Observations on limnological conditions associated with a fish kill of *Oreochromis niloticus* in Lake Chivero following collapse of an algal bloom

Lindah Mhlanga[1,2]*, Jenny Day[2], Moses Chimbari[1], Nqobizitha Siziba[3] and Gertrud Cronberg[4]

[1]*University Lake Kariba Research Station, PO Box 48, Kariba, Zimbabwe,* [2]*Freshwater Research Unit, Department of Zoology, University of Cape Town, Rondebosch 7701, Western Cape, South Africa,* [3]*Lake Kariba Fisheries Research Institute, PO Box 75, Kariba, Zimbabwe and* [4]*University of Lund, Department of Ecology/Limnology, Ecology Building, S-223 62 Lund, Sweden*

Abstract

Possible causes of deaths of *Oreochromis niloticus* in Lake Chivero were examined in relation to changes in limnological conditions monitored over a 25-month period. The fish deaths coincided with the collapse of an algal bloom that had developed and builtup in the lake for 8 months. Chlorophyll *a* and dissolved oxygen increased to average concentrations of 42.4 μg l^{-1} and 10.9 mg l^{-1} respectively prior to the collapse of the bloom. Dissolved oxygen decreased when the bloom started to die off and coincided with the fish deaths when the average surface dissolved oxygen concentration in the lake was 3.9 mg l^{-1} and was at a depth of 5 m <2 mg l^{-1}. Mortality probably resulted from depressed oxygen levels caused by the high oxygen demand from the massive algal die-off and released algal toxins. This is the first time that die-off of algae has been linked to fish-kills in Lake Chivero as occurs in other hypereutrophic systems.

Key words: algal bloom, fish kill, *Oreochromis niloticus*

Résumé

Les possibles causes de la mortalité de l' *Oreochromis niloticus* dans le Lac Chivero furent examinées par rapport aux changements dans les conditions limnologiques contrôlées sur une période de 25 mois. La mort des poissons coïncida avec la chute d'une prolifération (bloom) des algues qui avait développé et accumulé dans le lac au cours de huit mois. Le taux de chlorophyll *a* et d'oxygène dissolu atteignit les concentrations moyennes de 42,4 μg l^{-1} et 10,9 mg l^{-1} respectivement avant la chute de la prolifération. L'oxygène dissolu diminua quand la prolifération commença à mourir et coïncida avec la mort des poissons quand la concentration moyenne d'oxygène dissolu à la surface du lac fut 3,9 mg l^{-1} et <2 mg l^{-1} à une profondeur de 5 m. En toute probabilité, la mortalité résultait de niveaux bas d'oxygène dus à la demande élevée par la mortalité massive des algues et les toxines algiques déchargées. Ceci est la première fois que la mortalité des algues a été liée à la mortalité de poissons dans le Lac Chivero, comme dans les systèmes hypereutrophes.

Introduction

Hypereutrophic systems characterized by mean phosphorus concentration of approximately 100 μg l^{-1}, and mean and maximum chlorophyll *a* levels of 25 and 75 μg l^{-1} respectively [Organization for Economic Cooperation and Development (OECD), 1982] are highly productive and tend to be ecologically unstable (Barica, 1981). They experience periods of rapid phytoplankton development followed by population crashes when production becomes unsustainably high (Barica, 1981). Periodic crashes of algal populations or huge oxygen demands from anaerobic hypolimnia during turnover can cause anoxia in hypereutrophic systems and this can result in fish mortalities (Robarts, 1985).

Lake Chivero in Zimbabwe is hypereutrophic (Magadza, 2003). Unchecked eutrophication occurs primarily as a result of discharge of nutrient-rich sewage and industrial effluents. Eutrophication first became evident in the form of algal blooms in 1960 (Munro, 1966) when the city of Harare began discharging insufficiently treated sewage into the lake (Marshall, 1997). During this period conductivity was 100 μS cm^{-1}, soluble reactive phosphorus 0.04 mg l^{-1} and ammonia 0.1 mg l^{-1} (Marshall, 2005). The lake became hypereutrophic in the mid-late 1960s (Magadza,

Correspondence: Email: wmhlanga@africaonline.co.zw

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2003). Conductivity then had increased to 170 μS cm^{-1}, soluble reactive phosphorus to 0.22 mg l^{-1} and ammonia to 0.4 mg l^{-1} (Marshall, 2005) and algal blooms became a permanent feature in the lake (Marshall, 1997). Algal blooms declined in response to reduction in nutrient loading between 1970 and 1980 when sewage effluent was diverted through irrigated pastureland. Conductivity decreased to 126 μS cm^{-1}, soluble reactive phosphorus to 0.04 mg l^{-1} and ammonia to 0.04 mg l^{-1} (Marshall, 2005).

An extensive outbreak of water hyacinth (*Eichhornia crassipes*) between 1985 and 1991 that covered about 35% of the lake's surface (Chikwenhere & Phiri, 1999) indicated a further deterioration in water quality (Marshall, 1997). The infestation by water hyacinth temporarily masked deterioration in water quality because the weeds removed nutrients from the water, as shown by the decrease in the concentrations of soluble reactive phosphorus to 0.03 mg l^{-1} and ammonia to 0.06 mg l^{-1} (Marshall, 2005). Destruction of weed mats using 2–4D in 1990 led to an increase in soluble reactive phosphorus to 0.1 mg l^{-1} and ammonia to 0.13 mg l^{-1} because nutrients were released into the water from decomposing plants. This resulted in the reappearance of continuous algal blooms, mostly of unpalatable and often toxic species (Rommens *et al.*, 2003).

Fish kills of *Oreochromis macrochir* have been reported in Lake Chivero from as early as 1971 (Burke & Thornton, 1982), when the lake became hypereutrophic (Moyo, 1997). Fish kills were caused by sudden turnovers that occurred between February and March, bringing deoxygenated bottom waters to the surface and causing anoxia. Moyo (1997) linked a fish kill of approximately 100 tonnes of *O. macrochir* in the lake in March/April 1996 to deoxygenation that was compounded by ammonia toxicity which was unacceptably high at 3.5 mg l^{-1}. Frequent fish-kills now characterize Lake Chivero, especially between November and April (Magadza, 2003) although the cause has not previously been established. Fish-kills occurred in the lake during the 1999–2002 period starting in January and ending in March (C. Phiri, pers. comm.).

The link between collapse of an algal bloom and fish kills has not been reported for Lake Chivero. During a 25-month study period between February 2003 and February 2005 a fish kill was observed during February and March 2005. This paper reports the limnological conditions leading up to the fish-kill and discusses possible causes in relation to the development of an algal bloom and changes in dissolved oxygen, ammonia, pH and temperature, which have previously been implicated in fish deaths in the lake.

Materials and methods

Study area

Lake Chivero (formerly Lake Mcllwaine) is a manmade lake that is situated on the Manyame River and is located 37 km to the southwest of Harare, the capital city of Zimbabwe at an altitude of 1363 m a.s.l. It is located at latitude 17°54′S and longitude 30°48′E (Fig. 1). It is a warm monomictic lake (Munro, 1966). At full supply level it has an area of 26.3 km^2, a volume of 250.4 $\times$ 10^{6}m^3 and a mean depth of 9.4 m. Table 1 lists the main hydrological and morphometric features of the lake (Burke & Thornton, 1982). The deepest part with a maximum depth of 27.4 m occurs near the lake spillway (Munro, 1966) but because of siltation this has changed and the maximum depth recorded during this study was 20 m. Station 1, located near the spillway, was the deepest while the maximum depth at station 2 was 3 m and at station 3 was 3.5 m.

When the lake was constructed in 1952 its main function was to provide water to the city of Harare but it now also supports a fishery and is used for recreational activities. Twenty-nine species of fish have been recorded in the lake but the most successful have been the cichlids, which are adapted to living in standing waters (Marshall, 2005). Two introduced species, *O. macrochir* and *Oreochromis niloticus*, dominate the fishery. *Oreochromis macrochir* accounted for 70% of the commercial catch until the 1990s (Marshall, 1978) while *O. niloticus* is now dominant.

Sewage and industrial effluent are discharged into the lake through the Mukuvisi and Marimba Rivers (Marshall, 1997). Mukuvisi River receives approximately 36 million litres of treated effluent daily (Mathuthu *et al.*, 1995) and contributes the highest nutrient load to the lake (Thornton, 1980). About 80% of the total inflow to the lake is from Manyame River. Abstraction for the city of Harare and downstream users take 60% while evaporation account for 30% of the outflow (Ballinger & Thornton, 1982).

Sampling procedures

Chlorophyll *a*, dissolved oxygen, ammonium, pH and temperature were monitored monthly at three stations (Fig. 1) between February 2003 and February 2005 except for January 2005. Water samples were collected using a Ruttner sampler from the following integrated depth intervals: 0–2, 2–4 and 4–6 m at station 1, 0– 2 and 2–3 m at station 2 and 0–2 and 2–3.5 m at station 3

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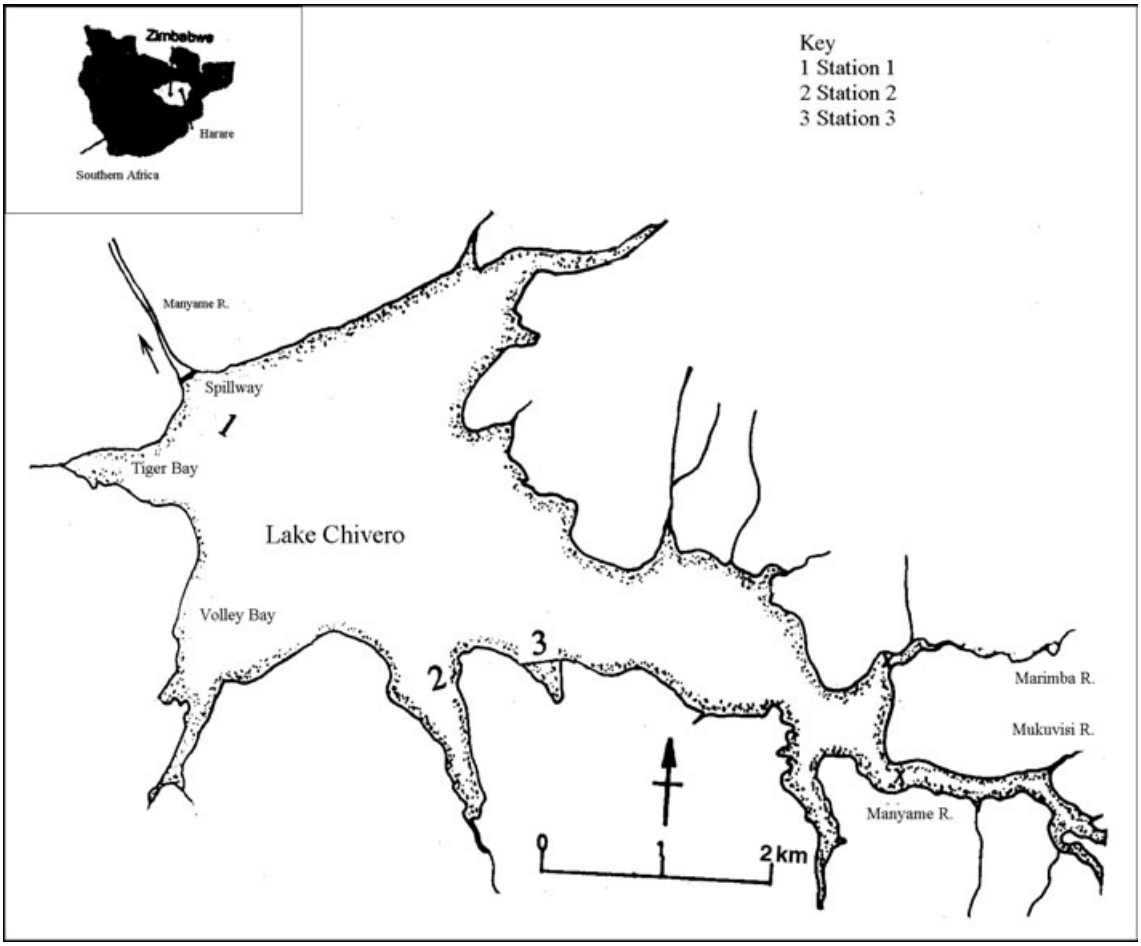


Fig 1 Location map of Lake Chivero showing the sampling stations (1–3)

Table 1 Hydrological and morphometric features of Lake Chivero

Characteristic	
Full supply volume ($\times 10^6$ m^3)	250
Full supply surface area (km^2)	26.30
Catchment area (km^2)	2227
Shoreline length (km)	74
Maximum depth (m)	27.43
Mean depth (m)	9.4
Maximum breadth (km)	8.0
Mean breadth (km)	1.68
Length (km)	15.7
Renewal time (years)	0.82

(Fig. 1). Following the onset of the bloom in May 2004 samples were taken down vertical profiles at 5-m depth intervals at station 1 to ascertain the vertical distribution of the same variables during the bloom period.

Temperature (°C), pH and dissolved oxygen (mg l^{-1}) were measured with a mercury thermometer, a WTW 330i pH meter (Geotech Enviromental Equipment, Inc., Denver, Colorado, USA) and a WTW Oxi 330 oxygen meter (Geotech Environmental Equipment, Inc., Denver, Colorado, USA) respectively. The concentration of ammonium (mg l^{-1}) was determined by the indo-phenol blue method (Golterman, Clymo & Ohnstad, 1978) and chlorophyll *a*(μg l^{-1}) by the acetone extraction method (Golterman *et al.*, 1978). Chlorophyll *a* concentration was used as an estimate of phytoplankton biomass.

Fish are reported to have started dying around the first week of February 2005. Fish that had just died or were gulping for air on the water surface were collected using a scoop net. A total of 55 fish were collected between 24th February and 1st March 2005. The total length, standard length and weight of each fish were measured and all

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specimens were identified. The condition factor was calculated in order to determine the health status of the fish. The condition factor (c.f.) was calculated using the equation $c = W \times 100/l^3$ (Pauly, 1984) where W = weight in grams (ungutted weight) and l = length in centimeters (standard length).

Results

Over the whole sampling period from February 2003 to February 2005, a fish-kill occurred only around February/March 2005 and members of only one species, *O. niloticus*, died. Many dead fish were observed along the shoreline during two lake-wide surveys undertaken on the 24th and 27th February. Fish dead or gulping for air on the surface were observed within the lake suggesting that the fish kill was lake-wide and not localized. Waves washed dead fish ashore. The mean total length of the fish was 25 ± 1.6 cm (range 21–33 cm). The mean standard length was 21 ± 1.44 cm (range 17–27 cm) and average weight was 282 ± 60 g (range 161–530 g) while the mean condition factor (c.f.) of the fish was 3.2 ± 0.48 (range 2.44–4.66).

An algal bloom was present in Lake Chivero from May until December 2004, during which time surface scums covered the whole lake. By February/March, when the fish-kill occurred, the bloom had crashed. Comparison of algal development in Lake Chivero from February 2003 to February 2005, using chlorophyll *a* as a measure of algal biomass (Fig. 2a), suggests a link between the development and subsequent collapse of the algal bloom and the fish-kill. A mild bloom developed in February 2003, with an average chlorophyll *a* concentration of 18.5 ± 7.8 μg l^{-1} (Fig. 2a). Algal cells were visible on the water surface but the bloom did not develop further and fish did not die. In February 2004 the chlorophyll *a* concentration was low (average concentration 7.5 ± 1.3 μg l^{-1}). A gradual build-up in algal biomass occurred from May 2004 (chlorophyll a = 7.1 ± 0.1 μg l^{-1}) when another bloom started developing and reached a maximum chlorophyll *a* concentration of 42.2 ± 2.4 μg l^{-1} in November. The bloom persisted in the lake until December. It must have crashed between January and February, probably because it had reached an unsustainable maximum level of productivity. When the fish started dying in February 2005, the average

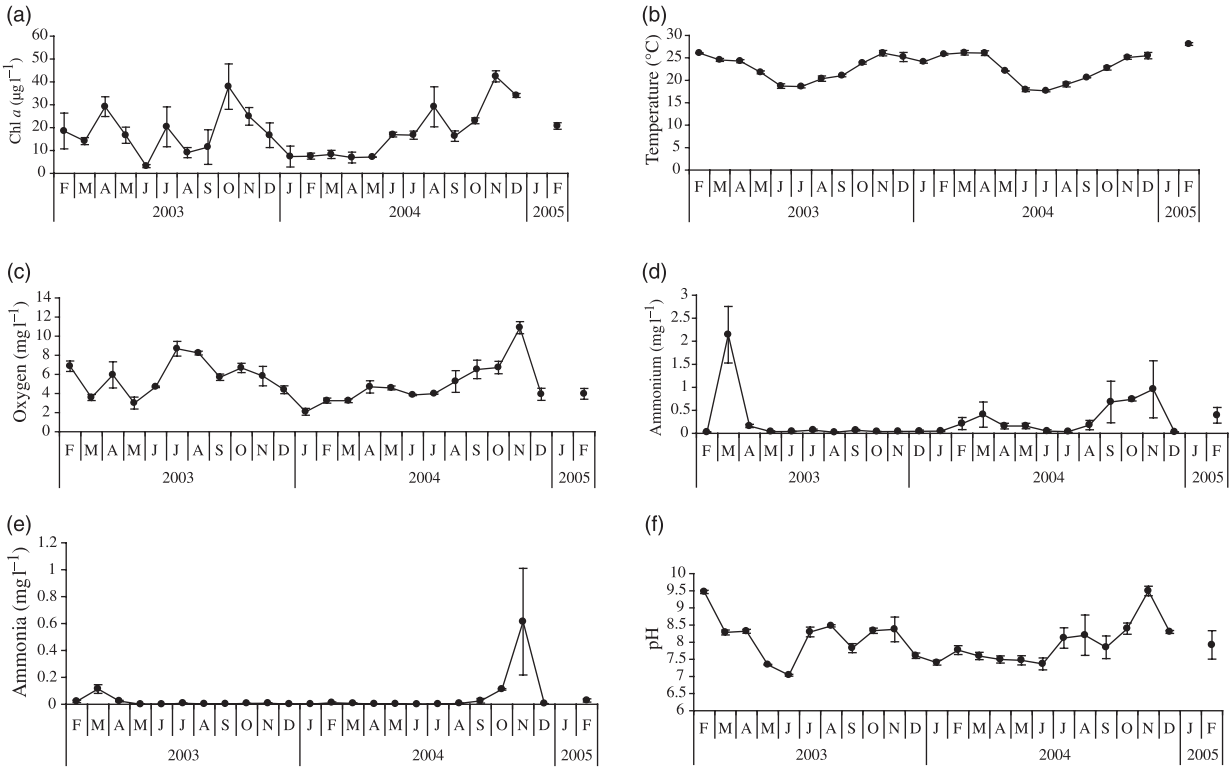


Fig 2 Temporal change in (a) chlorophyll *a*, (b) temperature, (c) dissolved oxygen, (d) ammonium, (e) ammonia and (f) pH (mean of three stations) in Lake Chivero from February 2003 to February 2005

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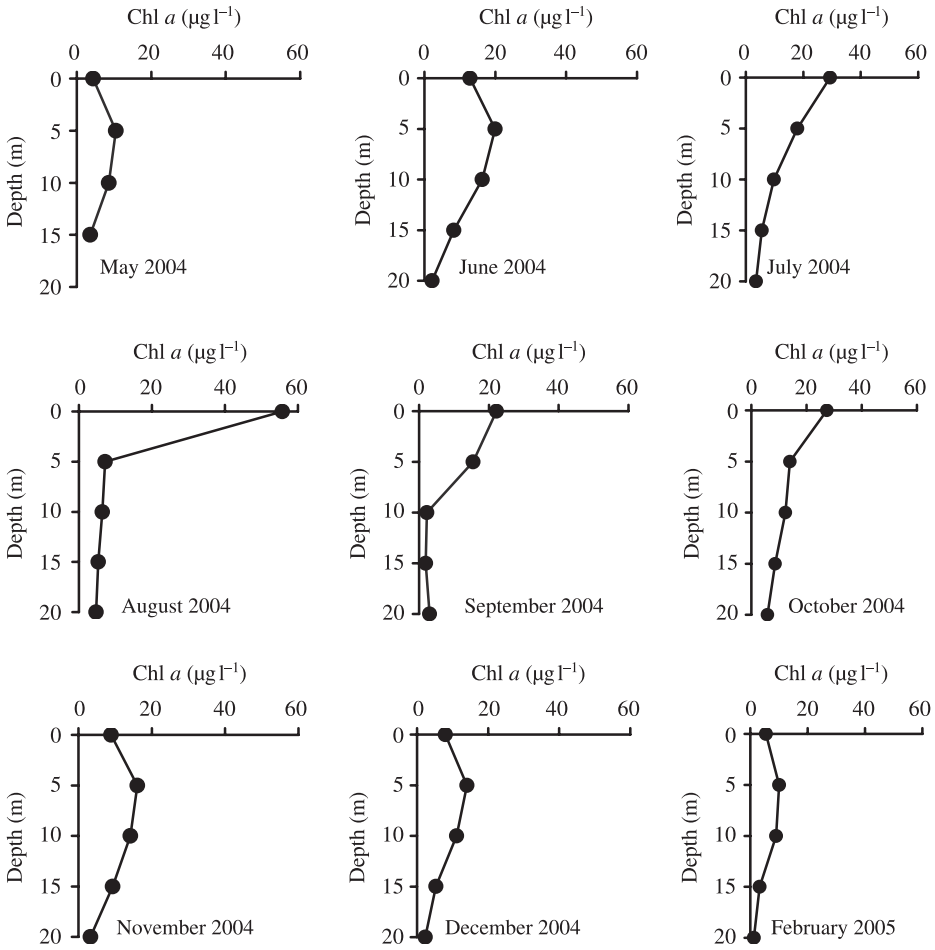


Fig 3 Depth profiles of chlorophyll *a* at station 1 before and during the fish kill

chlorophyll *a* concentration was 20.7 ± 1.4 μg l^{-1}. Depth profiles of chlorophyll *a* are shown in Fig. 3; concentrations decreased from surface to bottom and were highest between 0 and 10 m.

The period around February to March was when surface temperatures were highest in the lake (Fig. 2b). The average surface water temperature was lowest (22.1 ± 0.04°C) when the bloom started developing in May 2004, rising to a maximum of 28.1 ± 0.3°C in February 2005 (Fig. 3b). Temperature profiles during the period May 2004 to February 2005 are presented in Fig. 4. Surface water temperature varied between 17.4°C and 28°C and the bottom temperature between 16°C and 20.5°C. The lake was stratified during the fish kill in February, temperatures at the surface and the bottom being 28°C and 25°C respectively (Fig. 4). In June the lake was isothermal while for the rest of the time the lake was stratified (Fig. 4).

The temporal changes in the concentrations of dissolved oxygen between February 2003 and February 2005 are shown in Fig. 2c. During the period when there was no bloom, concentrations ranged between 1.8 and 9.5 mg l^{-1}. Dissolved oxygen positively correlated with chlorophyll *a* ($r = 0.51$) and pH ($r = 0.52$) (Fig. 2a,c,f). Following the onset of the bloom in May 2004, dissolved oxygen increased to a maximum concentration of 10.9 ±0.6 mg l^{-1} in November, after which it dropped to an average concentration of 3.9 ± 0.6 mg l^{-1} in February 2005 when the bloom had crashed, a concentration similar to that in December 2004, yet fish kills occurred only in February/March 2005. Figure 2c shows that the period around January to March (except for February 2003, when a mild bloom occurred) was characterized generally by the lowest dissolved oxygen concentrations in the lake during the study period. Ammonium and temperature

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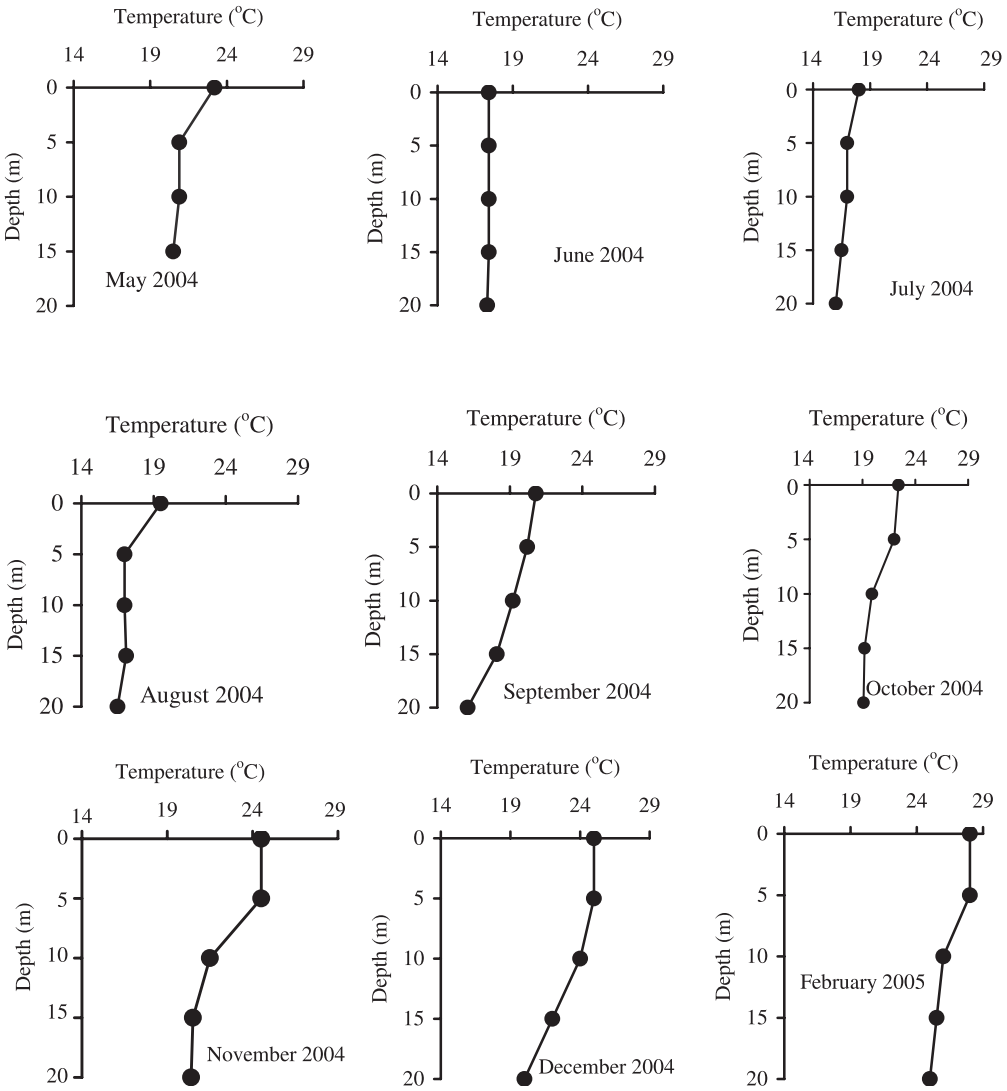


Fig 4 Depth profiles of temperature at station 1 before and during the fish kill

levels were also highest during this period (Fig. 2d,b). Changes in dissolved oxygen concentration down the water column from May 2004 to February 2005 are shown in Fig. 5. At the onset of the bloom in May the dissolved oxygen concentrations at the surface and bottom were 4.1 mg l^{-1} and 1.9 mg l^{-1}, respectively (Fig. 5). A slight decline in surface values occurred in June and July, after which the concentrations increased to a maximum of 9.3 mg l^{-1} in November. Concentrations at the bottom ranged from 1 to 3.1 mg l^{-1} prior to the collapse of the bloom, the maximum occurring in September. After the bloom collapsed, dissolved oxygen levels dropped to 4.2 mg l^{-1} at the surface and 0.6 mg l^{-1} at the bottom. From 5 m to 20 m dissolved oxygen levels ranged from 1.9 to 0.6 mg l^{-1}.

As ammonia toxicity has been implicated in fish kills in Lake Chivero, ammonium levels were monitored and ammonia calculated using a conversion factor (Dallas & Day, 2004). Temporal changes in ammonium and ammonia concentrations in the lake between February 2003 and February 2005 are shown in Fig. 2d,e respectively. Except for the peak in March 2003, concentrations were low (<0.5 mg l^{-1}) between February 2003 and January 2004. A slight increase occurred in March 2004, after which levels declined then increased again from August to November 2004 but not above the March 2003 concen-

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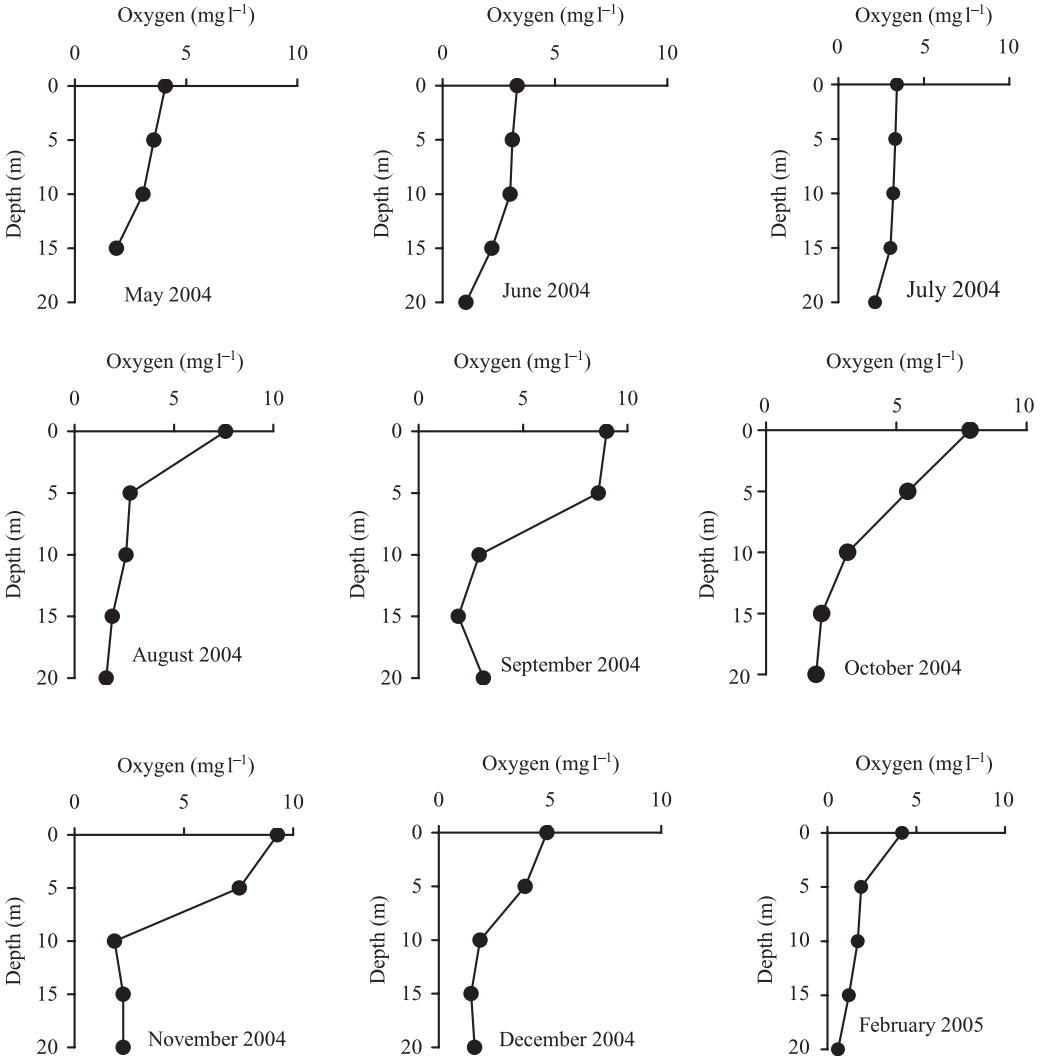


Fig 5 Depth profiles of dissolved oxygen at station 1 before and during the fish kill

trations. Changes in ammonium profiles during the development of the bloom are shown in Fig. 6. Except in May, concentrations were highest in bottom waters and increased as the bloom developed. Highest concentration occurred in November. The concentrations at the surface and the bottom in November were 1.1 and 4.4 mg l^{-1}, respectively, while the surface and bottom concentrations in February during the fish kill were 0.2 and 4.1 mg l^{-1}, respectively (Fig. 6).

Changes in pH between February 2003 and February 2005 are shown in Fig. 2f. Patterns differed during the bloom and no-bloom periods. As the bloom developed, pH increased to reach the highest average level of 9.5 ± 0.1 in November 2004 (Fig. 2f). Change in pH was positively correlated with chlorophyll *a* ($r = 0.52$) and dissolved oxygen ($r = 0.70$). The pH was highest at the surface and lowest at the bottom between May and February. In February, when the fish-kill occurred, the pH at the surface was 7.6 and at the bottom 6.6.

Discussion

The fish kill coincided with the collapse of an algal bloom suggesting that the dead algae were the likely causative factor. Decreased algal productivity and subsequent breakdown of algal material by aerobic bacteria after the

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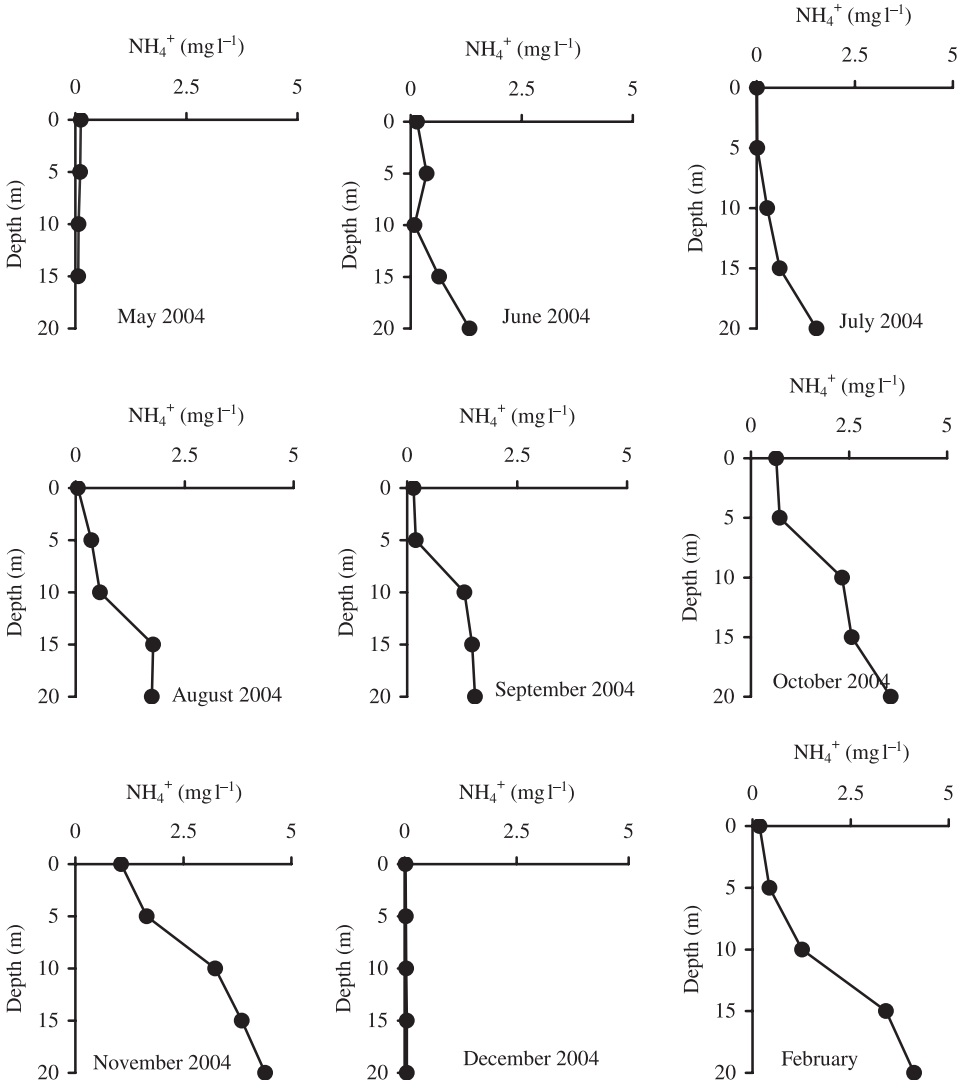


Fig 6 Depth profiles of ammonium at station 1 before and during the fish kill

bloom collapsed could have depressed dissolved oxygen to below 1.9 mg l^{-1} from 5 m depth, a level at which cichlid fish become sensitive (Magadza, 1997). Low dissolved oxygen concentrations probably contributed to fish mortality as observed in the Nyanza gulf of Lake Victoria, where fish kills of *O. niloticus* occurred when levels dropped to 3.2–4.8 mg l^{-1} following collapse of an algal bloom (Ochumba, 1990). Oxygen levels could have been as low as 0.4 mg l^{-1}, as in the *O. microchir* fish kill (Moyo, 1997) such that fish could have been affected mainly at night.

Interestingly, fish did not die in March and May 2003, between January and March 2004 and in December 2004 when dissolved oxygen concentrations were as low as during the fish kills. This indicates that factors other than low dissolved oxygen concentrations probably algal toxins released following the collapse of the algal bloom (Carmichael & Falconer, 1993; Chorus & Bartram, 1999), could have also contributed to fish deaths. Microcystins have been reported to be toxic to fish (Andersen *et al.*, 1993) and *Microcystis aeruginosa*, a common producer of microcystin (Falconer *et al.*, 1994) and dominant in Lake Chivero during blooms (Mitchell & Marshall, 1974; Robarts & Southall, 1977), could have been a source of toxins. Previously an enlargement of fish livers of dead *O. macrochir*

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had been attributed to possible algal toxicity (Moyo, 1997) such that the contributory role of algal toxicity cannot be excluded.

Lowest dissolved oxygen levels occurred around January – March, similar to the period of severe oxygen deficiency indicated by Magadza (1997). From information now available it is probable that high levels of dissolved oxygen occur as a result of high photosynthetic rates when algal biomasses are large. As the algae die off around February to March, dissolved oxygen declines in the lake, sometimes to a point at which fish-kills occur, as observed by Barica (1981) in other hypereutrophic lakes. In such systems oxygen regimes are often unbalanced, with periods of supersaturation followed by complete anoxia as large populations of phytoplankton collapse (Barica, 1981). The variations in oxygen concentrations from May 2004 to February 2005 illustrate this phenomenon and the variations in chlorophyll *a* link them to changes in phytoplankton biomass in the lake.

Previous fish kills in Lake Chivero have been attributed to deoxygenation of the water during turnover when oxygen deficient, ammonia-rich water is brought to the surface (Burke & Thornton, 1982; Moyo, 1997). Sudden overturns between February and March are the major cause of fish kills in Lake Chivero (Moyo, 1997). In February 2005, when the fish died, the lake was stratified so fish deaths are unlikely to have been because of turnover unless a sudden turnover was not detected.

Ammonia poisoning has been implicated in fish deaths in Lake Chivero (Magadza, 1997; Moyo, 1997) and massive mortality of tilapia is reported to occur when concentrations exceed 2 mg l^{-1} (Popma & Masser, 1999). The concentrations during this study were below the level toxic to fish. Low levels of ammonia in surface water also indicate that turnover had not occurred because ammonia would be expected to be high after turnover (Marshall, 1997).

The pH of the water has not been directly implicated in fish kills, its influence being mainly through shifting the equilibrium from nontoxic ammonium to un-ionized ammonia, which is toxic (Magadza, 1997). According to Dallas & Day (2004) at a pH of 9 and a temperature of 25°C, slightly more than a third of the ammonium ions in water are converted to toxic un-ionized ammonia. The pH during the fish kill did not exceed a range 5–10, within which tilapia can survive (Popma & Masser, 1999) and is unlikely to have contributed to fish deaths.

All previously recorded fish kills in Lake Chivero have been of *O. macrochir* (Burke & Thornton, 1982; Moyo, 1997). Another observation regarding fish-kills in Lake Chivero is the differential size mortality. During this kill, the only fish to be found dead ranged in standard length between 17 and 27 cm. The specimens of *O. macrochir* that died in 1996 ranged in length from 12 to 26 cm. It is not known why in this case *O. niloticus* of a particular size died, instead of *O. macrochir*. In trying to explain why *O. macrochir* was the only species to die during the massive 1996 fish-kill, Moyo (1997) suggested that it might have been the species most susceptible to deoxygenation and ammonia toxicity but it has yet to be established why the fish kills are species- and size- specific (Marshall, 2005) and why this particular genus is affected. The average condition factor for this size class was 3.2 and it ranged from 2.44 to 4.66. The fish were in good condition as the average condition factor was within the range reported for *O. niloticus* (Abubakar, 1988). This indicates that external factors might have caused fish mortality.

Although a link between fish kills of *O. niloticus* with the breakdown of an algal bloom is reported for the first time in Lake Chivero, elsewhere this phenomenon is well documented (Ochumba, 1985, 1987, 1990). This is an issue of concern which requires the institution of appropriate management measures because as the severity of algal blooms in the lake increases more kills may occur when the blooms crash.

Acknowledgements

This study was supported by a research grant received from Water Research Fund for Southern Africa and technical support from the University Lake Kariba Research Station.

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(Manuscript accepted 16 December 2005)

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