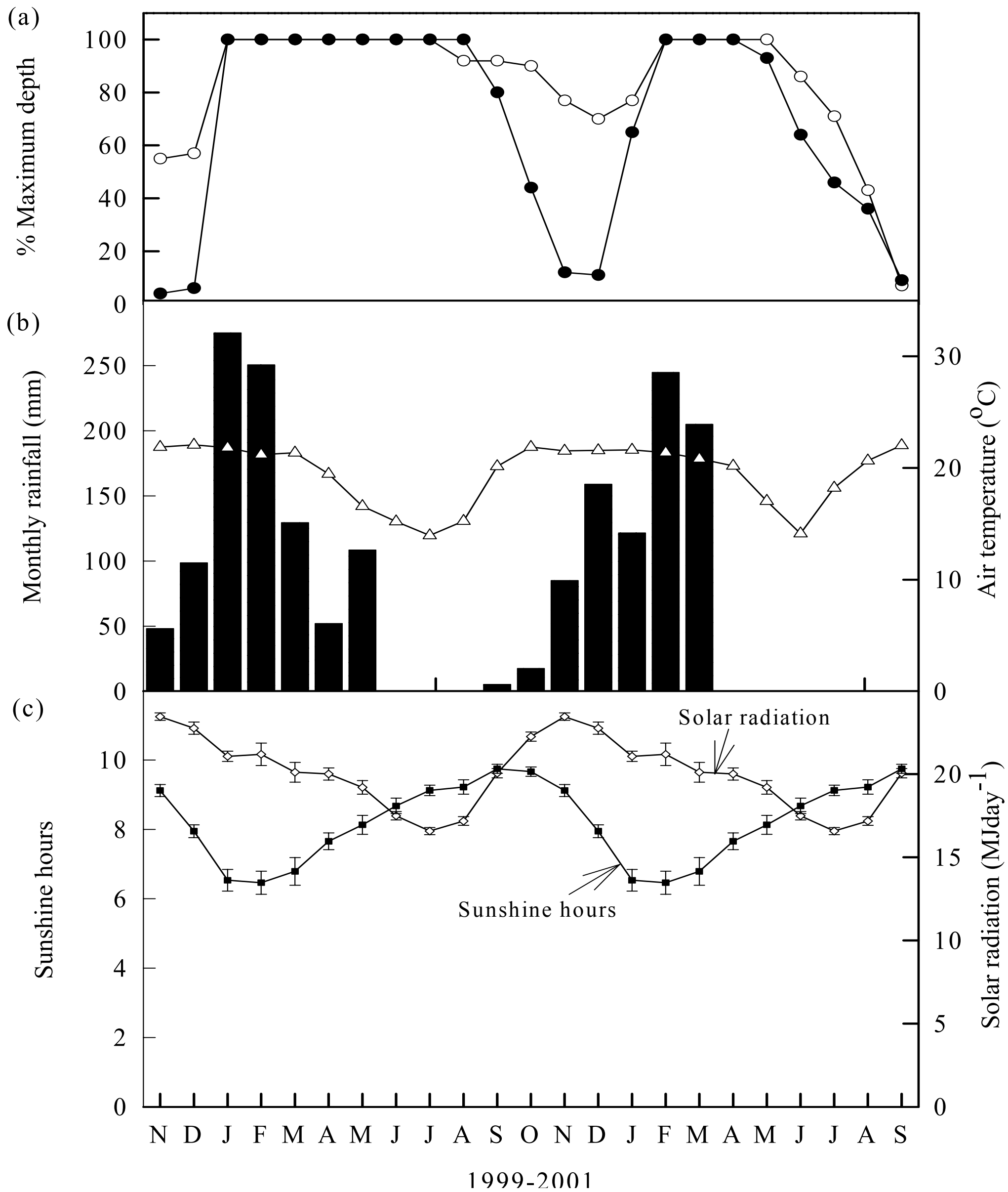


Figure 1.3. (a) Changes in water level (% maximum depth) in the two dams (Upper dam = ●; Lower dam ○); (b) monthly rainfall (mm) and mean monthly air temperature (Δ) during the study period; (c) mean monthly values for solar radiation (MJ day^{-1}) (◊) and sunshine hours (■) over a 20 year period recorded in Belvedere, Harare. (Error bars = SD). Data from the Metereological Department, Harare.

characteristics, slope, soils and other parameters that influence runoff were also assumed to be similar. It was assumed that all the water from the Upper dam goes into the Lower dam where it is removed by abstraction and overflow. This was only true for the year 2001 but not 2000 mainly because it was a drought year. It was also assumed that all the runoff from the catchment was either translated into river discharges or other forms of surface flow into the two dams.

Data from the Muzururu catchment spanned 21 years from 1973-1994 (Zimbabwe National Water Authority, 2004) from which the means for some of the variables were derived. The estimated runoff coefficient for the Munwahuku catchment was calculated by dividing mean total unit runoff (mm) by the annual rainfall received at the dams (906mm) to give a value of 0.181 or 18.1%. The runoff coefficient for the Munwahuku catchment was compared to that of other countries (Figure 1.4) and this gave a reasonable comparison with some of the major river basins in Southern Africa.

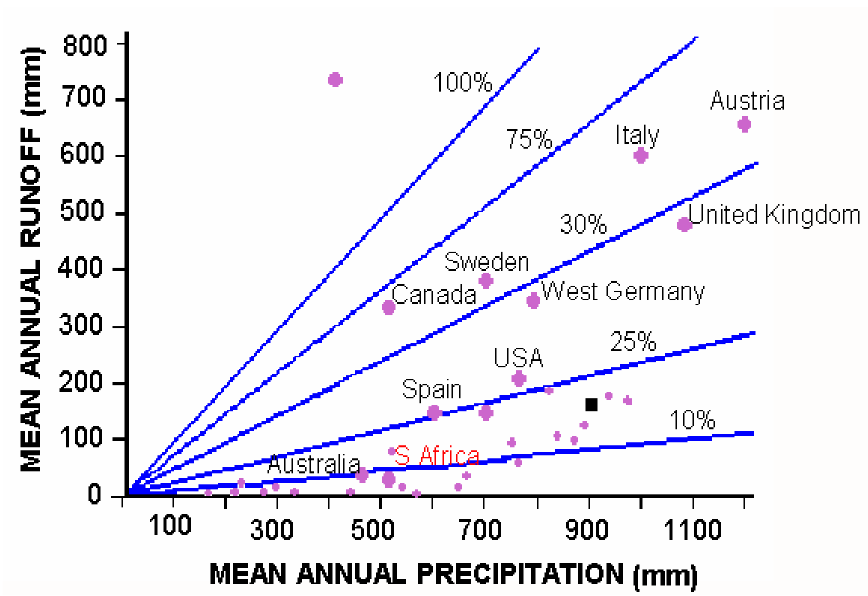


Figure 1.4. Mean annual rainfall (MAR) and mean annual precipitation (MAP) for selected countries in the Northern and Southern hemisphere (named) as well as major river basins in Southern Africa (small circles).The Munwahuku catchment indicated by (■). (Modified from Allanson *et al.*, 1990)

The mean annual total runoff for the Muzururu catchment (221km^2) was 3. 6366 * 10^7 m^3 per year and so a runoff rate of 450.83m^3 km^{-2} day^{-1}. This was a higher value compared to the mean annual runoff rate for the entire Lake Manyame catchment estimated to be 215 m^3 km^2 day^{-1} for the period 1985-1994 (Jica Report, 1996). The estimated runoff rate for the Munwahuku catchment (18km^2) was therefore 8. 115 * 10^{3}m^3 day^{-1}.The estimated theoretical water residence time for each of the two dams was calculated by dividing maximum volume by mean annual inflow rates to give:

Upper dam = 73 300m^3 / 8 115m^3 day^{-1} = 9.0 days

Lower dam = 29 300m^3 / 8 115m^3 day^{-1} = 3.6 days

The Upper dam has about 40 renewals per year and the Lower dam 101 per year. These residence times are very short compared to that of Lake Chivero (0.8 years or 9.6 months) (Marshall and Falconer, 1973) and a little more than 3 years for Lake Kariba (Coche, 1974). The hydrological regime of these two dams was that water residence time is less in the Lower dam than the Upper one and water level fluctuations more severe in the Upper dam compared to the Lower one.

CHAPTER TWO

SEASONAL AND DIURNAL STRATIFICATION REGIMES

2.1. Introduction

The stratification of a lentic system is the single most important factor regulating ITS biotic processes because it creates density differences that influence vertical mixing and regulates the distribution of chemical ions and suspended particles with respect to depth (Richerson, 1992). When a water body is stratified nutrient ions and particles including phytoplankton that sink from the epilimnion are retained and recycling is minimized by reduced vertical mixing. When stratification breaks down, they are returned to the surface in a cyclic process that influences the overall productivity of a water body.

Stratification in lentic systems in four time scales: interannual, annual, polymictic and diurnal (Richerson, 1992). Although the seasonal cycle of stratification has been investigated in numerous African waters little attention has been given to the cycle over shorter time scales. A daily cycle of stratification occurred in Lake George, Uganda (Ganf, 1974; Ganf and Horne, 1975) and in a small reservoir in Ghana (Thomas and Ratcliffe, 1973) but not in three Zimbabwean reservoirs (Mitchell and Marshall, 1974).

The importance of stability on diurnal and seasonal stratification and its effects on plankton distribution and productivity has been demonstrated in some tropical waters. Diurnal stratification was recorded in three shallow water bodies (Jebel Aulia reservoir, Sudan; lagoon near White Nile, Sudan; Pilkington bay (not open water) Lake Victoria) where the regular mixing of water affected the distribution of blue green algae and thus of photosynthetic activity (Talling, 1957). The breakdown of thermal stratification in Eleiyele reservoir (Ibadan, Nigeria) caused regular and seasonal fluctuations in nutrients and had a direct influence on zooplankton and phytoplankton

(Imevbore, 1965). The diurnal patterns of thermal stratification, photosynthesis and nitrogen fixation has resulted in a short limnological cycle in Lake George, Uganda and these rapid changes do not give sufficient time for the phytoplankton species composition to change in response to the changing environment (Ganf, 1974; Ganf and Horne, 1975). Consequently changes in species composition in the lake are highly unlikely. A cycle of thermal stratification during the day and destratification at night occurred throughout the year in Opi Lake (Nigeria) and was attributed to seasonal differences in lake depth, wind and air temperature (Hare and Carter, 1984).

Most small dams undergo drastic seasonal water level fluctuations due to short water residence times. The influence of such hydrology on their stability and stratification regimes is unknown. This chapter describes the results of an investigation of the seasonal and diurnal cycle of stratification in two small dams in Zimbabwe and considers the differences between them and larger ones.

2.2. Methods

Measurements were done once a month in each dam for 20 months from November 1999 to August 2001 from a sampling point in the deepest part of each reservoir. This sampling strategy did not consider spatial variations because of limitations of time hence one sampling point was selected for this aspect of the study. The seasonal pattern of stratification was determined by measuring temperature and dissolved oxygen at midday at 0.5-m intervals at the deepest station in each dam using a dissolved oxygen meter (WTW Oxi 330). The diurnal variations in temperature and dissolved oxygen were determined by taking readings at 4 hr intervals over 24 hrs in December 2000 and January 2001 and at 2 hr intervals in February and July 2001.

The stability of a water body is the quantity of work or mechanical energy required to mix the entire volume of water to a uniform temperature without subtraction or addition of heat (Idso, 1973). It quantifies the resistance of stratification to disruption by the wind and therefore the degree to which the hypolimnion is isolated from the epilimnion (Wetzel, 2000).

It was calculated using the formulae:

$$S = \frac{1}{V}\int_{0}^{z_m}(z - z_g)(1 - p_z)A_z dz \text{ and } z_g = \frac{1}{V}\int_{0}^{z_m} zA_z dz$$

where S = stability (g-cm cm^{-2}), p_z = density of water at depth z, calculated using the temperature-density relationship of water at a pressure of 1 atmosphere, z_m = maximum depth of water column (m), z_g = depth to the centre of gravity of the lake (m), z = depth of thermocline, V = volume of water body, A_z = area enclosed by contour at depth of depth z (Hutchinson, 1957).

The depth-area hypsographs of the two dams are illustrated in Figure 2.1.

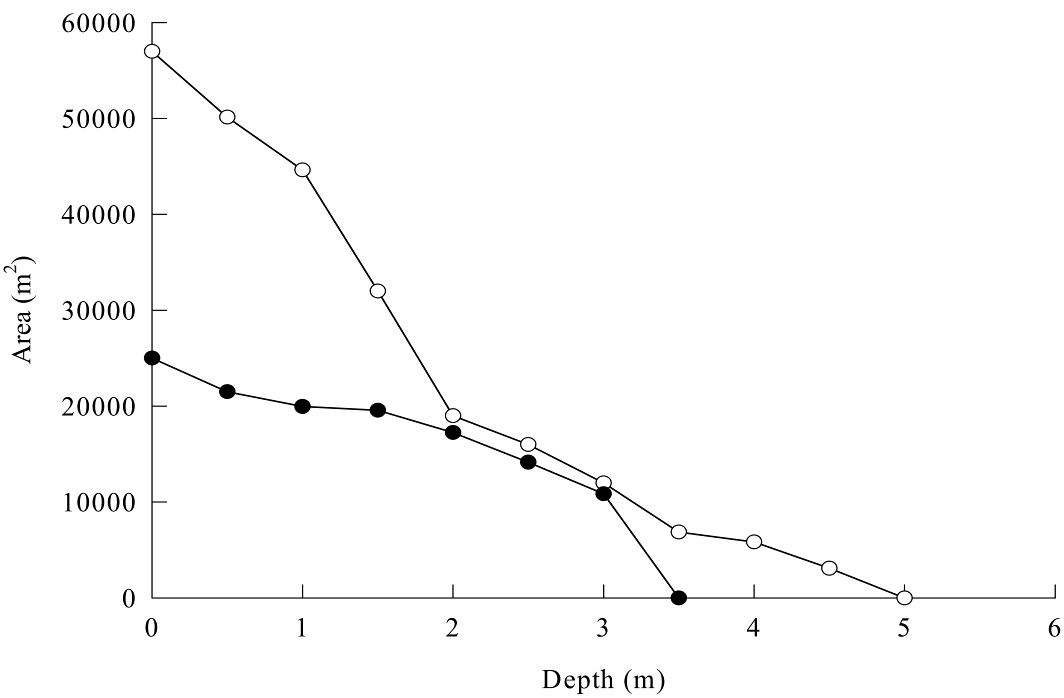


Figure 2.1. Hypsographic (depth-area) curves of the Upper dam (○) and the Lower dam (●) on Mumwahuku river

2.3. Results

Seasonal variation in temperature and dissolved oxygen

Thermal stratification was established in both dams during the summer (October – December) and was weakly established during the main rainy season period (January-April) (Figure 2.2). It then broke down completely in winter (May – September) (Figure 2.2). In summer the mean temperature of the water column was higher when the water levels were low than when they were high. For example, in December 1999 the mean temperature of the top dam was 25 °C but in January 2000 it was 22.5 °C. The pattern of oxygen stratification was similar with oxygen concentrations in summer falling to < 1.0 mgl^{-1} at the bottom (Figure 2.3). Stability was high in the Upper dam when it was full (January 2000 to June 2000) ranging from13.1 to 55.4 gcm cm^{-2}. It dropped to below 1 gcm cm^{-2} from July 2000 to December and started increasing in February 2001 to reach a value of 11 gcm cm^{-2} in April after which it declined again (Figure 2.4). The pattern in the Lower dam was different with stability values being generally below 10 gcm cm^{-2} from December 1999 to November 2000. Stability increased in December 2000 to 40 gcm cm^{-2} but thereafter declined to about 5 gcm cm^{-2} in April. Thereafter stability gradually declined to previous levels (Figure 2.4).

Diurnal variation in temperature and dissolved oxygen

The first 24-hour sampling was done on 17 December 2000 when the water level was low in both dams and there was no inflow of water into either of them. Both dams were thermally stratified during the day and early evening but stratification broke down at night and was only reestablished the next day at about 1100 hrs (Figure 2.5). This pattern was related to the air temperature which reached 31 °C at midday but fell to 16 °C at midnight. There were substantial temporal differences between the Upper and Lower dams with

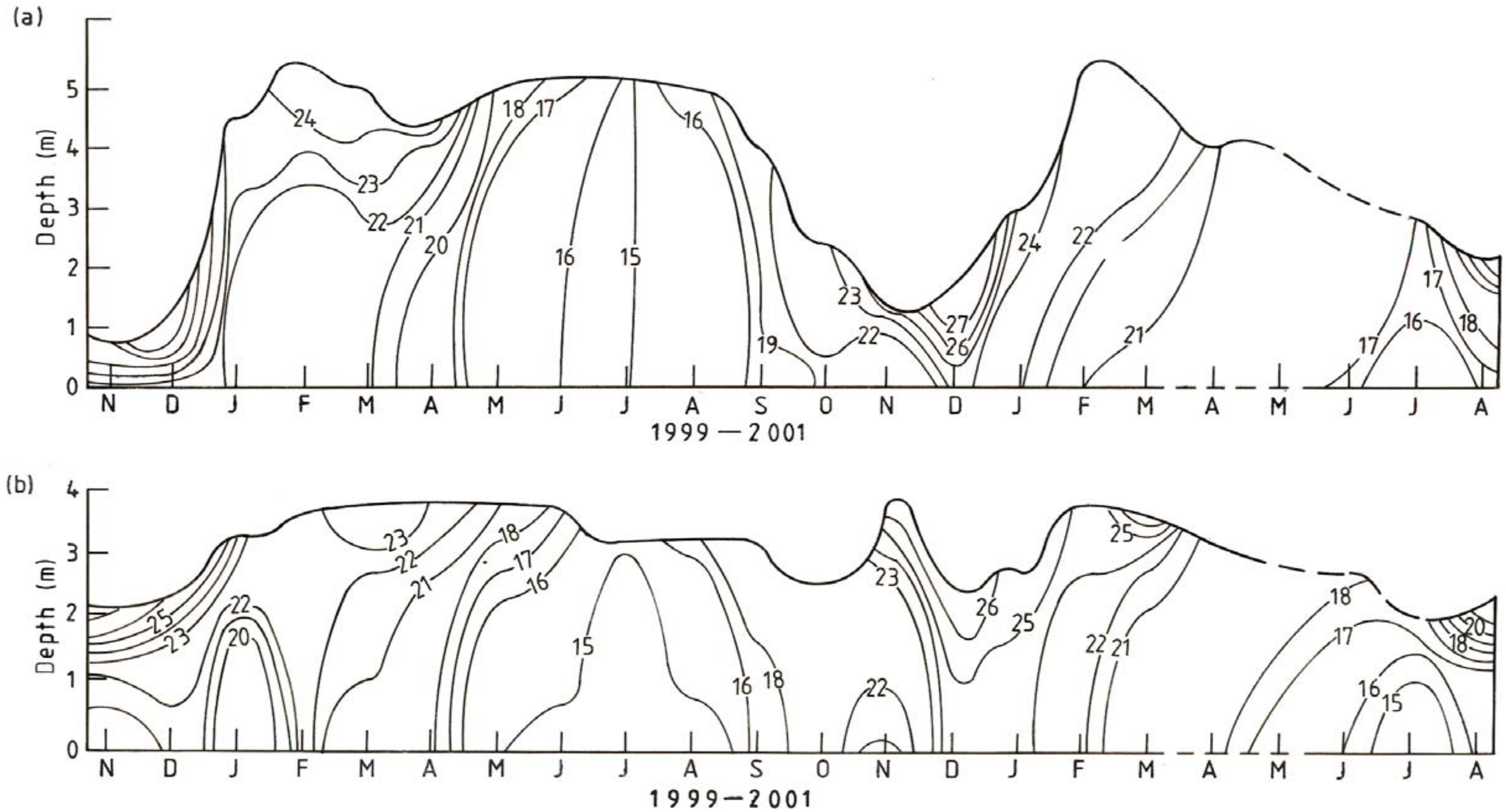
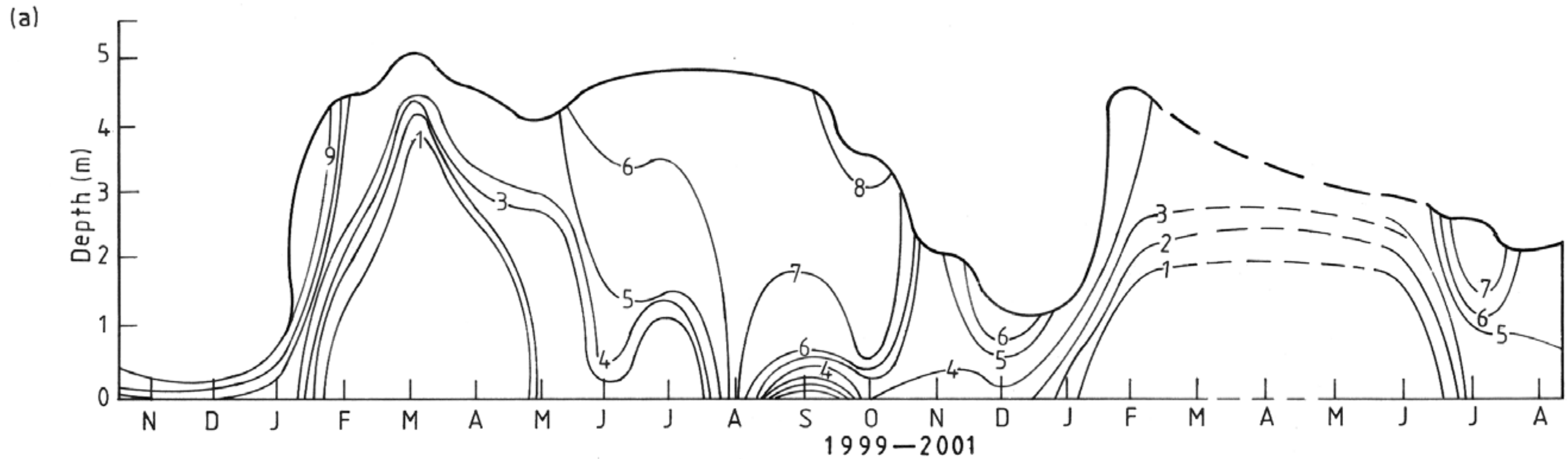


Figure 2.2. Isotherms (^{o}C) in the (a) Upper dam and (b) the Lower dam, November 1999 to August 2001 (Dashed lines = no measurements taken).



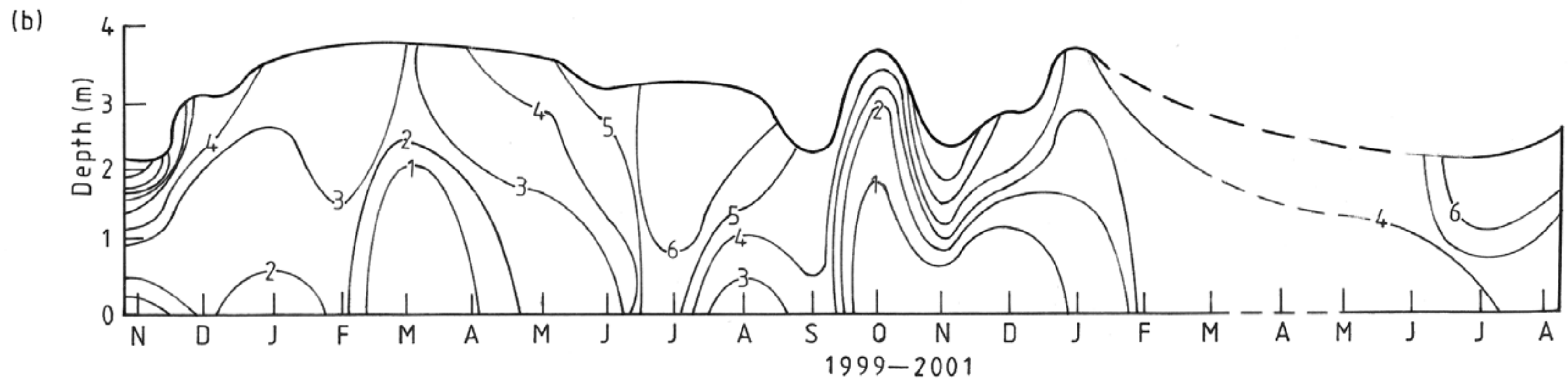


Figure 2.3. Dissolved oxygen isopleths in the Upper dam (a) and the Lower dam (b) from November 1999 to August 2001 (Dashed Lines = no measurements taken).

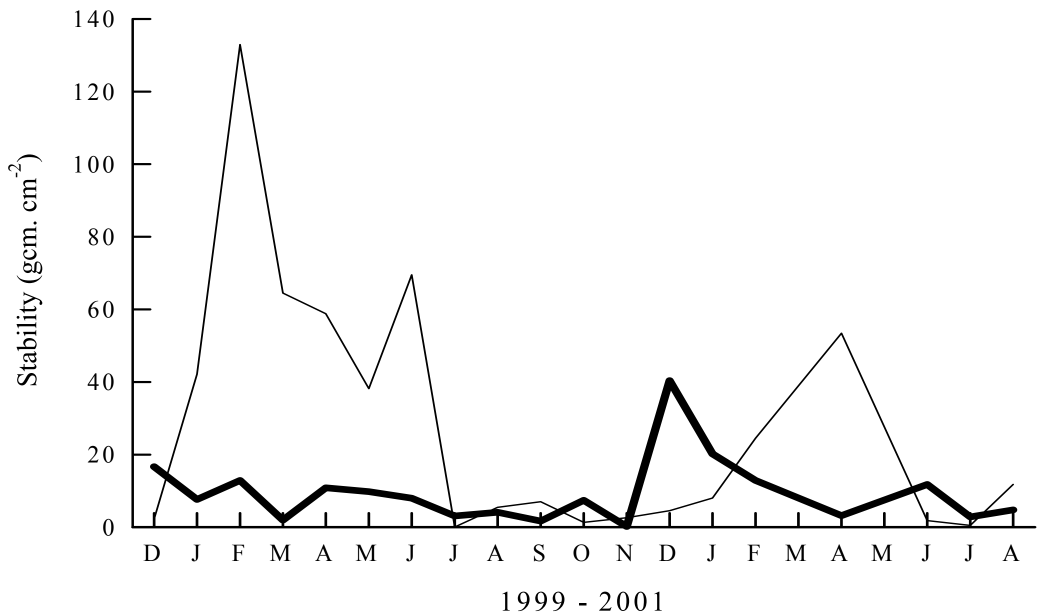


Figure 2.4. Changes in stability (gcm cm^{-2}) in the two dams, from December 1999 to August 2001. (Thin line = Upper dam; thick line = Lower dam)

Table 2.1. The stability (gcm cm^{-2}) of the water column in the two dams over 24 hrs in December 2000, January, February and July 2001 (UD = Upper dam; LD = Lower dam). * = No measurements done

	Dec 2000		Jan 2001		Feb 2001		Jul 2001	
Time(hrs)	UD	LD	UD	LD	UD	LD	UD	LD
1200	0.20	0.77	0.76	3.95	5.18	0.91	0.18	0.173
1400	*	*	*	*	24.07	6.18	0.37	0.64
1600	0.33	2.17	0.28	3.11	42.50	7.73	0.24	0.26
1800	*	*	*	*	34.57	6.18	0.49	0.43
2000	0.26	0.57	0.22	2.01	42.50	5.52	0.24	0.19
2200	*	*	*	*	28.81	16.33	0.06	0.09
0000	*	*	0.18	0.17	*	*	0.01	0.05
0200	*	*	*	*	23.77	3.75	0.18	0.03
0400	0.01	0.56	0.08	0.05	18.73	3.75	0.12	0.02
0600	*	*	*	*	21.38	2.71	0.43	0.01
0800	0.02	1.15	0.42	0.03	18.14	1.42	0.12	0.02
1000	*	*	*	*	19.09	1.80	0.01	0.19
1200	0.01	2.55	12.29	0.28	14.90	16.59	0.06	0.42
Mean	0.14	1.29	2.03	1.37	24.47	6.07	0.19	0.19
SD	0.14	0.86	4.53	1.64	11.06	5.29	0.16	0.20

destratification occurring 5-6 hours later (at about 0400hrs) than in the Upper dam (at about 2300hrs) (Figure 2.5). In the Upper dam there was a remarkable temporal disparity between temperature and dissolved oxygen cycles but there was concordance in the Lower dam (Figure 2.5).

The pattern of dissolved oxygen followed that of temperature being stratified in both dams during the day and breaking down during the night. The oxygen content in the whole water column fell in both dams after destratification occurred. Stability reached a peak at midday in the Lower dam (2.55 gcm cm^{-2}) but not in the Upper dam (Table 2.1). The variation in stability over 24hrs was 0.32 and 1.98 gcm cm^{-2} in the Upper and Lower dams, respectively (Table 2.1).

The next 24-hour sampling (15 January 2001) was done during the wet season when both dams were filling up and temperature stratification was weakly established and characterized by very small temperature gradients (Figure 2.6). The diurnal variation in air temperature was smaller (11 °C compared to 15 °C) than in December and the water was strikingly cooler with a mean of 21.7 °C and 20.5 °C in the Upper and Lower dams respectively. Oxygen stratification was pronounced and the concentration of dissolved oxygen fell to <1.0 mg l^{-1} from a depth of almost 2 m & 1.5m in the Upper and Lower dams respectively during the day (Figure 2.6). In the Upper dam this stratification broke down around 0200hrs and for a short time oxygen level improved in the bottom waters. This stratification slowly reestablished from about 0600hrs with oxygen levels below 2m falling again to <1.0m (Figure 2.6). Stability was higher in the Upper dam (mean = 2.03 gcm cm^{-2}) than in the Lower one (1.37 gcm cm^{-2}) but it fell in both dams between midnight and 0800 hrs (Table 2.1). The variation in stability over 24hrs was 12.21 and 3.92 gcm cm^{-2} in the Upper and Lower dams respectively (Table 2.1). Both dams were full on 16 February 2001 and there

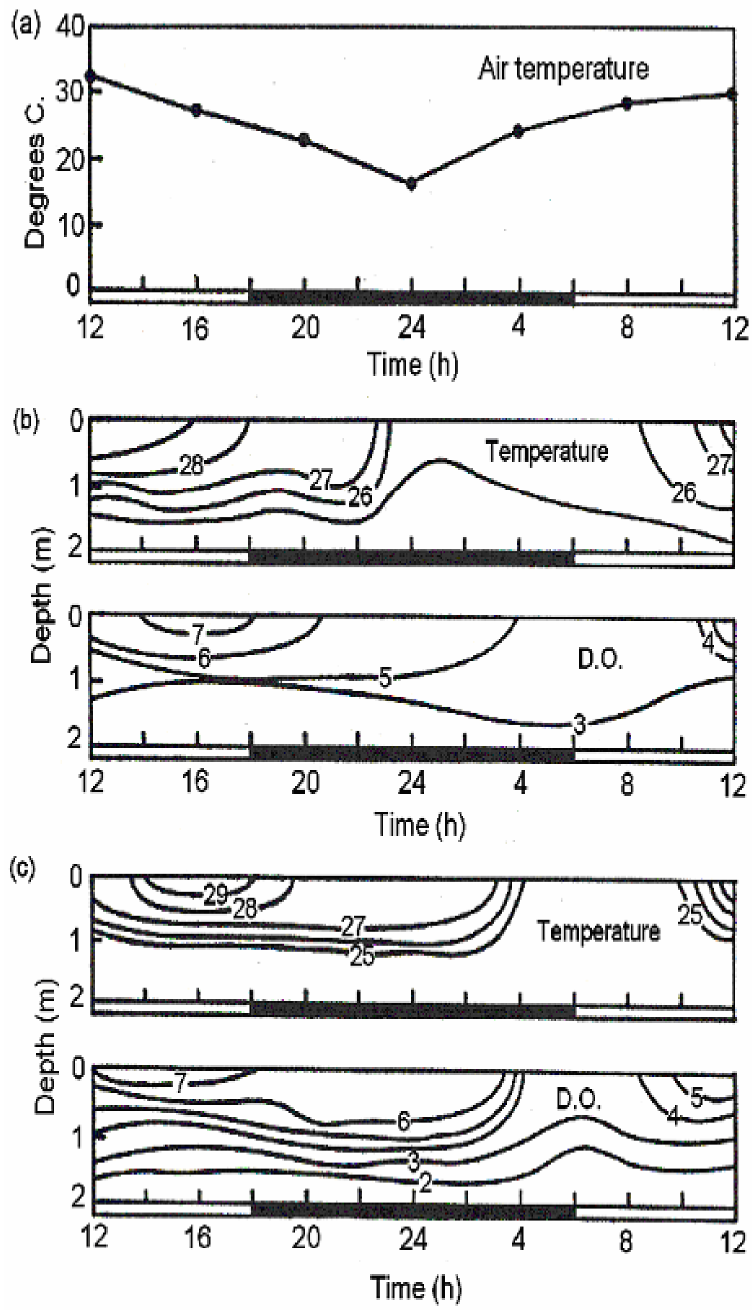


Figure 2.5. Diurnal variation in air and water temperature (^oC) and dissolved oxygen (mg l^{-1}) in the Upper and Lower dams on 17 December 2000

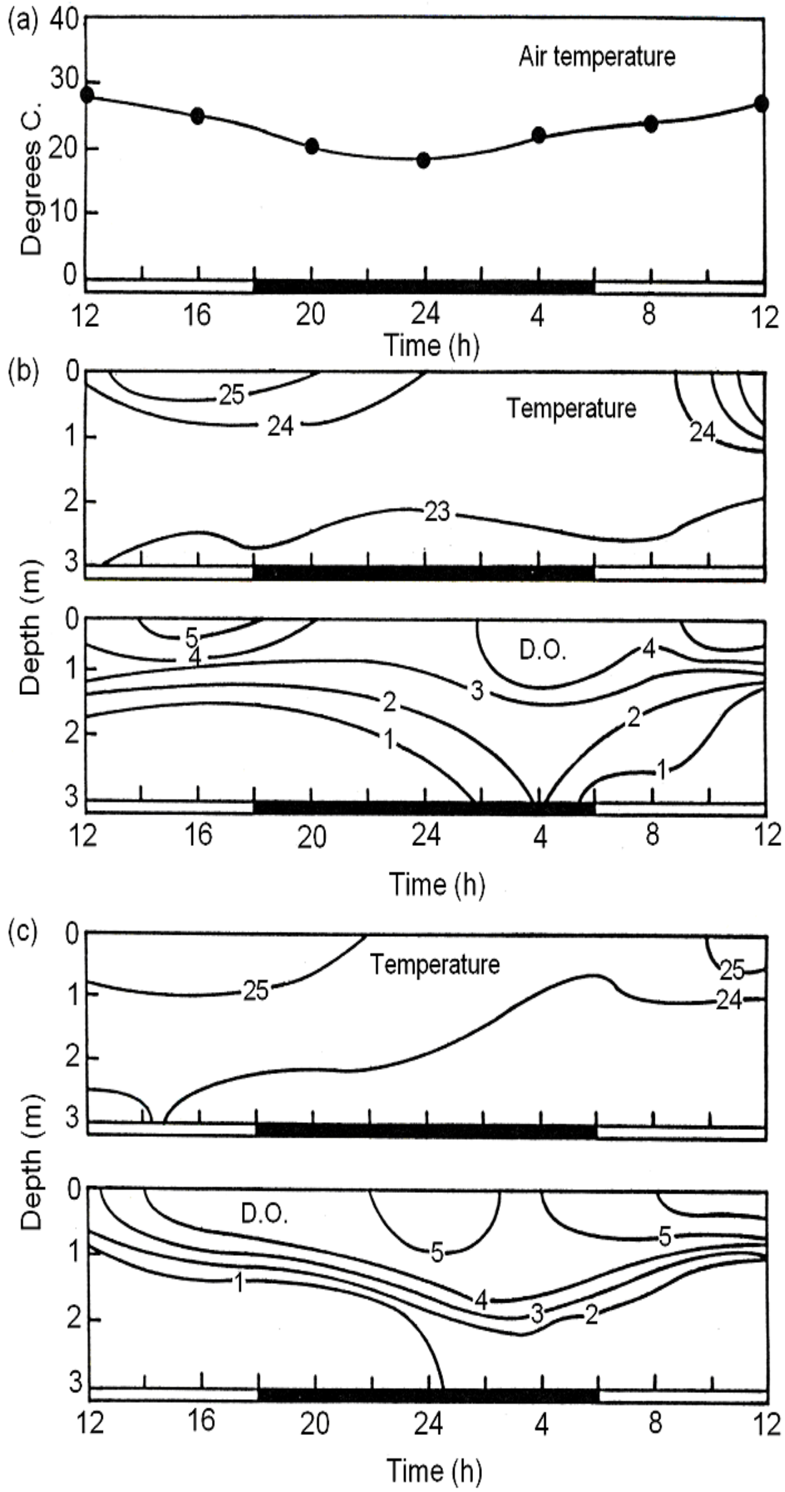


Figure 2.6. Diurnal variation in air and water temperature (^{o}C) and dissolved oxygen (mg l^{-1}) in the Upper and Lower dams on 15 January 2001

was weak thermal stratification. The temperature gradients were small; 4 °C (21 °C-25 °C) in the Upper dam and 3 °C (22 °C-25 °C) in the Lower one and stratification broke down between 0600 and 0800hrs when both were isothermal (Figure 2.7). Oxygen stratification in the two dams followed that of temperature, being weakly stratified between 1200 and 2400 hrs with deoxygenation near the bottom beginning at 0200 hrs. By 0400 hrs the anoxic water reached to 1m at 0400 hrs in the Lower dam, and to 1.75 m at 0600 hrs in the Upper dam (Figure 2.7). In both dams stability was greatest during the day, falling to its lowest values between 0200 and 0800 hrs but it was still much higher than it had been in the previous months (Table 2.1). Stability was relatively high (mean = 24.47 and 6.07 gcm cm^{-2}) and it varied by 37.32 and 15.69 gcm cm^{-2} in the upper and lower dams respectively (Table 2.1).

The last sample (10 July 2001) was done in winter when water levels had begun to fall, mostly in the Upper dam. The air temperature ranged from 7 °C at midnight to 21 °C at midday, giving a temperature range of 16 °C, which was greater than on any of the three previous occasions. There was no thermal stratification in either dam although the water warmed up by about 2 °C during the day (Figure 2.8) but there was pronounced oxygen stratification during the day and evening in the Upper dam. It was especially well developed between 2000 and 2400 hrs when the oxygen concentration ranged from >7 mg l^{-1} at the surface to <1 mg l^{-1} at the bottom.

This stratification broke down after midnight and the oxygen concentration was around 6 mg l^{-1} throughout the water column. The pattern was less obvious in the Lower dam and at times the differences between top and bottom were almost 2 mg l^{-1} (Figure 2.8). Stability was very low (mean = 0.19 gcm cm^{-2} in both dams) even during the day (Table 2.1) and less variable in both dams (Upper dam = 0.48 gcm cm^{-2}; Lower dam = 0.62 gcm cm^{-2}) than in any of the previous samples (Table 2.1).

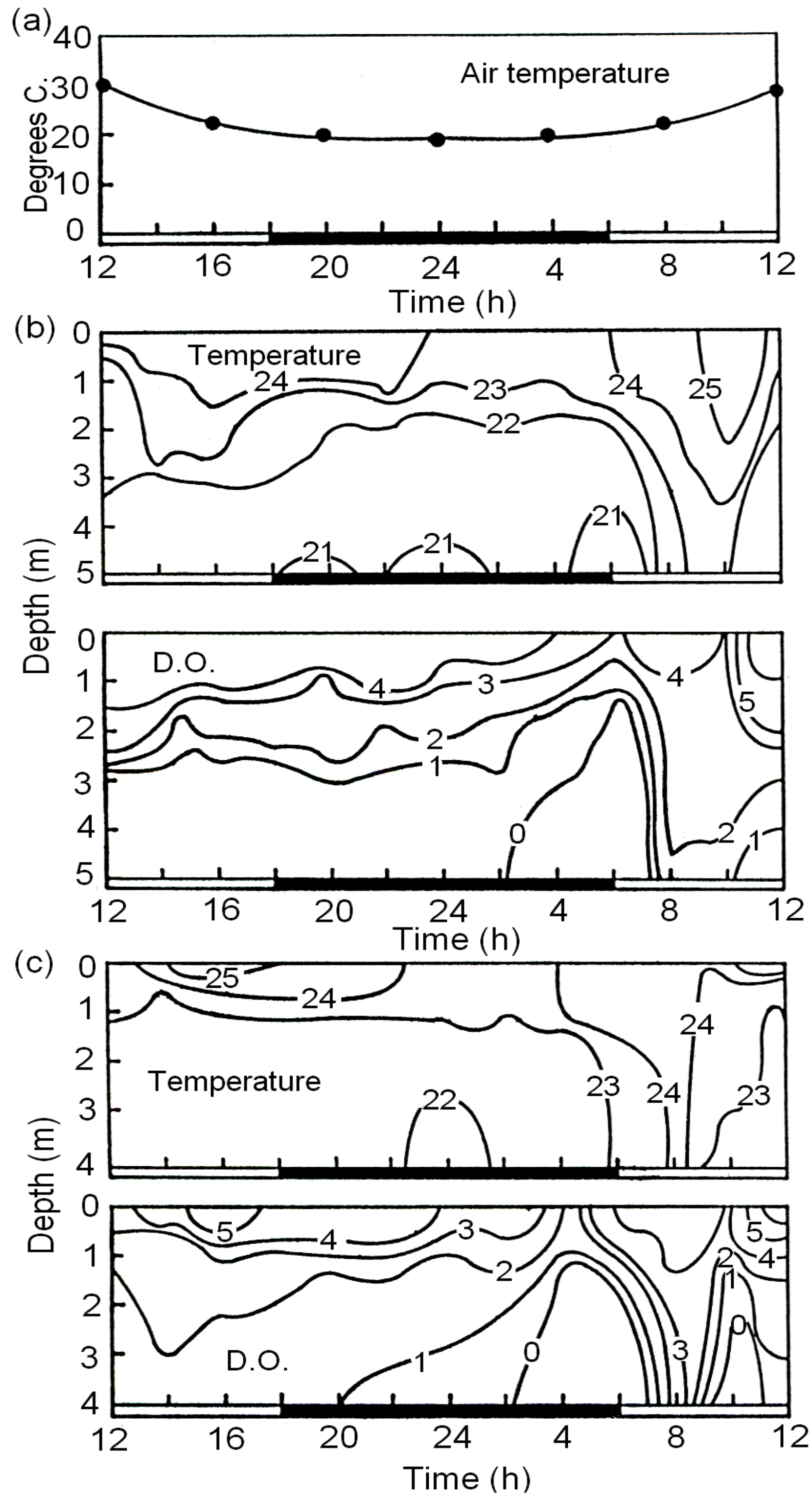


Figure 2.7. Diurnal variation in air and water temperature (^{o}C) and dissolved oxygen (mg l^{-1}) in the Upper and Lower dams on 16 February 2001

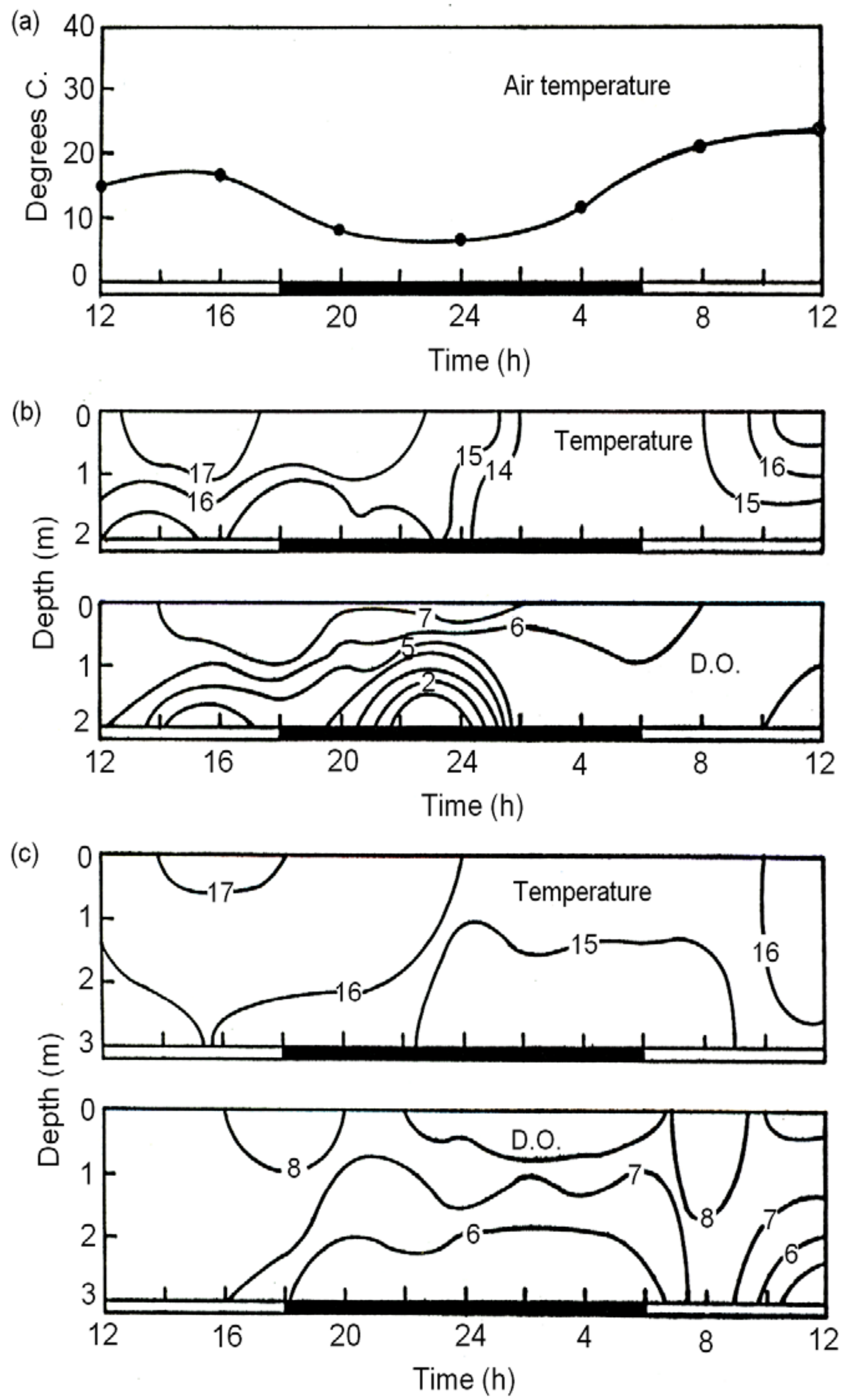


Figure 2.8. Diurnal variation in air and water temperature (°C) and dissolved oxygen (mg l^{-1}) in the Upper and Lower dams on 10 July 2001

2.4. Discussion

The occurrence of a diurnal cycle of temperature and oxygen stratification was an important finding of this study. The daily discontinuity of short-wave solar radiation income is the main cause of diel cycles of heat storage and thermal stratification (Talling, 1990) modified by wind regimes. Thus whilst temperatures were high in summer thermal stratification was firmer during daytime than in winter. Weak diel thermal and oxygen stratification during winter was mainly a result of lower air temperatures hence thermal gradients within the water column were also small. This cycle was observed in both dams with minor differences hence the effects of drawdown were not a major factor between them since diel thermal stratification occurred even when water levels were at their lowest. The flow-through regimes in the dams were not determined directly but evidence suggests that they could be the main factor that caused poor thermal stratification during the main rainy season from January to April in both dams.

Diel oxygen stratification was established in the dams as well. The amplitude of changes in oxygen concentration principally depends on the rates of photosynthetic production and respiratory removal (*in situ*) by the primary producers (Talling, 1957). In the case of algae their vertical distribution is in turn determined mainly by changes in thermal stratification and since neither primary production nor phytoplankton distribution were determined we can just speculate on likely scenarios. In both dams it was observed for the months of January and Februaury that when isothermal conditions occurred oxygen levels in the bottom waters rose as well usually in the early morning period emphasizing the role of thermal stratification in facilitating mixing of water and thus oxygen distribution in the dams.

Deoxygenation of bottom waters in both dams was observed in January-February 2001 while oxygen levels were relatively higher in the months of December and July. An oxygen deficit near the sediment indicates that oxygen demands are high and the opposite is true. These deficits are likely to be a result of increased oxygen demand by heterotrophic bacteria rapidly processing organic material deposited by river inflows hence the observation during the main rainy season. The seasonal importation of allochthonous organic material during the rainy season is a characteristic feature of most tropical reservoirs (Imevbore, 1967) but since the quality of water entering the small dams was not determined we can only speculate that similar effects could be possible subject to more investigations. During the dry season when shorelines became exposed in both dams (to a greater extent in the Upper dam); terrestrial vegetation grew on exposed sediment and was flooded when the dams filled. This could have been another important source of organic matter as it decomposed promoting intense bacteriological activity leading to oxygen deficits. The absence of such oxygen deficits in July could suggest that processing of organic matter occurs rapidly and bacteriological activity declines or it could be an effect of lower water temperatures during winter because temperature is a major determinant of the rates of processes such as decomposition.

A possible explanation for the small differences between the dams in terms of their stratification regimes is that the Upper and Lower dams are on the same scale in terms of size (i.e. surface area & volume) irrespective of changes due to drawdown. Thus the influence of external fluxes such as solar radiation income and wind run would bring about similar changes in both dams. Diel heat changes are inevitably less buffered in small volumes hence the occurrence of the observed thermal stratification in the small dams. Another contributory factor is that most small dams tend to be shallow (mean depth < 2m) and so the influence of external fluxes on a shallow water body would bring about greater changes in stratification patterns than in larger lakes (Talling, 2001).

The seasonal cycle in these two small dams were similar to those in larger Zimbabwean reservoirs, ranging from 0.042 km^2 to >5000 km^2 (Coche, 1974; Mitchell and Marshall, 1974). The diurnal cycle of temperature and oxygen stratification was in marked contrast to the larger reservoirs where there was little change between day and night (Mitchell and Marshall, 1974). An exception to this was the strong deoxygenation that occurred in Lake Chivero in 1996, but this was a special case because of the lake's severe organic pollution at that time (Moyo, 1997). In larger reservoirs for example Lake Kariba (Coche, 1974), Chivero, Mazoe and Mwenje (Mitchell and Marshall, 1974), stable and predictable seasonal stratification patterns were observed. A major outcome of these diel stratification patterns in small dams is that they have very low stability (max. 135 gcm cm^{-2} = Upper dam and 40 gcm cm^{-2} = Lower dam) compared to larger water bodies such as Lake Kariba where stability in Basin IV (lacustrine conditions; area = 2 563km^2) was 2 577 gcm cm^{-2} (Coche, 1974), Lake Pyramid, Nevada with an area 532 km^2 (S = 7 – 10 000 gcm cm^{-2}) and Lake Atitlan, Guatemala with an area of 136.9 km^2 (S = 21 500 gcm cm^{-2}) (Hutchinson, 1957). Therefore the work that needs to be done to circulate the small dams is obviously much less compared to large dams. The Lower dam was generally less stable compared to the Upper one and the reason for this could be the frequent discharges from the Upper dam with the effect of preventing stable stratification.

The cycle of diel thermal and oxygen stratification has implications for the exchange of nutrients between sediments and the water. It is likely that nutrient exchange is rapid and frequent in small dams and so their sediments may never quite take the role of a "nutrient sink". There is need for further investigation on sediment-water exchanges and how they are influenced by stratification regimes. Also further research on primary production, phytoplankton distribution, the effect of flow-through regimes on stability of stratification and the cycling of organic matter on dissolved oxygen levels would provide some of the answers to gaps in this study.

CHAPTER THREE

WATER AND ASPECTS OF SEDIMENT CHEMISTRY

3.1 Introduction

The limnology of large and medium-sized African reservoirs has been extensively studied (Allanson *et al*., 1990; Talling and Lemoalle, 1998) but relatively little is known about the limnology of small ones. These are numerous in some parts of the continent (Zimbabwe alone has 10 000 reservoirs < 5ha in extent; Marshall and Maes, 1994) and they have significant effects on aquatic ecosystems. These effects include changing the seasonal flow pattern of streams, and they often have extensive seasonal water level fluctuations that may influence their physical and chemical characteristics. They might also influence sediment characteristics and their interaction with the overlying water.

The effects of drought and water level fluctuations on physico-chemical variables have been investigated in a number of larger water bodies in Africa. In Lake Kariba Zimbabwe/Zambia), water level changes brought about major changes in nutrient concentrations that led to an immediate increase in the population of chironomid larvae (McLachlan and McLachlan, 1969; McLachlan, 1970). A similar phenomenon was noted in Lake Chivero, Zimbabwe (Marshall, 1978). The fall in water level in Lake Murray (Papua New Guinea) was accompanied by a marked increase in pH, conductivity, total hardness and total suspended solids (Osborne *et al*., 1987). Similar changes were recorded in the upper Okavango delta during low water levels (Hart, 1997) and in floodplain lakes such as Australian billabongs (Hillman, 1986; Bayley and Sparks, 1989). There have not been many documented cases of complete desiccation but a good example is the case of Lake Chilwa, Malawi, where changes in water chemistry especially conductivity, salinity and turbidity were drastic. Assessment was done as the lake dried out (Moss and Moss, 1968; Morgan and Kalk, 1970) and after its recovery from drought (McLachlan *et al*., 1972).

This study investigated the influence of water level fluctuations on the physical and chemical characteristic of the water and sediment of two small dams in Zimbabwe.

3.2. Methods

Samples and measurements were done around midday for 20 months (November 1999 to August 2001) in each reservoir from two sites one in the deepest part and the other the shallower end near the river mouth. A 3-litre Ruttner sampler was used to collect water samples from the bottom and through the water column at 0.5-m intervals from the two stations in each dam. Two sets of samples were collected and these were mixed (vertical series) and subsamples taken for field and laboratory analyses from this depth-integrated sample. Conductivity & total dissolved solids, pH, and transparency were measured with a conductivity meter (WTW LF330), a pH/mv meter (WTW pH 330) and a 20-cm diameter Secchi disc, respectively. Total suspended solids were estimated by filtering a known volume of water using Whatman Glass Fibre 47mm filters. These filter papers were dried at 105 °C for 5 hrs and the difference in mass before and after filtration was expressed as mg l^{-1}.

Biological oxygen demand was measured by incubating dark bottles in a water bath filled with water from the dams artificially saturated with oxygen for 5 days at 25 °C. Dissolved oxygen in the dark bottles was measured with an electronic probe (WTW Oxi 330) and BOD calculated by the difference between the initial oxygen level when the water was saturated and the level after five days. Alkalinity was determined by titration of the water sample against 0.01N hydrochloric acid with methyl red indicator (Mackereth *et al*., 1978).

Other chemical variables were measured with a Hach water analysis kit (DR/2010 portable data logging spectrophotometer) using filtered water samples (Whatman GF 47mm filters) except for total nitrogen and total phosphorus. The summary of each method is given but they are described in detail in Farber *et al*. (1960):

(a) Chemical oxygen demand - was determined by the reactor digestion method (Test no.8000). The sample was heated for two hours with a strong oxidizing agent, potassium dichromate. Oxidizable organic compounds reacted reducing the dichromate ion ($Cr_2O_7^{2-}$) to green chromic ion (Cr^{3+}). The amount of Cr^{6+} remaining was determined by measuring absorbance at 420nm. The COD reagent also contained mercury ions used to complex chloride interferences and silver ions acting as catalysts. The estimated detection limit (EDL) is 2mg l^{-1}. Precision is at a standard deviation of ± 2.7 mg l^{-1} COD.

(b) Nitrate-nitrogen – was determined by the chromotropic acid method (Test no. 10020). The nitrate in the sample reacted to give a yellow product with maximum absorbance at 410nm. Precision is at standard deviation ± 0.2 mg l^{-1} N.

(c) Nitrite – was determined by the diazotization (chromotropic acid) method (Test no. 8507). The nitrite in the sample reacted with sulphanilic acid to form an intermediate diazonium salt. This coupled with chromotropic acid to produce a pink colored complex directly proportional to the amount of nitrite present. Absorbance was measured at 507nm. Precision is at a standard deviation of ± 0.006 mg l^{-1} NO_2^- - N

(d) Ammonia – was determined by the salicylate method (Test no. 10023). Ammonia compounds in the sample combined with chlorine to form monochlorine. Monochlorine reacted with salicylate to form 5 – aminosalicylate. The 5- aminosalicylate was oxidized in the presence of a sodium nitroprusside catalyst to form a blue colored compound. The blue color was masked by the yellow color from the excess reagent present to give a final green colored solution. Absorbance was measured at 655nm. The precision levels are at standard deviation of ± 0.03mg l^{-1} N.

(e) *Total nitrogen* – was determined by the persulfate digestion method (Test no. 10071). The alkaline persulfate digestion converted all forms of nitrogen to nitrate. Sodium metabisulphate was added after the digestion to eliminate halogen oxide interferences. Nitrates then reacted with chromotropic acid under strongly acidic conditions to form a yellow complex with a maximum absorbance at 410nm. Precision is at standard deviation less than 1mg l^{-1} N.

(f) *Reactive phosphorus* - was determined by the PhosVer 3 method (Test no. 8048). Orthophosphate reacted with molybdate in an acid medium to produce a phosphomolybdate complex. Ascorbic acid then reduced the complex producing intense molybdenum blue color. Absorbance was measured at 890nm. Precision is at a standard deviation of ± 0.02 mg l^{-1} PO_4^{3-}.

(g) *Total phosphorus* - was determined by the PhosVer 3 with acid persulfate digestion method (Test no. 8190). The phosphates present in organic and condensed inorganic forms (meta-, pyro- or other polyphosphates) were 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 were converted to orthophosphates by the heating with acid and persulfate. Orthophosphate reacted with molybdate in an acid medium to produce a phosphomolybdate complex. Ascorbic acid then reduced the complex producing intense molybdenum blue color. Absorbance was measured at 890nm. The EDL for this test is 0.04 mg L-1 PO_4^{3-} and the precision level is ± 0.09 mg l^{-1} PO_4^{3-}.

Kruskal-Wallis tests ($p < 0.05$) were done to test for differences in physico-chemical characteristics (a) between the sampling stations (station 1 & 2) in each of the two dams (H_0 = no difference between the two sampling stations) and (b) between the upper and lower dams (H_0 = no difference between the two dams) for each physico –chemical variable. These analyses were for data covering the period December 1999 – August 2001. The computer software MINITAB (ver 3.0) was used.

Sediment samples were collected from November 2000 to August 2001 from two stations in each dam (deep water & shallow water station) with a Petersen dredge to assess possible differences between drawdown and permanently inundated zones and changes that occur when the exposed areas are flooded. These samples were analyzed by the Institute of Environmental Science at the University of Zimbabwe, using methods described by Okalebo *et al*. (1993). Variables that were determined included sediment texture (sand, silt, clay content), carbon content (%C), total phosphorus (%P), and total nitrogen (%N).

(a) Sediment texture was determined by the Bouyoucos or Hydrometer method. The method analyses particle size from which estimates of the sand, silt and clay content which is expressed as % by weight of oven-dry organic matter free soil (Okalebo *et al*., 1993).

(b) Organic carbon (%C) was determined by the sulphuric acid and aqueous potassium dichromate ($K_2Cr_2O_7$) mixture method. After complete oxidation of the sample from the heat of solution and external heating, the residual $K_2Cr_2O_7$ (in oxidation) was titrated against ferrous ammonium sulphate. The used $K_2Cr_2O_7$ which is the difference between added and residual $K_2Cr_2O_7$ gave a measure of organic carbon content of the sediment sample as % by weight of sample (Okalebo *et al.*, 1993).

(c) Total nitrogen and phosphorus – Analysis of total nutrients required complete oxidation of organic matter. This was accomplished through dry ashing or acid/alkaline digestion of the sediment. Wet acid oxidation (Kjeldahl oxidation) left a sulphuric acid solution to which hydrogen peroxide was added as an additional oxidizing agent, selenium was used as a catalyst while lithium sulphate was added to raise boiling point of mixture. Colorimetry was then used to determine P & N. Results are calculated as % content by weight of sample (Okalebo *et al*., 1993).

3.3. Results

Water chemistry

In both dams, conductivity was low (about 10 μS cm^{-1}) after the first rains in December 1999 (Figure 3.1a) but it increased afterwards to reach a peak of 160μS cm^{-1} in the bottom dam (in August 2000) and 140 μS cm^{-1} in the Upper dam (in October 2000). The pattern was repeated after the rains began in November 2000 when conductivity fell to 30 μS cm^{-1} rising to 120 μS cm^{-1} in February 2001. It dropped sharply in March following heavy rains, but rose to around 140μScm^{-1} until August 2001, after which both dams were more or less empty. The trend in total dissolved solids (Figure 3.1b) was similar to that of conductivity. There were two peaks for total suspended solids in both dams, in March 1999 and November-December 2000 ((Figure 3.1c), following heavy rains and sharp inflows of water. At other times, the concentration of total suspended solids was relatively low (<100 mg l^{-1}). Transparency (Figure 3.1d) was generally low during the rainy season because of suspended material carried in by the river but it rose during the winter months (May-September) after the river had ceased to flow and the suspended material settled out. It tended to be less in the Lower dam than the Upper one.

The pH (Figure 3.1e) ranged from 6.2 to 9.3 in both dams, with some periodicity being evident. Five clear pulses were observed during the 20 months and they could be related to algal abundance (Figure 3.1e). The total alkalinity fluctuated without any seasonal trends in both dams (Figure 3.1f). There seemed to be an annual cycle of biological oxygen demand in both dams with the highest values being recorded in December 1999 and December 2000-February 2001 (Figure 3.1g). The lowest values were recorded from May-September 2000 and in May 2001 but this trend were not maintained owing to the fact that both dams were drained rapidly from May 2001 onwards.

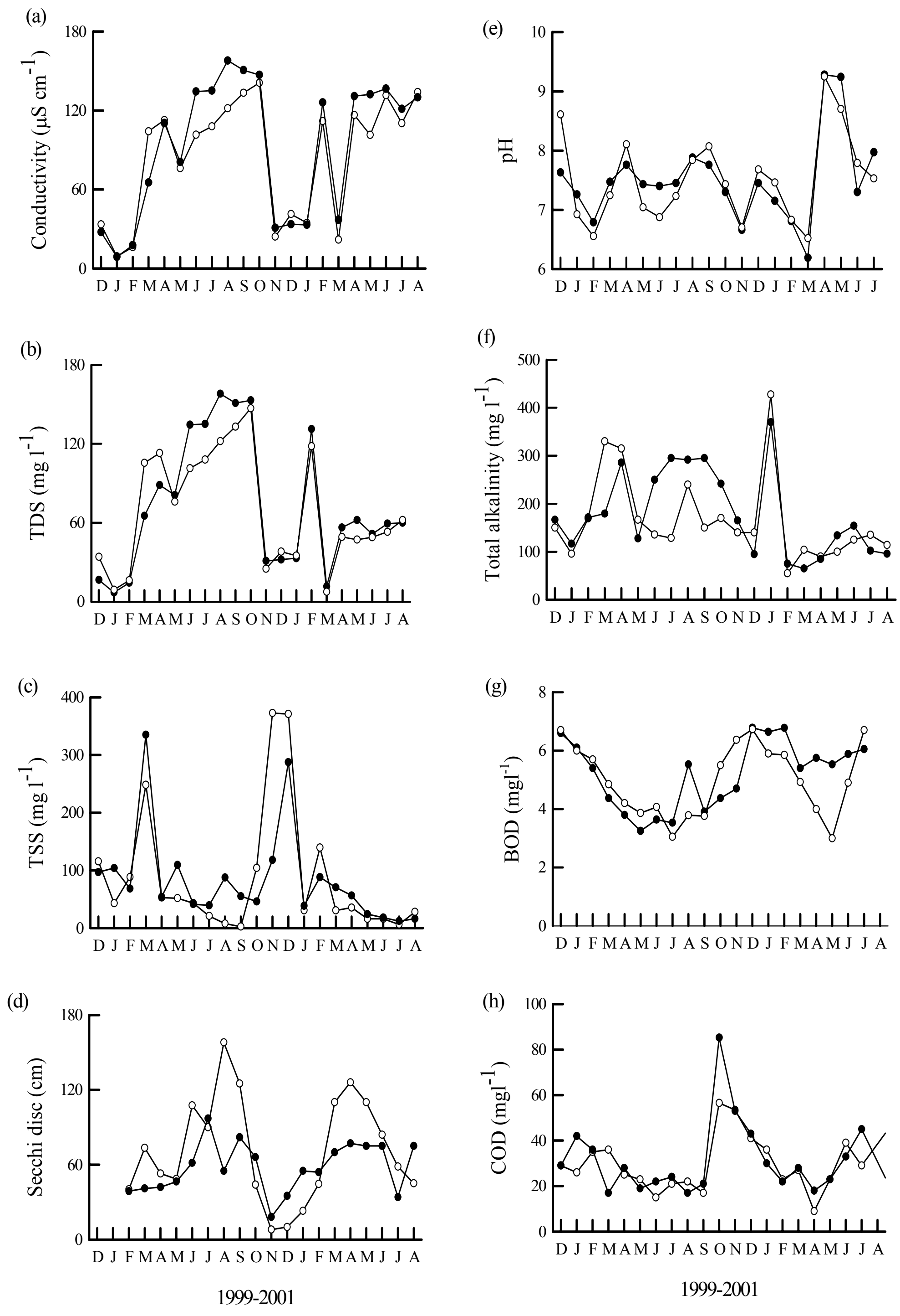


Figure 3.1. Changes in (a) conductivity (μScm^{-1}), (b) total dissolved solids (mg l^{-}), (c) total suspended solids (mg l^{-1}), (d) Secchi disc transparency (cm), (e) pH, (f) total alkalinity (mg l^{-1}), (g) biological oxygen demand (mg l^{-1}) and (h) chemical oxygen demand (mg l^{-1}) in the two dams from December 1999 to August 2001). (Upper dam = ○; Lower dam = ●)

The chemical oxygen demand was relatively stable in both dams except for a sharp peak in late 2000, when it rose from around 20 mg l^{-1} in September to 80 and 60 mg l^{-1} in the Upper and Lower dams respectively (Figure 3.1h).The concentrations of nitrate-nitrogen were similar in both dams, varying from about 0.1-1.0 mg l^{-1} (Figure 3.2a). There was a slight increase in January-March 2001 followed by a decrease with the lowest levels being recorded in May and June 2001, after which the concentrations increased. Ammonia concentrations were about one order of magnitude lower than nitrate and varied much more, from a minimum of 0.01 mg l^{-1} to a maximum of about 1.0 mg l^{-1} (Figure 3.2b).

Ammonia tended to decrease during the winter months (May-August) although there was no clear seasonal pattern. Nitrite concentrations were about one order of magnitude lower than ammonia, ranging from about 0.01-0.03 mg l^{-1} and fluctuated widely without any seasonal pattern although it was higher in the Upper dam most times (Figure 3.2c). The concentration of total nitrogen fluctuated widely, from about 0.3 to about 5.0 mg l^{-1}, in both dams but without any clear seasonal pattern (Figure 3.2d).

The concentration of reactive phosphorus was relatively low in both dams during the latter half of the 2000 rainy season and early dry season (February-July) but rose to a peak in September, after which it remained relatively stable at 0.5-1.0 mg l^{-1} (Figure 3.3a). From September 2000 to December 2000 reactive phosphorus was much higher in the Upper dam than the Lower one and again in March 2001 (Figure 3.3a). Total phosphorus varied much less, being around 0.15-1.5 mg l^{-1} for most of the year except for a sharp decrease in the Lower dam in June 2000 (Figure 3.3b). The Upper dam again had higher concentrations compared to the Lower one from October 2000 to December 2000 (Figure 3.3b). There was no seasonal pattern in the TN: TP ratio in both dam and it was well below 10 in both of them apart from peaks of 11.0 and 14.0 in the Lower dam in June 2000 and April 2001 (Figure

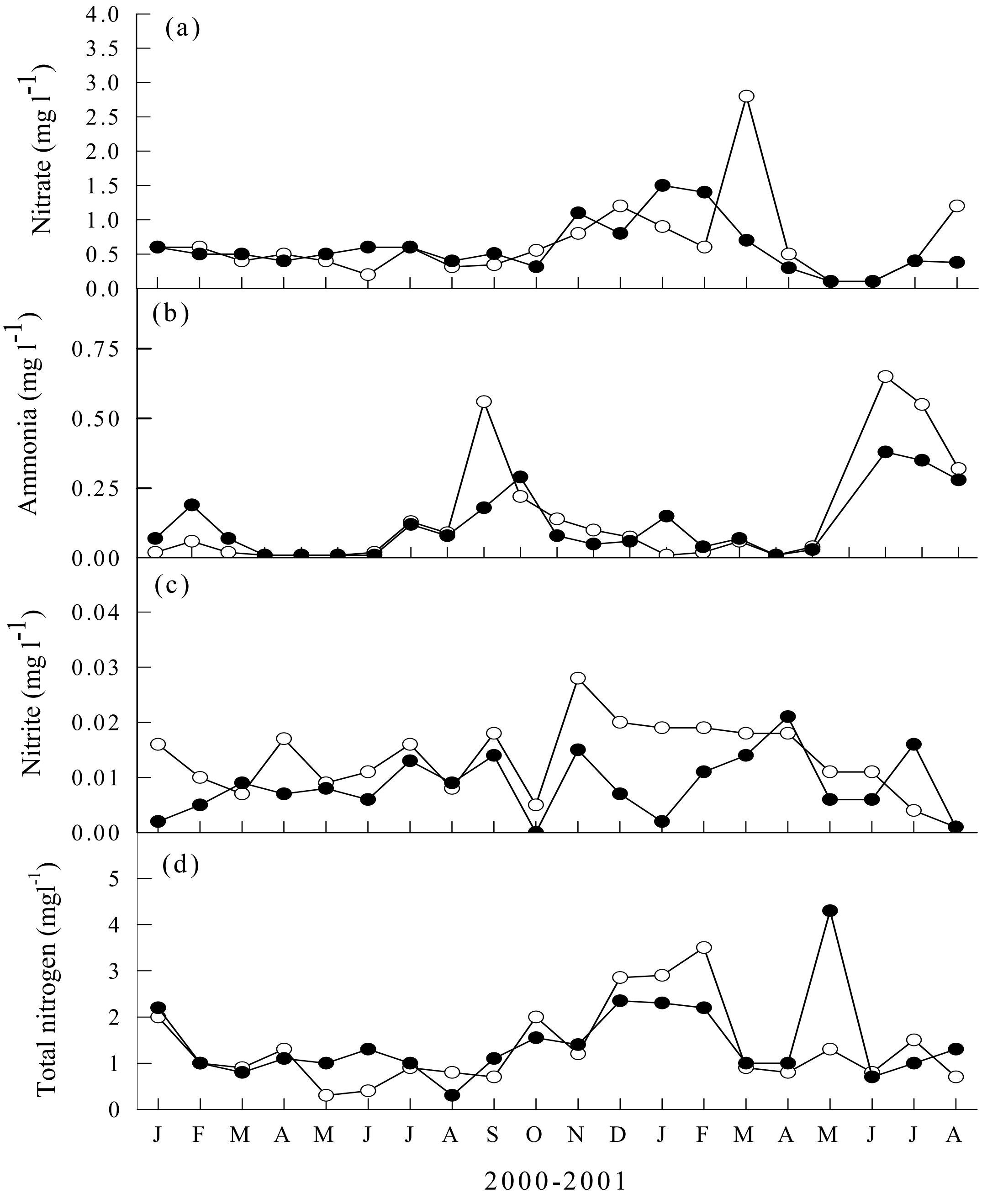


Figure 3.2. The concentration (mg l^{-1}) of (a) nitrate (b) ammonia and (c) nitrite (d) total nitrogen in the Upper (○) and Lower (●) dams, (January 2000 to August 2001).

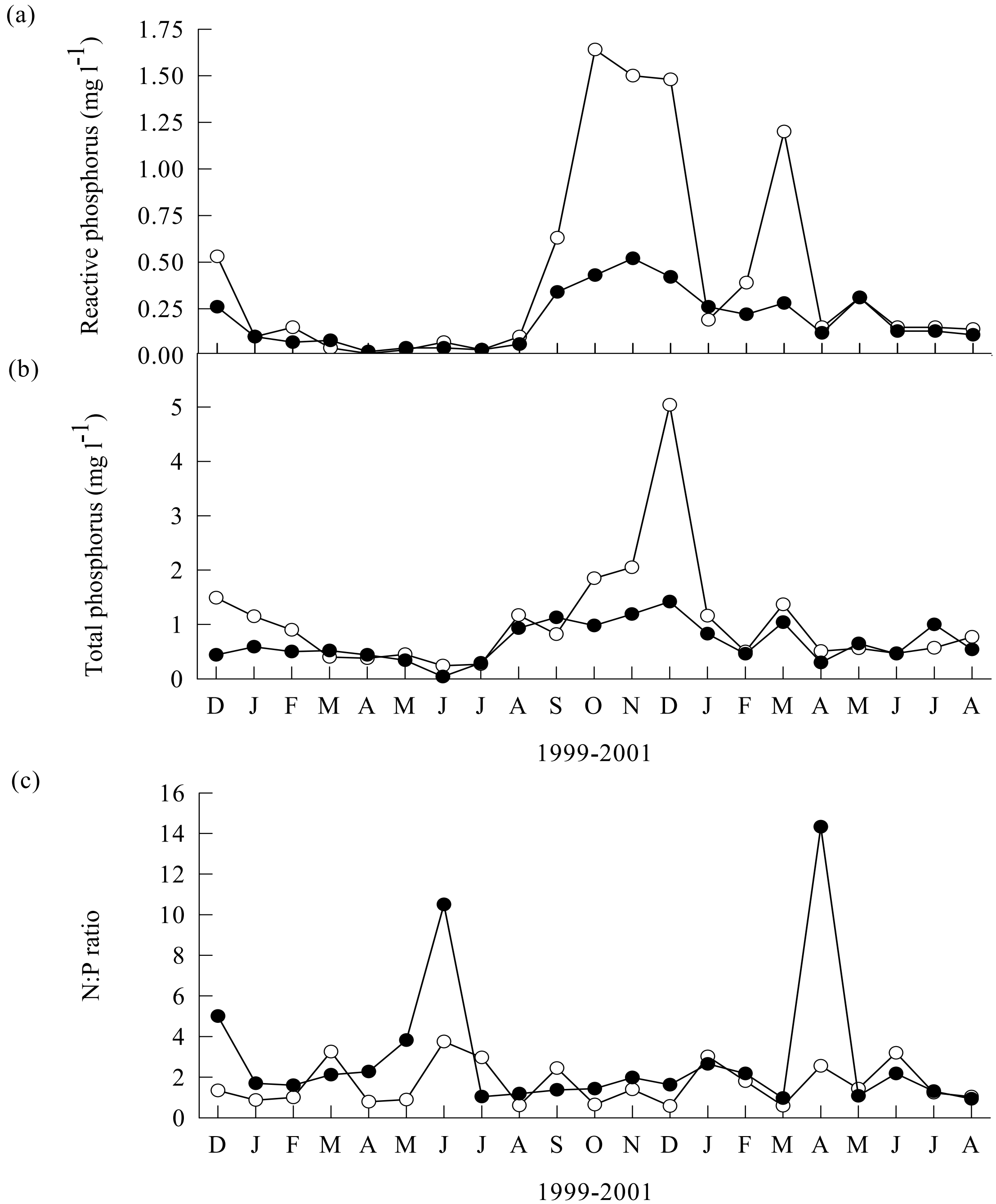


Figure 3.3. The concentration (mg l^{-1}) of (a) reactive phosphorus, (b) total phosphorus and (c) the N: P ratio in the Upper (○) and Lower (●) dams, December 1999 – August 2001.

Table 3.1. Kruskal-Wallis test for differences in physico-chemical characteristics between the sampling stations (station 1 & 2) in each of the two dams and test for differences between the Upper and Lower dams. (H = adjusted for ties). These data are for the period December 2000 – August 2001. None of the comparisons were significant ($p < 0.05$).

Variable	Upper dam (Station 1vs 2)	Lower dam (Station 1 vs 2)	Upper dam vs Lower dam (H = adjusted for ties)
Conductivity	H = 0.02 p = 0.892	H = 0.00 p = 0.957	H = 0.02 p = 0.892
pH	H = 0.29 p = 0.588	H = 1.38 p = 0.240	H = 0.29 p = 0.588
Total dissolved solids	H = 0.01 p = 0.903	H = 0.01 p = 0.935	H = 0.01 p = 0.903
Secchi disc transparency	H = 0.02 p = 0.892	H = 0.00 p = 0.968	H = 0.02 p =0.892
Total suspended solids	H = 0.00 p = 0.957	H = 0.00 p = 0.946	H = 0.00 p = 0.957
Chemical oxygen demand	H = 0.01 p = 0.903	H = 0.03 p = 0.871	H = 0.01 p = 0.903
Biological oxygen demand	H = 0.00 p = 1.000	H = 0.01 p = 0.925	H = 0.00 p = 1.000
Reactive phosphorus	H = 0.12 p = 0.725	H = 0.00 p = 0.957	H = 0.12 p = 0.725
Total phosphorus	H = 0.02 p = 0.882	H = 0.06 p = 0.808	H = 0.02 p = 0.882
Nitrate	H = 0.00 p = 0.989	H = 0.31 p = 0.579	H = 0.00 p = 0.989
Ammonia	H = 0.01 p = 0.903	H = 0.01 p = 0.935	H = 0.01 p = 0.903
Nitrite	H = 0.84 p = 0.361	H = 0.00 p = 1.000	H = 0.70 p = 0.361
Total nitrogen	H = 0.05 p = 0.828	H = 0.02 p = 0.892	H = 0.05 p = 0.828
Total alkalinity	H = 0.03 p = 0.860	H = 0.01 p = 0.935	H = 0.03 p = 0.860

3.3c). The two peaks corresponded to declines in total phosphorus in the Lower dam at that time. There were no significant differences ($p < 0.05$) between the two sampling stations in the Upper and Lower dams for all the variables that were measured (Table 3.1). When the two dams were compared there were no significant differences ($p < 0.05$) in the physico-chemical variables measured in the two dams (Table 3.1).

Assessment of sediment characteristics

The sediments were predominantly sandy. Sand content ranged from 83-93% in the dams and was higher in the Upper dam. In the Upper dam silt content was 1% & 5% and clay content was 6% & 2% in the deep and shallow water sediments respectively (Table 3.2). In the Lower dam silt content was 3% & 5 % and clay content was 14% & 10% in the deep and shallow water sediments respectively (Table 3.2). There was no monthly variation in the proportions of carbon, nitrogen and phosphorus in the two dams but the levels of organic carbon, nitrogen and phosphorus were higher in the Lower dam (Figure 3.4). Thus the sediments of the two dams differed considerably with organic carbon, nitrogen and phosphorus being 1.8, 1.5 and 3.0 times higher in the Lower dam than in the Upper one (Figure 3.4).

Table 3.2. Sediment texture (%) at deep and shallow water stations in the two dams (February 2001).

	Upper dam		Lower dam	
	Deep	Shallow	Deep	Shallow
% sand	93	93	83	85
% silt	1	5	3	5
% clay	6	2	14	10

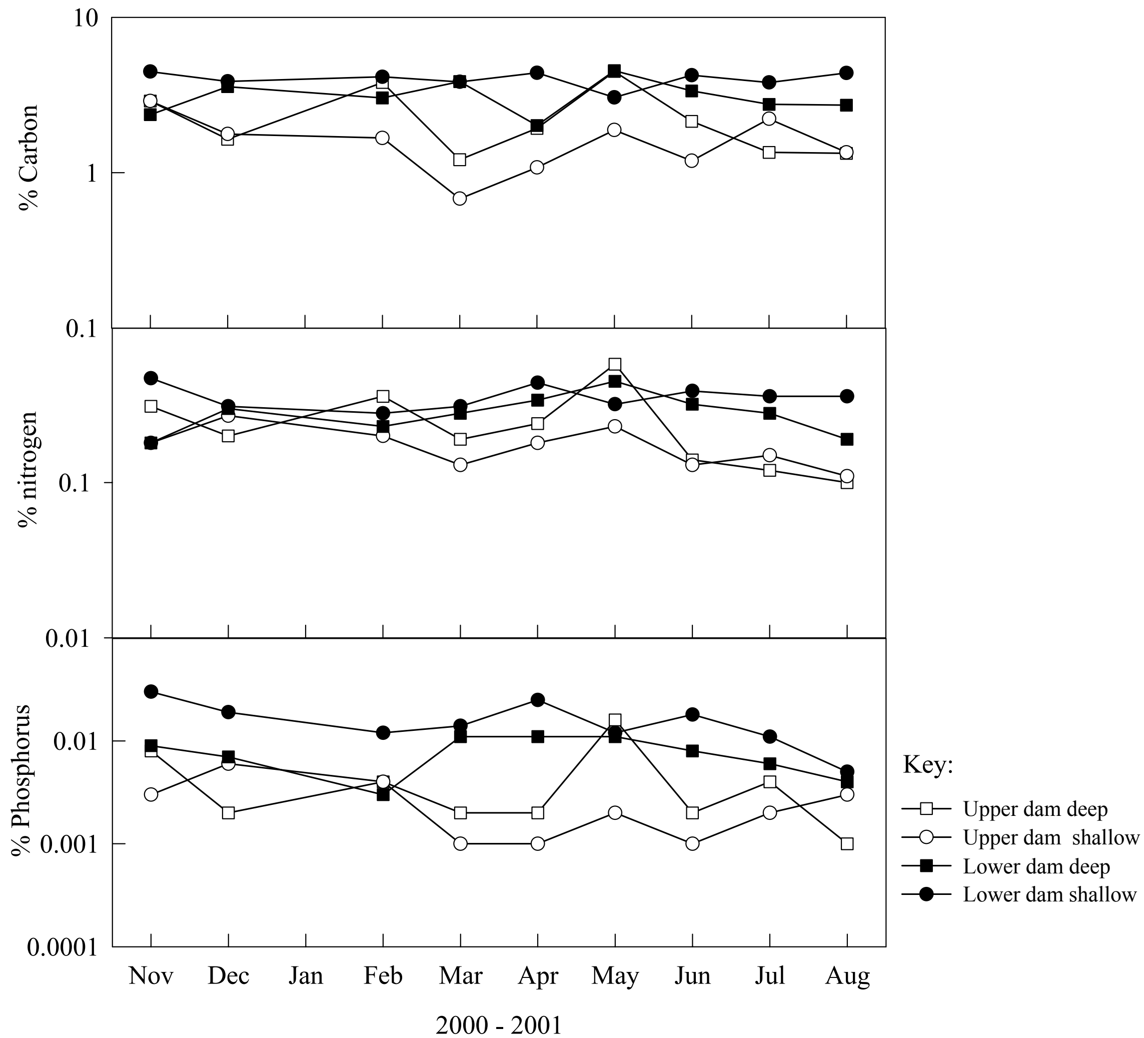


Figure 3.4. The proportions (%) of (a) carbon, (b) nitrogen and (c) phosphorus in the sediments of the two dams (November 2000 – August 2001).

3.4. Discussion

This chapter provides a descriptive account of the physico-chemical aspects of two small dams with different hydrological regimes. There were no significant differences between the water chemistry of the two dams because while they are large differences in drawdown levels, water residence times were short for both dams so it was unlikely that there would be major differences between the dams. As a result of this the influence of the Upper dam on water quality is minimal because there is just no time for sediment exchange processes to take place before water is pumped to the Lower dam. The rise in conductivity and total dissolved solids up to November 2000 could be a result of concentration effects as volumes decline and the fall in conductivity was a result of dilution by the early rains late in the dry season when water levels were lowest. Conductivity in Lake Chivero (2600ha) ranged from 6.5 – 23.2 mS cm^{-1} (Thornton, 1980) because of its enriched status. We might also consider that these small dams drain a much smaller catchment and importation of dissolved ions could be less but there is no direct evidence to support this.

The reason for the increase of dissolved ions after the dams fill was not adequately investigated but it could be that there was flushing of soluble ions from the exposed sediment. Imports of dissolved ions via river inflows were not investigated. The temporary re-suspension of sediment particles and the erosion of the exposed sediment by the early rains accounted for the low water transparency towards the end of the dry season. The effect was more pronounced in the Upper dam mainly because a greater surface area was exposed to the effect of these rains. Secchi disc transparency ranged from 0.6-1.6m in Lake Chivero (Robarts, 1979) so the two small dams' exhibit much greater variation in water transparency (<10cm - 160cm = Upper dam; 16cm-110cm = Lower dam). As a result during the early rainy season turbidity is so high to an extent that light availability could seriously limit photosynthetic activities. This high turbidity during the dry season was attributed to the shallow nature of the dams and hence their increased susceptibility to wind action. The period of dense algal growth correlated with periods of low transparency and hence algal growth also contributed to lowering of water transparency but to a lesser extent compared to inorganic & organic particles. The pH fluctuated in about five cycles and the reason for this is not known. A major determinant of pH cycles is photosynthetic activity but this was not measured. The pH was in a similar range compared with that of Lake Chivero (6.3 to 9.8) being slightly alkaline (Thornton and Nduku, 1982)

The sudden increase of reactive phosphorus (Sept. 2000-Dec. 2000) in the Upper dam could be the result of increased amount of suspended sediments. It has been established that they can be a significant source of phosphorus even under aerobic conditions and the rate of phosphorus release can also double when sediments are frequently disturbed (Watts, 2000; Wetzel, 2001). It could not be the result of river inflows because there the river was completely dry until January 2001. Reactive phosphorus in eutrophic Lake Chivero ranged from 0.267 mgl^{-1} to 0.486 mgl^{-1} (March 1996- May 1996) (Moyo, 1997) of which the small dams were also in that range and so external phosphorus loading from the agricultural catchment might have been a contributory factor for the dams to attain such high levels of reactive phosphorus. Unfortunately external phosphorus loading could not be determined. The low N: P ratio in the two small dams was due to relatively high levels of phosphorus in the two dams and the two small dams might have blue green algae dominated phytoplankton community. Nutrient ratios are an important predictor of the composition of phytoplankton communities and N: P ratios less than 29 were found to promote dominance of blue green algae (Flett *et al.*, 1980; Smith, 1983).

Chemical and biological oxygen demand showed increases during the hot season and lower values during the cool dry months. This suggests an increase in oxidisable inorganic and organic matter at such times because rates of decomposition and mineral processes are mainly controlled by temperatures rather than water level fluctuations. Higher air temperatures and small water column depths result in overally higher water temperatures during the dry season for the small dams. BOD ranged from 5.3 mg l^{-1} to 25.0 mgl^{-1} in Lake Chivero (Moyo, 1997) whereas the upper limit in both small dams did not exceed 7mgl^{-1}.

The work done on sediments was not an in-depth study of sediments in small dams but an attempt to get some idea on how extensive water level fluctuations could affect the distribution of some variables. The sediment variables differed except for silt content. Differences between the drawdown zone and permanently inundated zones were more apparent in the Upper dam with silt and clay content being different between the shallow and the deep sections whereas in the Lower dam only clay content differed. The Upper dam is more likely to retain larger suspended particles (rapid sinking rates) than the fine clays (slower sinking rates) kept in suspension and deposited in the Lower one and hence then differences in sediment texture. This has also been observed on a river with multiple dams built in series (Palmer and O'Keefe, 1990).

In the Upper dam, the deep water sediments generally had higher proportions of carbon, nitrogen and phosphorus compared to the shallow water sediments. The opposite was true in the Lower one. The theory by Hansen (1959), would suggest that the sources of sediment organic matter were largely allochtonic (submerged terrestrial and aquatic vegetation) for the Lower dam because of a C: N >10 and autochtonic (planktonic and bacterial productivity) in the Upper dam with a C: N <10. It was expected the Upper dam to serve as a sedimentation basin for allochthonous inputs since it receives the main discharge from the river directly and to see a shift to autochthony in the Lower dam. A possible explanation could be that since the drawdown level in the Lower dam was much less than that of the Upper dam and so processing of dead, submerged terrestrial and aquatic material represents a significant allochthonic contribution in contrast with the Upper dam. Another important factor that could explain this is the short residence time of the Upper dam which results in rapid flow through of incoming water into the Lower one hence most allochthonous inputs might be deposited there. The influence on the chemistry of overlying water by the sediments of small, shallow might be considerable in contrast with larger dams because of diel cycles of thermal and oxygen stratification that ensure frequent mixing but this requires further investigation. The maximum %C in both dams was almost 5% compared to a maximum of 12.8% in Lake Chivero in the centre of the lake and the least values (0.5%) along the shores in the shallow parts (Nduku, 1976). The amount of total phosphorus and total nitrogen in Lake Chivero also increased with depth but the relationship for total nitrogen was not linear. This was quite typical of the Upper dam but in contrast, the Lower dam generally recorded higher proportions of carbon, nitrogen and phosphorus in the shallow water sediments than the deep water sediments.

This chapter of the study described changes that occur in small dams mainly on a temporal scale but more in-depth investigations of each variable and determining process are necessary. Then all the observed trends can be adequately explained. A more comprehensive investigation of the sediments including sampling of interstitial water would come up with a clearer explanation of the sediment processes and their role in nutrient cycling in small dams. Diel cycles of physico-chemical variables such as pH, conductivity & various ions must also be determined and related to thermal stratification regimes prevalent in the dams & dynamics of the biota. The quality & quantity of inflows was not monitored because of inadequate resources but this important aspect would have clearly established if non-point pollution from agricultural catchment contributed to the nutrient load in the small dams. That information would also have helped to determine if these small dams play a significant role in nutrient cycling and the extent of nutrient loading in such dams. This information would be vital when developing management strategies small dams.

CHAPTER FOUR

PHYTOPLANKTON AND ZOOPLANKTON COMMUNITY STRUCTURE

4.1. Introduction

Small dams in southern Africa typically experience wide fluctuations in water level from one year to another because of variations in annual rainfall and the water management regime (Marshall and Maes, 1994). Some of the organisms inhabiting such water bodies have developed mechanisms for dealing with periodic droughts such as immigration or aestivation but plankton have low mobility and are therefore unable to avoid the environmental changes imposed by fluctuating water levels. These fluctuations cause wide variations in some physical and chemical variables (Moss and Moss, 1968; McLachlan, 1969; McLachlan, 1970, Morgan and Kalk, 1970; McLachlan *et al*., 1972; Marshall, 1978; Osborne *et al*., 1987) and these might have an effect on both phytoplankton and zooplankton communities.

In Lake Arancio, Italy the hydrological pattern of the reservoir affected the population dynamics of cladocerans via bottom-up processes while the predation efficiency of planktivores was regulated through top-down processes (Naselli-Flores and Barone, 1997). Periodic flooding and the frequent breakdown of thermal stratification at low water level in Eleiyele reservoir (Nigeria) caused fluctuations in nutrients that influenced the abundance and diversity of phytoplankton and zooplankton (Imevbore, 1965). In Zimbabwe, most investigations in plankton ecology have concentrated on the zooplankton of larger dams such as Cleveland Dam (Elenbaas and Grundel, 1994), Lake Chivero (Munro, 1966; Thornton and Taussig, 1982; Magadza, 1994; Elenbaas and Grundel, 1994) and Lake Kariba (Begg, 1974, 1976; Mills, 1977; Magadza, 1980; Masundire, 1989a, 1989b, 1991, 1992, 1994; Green, 1985; Marshall, 1991, 1997). Phytoplankton has been studied in less detail but some data have been reported from Lake Chivero (Munro, 1966;

Robarts *et al*., 1982) and Lake Kariba (Thomasson, 1965, 1980; Hancock, 1979; Ramberg, 1985, 1987; Cronberg, 1997). Relatively little is known about plankton in the country's smaller reservoirs. Thornton and Cotterill (1978) listed phytoplankton and zooplankton species in five small dams in the eastern highlands of the country while the zooplankton associations in 18 small impoundments, including those in the eastern highlands, were described by Green (1990). This work highlighted the effect of altitude on species composition, abundance and diversity.

This chapter describes the plankton community structure of the two small Mumwahuku river dams. The pattern of diurnal and seasonal stratification and aspects of water and sediment chemistry have been described in earlier chapters.

4.2. Methods

Sampling was done once a month from a station in the deepest part of the dam and another in shallow water from December 1999 to August 2001. The stratification regimes & water chemistry have already been described in the preceding chapters. Phytoplankton and zooplankton were collected by three vertical hauls using 20μm and 64μm mesh nets respectively of diameter 20cm from the bottom to the surface through the water column at an approximate speed of $0.5ms^{-1}$. These samples were mixed to form one bulked sample for each dam (Bottrell *et al*., 1976) and fixed by adding 4% iodine and 10% formaldehyde to phytoplankton and zooplankton respectively. The volume of sampled water that passed through the net was then estimated by:

$$V = \pi r^2 d$$

where V = volume of water filtered by the plankton net, r = radius of the mouth of the net and d = distance the net was pulled through (Downing and Rigler, 1982).

Phytoplankton samples were condensed by sedimentation for 48hrs and 10 subsamples were enumerated using Hawksley Cristalite B.S. 748 counting chambers. Identification was done to species level where possible using keys in Prescott (1970), Iltis (1980), Elenbaas (1994), and

Canter-Lund and Lund (1995). Water for chlorophyll *a* analysis was obtained from the deepest part of each reservoir by taking samples with a 3L Ruttner water sampler at 0.5m intervals.

Subsamples of 500ml were collected in plastic bottles and these were later filtered using Whatman GF 47mm filter paper. The filters were placed in centrifuge tubes and 10ml ethanol (96%) added to cover the filter. Thereafter the sample was stored for extraction at room temperature and in darkness for about 20 hrs. After incubation the centrifuge tube was shaken vigorously to ensure homogenous distribution and then centrifuged for 10 minutes at approximately 2000 m s^{-2}. The absorbance of the final extract was measured at chlorophyll absorption maximum (665nm) and correction for turbidity was made by measuring the absorption at 750nm, where chlorophyll is non-absorbing.

Calculations were done using the formula:

$$\text{Chlorophyll } a \text{ concentration (mg l}^{-1}\text{)} = \frac{(Abs665 - Abs750) \times e \times 10000}{83.4 \times V \times l}$$

e = volume of the extract
l = the length of the cuvette (100 mm)
V = filtered volume (0.5 litres)
83.4 = the absorption coefficient for ethanol (expressed in liters g-1cm-1). If methanol is used as solvent the absorption coefficient is 77.0, (Bronmark and Hansson, 1998).

Similarly, zooplankton samples were condensed and 10 subsamples taken for counting. Rotifers and cladocerans were identified to species using keys in Brooks (1957), Pontin (1978), Rey and Saint-Jean (1980), Dussart (1980), Pourriot (1980) Seaman *et al.* (1999). Unfortunately it was not possible to identify copepods to species level. The Shannon-Wiener index of diversity (*H*') (Krebs, 1989) was calculated for phytoplankton and zooplankton using the computer program PRIMER (ver. 5). It was calculated as follows:

$$H' = \sum_{i=1}^{s} (pi)(\log_2 pi)$$

where H' = index of species diversity (*nits* /individual)
P_i = proportion of total sample belonging to *i*th species
s = number of species, (Krebs, 1989).

Relationships between the plankton and the environmental factors were analyzed using Canonical correspondence analysis (CCA) looking at species-environment interactions only. The software CANOCO (ver. 3.1) was used for this purpose. CCA is a direct gradient analysis technique where the ordination axes are constrained to be linear combinations of environmental factors (ter Braak and Verdonschot, 1995).

All plankton species abundance data were square-root transformed to reduce skewness in the data and species included must contribute a relative abundance of at least 1% in each dam. It was necessary therefore to pool data of some species from the same genus and this was the case for most phytoplankton species. Nineteen zooplankton taxa and twenty-three phytoplankton taxa in both dams were considered for the analysis.

Eighteen environmental factors (and their abbreviations) were selected for zooplankton analysis and these were: Conductivity (COND), pH, Total dissolved solids (TDS), Secchi disc transparency (SD), Total suspended solids (TSS), Chemical oxygen demand (COD), Reactive phosphorus (RP), Chlorophyll *a* (Chl A), Biological oxygen demand (BOD), Water level (WL), Nitrate (NITRA), Ammonia (AMM), Total nitrogen (TN), Nitrite (NITRI), total phosphorus (TP), total alkalinity (TAL), water temperature (WT) and dissolved oxygen (DO).

For phytoplankton analysis, chlorophyll *a* was removed from the analysis. Forward selection (similar to stepwise multiple regression) was used to determine the minimum number of explanatory factors that explained statistically significant ($p < 0.05$) proportions of variation in the species data. The significance of these variables was assessed using Monte Carlo permutation (199 unrestricted) tests (ter Braak and Verdonschot, 1995).

4.3. Results

Phytoplankton

Species composition and seasonal variation

There was a summer peak in phytoplankton abundance in both dams but abundance was low at other times of the year (Figure 4.1a). Generally phytoplankton was more abundant in the Lower dam than in the Upper one at most times during the research program (Figure 4.1a). There was an unusually large increase in plankton numbers in September and October 2000 (Figure 1a) when the population reached $1.2 * 10^5$ ind.l^{-1} in the bottom dam and just above $4.0 * 10^4$ ind. l^{-1} in the Upper one. The concentrations of chlorophyll *a* followed a similar pattern, being generally higher in the Lower dam and with a pronounced increase in both dams from September 2000 to November 2000 to a maximum of 10 μgl^{-1} and 6 μgl^{-1} in the Lower and Upper dams respectively (Figure 4.1b).

Thirty-three phytoplankton species were identified in the two dams (Table 4.1 & 4.2). Species composition & abundance between the two dams was quite different even though the species list was similar as shown by the clustering of samples from the Lower dam away from those of the Upper dam (Figure 4.2). It is clear that the communities are not that similar and within each dam there was high similarity amongst samples collected in 2000 and those collected in 2001 (Figure 4.2). Chlorophytes were the dominant forms in both dams until October 2000 and then euglenophytes dominated especially in the Lower dam from about November 2000 to March 2001 (Figure 4.1 c &d). In the Lower dam the chlorophytes regained their dominance but in almost equal proportions to the euglenophytes up to August 2001 (Figure 4.1d). An almost similar picture emerged for the Upper dam but this time the Bacillariophytes contributed significantly from July to August 2001 (Figure 4.1c). This marked change in the composition of the phytoplankton occurred in both dams in October 2000, especially the Upper one, where chlorophytes were dominant from January-October, making up an average of 74% of the population (Table 4.3).

During the period November 2000–August 2001 the proportion of the chlorophytes fell to about half of that in the preceding period, while the proportion of euglenophytes and bacillariophytes was three times greater. A similar trend occurred in the lower dam, although it was not as pronounced (Table 4.3). There chlorophytes made up 64% of the total phytoplankton from January-October 2000, falling to 32% in the next ten months, while euglenophytes rose from 28% to 63% and bacillariophytes did not change significantly (Table 4.3).

Important species in terms of abundance were *Chlamydomonas* sp., *Trachelomonas* sp. and *Chlorococcus* sp. in the Upper dam (Table 4.1) and *Chlamydomonas, Trachelomonas* and *Ankistrodesdmus* sp. in the Lower one (Table 4.2). *Mougetia* sp. did not occur in the Lower dam and *Peridinium* sp. did not occur in the Upper dam. Dinophytes, chrysophytes and cyanophytes were of little importance in the dams as they occurred in much lower densities and low frequency compared to other taxa. Species diversity (H') was lowest in December 2000 (1.154) and highest in April 2001 (2.486) in the Upper dam (Table 4.1). The Lower dam had a least value in diversity of 0.996 in April 2000 and a highest value of 1.995 in January 2001 (Table 4.2). There was no monthly trend in species diversity and the monthly number of species in each dam although the highest number of species per month was recorded in 2001 in both dams.

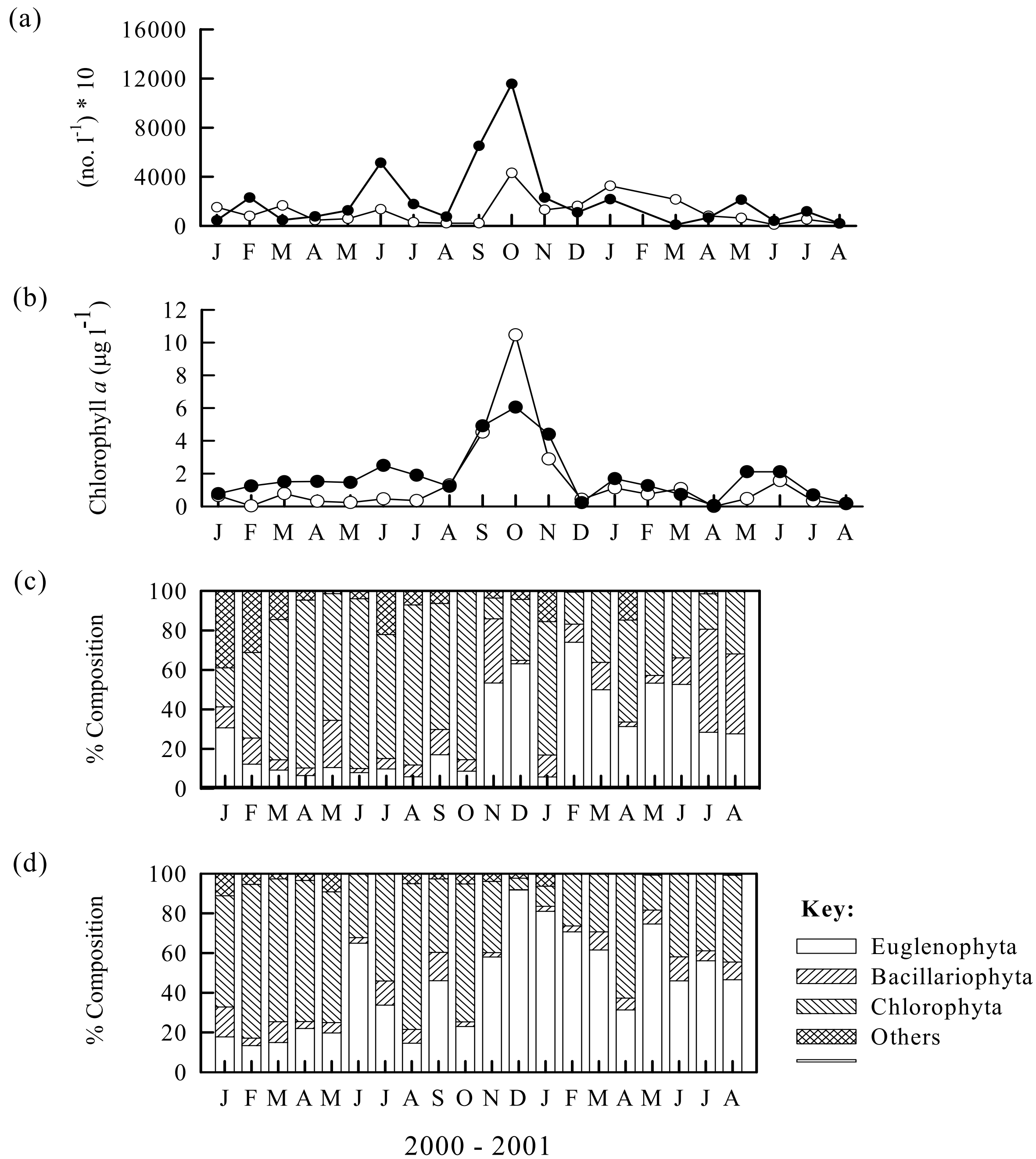


Figure 4.1. Temporal changes in (a) total abundance (no. l^{-1}) and (b) chlorophyll *a* content in the Upper (○) and Lower dams (●). Temporal changes in % composition of phytoplankton taxa in the (c) Upper and (d) Lower dams.

Table 4.1. The relative abundance (ind. l^{-1}) of some phytoplankton species occurring in the Upper dam, January 2000– August 2001 and the Shannon-Weiner index (*H'*) for each month.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Euglenophyta																				
Trachelomonas sp	4252	961	572	261	452	1001	66	38	196	758	4112	5556	2828	3784	835	2071	2940	468	1450	380
Euglena sp.	378	25	858	43	123	91	151	57	-	95	2878	816	2128	660	127	315	375	18	39	140
Phacus sp.	-	-	115	-	41	-	-	-	-	-	-	3807	392	484	113	165	105	30	-	-
Bacillariopyhta																				
Navicula sp.	4	58	74	-	-	-	-	-	5	6	-	45	23	12	6	-	7	9	23	14
Cymbella sp.	115	240	140	26	60	78	8	2	14	24	89	94	-	54	-	-	-	-	-	-
Melosira sp.	-	-	-	-	-	-	-	-	28	-	-	-	364	66	14	45	165	72	1599	-
Dinophyta																				
Dinobyron sp.	-	-	-	-	-	-	-	19	-	-	-	-	-	-	-	-	-	-	-	-
Chrysophyta																				
Ceratium sp.	-	-	-	-	-	-	-	-	-	95	-	-	-	-	-	-	-	-	39	20
Chlorophyta																				
Scenedesmus sp.	4725	253	115	-	-	-	-	-	-	-	69	233	140	-	-	-	-	-	-	-
Scenedesmus quadracauda	-	-	-	-	-	-	-	-	-	95	-	-	-	-	-	45	-	-	-	20
Scenedesmus acuminatus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15	-	-	20
Scenedesmus bijuga	-	-	-	-	--	-	-	-	-	-	-	-	-	66		25	75	14	-	20
Scenedesmus ercornis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12	45	-	118	-
Crucigenia tetrapedia	-	215	888	-	-	-	19	57	-	-	-	-	-	-	-	25	15	-	-	-
Volvox aureaus	-	-	-	45	23	14	-	-	-	-	-	-	13	-	25	-	-	45	-	-
Selenastrum gracile	-	-	-	-	-	-	-	-	-	-	-	-	112	22	-	45	30	-	-	-
Pediastrum sp.	-	4556	57	-	-	-	-	-	-	-	-	-	308	154	-	15	15	-	-	-
Pediastrum simplex	-	-	-	-	-	91	-	-	-	-	-	-	28	-	-	-	-	-	-	-
Pediastrum boryanum	-	-	-	-	-	-	-	-	-	-	-	-	84	-	-	-	-	-	78	-
Zygnema sp.	-	-	-	-	-	-	-	-	-	-	14	25	-	-	-	-	-	-	-	-
Staurastrum tetracerum	-	-	-	43	-	91	-	-	-	474	-	-	-	44	-	225	180	24	235	80
Closterium sp.	850	1467	-	43	370	91	123	38	-	3694	-	-	252	22	-	75	30	-	-	20
Spirogyra sp.	756	-	-	-	-	-	-	-	-	-	95	-	-	-	-	-	-	-	39	-
Chlamydomonas sp.	3874	11692	8642	3258	2055	6186	406	888	504	2557	822	1127	560	-	255	1860	1815	168	-	240
Chlorococcum sp.	756	759	-	217	1192	4731	113	113	252	3315	-	-	-	682	368	1592	-	-	-	-
Cosmarium granatum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	40
Geminella sp.	-	-	916	-	-	-	-	-	-	-	-	1321	-	44	-	-	105	42	-	-
Mougetia sp.	-	-	-	-	-	-	-	-	28	-	-	-	-	-	-	-	-	-	39	100
Tetraedon trigonum	-	127	-	--	-	-	-	-	-	-	-	39	-	44	-	120	15	-	78	-
Ankistrodesmus sp.	8504	557	801	130	82	364	709	113	84	-	-	854	56	-	42	12	-	6	-	-
Cyanophyta																				
Nostoc sp.	-	25	57	-	-	182	-	-	28	-	480	39	-	44	-	120	15	-	78	-
Anabaena sp.	5868	2480	2346	43	-	-	-	118	-	189	206	39	1176	44	-	270	240	72	431	60
Others	284	-	-	43	-	-	104	-	-	-	-	194	-	-	-	-	-	-	118	-
Total number of species (N)	**12**	**14**	**13**	**11**	**9**	**11**	**8**	**10**	**9**	**11**	**9**	**13**	**15**	**16**	**9**	**18**	**18**	**12**	**13**	**13**
Shannon-Weiner index (*H'*)	**1.928**	**2.217**	**1.367**	**1.267**	**1.636**	**1.235**	**1.579**	**1.894**	**2.093**	**1.828**	**1.995**	**1.154**	**1.499**	**1.612**	**1.474**	**2.486**	**2.352**	**1.741**	**1.750**	**1.841**

Table 4.2. The relative abundance (ind. l^{-1}) of some phytoplankton species occurring in the Lower dam, January 2000– August 2001 and the Shannon Weiner index (*H'*) for each month.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Euglenophyta																				
Trachelomonas sp	228	702	419	840	2424	32618	5705	702	18658	10438	6349	7246	6741	3896	370	1450	10809	1683	5942	824
Euglena sp.	279	432	294	682	61	290	94	327	4174	10438	3968	2408	9288	582	62	157	1205	33	131	32
Phacus sp.	304	1944	-	157	-	362	187	47	7120	5725	1417	388	1528	381	101	470	3810	165	522	64
Bacillariopyhta																				
Navicula sp.	458	123	56	14	147	-	-	-	45	-	-	20	-	-	-	126	-	-	-	-
Cymbella sp.	6836	8642	503	262	667	1450	2151	514	9329	2694	510	20	549	45	67	392	1477	495	392	176
Melosira sp.	-	-	-	-	-	-	-	-	-	-	-	367	196	157	12	-	-	-	196	-
Dinophyta																				
Dinobyron sp.	-	-	42	-	-	-	-	327	-	-	-	-	274	-	-	-	-	-	-	-
Chrysophyta																				
Ceratium sp.	-	-	--	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Chlorophyta																				
Scenedesmus sp.	2532	-	42	-	61	-	140	-	982	1683	340	41	235	-	-	-	233	-	-	-
Scenedesmus quadracauda	-	-	-	-	-	-	-	-	-	-	-	-	157	22	-	-	-	-	392	176
Scenedesmus acuminatus	-	-	-	-	-	-	-	-	-	-	-	-		6		39		165	196	176
Scenedesmus bijuga	-	-	-	-	-	-	-	-	-	-	-	-	-	72	-	118	-	-	-	24
Scenedesmus ercornis	-	-	-	-	-	-	-	-	-	-	-	-	-	24	-	235	-	-	392	-
Crucigenea tetrapedia	-	702	42	-	-	145	-	-	-	-	-	-	-	-	-	274	-	-	-	-
Volvox aureaus	-	-	21	-	13	10	-	-	-	-	-	-	14	-	47	-	26	-	-	-
Selenastrum gracile	-	-	-	-	-	-	-	-	-	-	-	-	-	22	-	274	39	-	196	-
Pediastrum sp.	-	3781	-	-	-	-	-	-	-	1347	-	-	-	-	-	78	-	-	65	-
Pediastrum simplex	-	-	-	-	-	-	-	-	-	-	-	-	78	179	-	-	-	-	65	8
Pediastrum boryanum	-	-	-	-	-	-	-	-	-	-	-	-	39	22	-	-	-	-	-	-
Zygnema sp.	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-
Staurastrum tetracerum	329	-	-	53	61	-	94	-	2210	1010	170	-	-	6	-	157	-	66	523	8
Closterium sp.	329	2377	-	53	242	1885	327	1543	3928	4040	1077	41	157	112	6	470	1322	132	65	-
Spirogyra sp.	51	-	-	-	-	-	-	-	-	1347	-	-	-	-	-	-	-	-	-	-
Chlamydomonas sp.	760	8479	2893	4985	4849	10438	4957	3086	12521	13468	1417	286	118	918	224	902	1128	825	1633	224
Chlorococcum sp.	2228	1890	-	105	2000	4422	4256	1356	10066	61280	1587	-	-	-	-	510	-	-	-	-
Cosmarium granatum	-	162	377	210	1152	-	-	47	-	-	-	-	-	22	6	39	-	31	-	120
Geminella sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	246	-	196	700	33	131	-
Mougetia sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Tetraedon trigonum	25	270	-	-	-	-	-	-	737	1347	-	204	510	45	6	392	39	-	-	48
Ankistrodesmus sp.	-	-	-	-	-	-	-	-	-	-	567	41	549	-	-	-	156	-	457	-
Cyanophyta																				
Nostoc sp.	50	702	84	53	-	-	-	-	246	-	57	-	-	-	-	-	-	-	-16	
Anabaena sp.	-	3294	42	53	182	218	-	94	1473	5724	1871	123	549	179	12	862	39	462	849	128
Others	-	-	-	-	-	-	-	-	-	-	-	567	-	-	6	-	156-	-	-	-
Total number of species (N)	**12**	**13**	**11**	**11**	**11**	**9**	**9**	**10**	**14**	**13**	**13**	**12**	**16**	**19**	**11**	**19**	**13**	**11**	**18**	**13**
Shannon-Weiner index (*H'*)	**1.720**	**1.740**	**1.560**	**0.996**	**1.575**	**1.179**	**1.725**	**1.498**	**1.352**	**1.726**	**1.426**	**1.618**	**1.995**	**1.499**	**1.572**	**1.797**	**1.568**	**1.712**	**1.661**	**1.709**

Table 4.3. The mean relative abundance (% number) of the major phytoplankton groups in the two dams showing the changes that took place in October 2000. The data are means, with the range for each ten-month period in brackets

	Upper dam Jan – Oct 2000	Nov 2000 – Aug 2001	Lower dam Jan – Oct 2000	Nov 2000 – Aug 2001
Chlorophyta	74 (32-90)	36 (11-80)	64 (32-82)	32 (6-63)
Euglenophyta	15 (6-50)	45 (7-74)	28 (14-65)	63 (24-94)
Bacillariophyta	11 (4-24)	19 (2-53)	8 (2-17)	6 (0-12)

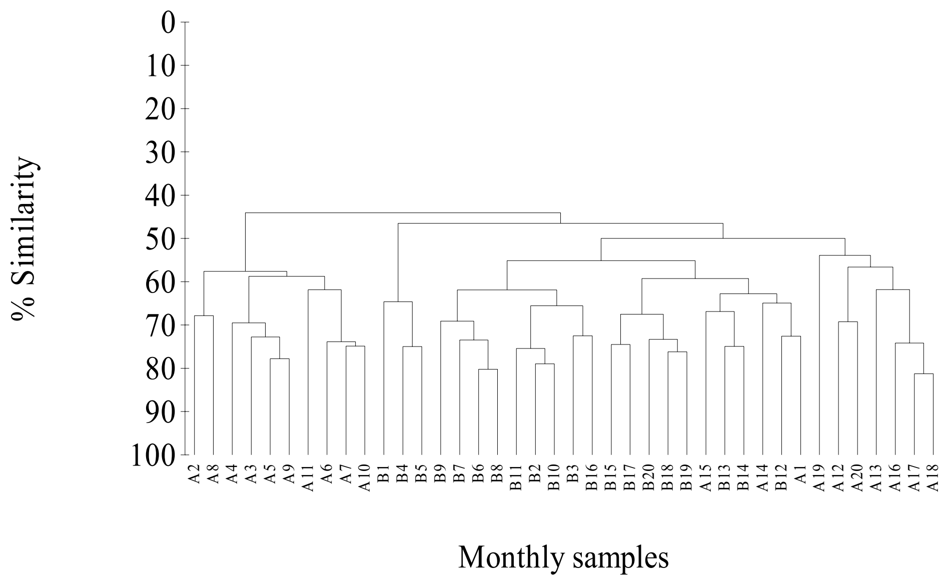


Figure 4.2. Similarity %) among phytoplankton samples collected at different times in the Upper and Lower dams. Key: Upper dam (A1 = Jan.2000, A2 = Feb. 2000…….A20 = Aug. 2001); Lower dam (B1 = Jan. 2000, B2 = Feb. 2000…….B20 = Aug. 2001).

The influence of 17 environmental factors on the phytoplankton community in the Upper dam was assessed using CCA. The results of this analysis are in figure 4.2. CCA axis1 (12.3%), axis 2 (8.7%), axis 3 (4.8%) and axis 4 (3.4%) explain the total variation the data set (Table 4.4). This was also equal to the total variation in the phytoplankton data that could be explained by environmental variation (29.2%) (Table 4.4). The first two axes together display 72% of the variation in phytoplankton abundance explained by the environmental variables (Table 4.4).

Four environmental variables that explained significant ($p < 0.05$) proportions of the variation in species data were identified and only these variables are shown in the biplot (Figure 4.3). In increasing order of significance they were Water level (WL), total alkalinity, mean dissolved oxygen (DO) and total suspended solids (TSS). In the biplot TAL was positively correlated to TSS & WL while the remaining variables were all negatively correlated (Figure 4.3). Along the WL axis is *Dinobyron* sp., *Pediastrum* sp. *Anabaena* sp., *Chlamydomonas* sp., *Ankistrodesmus* sp., *Scenedesmus* sp. & *Chlorococcum* sp. On the DO axis are *Nostoc* sp. & *Trachelomonas* sp.; and on the TSS axis are diatoms, *Phacus* sp. & *Geminella* sp. (Figure 4.3). No species were strongly associated with TAL axis. Analysis reflects a poor relationship between environmental variables and phytoplankton dynamics as only 29.2% of total variation could be accounted for.

Table 4.4. Summary of results of CCA output for the relationship between phytoplankton and environmental factors in the Upper dam

	CCA Axes				
	1	2	3	4	
Eigenvalues	0.149	0.105	0.058	0.041	
Cumulative % variance:					
of species data	12.3	21.0	25.8	29.2	
of species-environment relation	42.3	72.0	88.4	100.0	
Sum of all unconstrained eigenvalues					1.204
Sum of all canonical eigenvalues					0.352

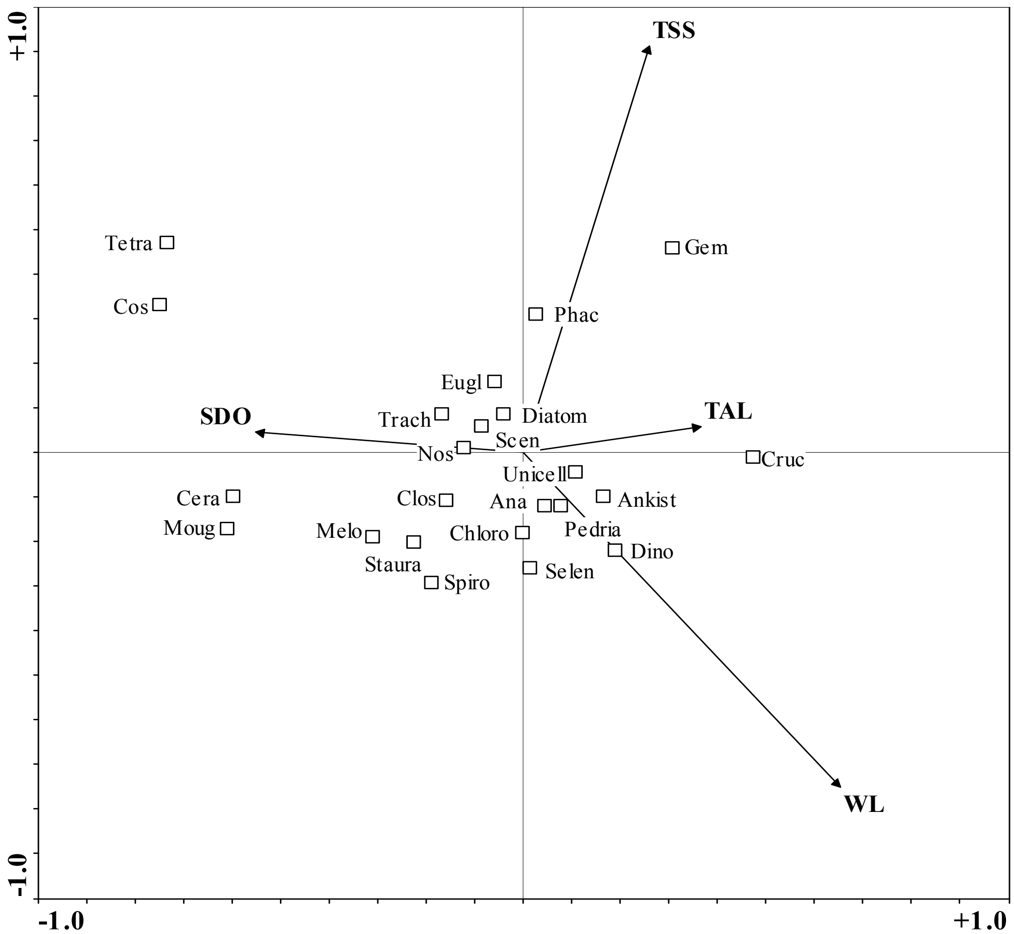


Figure 4.3. CCA biplot of the relationship between phytoplankton taxa (□) and the four statistically significant environmental factors (→) in the Upper dam. Taxa abbreviations are: *Crucigenea sp.* (Cruc), *Dinobyron* sp. (Dino), *Pediastrum* sp. (Pedria), *Ankistrodesmus* sp. (Ankist), *Chlamydomonas* sp. (Chlam), *Anabaena* sp. (Ana), *Selenastrum* sp. (Selen), *Chlorococcum* sp. (Chloro), *Spirogyra* sp. (Spiro), *Straurastrum* sp. (Staura), *Closterium* sp. (Clos), *Melosira* sp. (Melo), *Ceratium* sp. (Cera), *Mougetia* sp. (Moug), *Nostoc* sp. (Nos), *Scenedesmus* sp. (Scen), Bacillariophytes (Bacill), *Trachelomonas* sp. (Trach), *Euglena* sp. (Eugl), *Phacus* sp. (Phac), *Geminella sp.* (Gem), *Cosmarium* sp. (Cos), *Tetradium* sp. (Tetra).

Similarly, the influence of the same 17 environmental factors on the phytoplankton community of the Lower dam was assessed using CCA. CCA axis 1 (14%), axis 2 (10%), axis 3 (6.1%) and axis 4 (4%) explain the total variation in the species data (Table 4.5). Axes 1& 2 explain 65.3% of the variation explained by the environmental variables (Table 4.5). The total variation in the phytoplankton community that could be explained by environmental variation was 36.8% (Table 4.5).

Five environmental variables were identified as significant ($p < 0.05$) using forward selection and Monte Carlo permutation tests and these are displayed in the biplot (Figure 4.4). Reactive phosphorus (RP), Secchi disc transparency (SD), nitrite (NITRI) were positively correlated while Ammonia (AMM) & dissolved oxygen (DO) were negatively correlated to the rest of the variables (Figure 4.4). Along the NITRI axis was found *Peridinium* sp., *Spirogyra* sp. & *Chlorococcum* sp. Along DO axis are *Dinobyron* sp. & *Chlamydomonas* sp.; on the RP axis are *Pediastrum* sp., *Phacus* sp. and *Tetradium* sp. and along the SD axis *Euglena* sp.; along the AMM axis *Selenastrum* sp., *Melosira* sp., *Ankistrodesmus* sp., *Geminella* sp., *Scenedesmus* sp., *Trachelomonas* sp. & *Staurastrum* sp. (Figure 4.4). As in the Upper dam environmental factors do not sufficiently explain phytoplankton dynamics in the Lower dam but with the exception of DO all other factors identified as important in this analysis were different between the dams.

Table 4.5. Summary of results of CCA output for the relationship between phytoplankton and environmental factors in the lower dam

	CCA Axes				
	1	2	3	4	
Eigenvalues	0.110	0.079	0.048	0.032	
Cumulative % variance:					
of species data	14.0	24.0	30.1	34.2	
of species-environment relation	38.0	65.3	81.9	93.2	
Sum of all unconstrained eigenvalues					0.786
Sum of all canonical eigenvalues					0.289

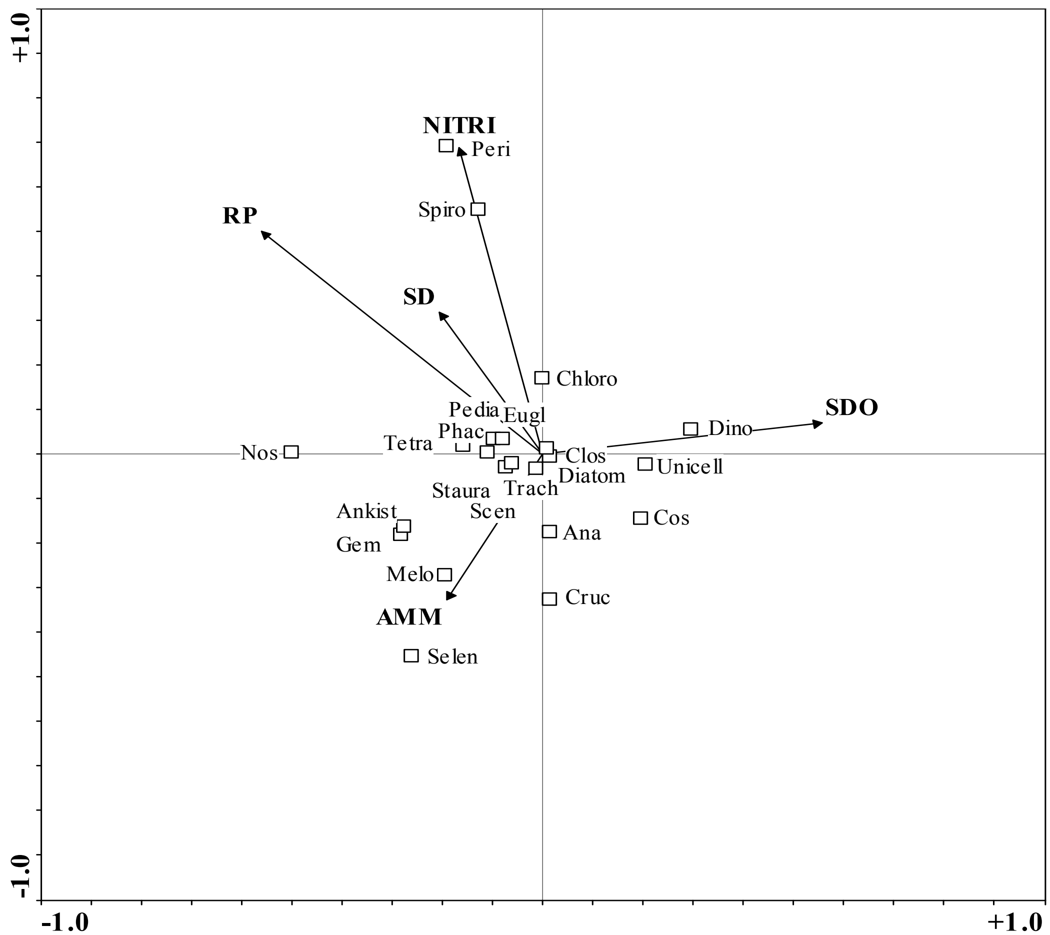


Figure 4.4. CCA biplot of the relationship between phytoplankton taxa (□) and the five statistically significant environmental factors (→) in the Lower dam. Taxa Abbreviations are as in Figure 4.3 except: *Peridinium* sp. (Peri)

Zooplankton

Species composition and seasonal variation

In contrast to the phytoplankton, there was little difference in the zooplankton density in the two dams (Figure 4.5). The population of zooplankton in both dams was low from January-October 2000, rarely exceeding 10 ind. l^{-1} and then a summer peak occurred in both dams (Figure 4.5a). The peak in the Upper dam suddenly occurred in Nov. 2000 with densities reaching 300 ind.l^{-1} and then steadily declined to a low in March 2001. There was a slight recovery in April and May 2001 and then values declined again (Figure 4.5a). In the Lower dam densities started to increase in Oct. 2000 and a peak was reached in January 2001 (400 ind. l^{-1}) and then a decline to follow a pattern & densities similar to the Upper dam. Similarity between the zooplankton communities of the two dams was assessed using hierarchical cluster analysis and while the highest level of similarity was observed among samples from the same dam, there was greater similarity in samples of both dams collected at the same time of the year compared to the situation with phytoplankton (Figure 4.6).

The change in the phytoplankton community composition that occurred in September/October 2000 was even more marked in the zooplankton and was observed in both dams (Figure 4.5 b & c). Before this period the dominant forms were adult copepods and cladocerans in both dams but after October 2000 their decrease was matched by an increase in the abundance rotifers and nauplii (Table 4.6). The dominant cladoceran species were *Daphnia laevis* (< 1 – 5 ind. l^{-1}) and *Diaphanosoma excisum* (< 1 – 2 ind. l^{-1}). Moina micrura was recorded a few times but in very low densities - < 1 ind. l^{-1} (Table 4.7) and the rotifers, present in small numbers, included *Brachionus falcatus, Keratella quadrata* and *Asplanchna priodonta* (Table 4.8).

In the Upper dam, nauplii appeared in large numbers in September 2000 and then rotifers dominated in Oct. - Dec 2000. (Figure 4.5b). From December 2000 cyclopoids start recovering and there was also a steady increase in nauplii larvae until rotifers lose their dominance until August 2001 (Figure 4.5b). In the Lower dam rotifers and nauplii appeared in September 2000 and the rotifers dominated until March 2001 (Figure 4.5c). Like in the Upper the proportion of nauplii increased gradually as the rotifers declined but recovery in copepods was not to the extent observed in the Upper dam. The cladocerans and calanoids were most severely affected by these changes in community composition and almost disappeared from the dams (Table 4.6). A striking feature of the cladocera was the complete absence of *Daphnia laevis* from both dams from November 2000 onwards while *Diaphanosoma exiscum* and *Moina micrura* increased in abundance during the same period (Table 4.7).

Table 4.6. The mean relative abundance (%) of the major zooplankton groups in the two dams showing the changes that took place in October 2000. The data are means, with the range for each ten-month in brackets

	Upper dam		Lower dam	
	Jan – Oct 2000	Nov 2000 – Aug 2001	Jan – Oct 2000	Nov 2000 – Aug 2001
Cyclopoida	42 (22-64)	30 (5-51)	49 (25-68)	18 (6-33)
Calanoida	14 (3-26)	0.4 (0-1)	23 (0-48)	1 (0-2)
Cladocera	27 (3-72)	3 (1-7)	13 (1-41)	3 (0-9)
Nauplii	7 (0-56)	36 (0-63)	5 (0-17)	32 (5-60)
Rotifera	10 (0-58)	30 (3-94)	11 (0-51)	47 (12-81)

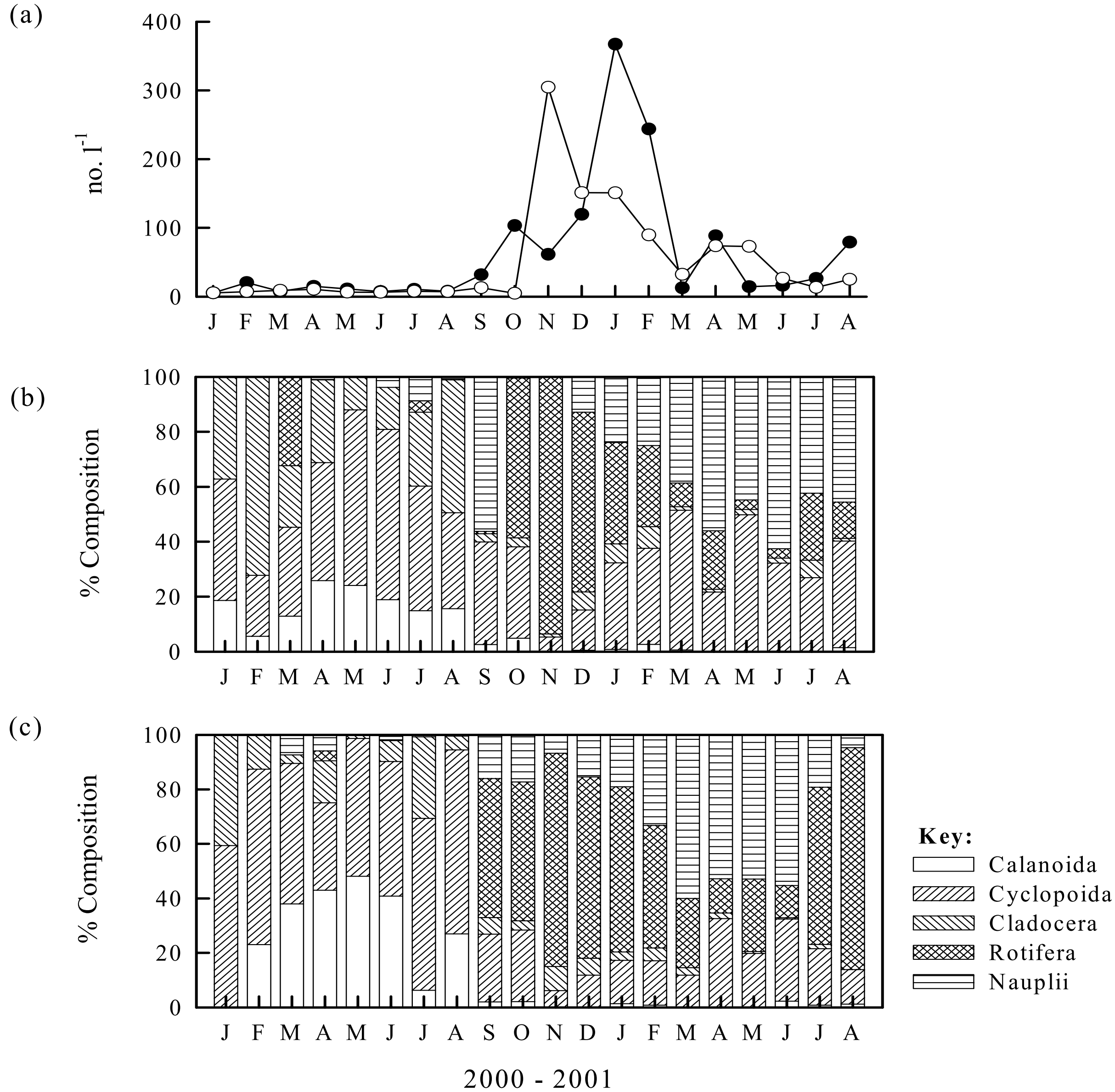


Figure 4.5. Temporal changes in (a) total abundance (ind. l^{-1}) and % composition of zooplankton taxa in (b) the Upper and (c) Lower dams

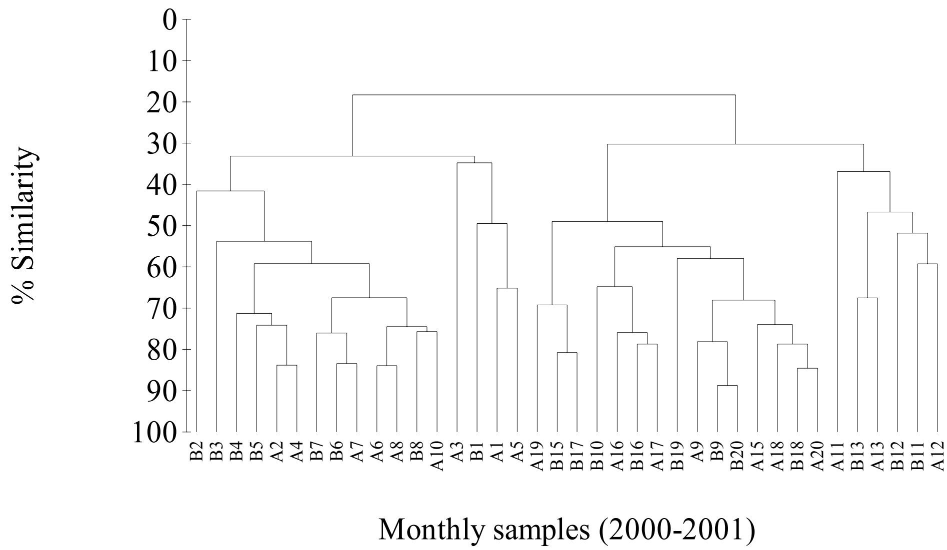


Figure 4.6. Similarity %) among zooplankton samples collected at different times in the Upper and Lower dams. Key: Upper dam (A1 = Jan.2000, A2 = Feb. 2000…….A20 = Aug. 2001); Lower dam (B1 = Jan. 2000, B2 = Feb. 2000……B20 = Aug. 2001).

Twelve rotifer species were identified in the dams and the dominant species were *Keratella cochlearis*. *K. tropica*, *K. quadrata*, *Filinia opolensis*, *Polyathra dolichoptera* and *Asplanchna priodonta* but the dominant species varied in the two dams. In the post rainy season *K. cochlearis* and *K. quadrata* were dominant in both dams (table 4.8). *Brachionus caudatus*, *B. annularis, Filinia opolensis* occurred in the highest abundance (> 150 ind. l^{-1}) and only in the Upper dam (Table 4.8). Zooplankton species diversity (H') ranged from 0. 501 to 1.606 in the Upper dam and 0.298 – 2.163 in the Lower one.

Table 4.7. The relative abundance (ind. l^{-1}) of five species of cladocerans and copepods recorded from the two small dams, March 2000 – August 2001. (U = Upper dam and L = Lower dam; + = present but less than 1 individual per litre and - = absent).

		J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A
Cladocera																					
D. laevis	U	3	5	2	1	1	+	2	4	-	+	-	-	-	-	-	-	-	-	-	-
	L	4	2	+	1	+	1	3	+	+	+	-	-	-	-	-	-	-	-	-	-
D. exiscum	U	-	-	-	-	-	+	-	+	+	+	4	8	10	-	-	+	2	+	+	-
	L	-	-	+	-	+	-	-	+	+	2	1	1	7	-	+	2	+	+	-	-
M. micrura	U	-	-	-	-	-	+	-	-	-	-	-	2	+	-	-	-	-	-	-	-
	L	-	-	-	-	-	+	-	-	-	2	4	+	4	-	-	+	-	-	-	-
Chydorus sp.	U	-	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	+
	L	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
B. longirostris	U	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	+	-
	L	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	-	-	+	-
Calanoida	U	+	+	+	+	+	+	+	+	+	+	-	1	1	-	+	-	+	-	-	+
	L	-	4	+	+	+	1	+	1	1	2	+	-	5	-	-	-	-	-	-	1
Cyclopoida	U	+	+	+	+	+	1	1	1	5	2	16	22	48	-	16	16	36	9	2	8
	L	+	10	+	+	+	1	4	2	8	27	4	2	59	-	1	29	2	5	4	10
Nauplii	U	-	-	-	+	-	+	+	+	7	+	-	20	36	-	13	41	33	17	4	11
	L	-	-	+	+	-	+	+	+	5	18	4	3	70	-	5	47	5	9	4	4

Table 4.8. The relative abundance (ind. l^{-1}) of rotifer species occurring in the two dams, 2000-2001. (U = Upper dam, L = Lower dam, + = present but less than 1 individual per litre, - = absent).

Species		M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A
Brachionus caudatus	U	-	-	+	+	-	-	-	+	181	3	+	-	-	-	-	-	-	-
	L	-	-	-	+	-	-	-	+	6	7	25	-	-	-	+	-	-	-
Brachionus falcatus	U	+	-	-	-	+	-	-	-	2	-	+	-	-	+	-	-	+	-
	L	-	-	-	+	+	-	-	-	+	+	8	-	+	+	+	-	-	-
Brachionus quadridentatus	U	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	L	-	-	-	-	-	-	-	-	2	-	9	-	-	-	-	-	+	-
Brachionus annularis	U	-	-	-	-	+	-	-	-	162	4	+	-	-	-	-	-	-	-
	L	-	-	-	+	-	-	-	-	-	16	-	-	-	-	-	-	-	-
Keratella cochlearis	U	-	-	-	-	-	-	-	-	3	2	1	-	+	13	2	+	-	+
	L	-	-	-	-	-	-	-	-	2	+	22	-	+	1	+	+	6	-
Keratella tropica	U	-	-	-	-	-	-	-	-	-	+	24	-	-	-	-	-	+	-
	L	-	-	-	-	-	-	-	-	1	7	-	-	+	-	-	-	-	-
Keratella quadrata	U	-	-	-	-	-	-	-	-	-	6	25	-	+	+	+	+	+	2
	L	-	-	-	-	-	-	-	-	+	3	25	-	+	2	+	+	7	-
Lecane sp.	U	-	-	-	-	-	-	-	-	-	-	1	-	+	-	-	-	-	-
	L	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Filinia opolensis	U	-	-	-	-	-	-	-	-	179	125	+	-	+	-	+	+	+	+
	L	-	-	-	-	-	-	-	-	34	2	17	-	+	2	+	-	-	-
Polyathra dolichoptera	U	-	-	-	-	-	-	-	-	-	2	58	-	+	2	+	-	+	-
	L	-	-	-	-	-	-	-	-	4	-	40	-	1	-	1	-	-	-
Asplanchna priodonta	U	+	-	-	-	+	-	+	-	26	30	2	-	+	+	-	-	-	-
	L	+	+	-	-	-	-	+	-	22	2	5	-	+	2	+	-	-	-
Other	U	-	-	-	-	-	-	-	-	-	7	+	-	-	-	-	-	-	-
	L	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-	-	-	-

The influence of 18 environmental factors on the zooplankton community in the Upper dam was assessed using CCA. Five environmental factors were identified as significant ($p < 0.05$) using Monte Carlo permutation tests and forward selection in explaining significant proportion of the variation in the species data. These are displayed in the biplot (Figure 4.8).The total variation that could be explained by the environmental variables was 74.3% and axes 1 & 2 (displayed) accounted for 72% of the explained variation (Table 4.9). CCA axis 1 accounted for 36.8%, axis 2 (16.1%), axis 3 (11.1%) and axis 4 (4.5%) of the total variation (Table 4.9.)

The environmental variables were Secchi disc transparency (SD), chemical oxygen demand (COD), nitrite (NITRI), ammonia (AMM) and water temperature (WT). AMM, NITRI, COD and WT were positively correlated while SD was negatively correlated to all the other variables (Figure 4.7). Along the SD axis were located Calanoids, Cyclopoids, *B. longirostris*, *K. cochlearis* & *Chydorus* sp.; on the AMM axis was *Brachionus* sp., *B. caudatus* and *B. falcatus*; on the NITRI axis was *F. opolensis;* along the COD axis was *A. priodonta* and along the WT axis was other rotifer species, *M. micrura* and *Lecane* sp. (Figure 4.7). Other species did not show any association with the environmental variables.

Table 4.9. Summary of results of CCA output for the relationship between zooplankton and environmental factors in the Upper dam

	CCA Axes				
	1	2	3	4	
Eigenvalues	0.573	0.250	0.173	0.069	
Cumulative % variance:					
of species data	36.8	52.9	64.0	68.5	
of species-environment relation	50.1	72.0	87.2	93.2	
Sum of all unconstrained eigenvalues					1.557
Sum of all canonical eigenvalues					1.143

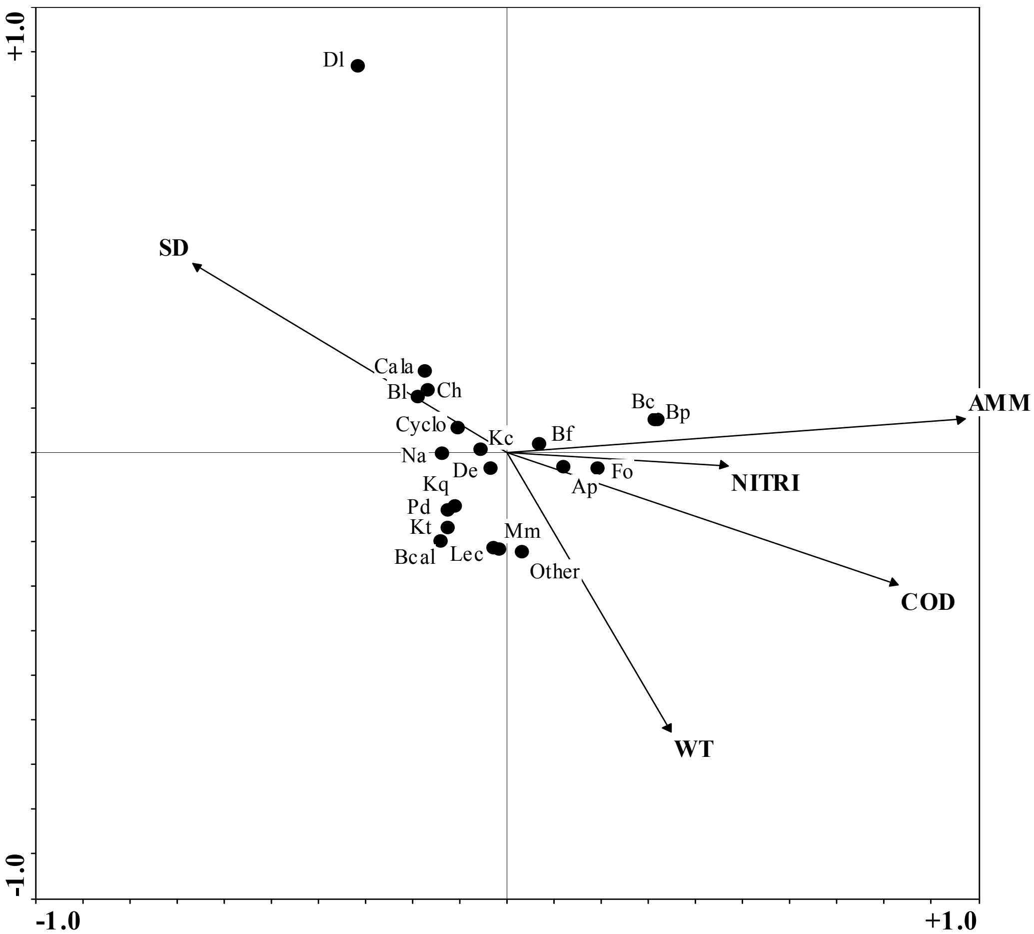


Figure 4.7. CCA biplot of the relationship between zooplankton taxa (□) and the six statistically significant environmental factors (→) in the Upper dam. Taxa abbreviations are: *Brachionus calyciflorus* (Bcal), *Keratella tropica* (Kt), *Keratella quadrata* (Kq), *Keratella cochlearis* (Kc), *Lecane* sp. (Lec), *Brachionus falcatus* (Bf), *Asplanchna. priodonta* (Ap), *Filinia opolensis* (Fo), *Brachionus caudatus* (Bc), *Brachionus* sp. (Bp), *Bosmina longistoris (*Bl), *Chydorus* sp. (Ch), Cyclopoida (Cyclo), Calanoida (Cala), Na (Nauplius), Daphnia *laevis* (Dl), *Daphnia exiscum* (De), *M. micrura* (Mm) and Other (unidentified rotifer species)

The relationship between zooplankton and measured physico-chemical variables was examined using CCA in the Lower dam. Four environmental factors of the original 18 were found to explain significant ($p < 0.05$) proportions of the variation of the plankton data using Monte Carlo permutation tests and forward selection and are displayed in the biplot (Figure 4.9). CCA axis 1 accounted for 29.8%, axis 2 (14.6%), axis 3 (10.1%) and axis 4 (5.5%) of the total variation in the species data (Table 4.10). The total variation that could be explained by the environmental variables was 64.8% (Table 4.10). The displayed axes 1 & 2 accounted for 68.5% of the variation that could be explained by environmental factors.

The environmental factors were total suspended solids (TSS), chemical oxygen demand (COD), water temperature (WT) and water level (WL). All variables were positively correlated except WL and TSS & WT (Figure 4.8). On the WT axis was *B. falcatus* & *B. caudatus*. On the COD axis was *A. priodonta*, *M. micrura*, *F. opolensis* and *B. annularis* to a lesser extent. Along the WL axis was *P. dolichoptera* and finally on the TSS axis at the point of greatest influence was *Brachionus* sp. & other unidentified rotifers (Figure 4.8).

Table 4.10. Summary of results of CCA output for the relationship between zooplankton and environmental factors in the Lower dam

	CCA Axes				
	1	2	3	4	
Eigenvalues	0.349	0.171	0.119	0.064	
Cumulative % variance:					
of species data	29.8	44.4	54.5	60.0	
of species-environment relation	45.9	68.5	84.1	92.5	
Sum of all unconstrained eigenvalues					1.172
Sum of all canonical eigenvalues					0.760

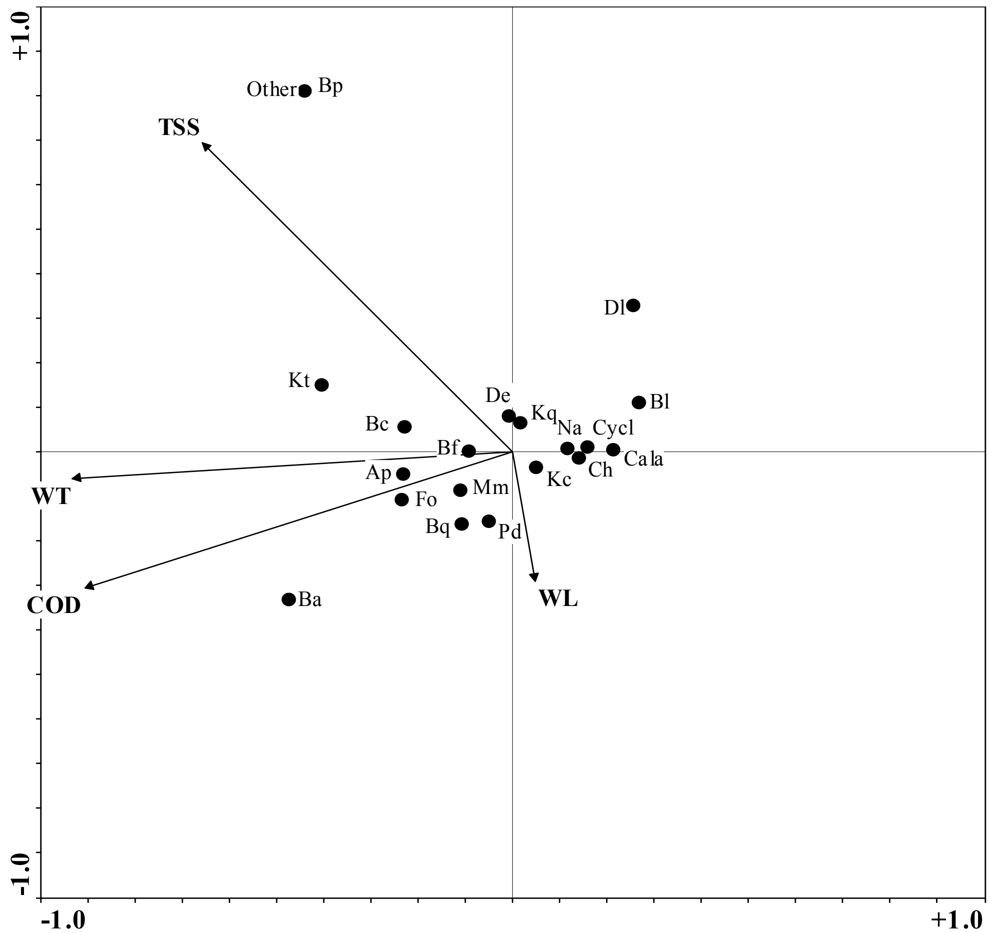


Figure 4.8. CCA biplot of the relationship between zooplankton taxa (□) and the six statistically significant environmental factors (→) in the Lower dam. Taxa abbreviations are as in Figure 4.8 except *Brachionus quadridentatus* (Bq) and *Brachionus annularis* (Ba)

4.4. Discussion

The differences between the plankton community in the first ten months of this study (January-October 2000) and in the second ten months (November 2000-August 2001) was the striking outcome of this investigation. So different were the phyto- and zooplankton communities during each of these two periods especially in the Lower dam that the results might have come from different reservoirs except that both reservoirs showed broadly parallel changes. What could have caused such a dramatic change?

There was no obvious answer to this question. The water chemistry of the two dams was highly variable but the pattern of variation was, at least for some variables, linked to water level fluctuations and tended to be consistent from one year to the next. The relationship between phytoplankton & environmental factors was very weak as shown by CCA analysis suggesting that some other factors were critical in driving change in the populations in the dams. Species composition was similar between the dams mainly because they shared water but abundance was greater in the Lower dam.

The highest densities of phytoplankton were recorded during periods of long water residence time and the least during the infilling period. This is because there is washout of the community in each dam mainly through surface water overflow. Algal abundance declined significantly but species composition and dominant assemblages remained more or less unchanged at the peak of the rainy season. An indirect effect of wash outs is that nutrients are also removed at the same time and hence their depletion contributes to declines in abundance and slow recovery of the populations later on in the year. Thus seasonal variation in phytoplankton abundance was related to hydrological changes within the dams and short residence times do not allow for the establishment of stable algal populations. This is probably the major reason why algal abundance in the Upper dam was much lower than that of the Lower one despite the two dams having very similar water chemistry.

The increase in zooplankton density that occurred in October 2000 followed an increase in phytoplankton, especially in the top dam, and there was an inverse relationship between phytoplankton and zooplankton densities from September 2000 to February 2001. The effects of grazing by zooplankton can influence phytoplankton community structure (McCauley and Murdoch, 1987) but there was no evidence it was significantly influenced by grazing since there was no consistent correlation between the density of zooplankton and phytoplankton. Moreover, in the Lower dam, the zooplankton peak occurred almost four months after the peak in the phytoplankton. Nevertheless, the possibility that these changes came about because of interactions between zoo- and phytoplankton cannot be excluded as this has been shown in other studies (Reynolds, 1989; Naselli Flores and Barone, 1996). But while cladocerans, copepods and rotifers feed on a large range of algae with a degree of selectivity, factors such as size, shape, availability, palatability and toxicity of the algae and their effect on selectivity are too complex to elucidate in a field study (Wetzel, 2001). Comments on the possible effects of predation by zooplankton on the phytoplankton of these two dams are therefore likely to be speculative until more research has been done.

Phytoplankton diversity was much higher than that recorded for eutrophic Lake Chivero as has been found in the study by Robarts (1979) and Ndebele (2003) as it is dominated entirely by one species *Microcystis* sp. and *Melosira* sp. In the small dams several species dominated and this changed from time to time hence higher species diversity. Persistence of diel cycles of thermal stratification creates conditions causing phytoplankton to spend less time in the autotrophic zone where they can photosynthesize. This frequent mixing can result in lowered productivity as a result but may also give the advantage of rapid nutrient cycling. There is need for further research on this aspect.

During the first part of this study (January – October 2000), the zooplankton population was low in both dams and dominated by large cladocerans such as *Daphnia laevis*, and by adult copepods. It is possible that predation by these animals suppressed the numbers of nauplius larvae and rotifers, while grazing may have reduced the numbers of euglenophytes and bacillariophytes. Something then seems to have occurred in both lakes in October 2000 to bring about a change in the plankton although it is not clear what that might have been. Both dams were very low at the time and there was a sharp peak in oxygen demand in October implying increased availability of oxidizable organic matter. Associated with this increase is that of suspended solids that reached a peak at the time as well. Particulate organic matter is a component of suspended solids and the suggestion is that if zooplankton can feed directly on POM as a food source then this would explain the sudden increase in abundance during the early rainy season period. Observations have been made by Darnell (1961) in Lake Pontchartrain where suspended organic matter rich with bacteria was the preferred food source for zooplankton rather than phytoplankton. A similar explanation was advanced by Imevbore (1967) on the relationships between plankton and the hydrology of Eleiyele reservoir in Nigeria.

The marked increase in the zooplankton populations in both dams was characterized by the loss of cladocerans and adult calanoids and the dramatic increase in the numbers of rotifers and nauplius larvae. The decrease in chlorophytes and their replacement by euglenophytes and bacillariophytes may suggest a change in the pattern of grazing and predation by zooplankton. Whatever the reasons for these changes, they highlight the fact that these small dams seem to be relatively variable environments and when enough studies have been done their predictability may improve even if their temporal variability does not. In comparison to other small dams investigated by Green (1990), the two small dams had more rotifer species (11+) and 6 cladoceran species but this is likely to be a result of a very short sampling period by Green (Elenbaas and Grundel, 1994) because seasonality is evidently an important factor determining abundance and diversity of zooplankton. Lake Chivero & Cleveland dam share similarities in the seasonal abundance of zooplankton with a summer peak dominated by rotifers as in the two small dams. There were 7 cladocerans, 17 rotifers and 18 rotifers in Lake Chivero and Cleveland dam respectively (Elenbaas and Grundel (1994). If we consider the unidentified rotifer species in the two small dams, perhaps the number of species would be almost similar. Abundance was comparable amongst the 4 dams, although rotifers during the summer peak were more abundant in the two Munwahuku dams.

CCA revealed identified that environmental factors were very important in the structuring of the zooplankton communities of both dams hence explaining a significant proportion of their variation. Chemical oxygen demand (COD) and temperature (WT) were important variables in both dams in determining zooplankton species composition and abundance. Suspended solids were also identified as significant factor in the Lower dam while Secchi disc transparency was important in the Upper one but the two factors are related. It might be suggested that increases in suspended solids interfered with the success of cladocerans noted especially by the disappearance of *D. laevis* and the decline of all the remaining species after a period of high turbidity.

Mechanical interference by suspended clay particles was found to reduce feeding rates by as much as 70% and to also severely restrict growth and reproduction of cladocerans although large-bodied cladocerans were deemed less vulnerable (Kirk, 1991, 1992) and this could also happen in small dams. Cladocerans are less selective in differentiating between inorganic particles and algae but rotifers are more efficient hence they are less affected by high amounts of suspended particles (Kirk, 1992; Kirk and Gilbert, 1990). The guts of filter-feeding cladocerans filled with clay at a time of high turbidity (Hart, 1987, 1988) and this will also have the negative impact on their buoyancy as more energy is then required to remain at selected depths. A positive effect of increased turbidity for cladocerans is that predation pressure by visual planktivorous fish becomes less but this is of benefit if there are no corresponding declines in photosynthetic activity and phytoplankton standing crop. In turbid reservoirs or periods of high turbidity rotifers would theoretically thrive better than cladocerans and this could explain their sudden dominance when the two small dams became turbid towards the end of the dry season.

Climatic and hydrological factors which influence internal cycles of plankton productivity could be the most important determinants of community structure in small dams. This is because their effects are more pronounced than in larger dams so changes in water chemistry and thermal stratification are rapid while the short generation times of plankton enables them to respond quickly to a changing environment. The duration of water in ponds was found to account for significant shifts in zooplankton community structure (Schell *et al*., 2001) and this might have important implications for small dams as well.

In Lake Arancio, Italy, strong water level fluctuations was one of the most important factors governing cladoceran dynamics via bottom-up and top-down forces (Naselli-Flores and Barone, 1997). The frequent variations in water depth may cause changes to the mixing depth-euphotic depth ratio as well as changes in turbulence which affects phytoplankton succession and thus impacting on the zooplankton community (Reynolds, 1989). Fluctuations in water level had an influence on predation pressure exerted by planktivorous fish as a decrease in transparency may have caused a decline in the reproductive success of the fish in the lake. As a result predation pressure exerted by fish fry on zooplankton was decreased (Naselli-Flores and Barone, 1997).

The structure of other trophic levels, especially the effects of fish, can also alter the seasonal responses of plankton (Sondergaard *et al.*, 1990; Korponi *et. al.*, 1997). Fish were present in both dams but their possible impact on the plankton could not be determined. Adult catfish *Clarias gariepinus* feed on zooplankton through indiscriminate sieving of water (Munro, 1966: Allanson *et al*., 1990) while others such as *Barbus paludinosus, Barbus trimaculatus, Tilapia sparrmanii* and *Oreochromis mossambicus* feed on zooplankton to some extent as juveniles (Munro, 1967). Intense predation by fish and dipteran *Chaoborus* larvae was shown to be more important in controlling cladoceran abundance than resource limitation in ephemeral water bodies (Twombly and Lewis, 1989). Therefore predation by fish might explain some of the changes in the plankton community dynamics but more work remains to be done before this explanation can be applied to these dams. Cladocerans and rotifers can produce diapause stages (May, 1986; Wolf & Carvalho, 1989; Nelson, 1996) that enable them to survive adverse conditions such as desiccation and more investigation is needed to assess the importance of recruitment from resting stages for plankton living in transient aquatic environments. Changes in water temperature would be an important cue for the hatching of resting eggs deposited in the sediment.

The intermediate disturbance hypothesis (Sommer *et al*., 1993) could be useful to explain plankton dynamics in small dams compared to large ones. Phytoplankton species diversity was higher in the small dams compared to Lake Chivero for example which is dominated by two species. But of course Lake Chivero has problems with eutrophication. In Brazil, a comparative study between a monomictic lake (Lake Dom Helvecio) and a polymictic reservoir (Barra Bonita) showed that the lake with its strong stratification was dominated by *k-selected* species (Copepoda and Cynophyta) (Tundisi and Tundisi, 1994). On the other hand, Barra Bonita reservoir which was characterized by temporal patterns of turbulence, stability, short retention time & other disturbance factors had much higher species richness and was dominated by *r-selected* species such as Rotifera (Tundisi and Tundisi, 1994). Further research on this aspect would be of great importance to our understanding of plankton dynamics in small reservoirs.

CHAPTER FIVE

THE STRUCTURE OF THE FISH COMMUNITIES

5.1 Introduction

Small dams experience extensive fluctuations in water level as well as their physical and chemical characteristics (chapter 2&3) but the effects of periodic droughts on fish communities has been little studied. In nature, most local fish can withstand fluctuations and this is what happens in rivers. Fish communities of some African lakes have species that can survive the harsh conditions of the drawdown and can colonize new habitats once the lakes are inundated again and the natural floods act as cues for large scale spawning of these species (Merron *et al*., 1993). Is this the same scenario in small dams?

Studies on Lake Chad during drought showed that the decrease in water volume concentrated fish present and correspondingly increased inter and intra-specific competition and their vulnerability to fishing gear (Benech *et al*., 1983). Increased shallowness in that lake resulted in resuspension of sediment that caused mass mortalities by damaging and clogging branchial tissues and also led to asphyxiation as release of organic matter created a high biological oxygen demand causing anoxic conditions to prevail (Benech *et al*., 1983). In a large dam like Lake Kariba, seasonal fluctuations in physico-chemical conditions affect abundance of *Limnothrissa miodon* through influences of river flow on their obligate food resource, zooplankton (Marshall, 1988).

Seasonal fluctuations in water level were the most important environmental factor affecting the fish community of Everglades marshes (Kushlan, 1976). When the water level was stable, fish density decreased but the biomass, average size, species richness and diversity increased. In a study on a Sri Lankan reservoir, the fluctuations in catch per unit effort of the cichlid *Oreochromis mossambicus* were dependent upon yearly fluctuations in water level the changes being manifested in the fishery after three years (De Silva, 1985). The effects of water level fluctuations on year-class strength of cichlids have also been reported on *Oreochromis macrochir* and *Oreochromis andersonii* from the Kafue floodplain in Zambia (Dudley, 1979).

The construction of a small dam has immediate consequences on thermal regimes and physical stability, food availability, breeding requirements and the loss of specialized habitats for riverine fish (Allanson *et al.*, 1990) and fish must adapt in order to succeed. Factors such as early maturation, high reproductive effort and shorter inter-brood intervals contributed to the successful colonization of the marginal areas of Lake Le Roux by *Barbus anoplus* (Cambray and Bruton, 1984).

Environmental stability, which is influenced by factors such as thermal regimes and stratification, availability of food and suitable breeding areas and the existence of specialized habitats determine suitability of dams for fish (Allanson *et al*., 1990). Very little is known about fish communities in small dams and their ecology in a rather unstable environment. Are the fish communities in the dams different in species composition, abundance, average size, reproductive activity and diversity? What are the possible factors that could account for any differences between the two dams? The changes in the fish communities of two small dams before, during and after the dry season were monitored to determine relationships between fish biology and the hydrological regime.

5.2 Methods

Sampling was done monthly for 20 months from November 1999 to August 2001. Before sampling commenced in 1999, all the pools upstream of the upper dam were cleared of any fish that were present in order to see what happens before, during and after the rains. This was done by electro-fishing. The sampling program in the dams was done using three types of fishing gear: a Smith Root VI-A electrofisher and fyke nets were used to sample the shallows while gill nets were set in the deep sections (> 1.5m) of each dam. Current settings used for the electrofisher varied depending on conductivity but frequent breakdowns of equipment and very low conductivity during the rainy season produced a discontinuous data set.

Two double fyke nets with a stretched mesh of 2.4 cm were set overnight at randomly selected sites in each dam up to a depth on 1.5m, from August 2000 onwards. The fleet of nylon monofilament gill nets with stretched meshes of 12, 20, 40, 80, 100, 140 mm and cotton multifilament nets with stretched meshes of 50, 75, 102, 127, 141, 152 and 165 mm were set overnight in each dam for 12 to 14 hours. They were used more extensively compared to the other two gears. Fish were identified and measured to total length (TL) to the nearest cm and weighed (g). Keys used for this purpose were found in Skelton (1993) and Bell-Cross and Minshull (1988). Gonad maturation was assessed on a simplified scale (Bagenal, 1968) as:-

(I) *Inactive* – includes the immature fish and adults in the resting stage. Sexual products have not yet begun developing, gonads are very small and eggs indistinguishable to the naked eye.

(II) *Active* – Stages when the eggs become distinguishable to the naked eye and testes change from transparent to a pale white color to when the sexual products can be discharged in response to very light pressure on the belly (' milking")

(III) *Spent* – sexual products have been discharged and gonads appear deflated. The ovaries may contain a few left over eggs or testes may have residual sperm.

If ripe gonads were present these were weighed and the eggs preserved in 4% formalin. Fecundity was estimated by weighing all the eggs and then determining weight of samples of 200 eggs. The total number was estimated by:

$$N = \frac{W_t}{W_g}$$

where N = total number of eggs, Wt = the total weight of the gonad and Wg = the weight of 200 eggs (Bagenal, 1968).

Condition factor was calculated by:

$$CF = \frac{W}{L^b}$$

where CF = condition factor, W = weight of fish (g) and L = total length (cm), (King, 1995). Hierachichal cluster analysis with group average linkage was used to determine similarity amongst different samples from the two dams. The computer software package PRIMER was used.

5.3 Results

Six fish species belonging to only 3 families: Cyprinidae, Clariidae and Cichlidae were caught in the two dams. The Clariidae were represented by *Clarias gariepinus* (Burchell, 1822), the Cichlidae *Oreochromis mossambicus* (Peters, 1852), *Tilapia sparrmanii* A. smith, 1840 and the Cyprinidae *Barbus paludinosus* Peters 1852, *Barbus lineomaculatus* Boulenger 1903 and *Barbus trimaculatus* Peters 1852, were caught during the entire sampling period with *B. paludinosus* being the most numerous in both dams. From November 1999 to January 2000 *B. paludinosus* was only caught in the Lower dam although the numbers declined significantly in January & Februaury 2000. In Februaury 2000 this barb was then caught in the Upper dam in greater numbers compared to the population in the lower dam. The populations in each of the dams later began to increase in number but for the entire sampling period and with a few exceptions, numbers of *B. paludinosus* were higher in the Upper dam than the Lower one (Table 5.1).

Clarias gariepinus first appeared in the Upper dam in January 2000 after it filled and it was the only species there at that time but in the Lower dam it had been caught in November 1999. The greatest number of catfish was caught in the Upper dam in July 2001 (31) and were mostly juveniles whilst in the Lower dam, 27 were caught in May 2001 (Table 5.1). The catfish occurred in much smaller numbers compared to the barb but after *B. paludinosus*, the most frequently occurring fish during the sampling program (Table 5.1). The other species occurred occasionally and were not significant in terms of numbers. The cichlids *O. mossambicus* and *T. sparrmanii* occurred more frequently in the catch from August 2000 to August 2001 (Table 5.1). In the Lower dam, *C. gariepinus* was sometimes the most abundant species when few or no *B. paludinosus* were caught (Table 5.1).

C. gariepinus generally made up the bulk of the ichthyomass especially in the Lower dam although *B. paludinosus* sometimes replaced when few *Clarias* were caught (Table 5.2). This is simply because of they catfish grow to much larger sizes compared to barbs. A slightly different trend occurred in the Upper dam where, even when both species were present, *B. paludinosus* sometimes contributed the greater percentage of the ichthyomass (Table 5.2). This is because *B. paludinosus* were more numerous in the Upper dam compared to the Lower one, while *C. gariepinus* were smaller in the Upper dam (Table 5.2). *O. mossambicus* and *T. sparrmanii* sometimes contributed significantly to the ichthyomass because they grow to a larger size than *Barbus* species but they were always few in numbers (Table 5.1 & 5.2). Male *B. paludinosus* were only caught by the electrofisher in small numbers because of their smaller size. Only a few juvenile catfish were caught with the electrofisher as larger fish probably evaded capture.

There was high similarity among samples of the fish community of the Upper and Lower dams (Figure 5.1) and so differences between the fish populations were small.

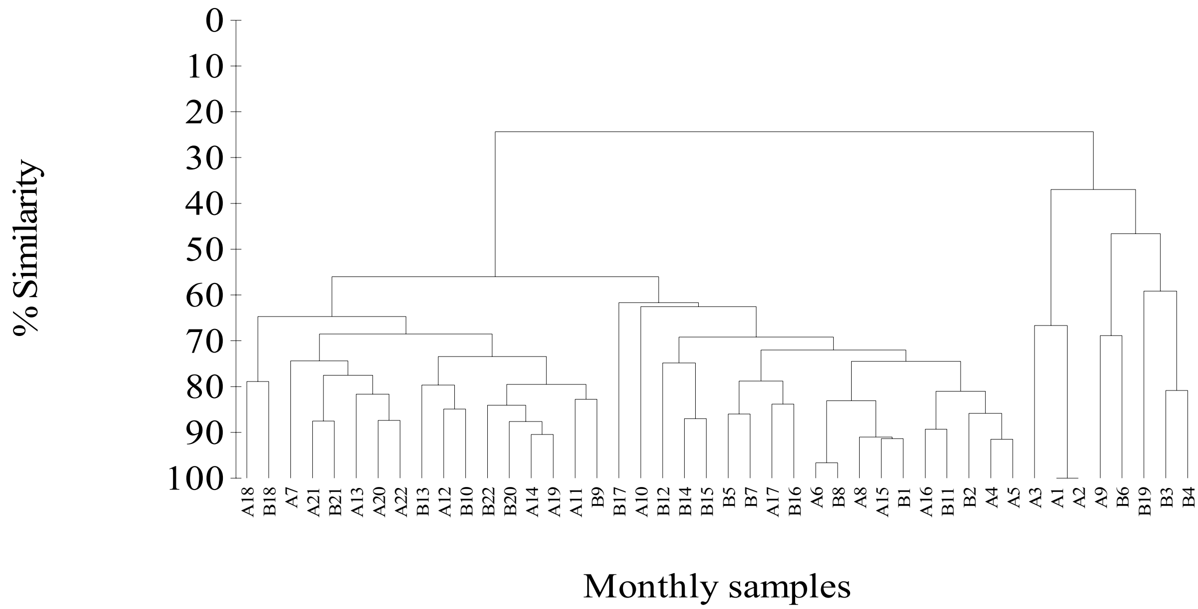


Figure 5.1. The similarity (%) of fish communities in the two dams, Nov. 1999-Aug 2001.
Key: Upper dam (A1 = Nov. 1999, A2 = Dec. 1999…….A22 = Aug. 2001);
Lower dam (B1 = Jan. 2000, B2 = Feb. 2000…….B20 = Aug. 2001)

Table 5.1. The total number of each fish species caught in the two dams combining catches by gill nets, fyke nets & electrofisher, November 1999 – August 2001

Species		N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A
B. paludinosus	U	-	-	-	68	62	78	447	94	23	108	233	141	319	208	113	90	38	521	244	422	703	385
	L	80	71	7	1	47	6	47	76	173	135	95	51	66	67	85	46	60	284	-	202	679	311
C. gariepinus	U	-	-	4	4	12	-	1	-	2	10	20	2	4	4	-	9	2	25	3	7	31	8
	L	6	-	8	9	11	3	-	-	4	8	2	10	3	-	-	1	16	4	27	10	7	3
O. mossambicus	U	-	-	-	-	-	-	-	-	1	-	-	-	-	7	-	1	-	19	10	2	8	4
	L	-	-	-	-	-	-	-	-	-	5	1	7	2	6	24	3	5	35	-	24	7	17
B. trimaculatus	U	-	-	-	-	-	3	-	1	-	43	-	8	4	-	-	-	-	6	1	13	23	4
	L	-	-	-	-	-	-	5	4	-	2	-	8	2	-	-	-	-	-	-	2	-	-
B. lineomaculatus	U	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	63	-	-	-	-
	L	29	17	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	67	-	-	32	-
T. sparmanii	U	-	-	-	-	-	2	-	-	1	-	3	3	53	-	-	-	-	-	1	-	16	17
	L	-	-	-	-	-	3	-	1	-	2	1	-	5	1	1	-	24	12	-	-	7	7

Table 5.2. The total weight (nearest g) of each fish species caught in the two dams combining catches by gill nets, fyke nets & electrofisher, November – August 2001

Species		N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A
B. paludinosus	U	-	-	-	472	368	667	1622	640	125	682	464	3137	3167	1073	361	240	245	335	1252	2027	672	835
	L	84	79	30	4	364	8	302	530	643	125	374	1195	2016	200	319	126	259	220	-	6872	518	1831
C. gariepinus	U	-	-	160	1041	544	-	55	-	112	794	150	162	167	77	-	510	24	155	23	1770	9681	3195
	L	1786	-	6029	14670	6136	483	-	-	1897	1115	2480	3668	218	-	-	7	13194	165	31900	9694	4803	550
O. mossambicus	U	-	-	-	-	-	-	-	-	79	794	150	162	167	77	-	-	24	156	23	1770	9681	3191
	L	-	-	-	-	-	-	-	-	-	62	-	320	960	71	108	11	27	495	-	4158	294	849
B. trimaculatus	U	-	-	-	-	-	20	33	5	-	337	-	62	13	-	-	-	-	8	7	94	63	65
	L	1	21	-	-	-	-	38	22	-	3	-	58	-	-	11	17	-	-	-	-	-	4
B. lineomaculatus	U	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	L	16	12	-	-	-	-	-	-	-	-	-	-	6	-	-	-	-	20	-	-	36	-
T. sparmanii	U	-	-	-	-	-	-	-	-	29	72	5	904	-	-	-	-	-	-	41	-	42	820
	L	-	-	-	-	-	17	-	38	-	7	426	-	-	19	92	-	150	120	-	-	159	557

Table 5.3. The range of total lengths (TL, mm) of six fish species caught in the Upper and Lower dams, Nov. 1999-Aug. 2001.

	Upper dam	Lower dam
C. gariepinus	51 - 645	38 - 630
B. paludinosus	17 - 116	15 - 108
B. trimaculatus	65 - 123	39 - 114
O. mossambicus	25 - 131	18 - 206
T. sparrmanii	11 - 230	35 - 195
B. lineomaculatus	18 - 66	19 - 78

Data from electro-fishing, gill and fyke netting were used to plot length frequency diagrams. This analysis was done for only two species *B. paludinosus* and *C. gariepinus* because sample sizes of the other four species were too small to come up with meaningful diagrams. The data from electro-fishing was irregular and therefore patterns were difficult to discern so mainly data from the other gear are explained. In the Upper dam *B. paludinosus* appeared in then catches in Februaury 2000 as adults with the dominant size class being 85-90mm (Figure 5.2). In March 2000 the 75-80mm was not caught and while there was a shift to the right with an increased frequency of 90-95mm sizes the 85mm class remained dominant. In April 2000 the 90-95mm size class dominated. This pattern was disrupted in May 2000 after heavy floods late in the rainy season (Figure 5.2). This could have facilitated movement of fish from pools upstream back into the Upper dam and these were mainly adults as the size range did not change significantly. The pattern is similar for the Lower dam from January 2000 to May 2000 although larger size classes were dominant (Figure 5.3).

The 85-90 class strengthened in July 2000 and then in August 2000 there was a shift downwards so that the dominant classes were now 70-85mm range perhaps suggesting recruitment into the main stock. In September 2000 the 70-80mm size class remained dominant as larger size fish declined in occurrence and only became significant in April 2001. It is not clear what cause this but fishing pressure could be a factor. From November 2000 the dominant size class declined from 75-80mm to reach 55-60mm in January 2001. This observation in January when the river started flowing could be an indication of upstream migration but these changes were not matched by those in the Lower dam as the 70-75mm class remained dominant from November 2000 to January 2001 then March 2001 to April 2001. In the Upper dam when water levels were lowest fish catches were magnitudes greater than in the Lower dam simply because fish were concentrated in a smaller volume and hence the changes in the population structure as more susceptible large classes were caught and removed from the population. Larger size classes occurred in the Upper dam from April 2001 onwards. Fish catches again increased dramatically in both dams from April 2001 to August 2001 in both dams and this was again due to increased fishing pressure as this was a drought year and water levels fell quickly in both dams.

Nothing meaningful could be discerned from length-frequencies of *C. gariepinus* in either dam mainly as a result of few samples hence they were grouped into six-month samples (Figure 5.5). When the Upper dam filled after a year when it was dry the first catches comprised *C. gariepinus* most likely because of its ability to migrate over long distances on land. Changes in the *B. paludinosus* populations' of the two small dams were further analyzed using cumulative length frequencies plotted on a probability scale against length to get some idea of migration, spawning events and recruitment (Figure5.3a & b). In the Lower dam in January 2000, 3 classes were identified all confined to the 70-80mm length class and then in March 2000 about 5 classes were identified in the population all above 80mm (Figure 5.4a).

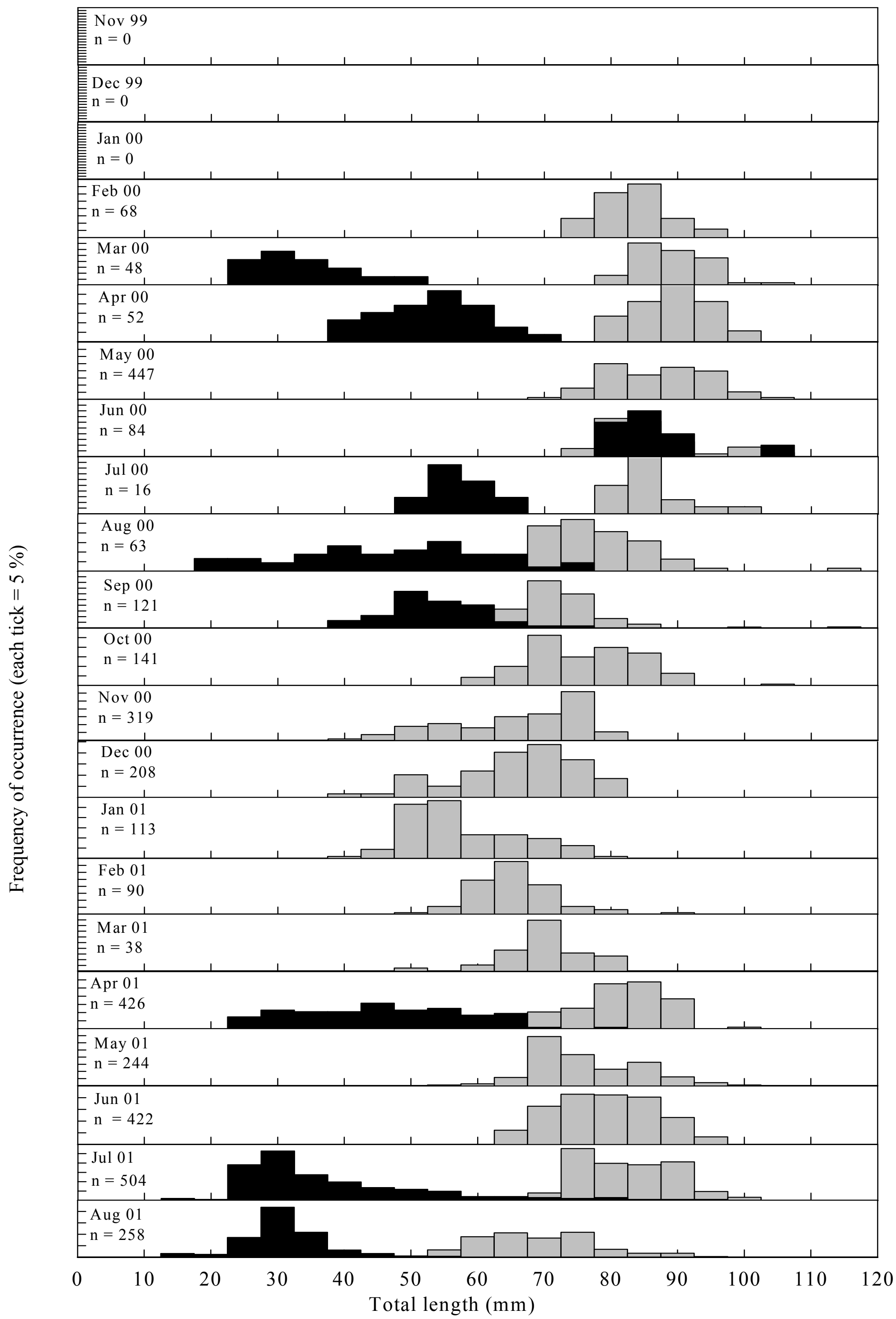


Figure 5.2. The length frequency distributions of *B. paludinosus* caught in the Upper dam with fyke and gill nets (grey shading) and the electro-fisher (black shading) (Jan. 2000 – Aug. 2001)

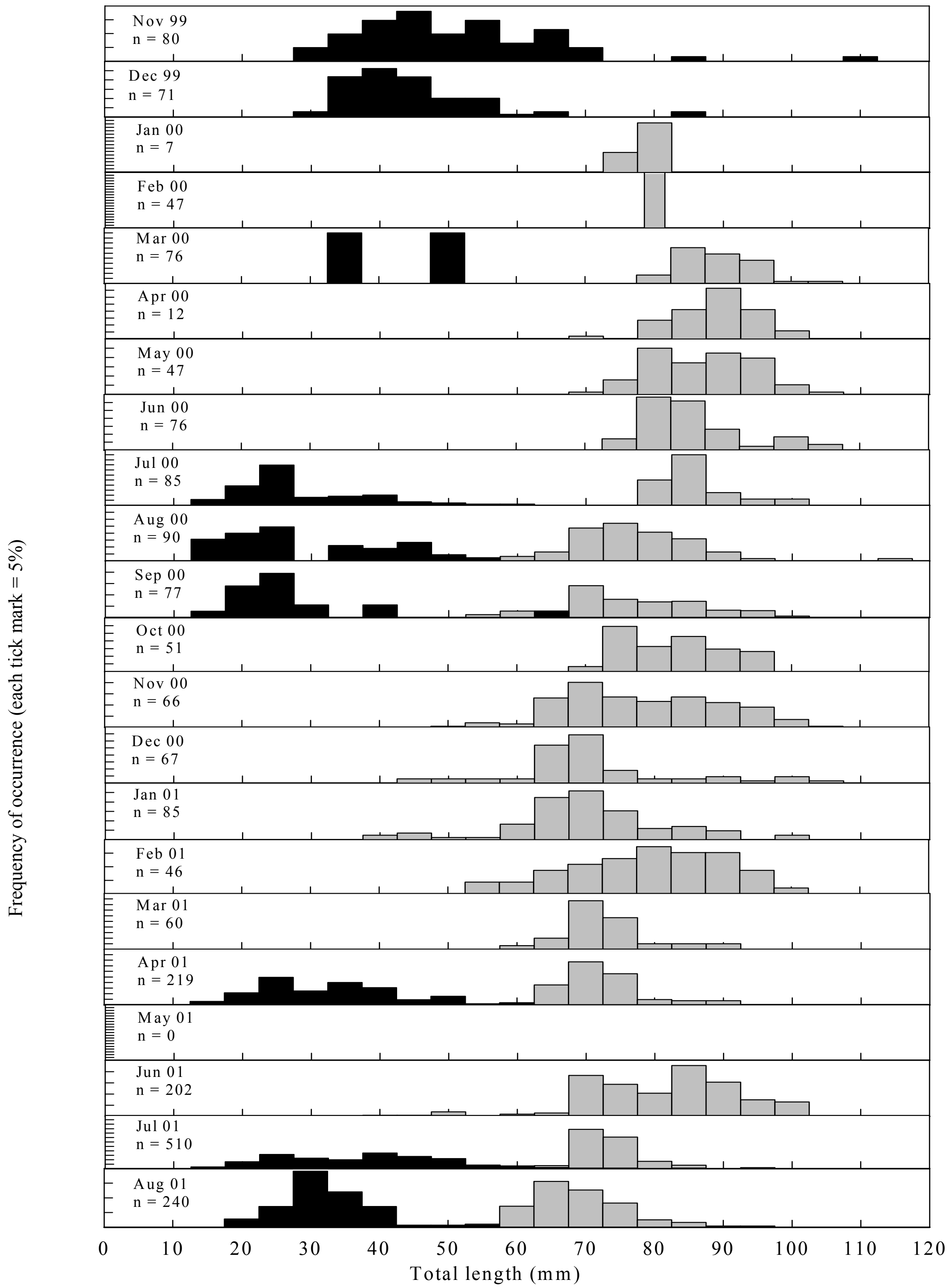


Figure 5.3. The length frequency distributions of *B. paludinosus* caught in the Lower dam with fyke and gill nets (grey shading) and the electro-fisher (black shading), (Jan. 2000 – Aug. 2001)

About 5 size classes of fish occurred in the Upper dam in February 2000 and in March 2000 only 3 classes could be clearly identified with the loss of the < 80mm class which might suggest growth rather than recruitment as these were large adults already. In April and May 2000 about 5 classes could be identified with the < 80mm class occurring in the Upper dam population again while in the Lower dam about 4 similar classes occurred during the same period and none was below 80mm (Figure 5.4a). These similar trends occurred in the June and July 2000 with higher frequencies in the Upper dam. In August 2000 smaller size fish appeared in the catch of both dams and this could be a result of recruitment into the fishable stock (Figure 5.4a). Small size classes (< 50mm) appeared in the catches in the months of November 2000 – January 2001 and this could be another recruitment phase. Also this period no fish > 80mm was caught in the Upper dam whilst there was a large population in the Lower dam. It might be suggested here that migration up the stream although flow was not significant could account for the changes observed (Figure 5.4b). Differences between the populations were small for the remaining months.

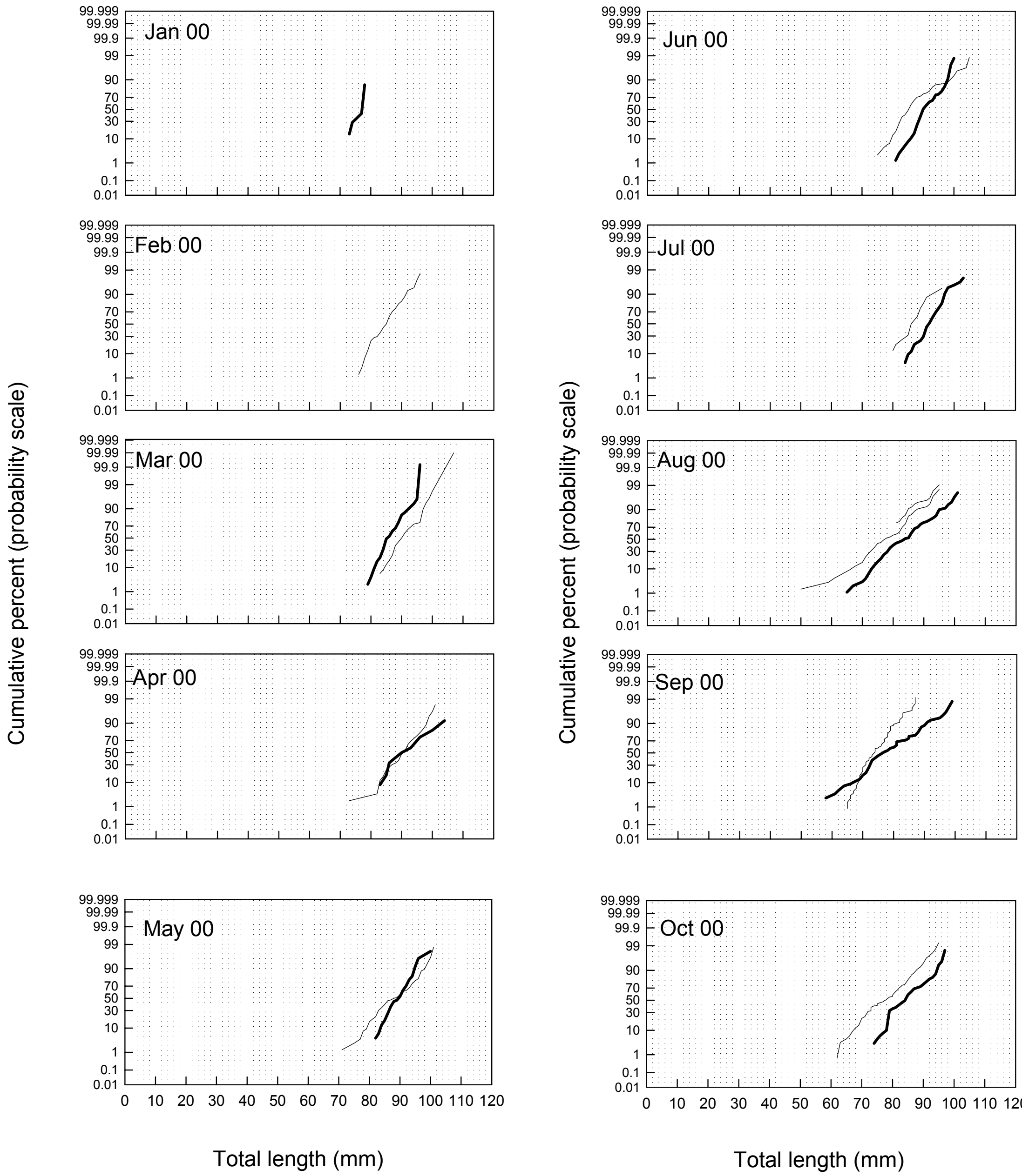


Figure 5.4a. Cumulative frequency (probability scale) plotted against size class of *B. paludinosus* in the two dams. (thin line = Upper dam, thick line = Lower dam), January 2000 – October 2000.

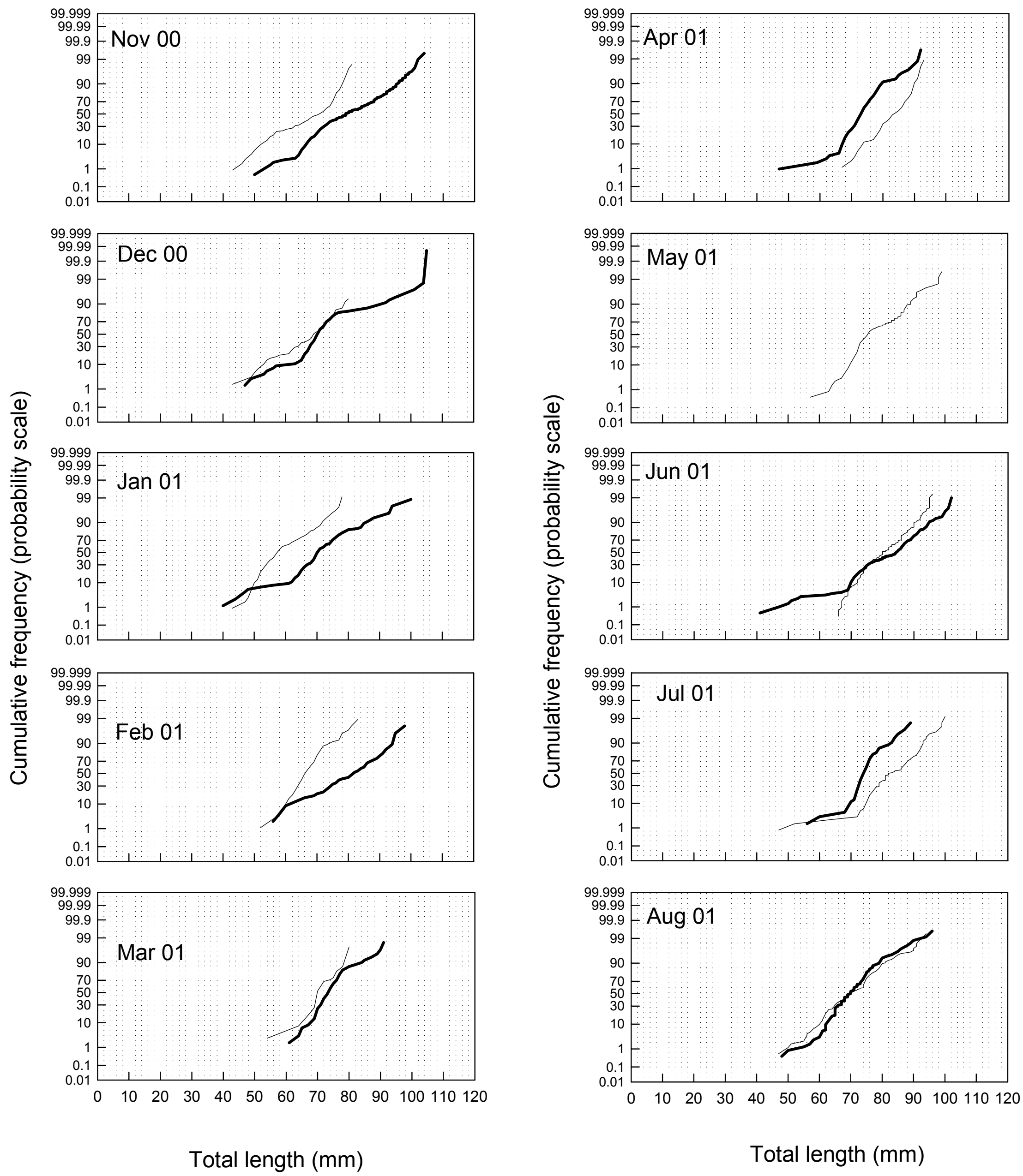


Figure 5.4b. Cumulative frequency (probability scale) plotted against size class of *B. paludinosus* in the two dams. (thin line = Upper dam, thick line = Lower dam), November 2000 – August 2001.

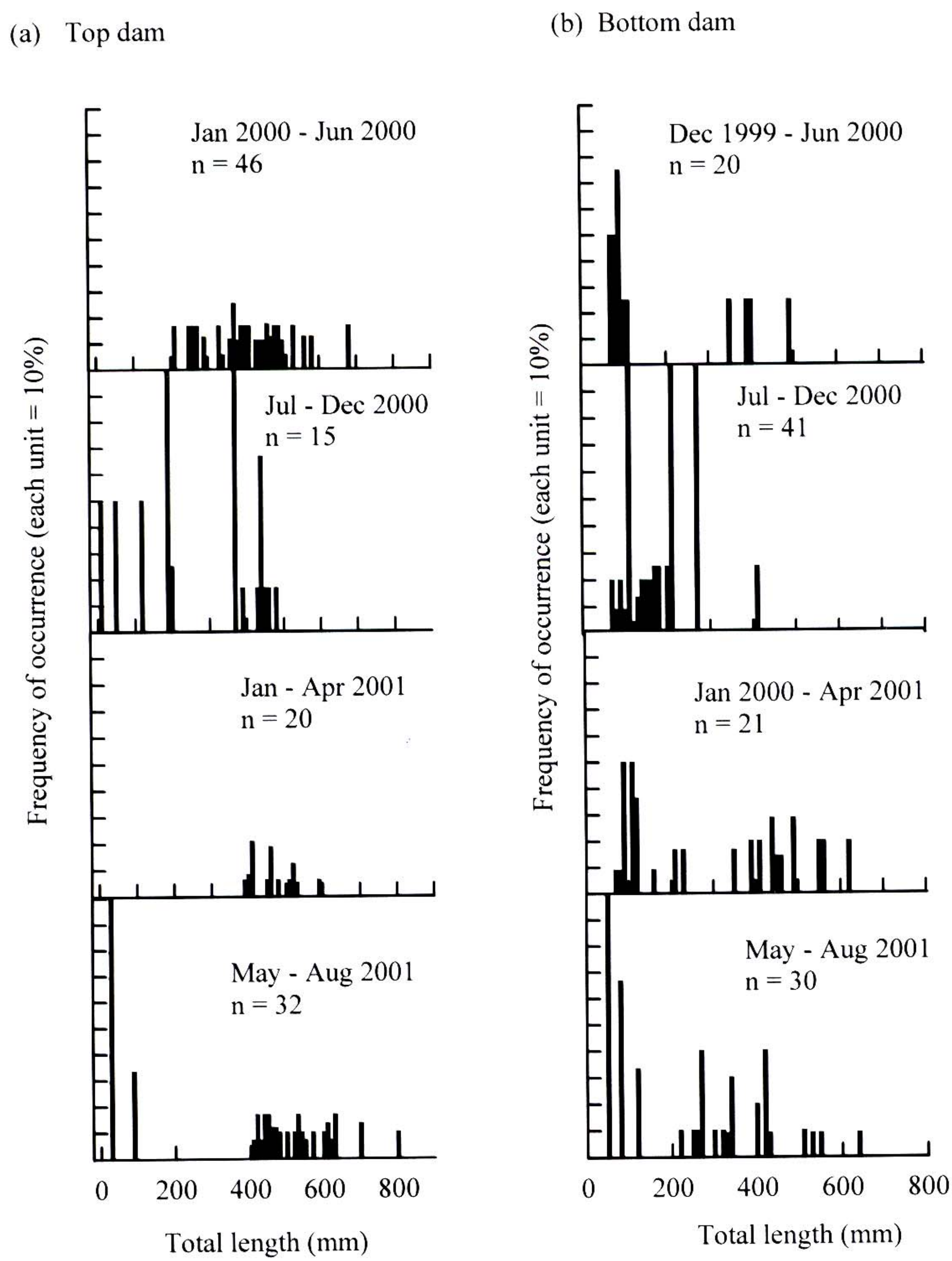


Figure 5.5. The length frequency distributions of *C. gariepinus* caught in (a) the upper dam and (b) lower dams with an electrofisher, fyke nets and gill nets (Nov. 1999 – Aug.2001)

There was an increase of mean fecundity with increasing fish size for *B. paludinosus* against length although wide ranges can be see as shown by the error bars (Figure 5.6). This is because of the seasonality in the breeding activity of the fish (Figure 5.7). Relative fecundity was highest when the dams were full or in the process of filling but the main breeding activity was suspected to be largely determined by onset of rains when spawning and migration begin (Figure 5.7). Condition factor did not change much and there were evidently no seasonal variations but for *B. paludinosus* and *C. gariepinus* (Figures 5.8 a & b). Growth was isometric (increasing in all dimensions at the same time) in *B. paludinosus* and *C. gariepinus* in the Upper and Lower dams with a cubic relationship between length and volume. In all cases *b* was close to 3 for both species in each of the dams (Figure 5.9). For both species deviation from the cubic relation was greatest fro fish caught in the Lower dam compared to the Upper one (Figure 5.9).

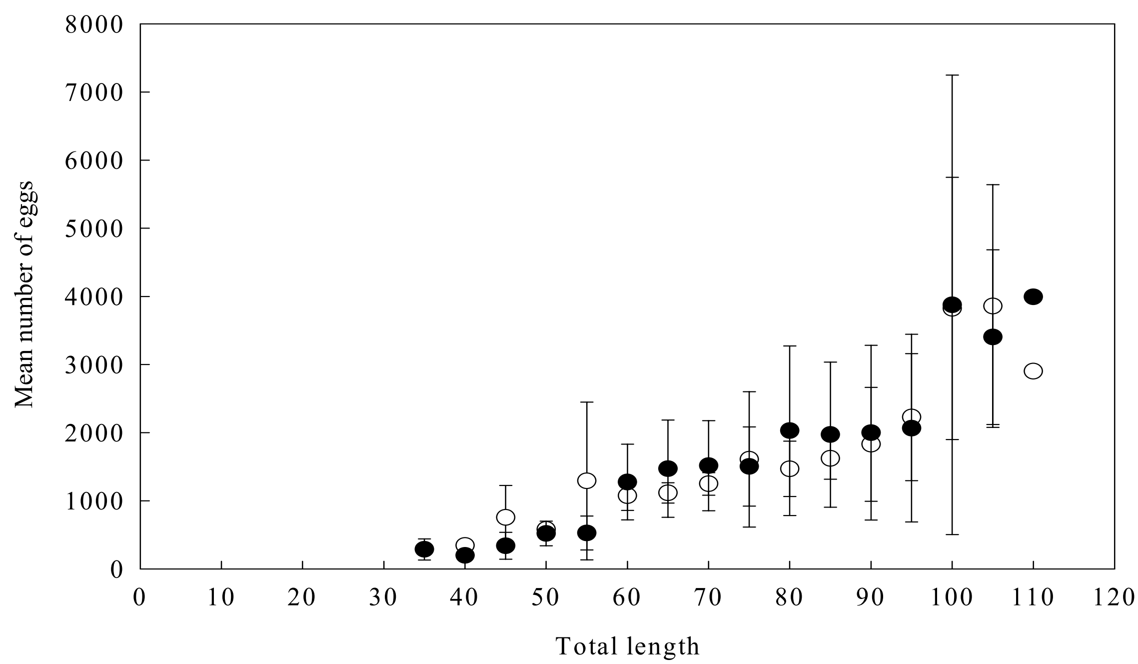


Figure 5.6. Fecundity (mean number of eggs ± SE) against the different size classes of *B. paludinosus* in the Upper (○) and Lower (●) dams

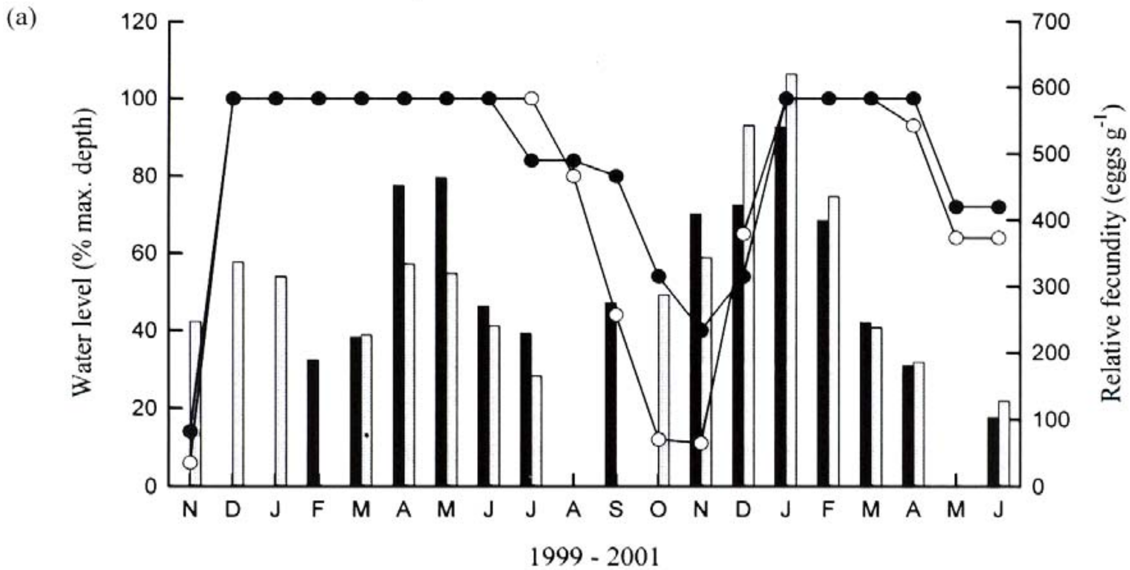


Figure 5.7. (a) Relative fecundity (upper dam ■; lower dam □) of female *B. paludinosus* in relation to fluctuating water levels (Upper dam ○; Lower dam ●).

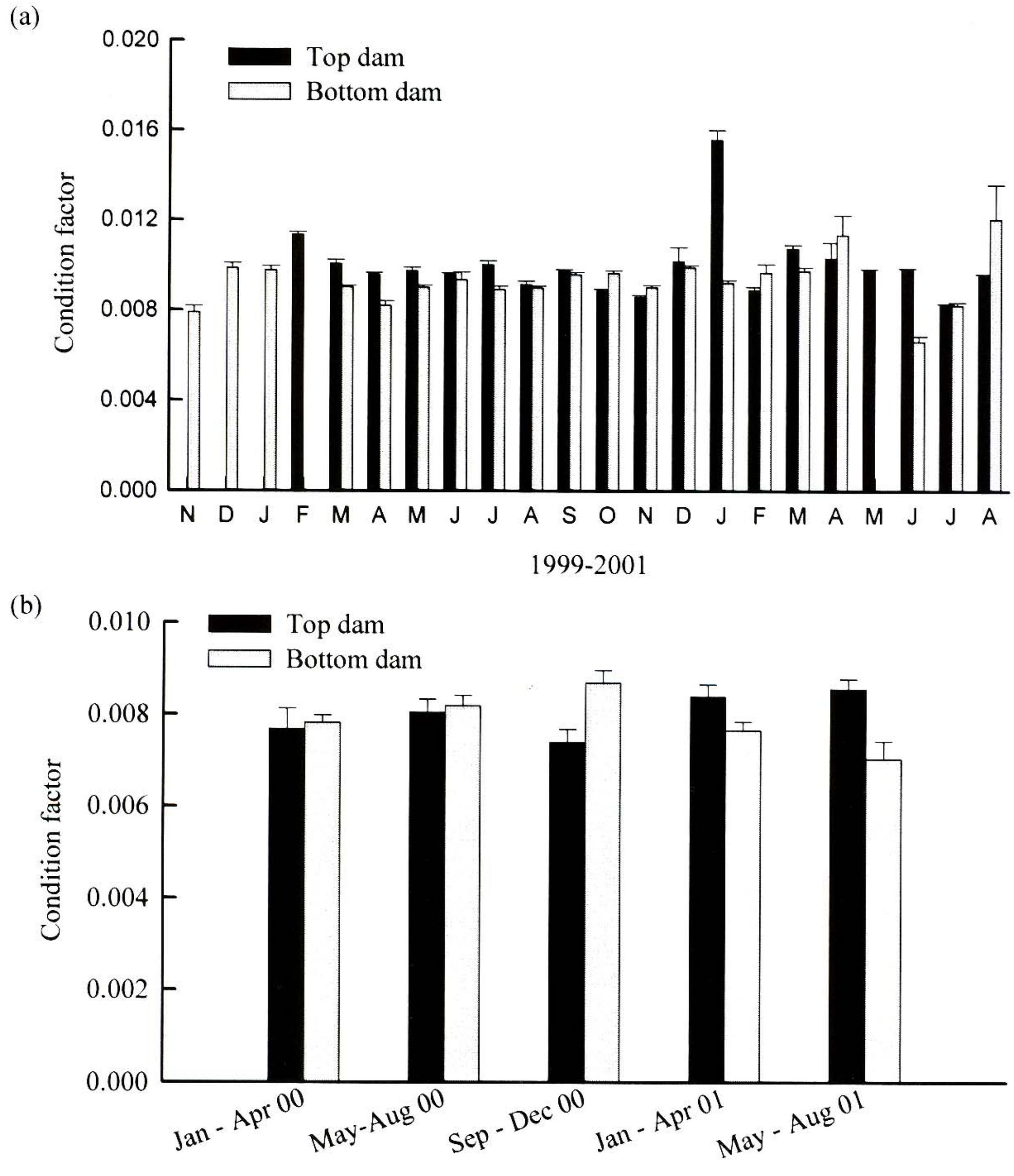


Figure 5.8. The condition factor of (a) *B. paludinosus* and (b) *C. gariepinus* in the upper and lower dams

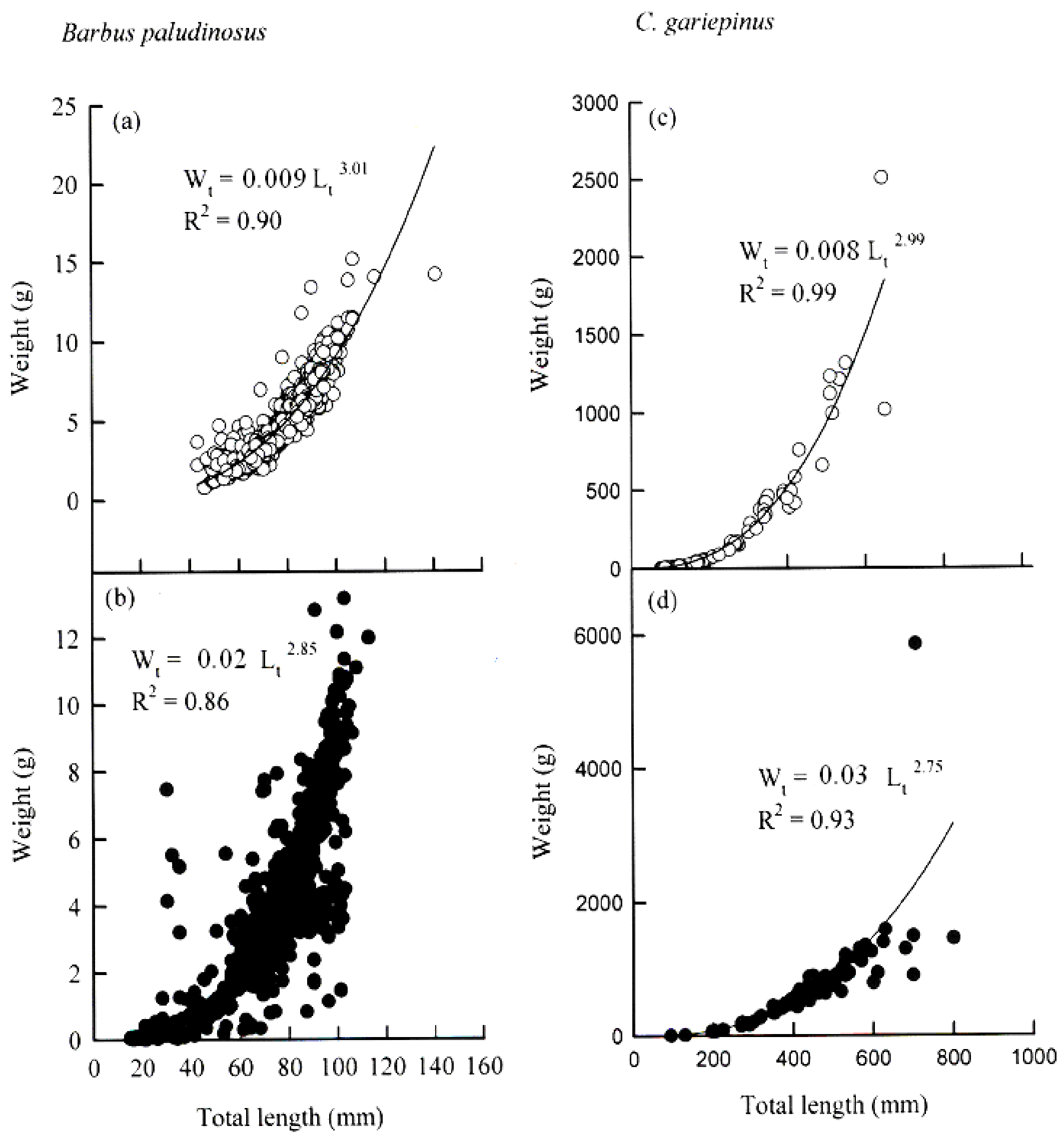


Figure 5.9. The length-weight relationships ($W = aL^b$) of *B. paludinosus* and *C. gariepinus* in the Upper dam (a & c) and Lower dam (b & d)

5.4. Discussion

Few fish species from only three families were recorded in the two dams and this is likely to be a result of fewer species occurring in the Munwahuku River itself compared to large rivers such as Zambezi River (21 families and 149 species) (Skelton, 1994). Consequently this has contributed to fish diversity in large dams such as Lake Kariba with over 40 species of fish recorded (Bowmaker *et al*., 1978). A fishing exercise was done on Munwahuku downstream of the two dams near Porta farm and 7 species were caught which were *Micropterus salmoides*, *B. paludinosus*, *B. lineomaculatus*, *C. gariepinus*, *Labeo cylindricus*, *T. sparrmanii* and *Serranochromis robustus* (Gratwicke, 2003). Only 4 of these were caught in the small dams. The occurrence of the two predators *M. salmoides* and *S. robustus* in Munwahuku River might be a key factor regarding fish diversity in these dams as their occurrence in many streams in Manyame catchment was correlated with low species diversity especially of cyprinids (Gratwicke, 2000). We conclude therefore that since the fish community in any man-made lake is primarily determined by the composition of the fish population in the river there were small differences between the dams and large differences when compared to other large reservoirs.

It is also unlikely that fish communities in such small dams will evolve into a diverse complex assemblage on the scale as that of large dams like Kariba where over 40 years the fish community has changed with the decline of such species as *Labeo congoro* and *L. altivelis* (species are closely associated with flowing water) after the dam filled and the successful introduction of *Limnothrissa miodon* to occupy the pelagic niche (Bell-Cross and Bell-Cross, 1971). These species are closely associated with flowing water and so loss of habitat was a contributory factor. The situation in small dams is different in that water residence times are very short and the suggestion is that as far as fish are concerned they could just be an extension of the main river. Extensive and frequent fluctuations of water levels and shoreline exposure up to 95% of the surface area suggest that they are frequent habitat shifts in small dams and hence the loss of niches that would allow for a diverse fish community to develop. These changes are such that the Upper dam with an area of 5ha and an established macrophyte community at full capacity was reduced within a few months to a muddy pool less than 1m deep and this is drastic for fish. Such changes might favour small organisms that quickly adapt such as plankton but for fish most of which require specialized habitats it might be a very difficult environment. Thus while fluctuations in water level are characteristic of every reservoir the loss of habitats is much less for larger dams. It is inconsequential to suggest a pelagic zone exists in the true sense for small dams (< 5m deep) thus the structuring of habitats seen in large lakes and reservoirs does not apply to the same extent.

Attempts to measure migration patterns in the river and on the spillway were not successful because when fish traps were set they were ripped off by high currents. The only indication that there were fish migrations is that the Upper dam which had no fish prior to the sampling exercise because it was completely dry was colonized first by *C. gariepinus* which migrates to breed on floodplains but can also breed in the shallow flooded grasslands near the dam (Kenmuir, 1976; Bruton, 1979; Clay, 1979; Bell-Cross and Minshull, 1988). Then *B. paludinosus* appeared next and the only logical conclusion was that they migrated from the Lower dam via the grassy, gently sloping spillway. Length frequency analyses for *B. paludinosus* did not clearly bring out aspects of migrations, spawning events and recruitment as might have been desirable because small size (< 40mm) fish could only be caught by the electrofisher which had too many breakdowns and its sampling efficiency was greatly reduced by low conductivity during the rainy season. Further investigation to determine to what extent migrations in and out of small dams account for the fish community structure is still required since we now know they take place.

The maximum size attained by *C. gariepinus* was 650mm (TL) and 630mm (TL) in the Upper and Lower dams respectively and this was much smaller to that in Lake Kariba (820mm TL)

(Kolding *et al*., 1992). Maximum size of *B. paludinosus* was 115mm and mm in the Upper and Lower dams respectively while another record has 130mm in (Bell-Cross, 1976). These data would have been more useful if the ages of the fish were known but it can mean that for these two species maximum size can be attained in the conditions prevailing in small dams. Length-weight relationships showed that *B. paludinosus* and *C. gariepinus* in the Lower dam increased less in weight than predicted by their increases in length compared to those in the Upper dam as the value of b ($W = aL^b$) was less comparatively. There were no seasonal variations in condition factor of *B. paludinosus* and *C. gariepinus* in the two dams and no differences of note between samples from the two communities. Since we used monthly samples and this suggests that for the fish, variations in food availability were not significant and there was little change in the average reproductive stage of the fish populations as this influences condition. Any differences in plankton abundance and composition between the two dams did not influence the condition of the fish.

Environmental factors in the small dams' fluctuated and most extreme fluctuations were in conductivity, suspended solids and water transparency. High amounts of suspended solids has been known to cause fish mortalities by clogging up gill filaments (Benech *et al*., 1983; Bruton, 1985) but in these two dams no fish were observed to die from such causes. The impact of increased turbidity and reduced water transparency on fish activity such as reproductive and feeding behaviour were not assessed in this study but this could have adverse effects.

In large dams macrophytes play an important role in increasing g species diversity as the establishment of such vegetation creates environments that are suitable for by increasing food availability, spawning areas and providing shelter from predators for minnows and cichlids. This saw an increase in species recorded in Lake Kariba from 28 before impoundment to over 40 (Bowmaker *et al*., 1978). This is unlikely to happen in small dams because the dynamics of this vegetation are very high because of severe drawdowns and might not be beneficial to juveniles because when water levels decline they are exposed to predators. Plant feeders such as *Tilapia rendalli* might not do well in small dams because of a likely shortage of food when water levels decline.

Water level fluctuations that are a result of seasonal river discharges and variable water residence times might not directly affect the populations but reproductive activity has been found to be adversely affected by massive increases in turbidity typical of small dams when water levels are low. This was not evident for these dam as relative fecundity actually increased when water level were lowest and the water most turbid. Perhaps because such events are not prolonged lasting 2-3 months the effects on reproductive success of fast growing and hardy species such as *B. paludinosus* would be less evident. The main spawning period for this species was identified as occurring from October-December in Lake Chilwa, Malawi (Howard-Williams *et al*., 1972) but from this study the measure of relative fecundity indicates a shift from January to April as the main breeding season. This might have been caused by the changes in the rainy season patterns as the main rains causing stream flow occur in January. Some measure of activity continues throughout the year but in the dry season there is a significant reduction in relative fecundity.

CHAPTER SIX

CONCLUSION

The results of a broad investigation into limnological and ecological aspects of small dams were presented. The main objective of the study was to determine the effects of different hydrological regimes in two small dams on their limnology. Several aspects were studied and the following conclusions were reached. With regard to thermal and oxygen stratification regimes it was established that a seasonal pattern of stratification typical of other large dams applies for small dams but furthermore small dams are characterized by a strong diel stratification cycles which are influenced by external fluxes mainly changes in short wave solar radiation income affecting water temperatures. During the hot summer diel stratification of temperature and oxygen could be observed but this was less marked during winter when temperatures were very low. This finding has important implications for nutrient exchanges between sediments and the water and phytoplankton distribution within the water column which in turn will affect primary productivity.

Water levels were important for stability because they influenced the depth of the thermocline and the Upper dam particularly was most stable when full but compared to some large dams, small dams are very unstable. Thus frequent mixing can be expected in small dams because of their low stability compared to larger lakes and dams. Flow-through regimes during the rainy season were also alluded to as an important factor preventing the establishment of stable stratification in small dams especially during heavy floods when retention times could be reduced to a few hours. This was also observed by Schrader (1958) that reservoir stratification got weaker when throughflows increased beyond certain limits. Thermal stratification also depends on retention time with temperature differences between surface and bottom water increasing with retention time (Groeger and Tietjen, 1998). Starskaba and Mauersberger (1998) also demonstrated this dependence for deep

(30m depth) Central European reservoirs and described the relationship by an equation. This has not been done for tropical dams but it explains why stable thermal stratification did not develop in these small dams for long periods. Overall differences between the two small dams were minor perhaps because in terms of size they are basically on the same scale and the effect of environmental fluxes would be similar.

The second objective of the study focused on water chemistry and some aspects of sediment chemistry. This study found seasonal trends in the variations of conductivity, TDS, Secchi disc transparency, total suspended solids, COD, BOD but not pH and alkalinity. It was suggested that these seasonal fluctuations are mainly a result of changes in hydrology i.e. dilution effects of rain, floods because of river inflows and low water levels with regards to water transparency and concentration of ions resulting in increased conductivity. The role of sediments on quality of water was not studied in detail but differences were found in the distribution of important minerals such as phosphorus and in sediment texture characteristics between sediments that are frequently exposed and those that are mostly inundated. This has important reference to the processing of allochthonous material brought into the dams. The were no significant differences between the dams in water chemistry and this was mainly because of short water residence times in the dams leading to questions about the role sediment plays in determining water quality. The consequences of diel stratification cycles on water quality were not clear except for the case of water transparency whereby frequent mixing at low water levels brought a lot of sediment into suspension. It was suggested that this sediment could be a factor contributing to high internal loads of phosphorus at the time. The effect of the reservoir is among other factors a function of its retention time and so these small dams with their short retention times are not likely to influence water quality significantly (Straskaba, 1999). Hence there were small differences between the water quality of the Upper dam and the Lower one

A third objective of this study was to look at phytoplankton and zooplankton communities in the dams and relate them to changes in the physical and chemical environment. Seasonal fluctuations in abundance were mainly attributed to the short water residence times leading to massive washouts of plankton during the peak of rainy season. Species composition remains largely unpredictable for the phytoplankton in small until we know more about them. Strangely, phytoplankton was weakly associated with environmental factors and apart from predation by zooplankton and juvenile fishes, hydrological factors were identified as likely determinants of the phytoplankton community. Physical factors such as temperature, water transparency, and suspended solids were identified as important in zooplankton dynamics. The role of suspended organic matter as food for zooplankton was suggested as a possible cause for the explosion in rotifers and the effect of suspended inorganic material on poorly selective feeders such as cladocerans explained their decline when water levels were low. The number of plankton species was comparable to other small dams and large dams.

The final objective of the study was to study fish communities in the two small dams. The conclusion reached was that the main determinant of species composition was the faunal composition in the river before it is impounded and this accounted for the fact that few species were recorded in these dams. There were no major differences in species composition and those observed could have been largely a result of random error during. The size structure of the fish communities did not show major changes that could be clearly be attributed to migration, recruitment or spawning events a major hindrance being that small size fish could not be adequately represented because of few samples from electro-fishing. Breeding activity was seasonal for *B. paludinosus* and largely commenced with the onset of major rains although some reduced activity was noted in both dams throughout the year. Changes in condition factor were minimal and there were no differences between the two dams hence the fish were not adversely affected by some of the extreme changes in the physical and chemical environment of the two small dams. High levels of suspended solids did not cause mortality in the fishes of the two dams although this has been observed in other systems. The short water residence time in the two dams could imply that small dams are more or less an extension of the main river and thus unlikely to have a significant impact on the biology and ecology of fish inhabiting them.

The main objectives of this study were satisfactorily attained considering the limitations in resources, except for the aspect on fish ecology where numerous sampling problems prevented the collection of a complete data set from which conclusions on patterns of migration, recruitment and spawning were supposed to be derived. Hence no firm conclusions could be reached about changes in size structure of the two main species populations found in these small dams.

Further research

Most recommendations for areas that could be further looked into have been stated in each of the chapter discussions. The limnology of small dams was never investigated in any detail until now and whilst there remain many gaps in our knowledge especially with regards to some of the specific mechanisms that drive change in small dams, this study was able to describe patterns and trends that occur in small dams from which hypothesis for further research can be formulated.

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