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How water exchange and seasonality affect the eutrophication of Murchison Bay, Lake Victoria

Suzan Luyiga[a], Sigrid Haande[b], Ronald P. Semyalo[a], Yusuf S. Kizito[a], Anne Miyingo-Kezimbira[a], Pål Brettum[b], Anne Lyche Solheim[b], Robinson Odong[a], Santa Maria Asio[c], Knut Helge Jensen[d], Petter Larsson[d,*]

[a] Department of Biological Science, CONAS, Makerere University, P.O. Box 7062, Kampala, Uganda
[b] Norwegian Institute for Water Research, Gaustadalléen 21, 0349 Oslo, Norway
[c] Department of Biological Sciences, Kyambogo University, P.O. Box 1, Kyambogo, Uganda
[d] Department of Biology, University of Bergen, P.O. Box 7800, 5020 Bergen, Norway

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ABSTRACT

Murchison Bay in the Northern part of Lake Victoria has for decades received a daily wastewater load of 0.2% of its volume from Kampala City, through the Nakivubo channel. In spite of this, the Water Treatment Works abstracts raw water from this bay and has been able to produce drinking water of sufficient quality for the capital. This study monitored various physical–chemical components within the bay during 2000–2003 to understand the processes responsible for the acceptable quality of raw water. Four sampling stations were located along a transect from the channel mouth towards the open lake.

Results: showed that the wastewater did not accumulate in the bay, instead was already strongly diluted 2.5 km from the channel mouth. This caused an abrupt reduction in conductivity and the concentrations of the nutrients total phosphorus (Tot-P), orthophosphate (PO_4-P) and total nitrogen (Tot-N).

Inshore–offshore exchange of water was mediated by flows from daily and sub-daily water level fluctuations and wind-driven currents. As a daily average, 2% of the Murchison Bay flowed in and out and the incoming wastewater was diluted 9.7 times.

During the dry season from June to August (D2), when the weather was influenced by the south-east monsoon, the thermal stratification in the main lake disappeared and cooler and deoxygenated water from deeper depths entered the bay influencing its water quality.

The daily flushing of water in and out of the bay due to water level variation was identified as the main factor diluting the bay water.

1. Introduction

The eutrophication of Lake Victoria is caused both by direct depositions of plant nutrients from the atmosphere and by run offs from point sources in the catchment area (Tamatamah et al., 2005). Murchison Bay in the North-Western part of Lake Victoria is one of the point sources receiving wastewater from Kampala City, the capital of Uganda with about 1.7 million inhabitants.

In spite of extremely polluted water entering Murchison Bay through the Nakivubo channel, the bay is less polluted than the Nyanza Gulf, also called Winam Gulf, on the Kenyan side of the lake (Sitoki et al., 2012; Gikuma-Njuru et al., 2013). This is visualized by the location of the city's waterworks 4 km south of the channel mouth. Although Cyanobacteria blooms all year round (Haande et al., 2011), and the amount of chemicals required to treat the water has increased considerably (Oyoo, 2008), the waterworks is still able to produce potable water.

The relatively good water quality in the bay was earlier expected to be an effect of the wetlands through which the wastewater was discharged. Previously, the Nakivubo channel ended in a papyrus swamp that retained a large proportion of the polluting compounds before the water reached the inner Murchison Bay (IMB) (Nalubega and Nakawunde, 1995). Today, however, the papyrus wetland has been converted to cocoyam fields, and the retention of nutrients and other pollutants is moderate to absent (Kansiime et al., 1994; Kansiime et al., 2005; Kansiime and van Bruggen, 2001; Kizito, 1986; Kyambadde et al., 2005). In addition the channel was widened in 2001–2003 to improve the city drainage. This

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* Corresponding author. Tel.: +47 95139226.
E-mail address: petter.larsson@uib.no (P. Larsson).

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has increased the potential loading of nutrients to the bay. However, since the waterworks are still able to extract raw water from the bay, some other contributing factors must determine the water quality in the bay.

Close to the waterwork's intake, is a land constriction causing a narrow water passage connecting the inner (IMB) and the outer Murchison bay (OMB) (Fig. 1). It could be expected that this narrow water passage could cause the pollution from the city to accumulate within the bay as in Nyanza Gulf (Sitoki et al., 2012). Since this seems not to be the case, the aim of this study has been to find and explain the processes responsible for the water quality.

One factor that could be significant is the daily water level fluctuations that take place. Earlier investigators of eutrophication in the bay have ignored this, but due to shallowness, even small water fluctuations could cause transport of considerable amounts of water in and out and dilute the wastewater. To study this idea in more detail, we monitored various physical and chemical components from the year 2000–2003 along a transect from the Nakivubo channel mouth to Makusu island in the outer bay.

Our study has been part of two larger projects studying the ecological conditions in the bay and the challenges Kampala City have in relation to water problems in general (Haande et al., 2011; Larsson et al., 2009; Semyalo et al., 2010).

2. Study area

Murchison Bay stretches about 30 km from the outskirts of Kampala to Entebbe. The narrowing that separates the IMB from the OMB is situated at Ggaba 4.8 km from the mouth of the Nakivubo channel (Fig. 1). The bay is shallow with a gently sloping bed to a depth of about 11 m at the Ggaba narrows and to about 12 m at our furthest sampling station. The IMB covers about 18 km^2, and its volume is about 6×10^7 m^3 with a mean depth of 3.2 m. The whole watershed for the IMB is 282 km^2 and the Nakivubo sub catchment represents only about 17% yet it contributes 76% of the effluent. This over representation is due to the fact that most of the water in the Nakivubo channel is direct runoff from the urban areas (Schröder et al., 1998).

The Nakivubo channel was widened to completion in 2002, from a width of about 5–20 m, spanning a length of about 10 km; from the Makerere hill slopes down to the floating Nakivubo wetland, to efficiently drain the city centre of floods. This has increased the concerns for the water quality in the bay (Birungi et al., 2007; Bomo et al., 2011; Kyambadde et al., 2006; Mugisha et al., 2007). In the present study, data are presented from both before and after the widening of the channel.

3. Methods

3.1. Sampling and analysis

Murchison Bay was monitored from November 2000 to December 2003 ($n = 52$). The data and samples were mostly collected every fortnight, but for various reasons there were two gaps (Nov–Dec 2001 and Jan–March 2003) in the sampling series. Sampling was conducted at four stations from IMB to OMB (Fig. 1, Table 1). St 1 was situated just outside the channel mouth to capture channel water before it mixed with the bay water, St 2 half way out of the inner bay, St 3 close to the Ggaba narrows, and St 4 much further out representing lake conditions.

All the stations were located with a GPS instrument (Garmin GPS12). Sampling took place between 10:00 and 15:00 h. Electrical conductivity, temperature and oxygen concentration were measured for every meter depth with a dissolved oxygen and conductivity instrument (YSI 85, Yellow Spring Instruments). Wind at the sampling stations was measured with an anemometer (Nerlien, Norway).

Water samples were collected with a 1 m tube column sampler of the Van Dorn type (Modified Ramberg Sampler) containing 5 L water. Samples were taken close to the surface and close to the bottom at all stations.

For chlorophyll-*a* (Chl-*a*) analyses water samples of 200–400 mL, were filtered through fibre glass filter (Whatmans GF/C) on board with a hand pump (<30 mm Hg). In 2001, the filters were put in 10 mL acetone, kept for 15–20 h dark in a dark refrigerator and thereafter centrifuged and analyzed photometrically both at the Makerere University and at the Norwegian

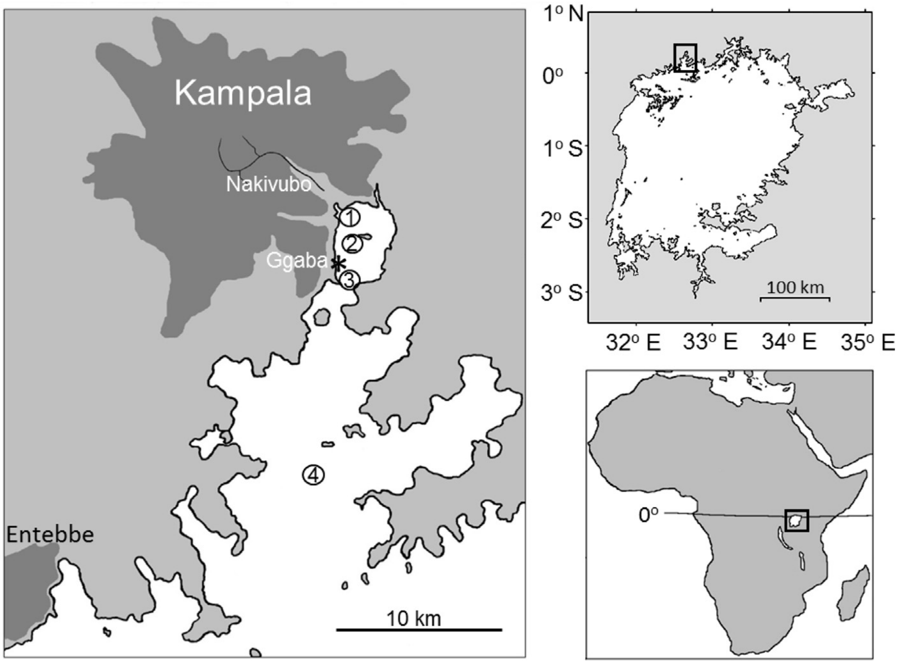

Fig. 1. Map of Murchison Bay with the sampling stations marked 1–4. The localization of the waterworks is marked with a star.

Table 1
Position of sampling stations in Murchison Bay.

Station No	Distance from St1 in km	Depth in m	Name	Co-ordinates
St 1	0	1.5	Mouth of Nakivubo channel	00°17.076′ N, 32°38.496′ E
St 2	2.5	5	South of Namalusu Island	00°15.727′ N, 32°38.749′ E
St 3	4.8	11	Ggaba Narrows	00°14.480′ N, 32°38.589′ E
St 4	15.6	12	South of Makusu island	00°08.715′ N, 32°37.580′ E

Institute for Water Research in Oslo, Norway (Haande et al., 2011). For bio-volume estimation and taxonomic determination of phytoplankton, 100 mL was preserved in Lugol's iodine solution on 8 separate occasions (2000–2001). Samples were analyzed under an inverted microscope with the Utermöhl technique (Brettum and Halvorsen, 2004; Haande et al., 2011; Utermöhl, 1958). Water samples for analysis of nutrients was collected unfiltered in 500 mL polyethylene bottles and preserved with 1 mL 5 M sulphuric acid until the analyses were performed in Norway. Nutrients were analyzed using Scalar autoanalyser according to Norwegian- and ISO-standard methods; total phosphorus (Tot-P, NS 4725), orthophosphate ($PO_4{}^{2-}$-P, NS 4724, filtered with membrane filter), total nitrogen (Tot-N, NS 4743). During the years 2000–2001, before the Nakivubo channel enlargement, the samples were taken once every two months. Samples were taken every fortnight in 2003, after the complete enlargement had terminated. There were only two dates with nutrient data in 2000 (November and December).

Hourly water level was recorded by the Directorate of Water Development in Entebbe for the period October 2002 to June 2003. Unfortunately, there were no hourly measurements available for the rest of sampling period. We did not measure the water level directly in the bay, but Schröder et al. (1998) found that the water level variation at Entebbe was almost identical to the water level fluctuations registered at the Ggaba Waterworks. Wind speed and direction were recorded by the Entebbe meteorological station at 15:00 daily. Data on precipitation from the Makerere hill in Kampala, provided by the Department of Geography, were taken as a representative for the watershed.

3.2. Calculations and statistics

The volume of daily water exchange between the bay and the lake was calculated as the average difference of water volume in the bay at daily maximum and minimum water level. For the input from the watershed, we used the average estimated by Schröder et al. (1998).

A non-linear least square regression model was used to describe the reduction of the conductivity and mixing of channel water with lake water in the transect from the inner to outer bay (Venables and Ripley, 2002):

$$Y = \beta_0 + \beta_1 \cdot 2^{-x/\theta} + \varepsilon \tag{1}$$

Here x is the distance in km from the mouth of the Nakivubo channel and y the estimated concentration at that distance. β_0 is the asymptotic value of the concentration ∞ km from the channel mouth and β_1 the total reduction from inner to outer bay. θ is the curvature parameter, i.e. the distance from 0 km to the point where the concentration is reduced to half, and ε represents the residuals of the model.

The daily in- and outflow would cause a mixing between lake water and the wastewater from the Nakivubo channel. Complete mixing was assumed at the intake for the waterworks (St 3) and should be reflected in the electrical conductivity there. If the mixing was according to the volumes from the two sources one could estimate the conductivity at St 3, C_{3e}, by the following formula:

$$C_{3e} = \frac{C_1 + D \cdot C_4}{1 + D} \tag{2}$$

where C_1 is the measured conductivity in the water from the Nakivubo channel (St 1), C_4 the offshore conductivity measured at St 4 and D the dilution factor estimated from the ratio between the volume exchanged with the main lake and the wastewater input. By doing so for, all the sampling events with hourly water level data, we could test whether the estimated water exchange and mixing actually took place. We tested the estimated against the measured conductivity at St 3 with a paired sample t-test. SPSS 15.0 and the statistics in Sigmaplot 11 were used to test for differences in the data. Differences between years and between seasons at the different stations were tested with both one- and two-way ANOVAs combined with Pairwise Multiple Comparison Procedures (Holm–Sidak method, Sigmaplot 11; (Glantz, 2011). The relationship between precipitation and conductivity at St 1 and St 2 and the relationship between wind and temperature differences of surface-bottom were described with a Pearson product–moment correlation. A paired sample t-test was used to test differences between surface and bottom samples. A Wilcoxon Rank Test was used to test the expected concentrations at St 3 versus the actual measured values. A Tukey's test (Zar, 1999) was used to identify the differences in Chl-a between years.

4. Results

4.1. Discharge from the watershed

There were two rainy seasons and two dry seasons through the year (Nicholson, 1996). The "long rain" (March–May, W1) normally peaks in April and the "short rain" (September–November, W2) in October, but there is a marked variation from year to year. During the study years (2000–2003), it rained every month and most in April (W1) and October (W2). The minimum was recorded in January–February (D1) and June–July (D2). The rain often fell as heavy showers causing very variable runoff into the IMB. During 2000–2003, the annual precipitation was 1086, 1490, 1631 and 1414 mm respectively. On average, this is 1405 mm or approximately the same as the total amount in 1997 (1398 mm) when Schröder et al. (1998) carried out their investigations. We therefore used their estimated average inflow to the bay from the watershed, 1.377 $m^3\,s^{-1}$, for our calculations. The estimate equals a daily input of about 1.190×10^5 m^3or about 0.2% of the IMB volume. The rain falling directly into the bay is on average 6.90×10^4 $m^3\,day^{-1}$. For 2008–2012, the average portable water abstracted was 8.5×10^4 $m^3\,day^{-1}$ (Oyoo, 2008), and the evaporation from the water surface was estimated by Schröder et al. (1998) to be 1752 mm y^{-1} or 8.64×10^4 $m^3\,day^{-1}$. Combining these factors, but ignoring the daily in- and outflow, gives a yearly positive outflow of 1.66×10^4 $m^3\,day^{-1}$. Remaining is a positive outflow of less than 1/10 of the sum volume of direct rainfall and the drainage from the catchment.

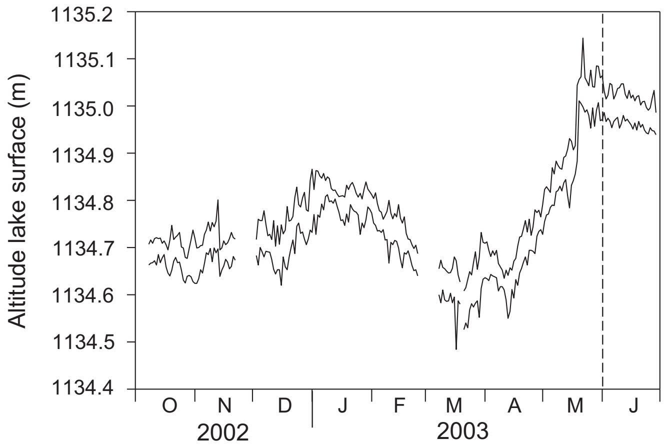

Fig. 2. Daily maximum and minimum water level from October 2002 through June 2003 measured as altitude (m). Vertical dashed line indicates beginning of dry season (D2).

4.2. Water level fluctuations

Between October 2002 and June 2003, the daily water level fluctuations ranged between 1.9 and 18.8 cm with an average of 6.4 cm. This average presents 1.15 × 10^6 m^3 of water or about 1.92% of the bay volume. Since the input from the watershed was only about 0.2% of the volume, then one unit volume of water from the Nakivubo channel will be mixed with about 9.7 units of lake water.

Fig. 2 shows the daily maximum and minimum water levels for the period we have data. During this period, the average water level raised by about 0.35 m with two maxima, one in December just after the rainy season (W2) and one in May when the rainy season (W1) was terminating. The difference between the daily maximum and minimum water level for the whole period is shown in Fig. 3. The largest variation occurred previous to the maxima in December and May. However, pronounced daily variations could also occur at other times of the year and they were frequent enough to eliminate any significant difference between the four seasons (Fig. 3).

In more detail, the hourly water level measurements at Entebbe is shown for a typical week in the wet season (March, W1) and dry season (June, D2) in Fig. 4a and b. The general picture depicted one main maximum and one main minimum every day. The maximum could appear at any time of the day, but most frequently between 10:00 and 12:00 h (21%) and between 00:00 and 02:00 h (20%). Superimposed onto the main oscillations were four to seven minor peaks and valleys. On average, the minor maxima and minima appeared in cycles of 3–5 h, and normally they were much smaller than the main oscillation. However, in the estimation of water exchange, only daily maximum and minimum were used.

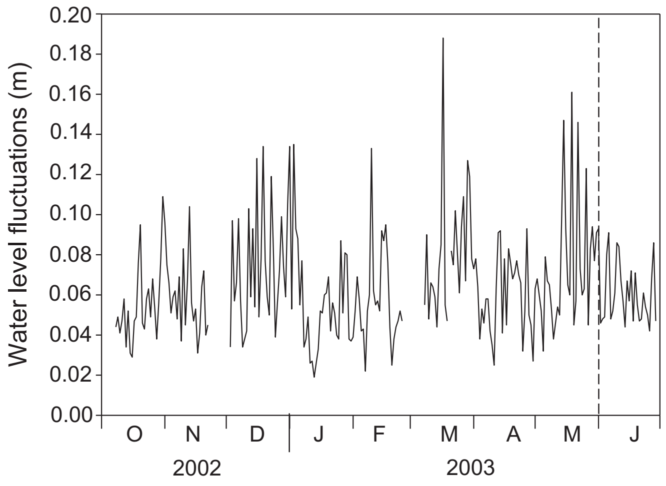

Fig. 3. Daily water level fluctuations from October 2002 to June 2003.

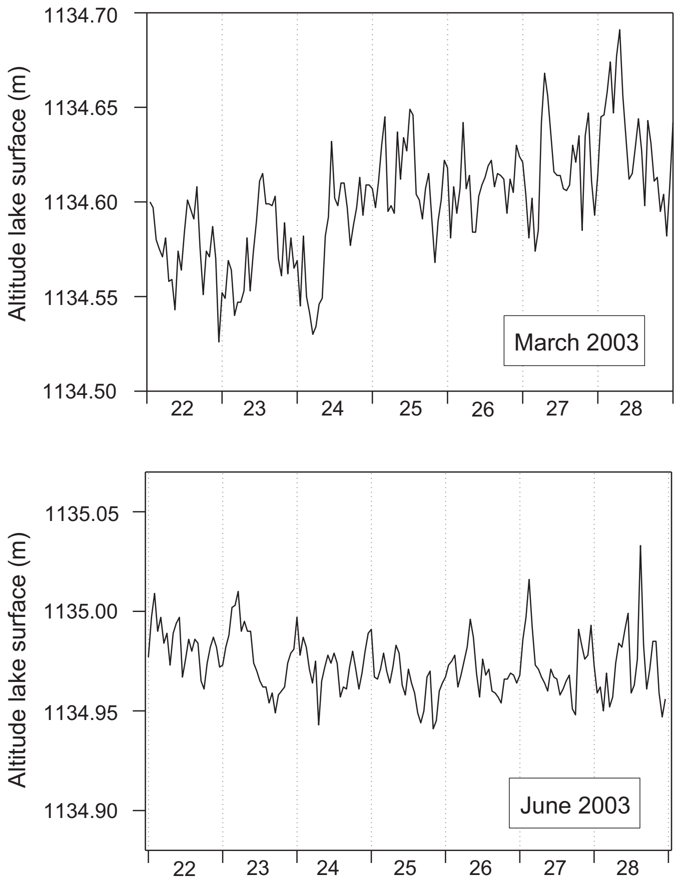

Fig. 4. Hourly measured water level at Entebbe during (a) one week in the wet season (March, W1) and (b) one week in the dry season (June, D2) 2003.

4.3. Wind and temperature

The wind had a diel rhythm with a weak northerly breeze blowing out of the bay in the morning, but at about noon it switches to a stronger southerly sea breeze. This pattern was found all year round with only a few exceptions. The highest wind speed we registered in the boat was 9 m s^{-1}. Stronger wind could occur at Entebbe, but the average wind speed at Entebbe at 15:00 was about 3.75 m s^{-1} (Fig. 5). During June–August (D2), the lake is strongly affected by the southeast monsoon (Niewolt, 1979) causing turnover in the lake (Talling, 1966).

Northerly wind blowing in the afternoon was registered only three times during the whole investigation period. Although water exchange in the bay was most likely affected by wind-driven currents, we were not able to quantify how much the local wind contributed to the water exchange in the bay. There was no correlation between the difference in sizes of the water level fluctuations and measured wind speed neither on-board the boat or at the station in Entebbe at 15:00; indicating that conditions at other times and localities, not covered by our sampling, contributed to the water level variation recognized in Murchison Bay.

A diel stratification and mixing within the bay caused a mid-day temperature differences between the water surface and bottom. During the calm morning hours, a thermocline usually built up and was most pronounced at noon. An afternoon sea breeze moved

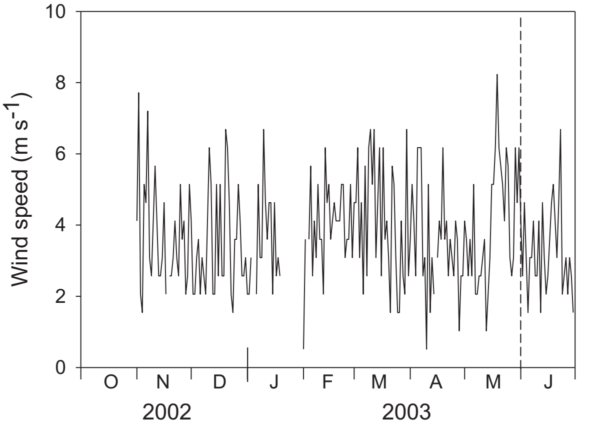

Fig. 5. Wind speed at Entebbe Meteorological station at 15:00 from October 2002 through June 2003.

the thermocline deeper and at midnight the whole water column was almost isothermal (Semyalo et al., 2009). The vertical stratification that built up during the morning resulted in temperature differences up to 3.5 °C between the surface and the bottom. However, most of the records were less than one degree (Figs. 6–8). The temperature difference between surface and bottom (at sampling time) showed a significant negative correlation with wind speed observed in the boat at St 4 ($n = 42$, $r = -0.465$, $p < 0.01$), but not at St 2 and 3. However, together with cooling, the afternoon breeze was usually strong enough to cause daily mixing of the water. The temperature was highest and the stratification most evident during the vernal and autumnal equinoxes (W1 and W2) (Figs. 6–9).

The mean temperature at St 2, 3 and 4 was very synchronized throughout the investigation period, while the temperature at St 1varied much more (23.5–28.0 °C) and has been excluded from Figs. 6–8.

4.4. Conductivity

The average conductivity is shown in Fig. 10. The conductivity in the surface layer was not significantly different from that at the bottom at St 3 ($p = 0.707$), and this indicated that the two types of water was mixed although there was a short mid-day stratification. Since we sampled at mid-day, with various degrees of thermal stratification, the average conductivity would compensate for possible differences between surface and near bottom recordings.

The nonlinear regression model (Formula (1)) described the development adequately and all three parameters (β_0, β_1, θ) were

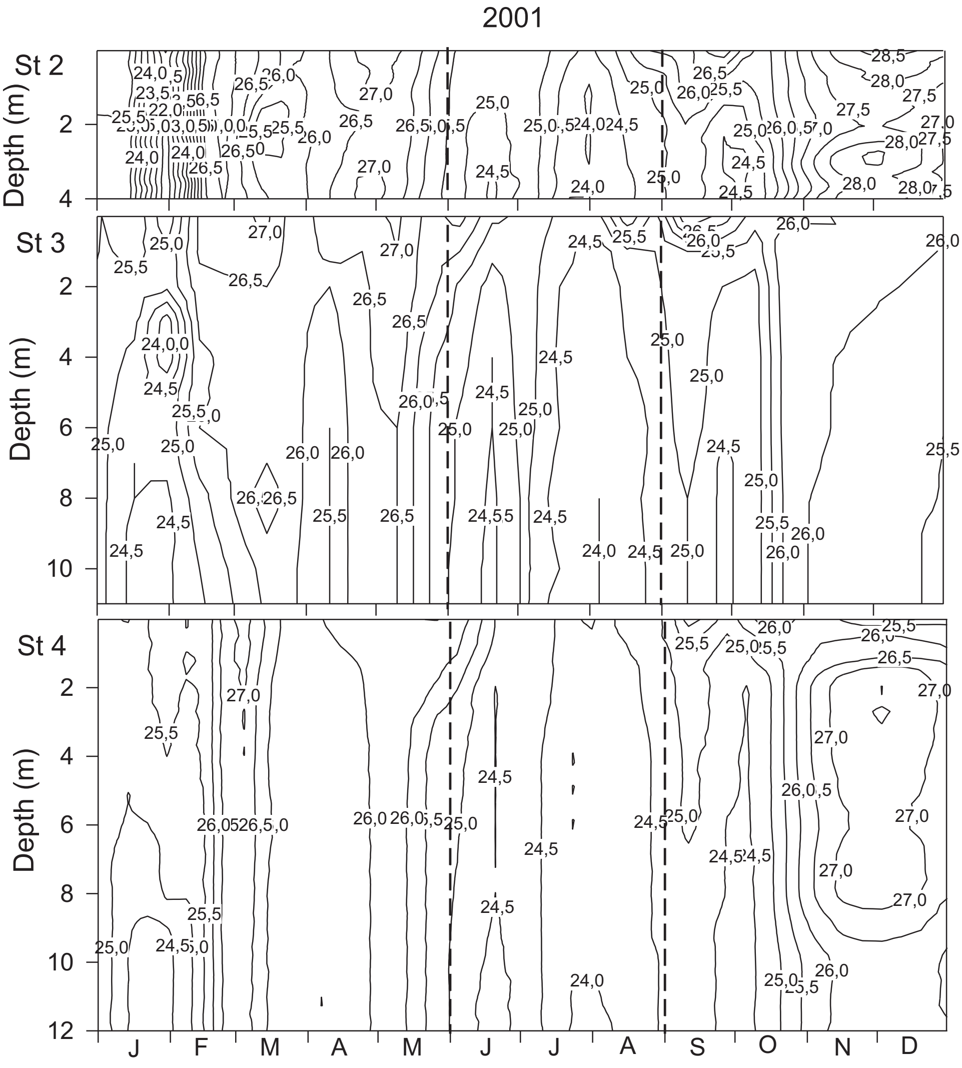

Fig. 6. Contour plot of water temperature in 2001 at St 2–4. Vertical dashed lines mark the dry season D2; the period influenced by the south-east monsoon.

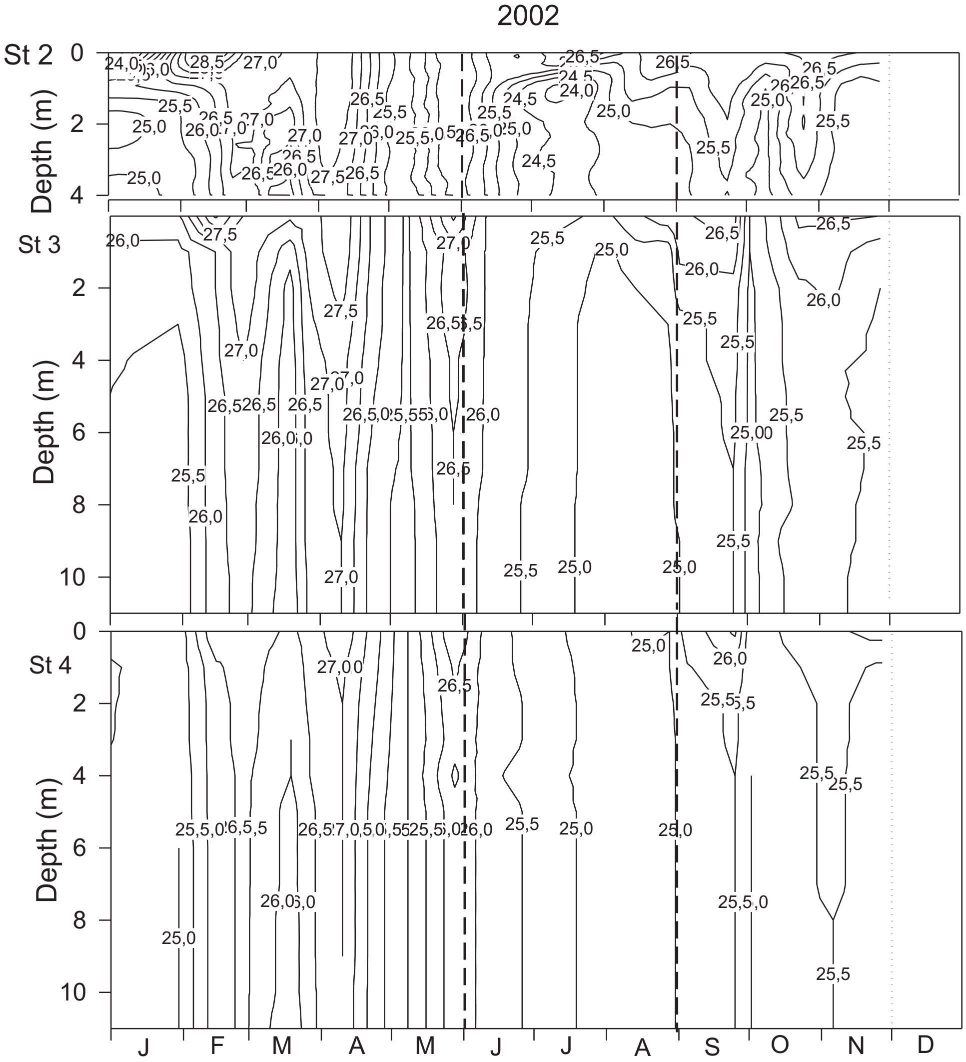

Fig. 7. Contour plot of water temperature in 2002 at St 2–4. Vertical dashed lines mark the dry season D2; the period influenced by the south-east monsoon.

highly significant ($p < 0.001$, df: 205; β_0: 98.39, $t = 31.12$; β_1: 64.65, $t = 13.08$; θ: 0.8642, $t = 3.49$, (Fig. 11). The main reduction in conductivity took place between St 1 and 2. Although there were only very small differences in conductivity between St 2, 3 and 4, the differences were statistically significant (Table 2). There was no significant difference between years at any of the stations (Table 3).

The conductivity at the channel mouth (St 1) varied considerably (Fig. 10), and this variation was correlated with precipitation.

Table 2
Differences between sampling stations tested with paired sample *t*-test. Conductivity and Chl-*a* were tested for 2001, 2002 and 2003. Two sampling days at the end of 2000 is added to the 2001 results. Concentrations of Tot P, PO_4-P and Tot N were analyzed only for 2001 and 2003. $p > 0.05$ NS, $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

Compound	Compared	Df	*t*	*p*	Sign. diff.
Conductivity	St 1 vs St 2	51	7.64	<0.001	***
	St 2 vs St 3	51	6.98	<0.001	***
	St 3 vs St 4	51	14.26	<0.001	***
Tot P	St 1 vs St 2	25	3.45	<0.005	**
	St 2 vs St 3	25	4.97	<0.001	***
	St 3 vs St 4	25	2.84	<0.01	**
PO_4-P	St 1 vs St 2	26	3.24	<0.005	**
	St 2 vs St 3	26	3.4	<0.005	**
	St 3 vs St 4	26	1.74	>0.05	NS
Tot N	St 1 vs St 2	26	2.91	<0.01	**
	St 2 vs St 3	26	1.84	>0.05	NS
	St 3 vs St 4	26	5.43	<0.001	***
Chl-*a*	St 1 vs St 2	41	0.46	>0.05	NS
	St 2 vs St 3	42	3.54	<0.001	***
	St 3 vs St 4	43	2.28	<0.05	*

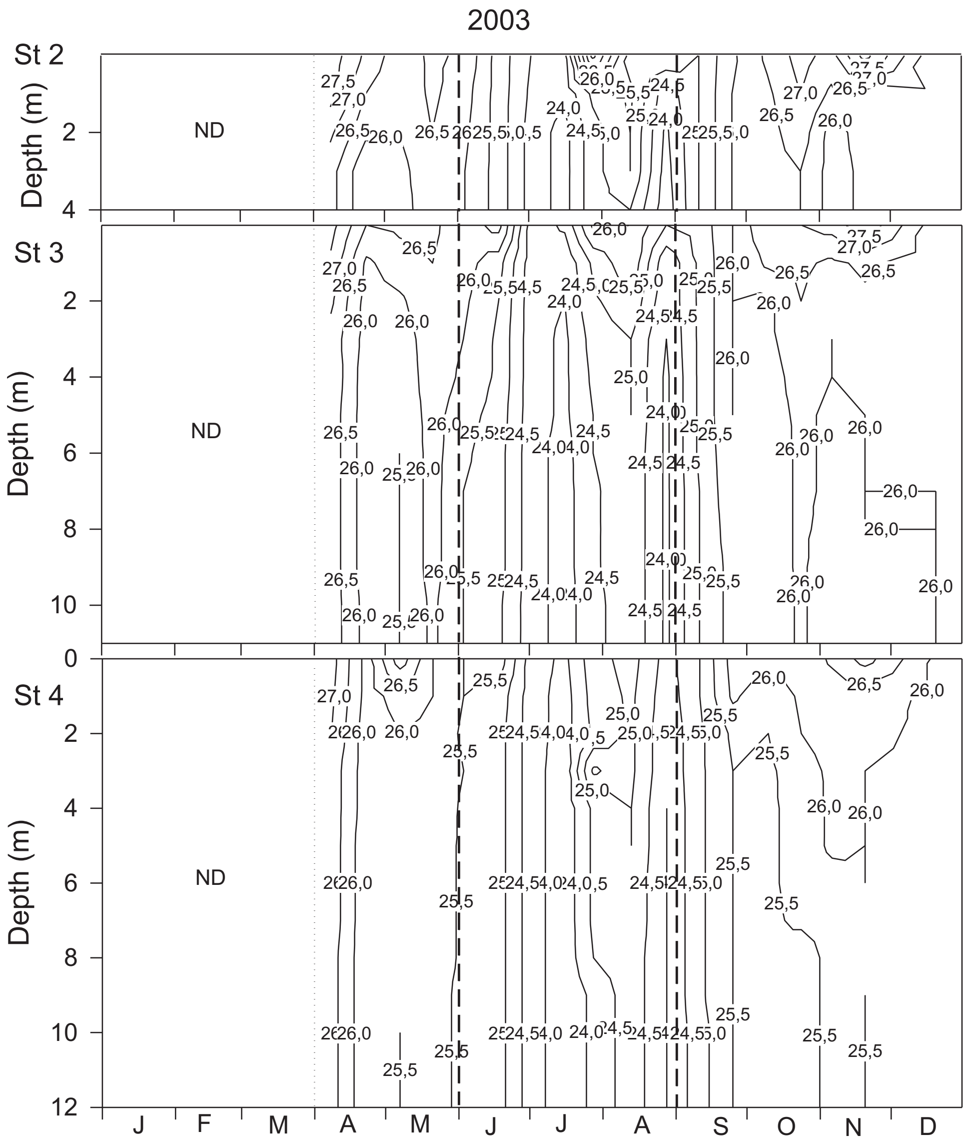

Fig. 8. Contour plot of water temperature in 2003 at St 2–4. Vertical dashed lines mark the dry season D2; the period influenced by the south-east monsoon. A period with no data is bounded with a dotted line and marked ND.

The best fit was the correlation with the accumulated rain over a 4 day prior to sampling ($n = 52$, $r = 0.49$, $p < 0.01$). Conductivity at the remaining stations was, however, almost unaffected by this variation. The expected conductivity at St 3 for all sampling dates was calculated from formula (2) using $D = 9.7$, and against the actual measurements, the Wilcoxon-Signed Rank Test did not find a significant difference between the two values (Table 4). To examine the robustness of the estimates above, we estimated the conductivity at St 3 (formula (2)) with various dilution factors ($D = 4$ up to 20). The development of p-value is shown in Fig. 12 and shows that the highest p was realized when the dilution factor was set about 8. This might indicate that the dilution factor was set somewhat too high in the first considerations, but since the 9.7 still falls within the $p > 0.05$ limits, the original estimate was used.

4.5. Nutrients

Also nutrients concentrations dropped dramatically between St 1 and St 2, and a small, but significant reduction from St 2 to the other stations was also shown (Table 2).

The nutrients concentrations at the surface were not significantly different from the concentrations close to the bottom (paired sample t-test, Tot-P: df = 61, $t = 1.27$, $p = 0.21$; PO_4-P: df = 80, $t = 0.62$, $p = 0.54$; Tot N: df = 68, $t = 1.52$, $p = 0.13$), and there was no significant difference between the years 2001 and 2003 for any of the compounds or stations (Table 3). The increase in nutrients concentrations at St 1 towards the end of in 2003 (Fig. 13a–c) had only a slight effect on the concentrations at St 2 and St 3.

The rate of reduction between the stations varied among the compounds. PO_4-P was reduced significantly more than conductivity, while Tot-P was not. The rate of reduction for Tot-N was on the other side significantly less than for conductivity (Table 4).

Most of the Tot-P in the channel water (St 1) was PO_4-P (72%), while at the other sampling stations PO_4-P contributed about 30%. Contrary to phosphate, the Tot-N was significantly higher than expected (Table 4).

4.6. Oxygen and lake turnover

The oxygen conditions in the bay are shown in Figs. 14–16. The trends in oxygen concentrations follow the patterns of

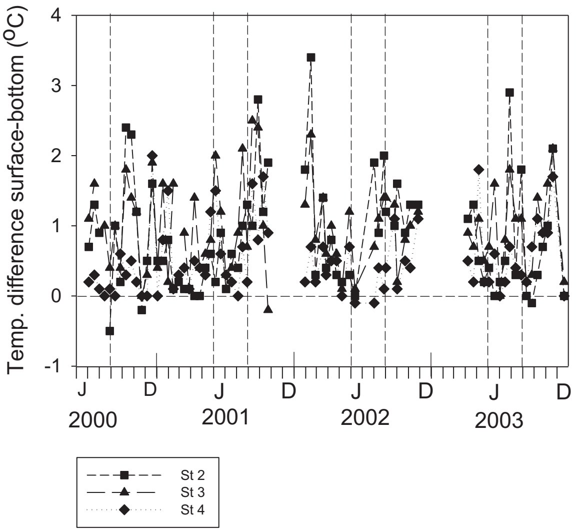

Fig. 9. The average temperature at St 2, 3 and 4 from June 2000 through December 2003. The vertical dashed lines show the dry season D2 when the lake is not stratified. The temperature at St 1 has been excluded from the graph since it was only 2 m deep and varied much more than the other stations.

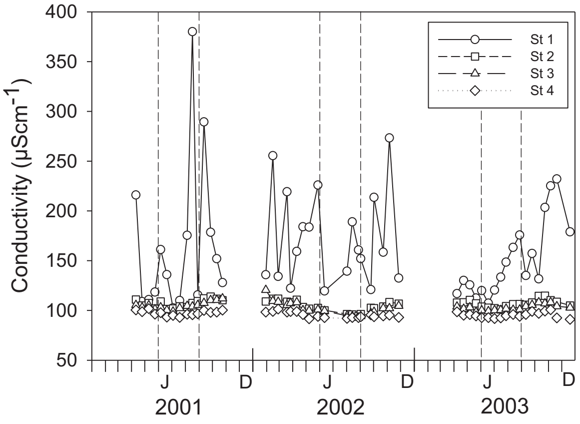

Fig. 10. The mean conductivity at all stations From April 2001 through December 2003. The vertical dashed lines show the dry season D2 when the lake is not stratified.

thermal stratification (Figs. 6–8). Oxygen concentrations were near uniform and ∼ 6 mg L^{-1} at St 4 during the D2 (June – July), due to greater wind-driven mixing and evaporation. Oxygen concentrations tended to decrease from offshore to inshore. Oxygen stratification occurred after the two dry seasons May (W1) and August, September and October (W2), when temperatures increased and vertical temperature differences were greater. During these periods, near surface concentrations were often above 9 mg L^{-1}, but the extent and location of hypoxia in the lower water column varied by year. For example, hypoxia developed in OMB but not in IMB in 2001 and 2003; the converse was true in 2002. Conditions in 2002 during the dry season (D2) were anomalous in comparison with the two other years. Oxygen remained stratified at St 4, while most of the water column was hypoxic at St 2 and 3 during June and July. Water temperatures during D2 were also highest during 2002 (Fig. 9). These altered conditions may have resulted from the anomalously low winds and higher relative humidity during D2 in 2002 (MacIntyre, 2012a) reducing the water exchange between the lake and the bay in this period.

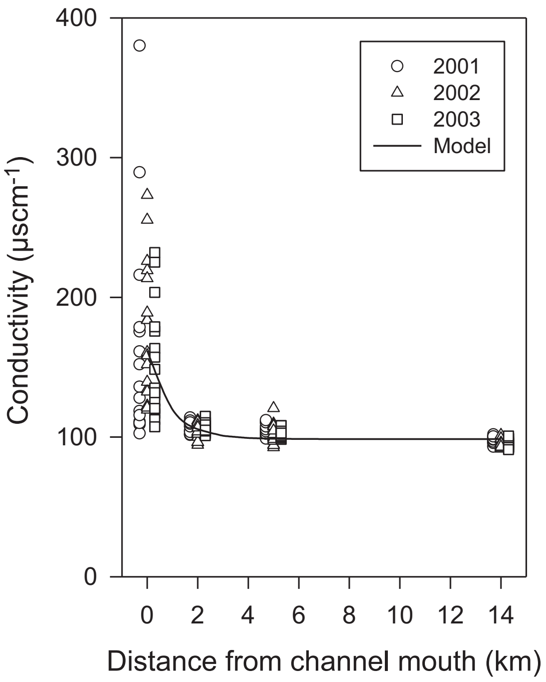

Fig. 11. A nonlinear regression model showing the reduction in conductivity from the inner to outer bay. Formular (1).

Table 3
Differences between years at the various stations are tested by ANOVA. Conductivity and chlorophyll-*a* (Chl-*a*) were tested for 2001, 2002 and 2003. Two sampling days at the end of 2000 was added to the 2001 results. Concentrations of Tot P, PO_4-P and Tot N were analyzed only 2001 and 2003.

Compound	St	df between groups	df within groups	*F*	*p*	Sign.diff.
Conductivity	1	2	49	0.76	0.48	NS
	2	2	49	2.98	0.06	NS
	3	2	49	1.41	0.25	NS
	4	2	49	3.02	0.06	NS
Tot P	1	1	25	1.95	0.18	NS
	2	1	25	0.21	0.65	NS
	3	1	25	0.4	0.53	NS
	4	1	24	0.09	0.77	NS
PO_4-P	1	1	26	2.11	0.16	NS
	2	1	25	1.59	0.22	NS
	3	1	26	1.01	0.32	NS
	4	1	25	0.01	0.98	NS
Tot N	1	1	26	3.27	0.08	NS
	2	1	25	1.87	0.18	NS
	3	1	26	0.6	0.44	NS
	4	1	25	1.18	0.29	NS
Chl-*a*	1	2	35	5.07	0.012	*
	2	2	41	7.05	0.002	**
	3	2	37	11.10	0.000	***
	4	2	36	7.70	0.002	**

Table 4
Comparison of expected and measured values of conductivity and concentrations of plant nutrients in Murchison Bay. Expected concentration at St 2 and 3 was calculated by mixing one part channel water (St 1, 0 km) with 9.7 parts lake water (St 4, 15.6 km). The means were compared in a paired sample *t*-test:$p > 0.05$ NS, $p < 0.05$ *. PO_4-P was more reduced than expected from only dilution and Tot-N was less reduced than expected from dilution.

Compound	Average expected St 3	Average measured St 3	*T*	*p*	sign	d.f.
Cond. µS cm^{-1}	102.3	103.2	0.92	0.360	NS	51
Tot. P µg P L^{-1}	73.72	64.67	1.71	0.100	NS	25
PO_4^- µg P L^{-1}	31.28	19.67	2.75	0.011	*	25
Tot. N µg N L^{-1}	868.6	997.0	2.11	0.045	*	27

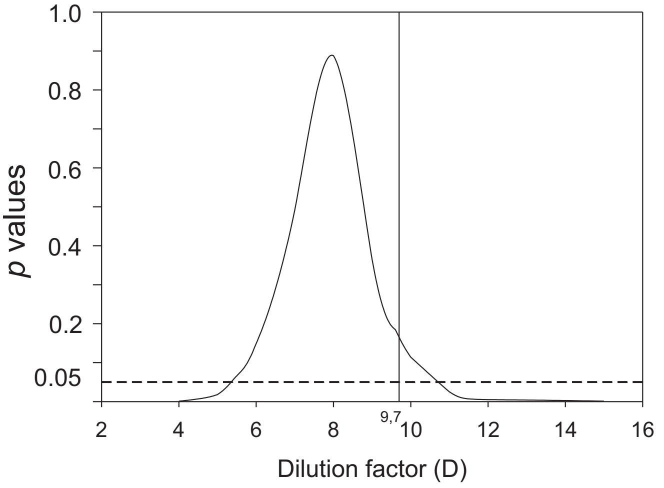

Fig. 12. The *p*-values from the test between expected and measured conductivity at St 3 with various dilution factors (*D*). The vertical line marks the *p*-value achieved when dilution factor was 9.7. This value was the dilution factor estimated from the water exchange taking place in the bay. The curved line shows the *p*-values achieved when various dilution factors (*D*) were used in the formula (2) for expected conductivity at St 3: $C_{e3} = (C_1 + D \bullet C_4)/(1 + D)$. The significance level of $p = 0.05$ is shown by a horizontal dashed line.

4.7. Chlorophyll-a and Secchi depth

We found no significant difference between the surface and the near bottom concentrations of Chl-*a*, indicating that the vertical mixing also affected the phytoplankton (paired sample *t*-test, df = 117, $t = 1.12$, $p = 0.27$).

Chl-*a* concentrations from the inner to the outer bay was much less reduced than the other compounds studied (Fig. 17). The difference between St 1 and St2 were not significant. Although the differences between the remaining stations were small, they were significant (Table 2).

The Chl-*a* concentrations at all stations were significantly higher in year 2001 (and 2000) than in 2002 and 2003 (Table 3). We therefore could not document any increase in Chl-*a* from before and after the completion of enlargement of the Nakivubo channel, rather a decrease was seen.

Seasonal variation was only found between D2 and D1 with the lowest concentration in D2 ($t = 3.040$, $p = 0.017$).

Secchi depth was negatively correlated with Chl-*a* concentrations ($r = 0.343$, $p < 0.001$) and was significantly deeper during D2 than at other times of the year (Fig. 18.). The Secchi depth was always higher in IMB than OMB.

4.8. Phytoplankton

In IMB, the phytoplankton assemblage was dominated by Cyanobacteria (Fig. 19), with an average of 81.0% of the total bio-volume. The second most abundant phytoplankton group was the Bacillariophyceae (diatoms) contributing 5.5%. In the outer bay (St 4), however, cyanobacteria contributed only 59% and the diatoms about 36% of the bio-volume (Fig. 19). The biovolume of the diatoms increased from IMB to OMB, while the cyanobacteria were more or less the same. This gave the highest total concentration of phytoplankton in OMB in contrast to the Chl-*a* that showed a weak decreasing trend. On average, about 35% of the cyanobacteria belonged to Nostocales group that are able to fix nitrogen. Although conductivity and nutrients showed a direct response to the dilution taking place in the bay, a much smaller reaction was found among the phytoplankton.

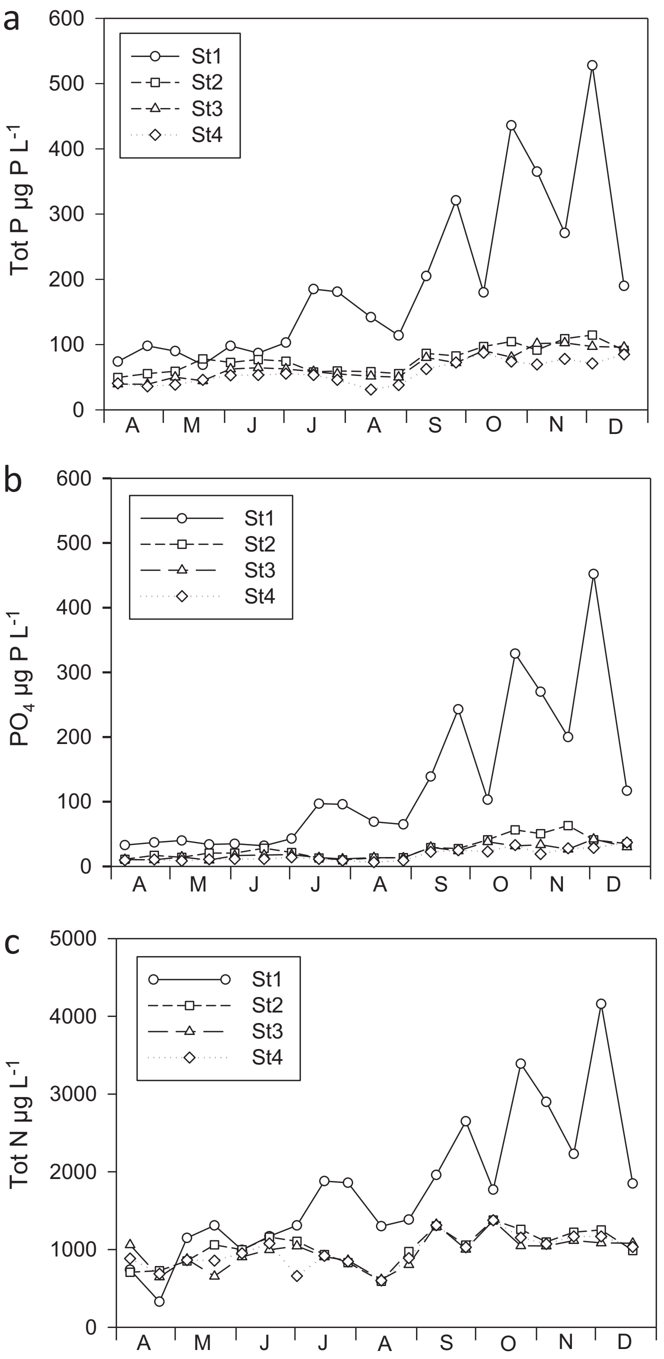

Fig. 13. Variation in concentrations of plant nutrients at all stations in 2003, (a) Tot – P, (b) PO_4–P and (c) Tot–N.

5. Discussion

There was a marked water exchange between IMB and OMB enhancing dilution of plant nutrients and other pollutants from the wastewater in Nakivubo channel. This exchange was due to a daily and sub-daily variability in water level and associated exchange flows that repeatedly flushed the bay. In spite of the narrows at Ggaba, the dilution was made possible by the existence of a gently sloping and sill-free bay bottom allowing water movement between IMB and OMB.

Although the daily water level fluctuations in Murchison Bay were in the same order of magnitude as found in other large lakes like Lake Okeechobee in Florida (Ji and Jin, 2006), and the Great

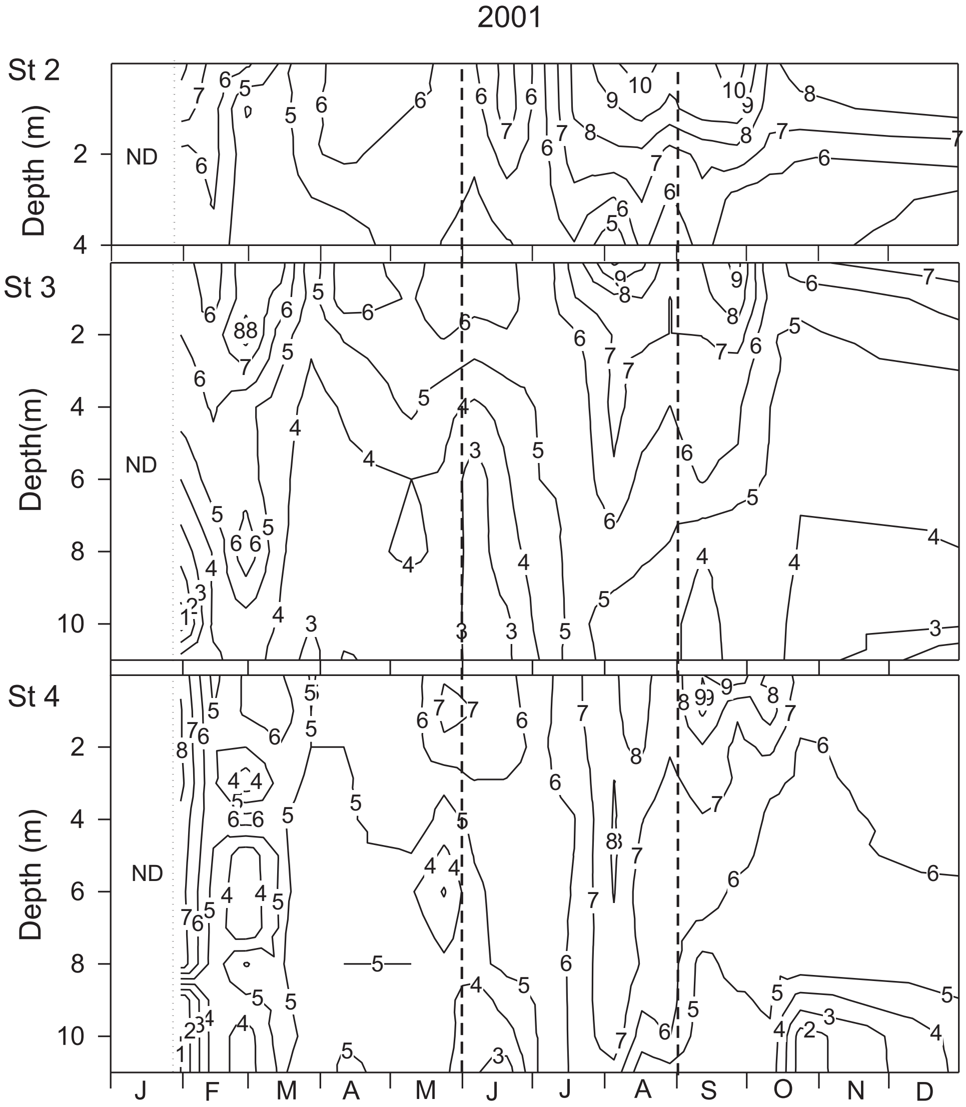

Fig. 14. Contour plot of oxygen concentrations at St 2–4 in 2001. Vertical dash lines mark the dry season D2 with south-east monsoon and the lake not stratified.

Lakes in North America (Trebitz, 2006), it has not earlier been used as factor affecting the water quality. Neither Schröder et al. (1998) nor Akurut et al. (2014) took this into account.

Since we found that the effect of the water level fluctuations were about 10 times larger than the input to the lake from the catchment, the water level fluctuations must have been the main cause for water exchange and the reason why the wastewater did not accumulate in the bay.

It seems likely that the short-time variation in water level is related to the very regular diel wind pattern over the lake (Fish, 1957; MacIntyre, 2012b). This also corresponds to the daily pattern in maximum water level, which most commonly occurred just before noon and/or around midnight. However, the daily maximum and minimum water level could occur also at other times of the day and we were not able to find any correlation between the water level variation and the wind speed measured neither on board our boat or measured at Entebbe (15:00 h). Therefore, the observed water level variations may be affected by additional factors such as surface seiches, tides, rainfall, runoffs from incoming rivers and outflow from the lake. These factors may sometimes work with and/or against the wind forces. Lake Victoria is so large that the lunar and solar tide is recognizable, and this is independent of the wind conditions. In the Great lakes of North America, the lunar and solar tide is found to be about 1–2 cm (Mortimer, 1974; Trebitz, 2006). If this is the range within Lake Victoria, then it is a recognizable part of the water level amplitude.

To estimate the amount of water moving in and out of Murchison Bay, we used the average of daily water level variation multiplied by the surface area of the inner bay to obtain the volume. To check whether this water exchange also led to water mixing, we used the decrease in electrical conductivity in the gradient from IMB to OMB. The conductivity was much higher in the channel water than in the lake and we consider its reduction from inner to outer bay to be a direct effect of dilution. Since we found no significant difference between the estimated and the direct measured values, we concluded that our dilution factor was close to being correct.

When using various dilution factors (D) in formula (2), we found that all values between 5 and 11 had $p > 0.05$ and the highest p-value appeared at about 8. There may be various reasons for this discrepancy and one important factor might be evaporation. In a recent study by MacIntyre et al. (2014) evaporation for various locations in Lake Victoria varied between 1.2 and 2.3 m^{-y}. The evaporation in IMB lays within this range, and the removal of water and not the compounds dissolved in it, could have increased conductivity and nutrients concentrations and reduced the dilution factor (D).

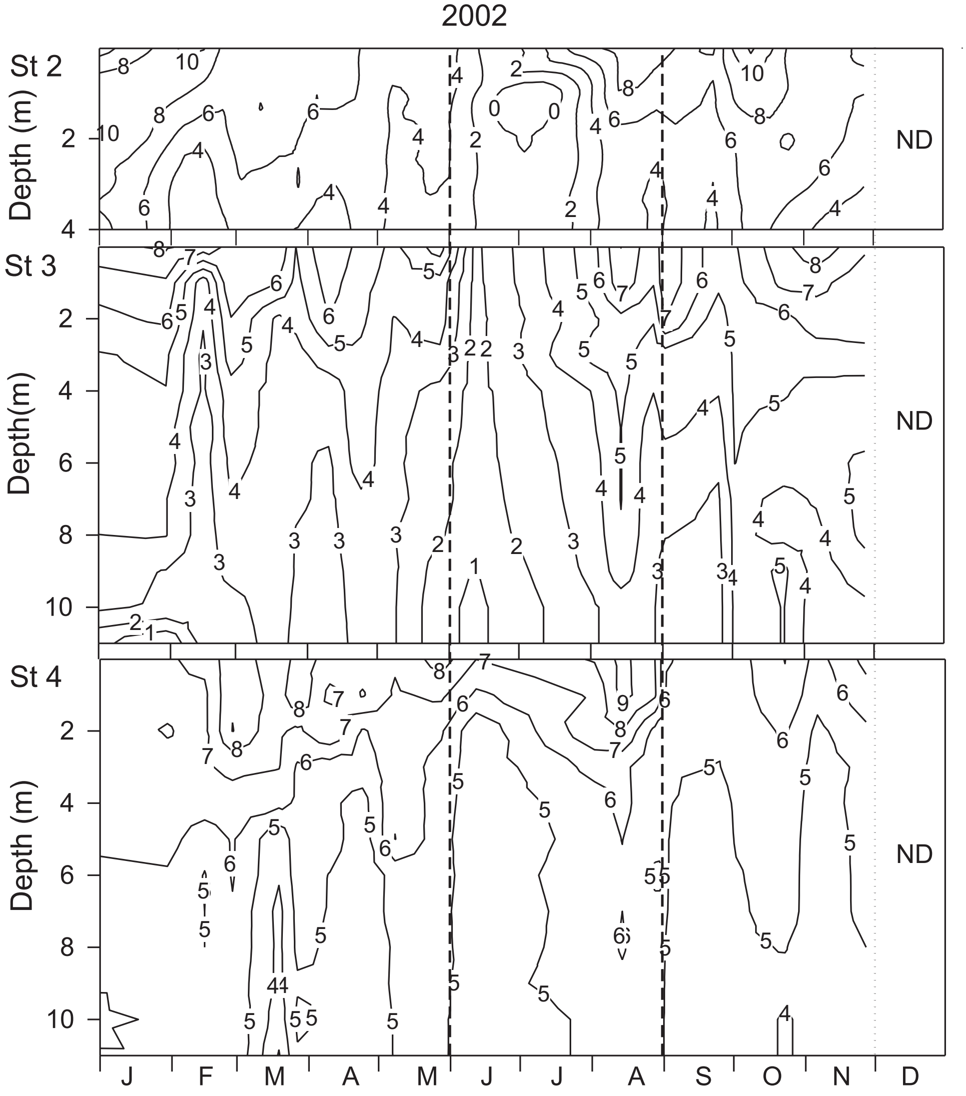

Fig. 15. Contour plot of oxygen concentrations at St 2–4 in 2002. Vertical dash lines mark the dry season D2 with south-east monsoon and unstratified lake.

Our results from Murchison Bay are in contrast to the findings in the Nyanza Gulf on the Kenyan side of Lake Victoria (Gikuma-Njuru et al., 2013). In their study, conductivity was used as a tracer for estimating the dilution of incoming municipal wastewater with lake water. They operated with a daily 10 cm water level fluctuation and that transported about 0.1% of the bays volume in and out of the bay daily. This was a larger water level variation than we had observed in Murchison Bay, but due to the larger volume in the Nyanza Gulf, the water exchange was less. The mixing did not take place within the Nyanza Gulf, but in a steep gradient at the outlet of the gulf. The river inflows and municipal sources were largely retained in the gulf and only a small fraction was transferred into the main lake. The difference between the two bays is most likely an effect of the morphology where the in and out flow are much more restricted for Nyanza Gulf than Murchison Bay. For other bays in Lake Victoria, the water exchange might be very different and therefore create other water qualities. We had assumed that the biological activity in the Murchison Bay should have affected the reduction in nutrients concentrations. However, that was only the case for PO_4–P. It was the dominating phosphorous compound in the channel water but at St 3 the measured concentrations were significantly less than expected from formula (2). Tot-P, however, showed a development similar to conductivity and the expected and measured concentrations at St 3 were not significantly different. Most likely, the PO_4-P in the channel water was taken up by the phytoplankton and converted to organic phosphorus.

The measured concentrations of Tot-N at St 3 tended to be slightly ($p = 0.045$) higher than expected from dilution only. This might be due to nitrogen fixation in the bay. About 35% of the cyanobacteria in the inner bay were nitrogen fixers (Haande et al., 2011), and Mugidde et al. (2003) found that nitrogen fixation in the northern inshore area of Lake Victoria had an annual average of 14.0 $\text{g N m}^{-2}\text{y}^{-1}$, implying an approximate rate of 7.99 g N s^{-1}. In the Nakivubo channel, 1500 $\mu\text{g L}^{-1}$ is a representative concentration and with an average mean water inflow of 1.377 m^3s^{-1}, this results in an input of about 2.07 g Ns^{-1}. Thus, with the same rate of nitrogen fixation as the Napoleon Gulf, it would be expected that Tot-N increased instead of being diluted. However, there are two counteracting processes; firstly, nitrogen fixation may be less than in the Napoleon Gulf because of a pre-existing surplus of nitrogen in the water, and secondly, nitrate and ammonium might have been taken up by macrophytes or released as nitrogen gas by denitrification.

Higher Chl-*a* concentrations were recorded in the IMB (St1 and 2) than in OMB. Since nutrient concentrations were high both within and outside the bay they would not be limiting (Haande

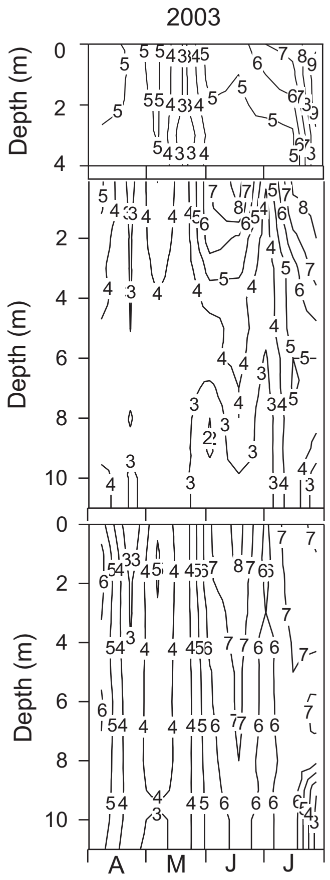

Fig. 16. Contour plot of oxygen concentrations at St 2–4 in 2003 with south-east monsoon and the lake not startified.

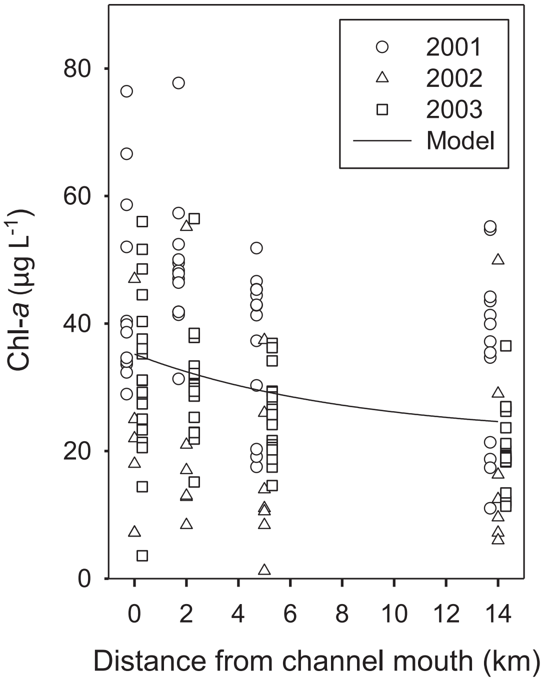

Fig. 17. Average concentrations of Chl-*a* at the various sampling stations in 2000–2003. Each data point represents the average between the concentrations measured at the surface and close to the bottom at each sampling station and date.The line is the estimated nonlinear regression model based on data for all the three years: $y = 22.43 + 12.64 \bullet 2^{-x/6.27\circ}$, where x is distance in km from channel mouth and y the estimated concentration.

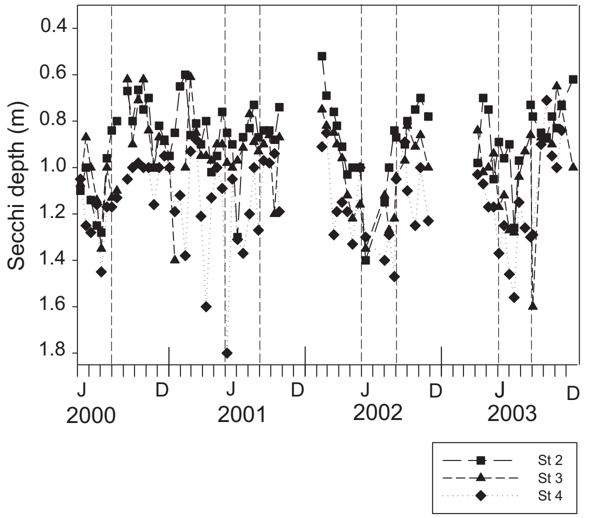

Fig. 18. The Secchi depth at the various sampling stations June 2000 through December 2003. Dashed vertical lines mark dry season D2.

et al., 2011; Hecky et al., 2010). The decrease in Chl-*a* was therefore most likely due to a deeper mixing depth. The Chl-*a* concentration in 2000–2003 was of the same order of magnitude as found by Schröder et al. (1998), but less than the levels recorded by Silsbe et al. (2006). We had expected an increase in Chl-*a*, particularly in 2003 after the widening of the Nakivubo channel. However, this was not the case, as both the median value from Schröder et al. (1998) and our measurements in 2001 and 2002 were higher than measured in 2003. In 2001, before the widening of the Nakivubo channel, Silsbe et al. (2006) recorded, some Chl-*a* values higher than ours. This underlines that the widened wastewater channel affected the conditions in the bay only to a minor degree. However, data from sampling and analyses conducted after 2003 indicate that there has been an increase in nutrient concentration in the bay (Oyoo, 2008). This is also emphasized by the increasing running costs by required by the water works since the widening of the channel (National Water and Sewerage Corporation, 2011).

The phytoplankton composition changed from IMB to OMB (Haande et al., 2011), with a total dominance of cyanobacteria in the IMB. In OMB, however, the Bacillariophyceae (diatoms) became more abundant with approximately the same biovolume

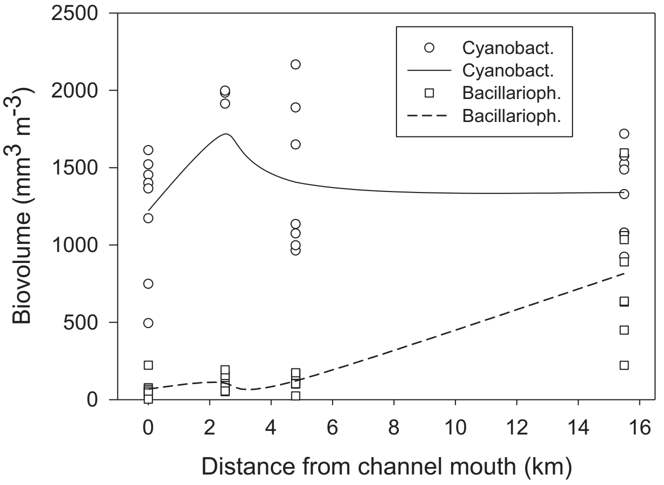

Fig. 19. Phytoplankton bio-volume of the two most common groups Cyanobacteria and Bacillariophycea in 2001. The solid line is a smoothed curve through the averages at each station.

as Cyanobacteria. This caused no decrease in biomass from IMB to OMB as we found for Chl-*a*.

The low number of samples from 2000 to 2001 was not sufficient to reveal any clear seasonal changes in the relative proportion of cyanobacteria and diatoms, but Haande et al. (2011) found that the relative proportion of diatoms was highest during the rainy seasons in 2003.

In Lake Victoria, at an altitude of 1135 m and a temperature about 23–27 °C, there is an equilibrium between air and water when the water contains about 7.3–6.7 mg O_2 L^{-1} (Wetzel, 1983). We found oxygen concentrations generally below saturation in Murchinson Bay indicating an overall poor water quality. The increased mixing during the southeast monsoon (D2) is expected to reoxygenate the water column. In 2001 and 2003, the concentrations reached 6 mg L^{-1} in much of the water column, and yet concentrations were less than 3 mg L^{-1} at the inshore and the offshore waters were stratified during the same period in 2002. Since the low oxygen co-occurred with the unstable period in the main lake, upwelling of deep offshore water could have caused this. However, this seems unlikely as wind direction was from the South (MacIntyre, 2012b) and with anomalously low speed and also because relative humidity was high during the southeast monsoon in 2002. The decreased oxygen concentrations inshore may result because flushing was reduced in that year. While our conductivity measurements indicated that dilution was similar all 3 years, the persistent low oxygen concentrations in 2002 suggest otherwise. Reduced winds are typical during the southeast monsoon under the El Nino Southern Oscillation (ENSO) conditions (MacIntyre et al., 2014). Consequently, more water treatment at the Ggaba treatment plant may be required during those conditions.

6. Conclusion

The water quality in IMB is, in spite of the very polluted catchment-derived wastewater, mainly a result of the water exchange between the main lake and the bay mediated by flows driven by short-term water level fluctuations.

The pollution from the city seems to be only slightly broken down in the IMB through biological activity. Most of the pollutants do not accumulate in the bay, but are transported more or less entirely out into the main lake contributing directly to the eutrophication of Lake Victoria.

The seasonal variation in temperature and wind cause marked differences in water level variation, thermic stratifications, water exchange and eutrophication. This calls for more detailed studies on the consequences of both short and long-term water level variation in Lake Victoria.

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