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Long-term variations of water quality in the Inner Murchison Bay, Lake Victoria

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Abstract The water quality in the Inner Murchison Bay (IMB) located in Uganda on the northern shores of Lake Victoria is affected by a complex mixture of processes and driving factors including pollution, river inflows, lake water levels, wetland management and flora and fauna populations. This study attempts to explain long-term variations of the IMB water quality and to provide a plausible water quality model. Because intermittent monitoring around the Bay hinders accurate determination of pollution, concentrations at the extreme northern shores (hotspots) are considered indicative of the pollutant loading into the bay. Delft3D-Flow was applied to study the Bay hydrodynamics and coupled with the Delwaq module to investigate water quality processes related to oxygen: organic and nutrient components i.e. dissolved oxygen (DO), biological oxygen demand (BOD) and ammonium (NH_4^+). It is found that the IMB water quality deteriorated exponentially in the period 2001–2014 due to increased pollution and the high residence time of water. The worst water quality was in 2010 when diffuse pollution intensified due to the lining of more drainage channels within Kampala City in addition to the declining *wetland effect*. The water quality towards the Outer Murchison Bay (OMB) deteriorated over time with dilution accounting for 40–60% of pollutant reduction. Although the effect of lake level variations is negligible compared to pollution into the IMB, increased lake levels after 2011 improved DO levels and mixing and hence BOD levels in the IMB.

Keywords Inner Murchison Bay · Lake Victoria · Pollution · Water quality

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Introduction

Lake Victoria, Africa's largest freshwater lake, covers a surface area of about 68,800 km^2 shared across three east African countries: Uganda (45%), Kenya (6%) and Tanzania (49%). The lake has a complex shoreline structure composed of gulfs and bays that receive municipal and industrial wastes from adjacent urban centres. Murchison Bay, situated in the north-western part of Lake Victoria, has been threatened by several anthropogenic and natural influences e.g. wetland degradation, overfishing, increased pollution from Kampala City, water hyacinth infestation and water level decline (Hecky et al. 1994, 2010; Scheren et al. 2000).

Pollutants enter the Inner Murchison Bay (IMB) mainly due to runoff leading to biodegradation of phytoplankton and macrophytes e.g. algae and water hyacinth. Over the past decade, the pollutant loading into the IMB from streams has increased tremendously due to population growth, wetland encroachment, economic growth, industrialization and paving of urban areas in

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Kampala City. Population pressures around Lake Victoria have led to land degradation, which increased sediment and nutrient loading into the lake aquatic system from non-point source pollution through streams (Machiwa 2003). Non-point pollution from catchments around Murchison Bay was reported to be higher than that from point sources into the bay even though no quantifiable justifications have been made (Banadda et al. 2009). Point source pollution is mainly from industrial wastes as well as organic and sewage disposal that ends up in the IMB through streams. Diffuse pollution collected through stormwater runoff ends up as point loads into the wetlands and lake through channels and streams (Banadda et al. 2009). Atmospheric deposition is another major source of pollution into Lake Victoria. Tamatamah et al. (2005) compared the existing estimates of phosphorus (P) from municipal and runoff inputs and concluded that atmospheric deposition represented 55% of the total P input to Lake Victoria.

Monitoring programs and studies for the IMB have reported persistent outbreak of algal blooms around the lake shores and increasing mass occurrences of cyanobacteria especially near inshore areas of the lake due to increased nutrient pollution (Cózar et al. 2007; Haande et al. 2011). Physicochemical tests along the shores and bays of Lake Victoria were found to differ greatly. Their water quality was mostly influenced by the extents of the bordering wetlands and the P and nitrogen (N) loading in runoff from the catchment and population centres upstream (Cózar et al. 2007; Bracchini et al. 2007).

Murchison Bay is heavily eutrophic, and N is regarded as the limiting nutrient (Haande et al. 2011; Ssebiyonga et al. 2013). The water treatment costs by the National Water and Sewerage Corporation (NWSC) have trebled in the last decade. This increase in costs is attributed to the fluctuating water levels and deteriorating water quality in the bay which has threatened the capacity to increase water production for the ever rising population in Kampala City (Kayima et al. 2008; Banadda et al. 2009). In this regard, the NWSC has extended water abstraction points to deeper and further portions of Lake Victoria. Consequently, a new water treatment plant is being developed out of the Murchison Bay, in the Katosi Peninsula of Lake Victoria located 60 km further east of Kampala City (NWSC 2015). Such investments ought to be environmentally and economically sustainable, and thus, there is a need to understand long-term water quality variations in the lake. Understanding long-term water quality variations in the bay is vital for decision making especially if routine monitoring is not adhered to. Due to lack of funds, systematic data collection and monitoring activities around Lake Victoria have not been carried out, implying that data on water quality is both scarce and scattered making it difficult to rank the degree of pollution in and around the lake (Machiwa 2003).

It is worth noting that no substantial flow and water quality models have been developed to study the variation of water quality in the bay with time despite its morphology and activities. Ecological models have been applied in several cases to communicate future stressor impacts (Janse 1997; Nielsen et al. 2014; van Gerven et al. 2015). Limited information is available on the hydrodynamics of the Murchison Bay and other Lake Victoria embayments. For example, in the eastern part of Lake Victoria, knowledge of the hydrodynamics of the Rusinga Channel with the main lake and Nyanza Gulf allowed for estimation of the net exchange of nutrients (Gikuma-Njuru and Hecky 2005; Gikuma-Njuru 2008). According to Gikuma-Njuru and Hecky (2005), mixing plays a major role in nutrient recycling and water quality in Lake Victoria. The Winam/Nyanza Gulf located in the northeast part of Lake Victoria provided a net source of dissolved silica and total N to the main lake, which was potentially a source of P to the gulf depending on the interchange through the Rusinga Channel. Khisa et al. (2006) attempted a first quantitative evaluation to determine the exchange dynamics between the offshore waters of Lake Victoria and Winam Gulf through the Rusinga Channel. They indicated that localised effects on circulation and flushing, which affected water quality patterns around Mbita Channel, were likely influenced by the construction of the Mbita Causeway. A mixing box model was used to quantify nutrient fluxes and ecosystem metabolism along the Winam Gulf and the Rusinga Channel and showed that nutrients entering the gulf through river inflows and municipal sources were largely retained in the gulf, with only a small fraction being transferred into the main lake (Gikuma-Njuru et al. 2013). Such lake embayment hydrodynamics are influenced by wind, temperature and water level changes. Wind-induced mixing in the Murchison Bay influenced its temperature and water quality (Ssebiyonga et al. 2013). Morana et al. (2014) indicated that environmental factors played a key role in the control of phytoplankton production of dissolved organic matter. Similarly, flushing and exchange

patterns between the IMB and offshore waters of Lake Victoria are vital in understanding water quality of the IMB. Akurut et al. (2014b) showed that the IMB volume is mainly controlled by the main lake water variations with more than 95% of the total volume determined by the Lake Victoria flux. Luyiga et al. (2015) attributed the short-term IMB water quality variations for the period 2000–2003 to water level fluctuations but did not quantify their hydrodynamic effects. It is important to study the long-term variations of water quality in the Murchison Bay in order to understand impacts of pollution and climatic conditions on the IMB. The main objective of this research is to explain the variations in IMB water quality over time. This research uses lake level variations to study the IMB hydrodynamics and consequently to provide a plausible water quality model in a bid to study the long-term variations in the IMB over the past decade.

Materials and methods

Murchison Bay

Murchison Bay, surrounded by the greater metropolitan Kampala area, is located south of Kampala City, Uganda. Kampala City is a hub of industrial and commercial activities, characterised by a very high population growth rate of 5.6% per annum and an estimated current population of about 2 million inhabitants (Vermeiren et al. 2012). Murchison Bay lies between latitudes (00°10′00″N–00°30′00″N) and longitudes (32°35′00″E–32°50′00″E) at an average elevation of 1134 m a.s.l. It covers a total area of about 62 km^2 and can further be split into the inner and outer Bay as their characteristics differ tremendously. IMB has an area of about 18.4 km^2 and an average depth of 3.2 m. It is the abstraction point for the potable water supply for Kampala and also the recipient for surface runoff, sewage effluent and industrial and municipal wastes from the city. The IMB has an average catchment area of 282 km^2 comprising both wetlands and urban areas of the city.

The major channels/rivers of the IMB, which are sources of significant pollution, include (Fig. 1) the following: Nakivubo, which drains Kitante and Lugogo channels with inlet into the IMB at M1; Kansanga, which stretches into the Gaba shoreline with outlet towards M4; Kinawataka, that drains industries of Nakawa and Kyambogo with outlet at M3; and Namanve area, which is predominantly a wetland and gazetted as an industrial park. In the past, a large portion of the Nakivubo Channel has been modified to a concrete-walled structure limiting pollutant purification through the ever dwindling wetlands surrounding the IMB (Kansiime and Nalubega 1999; Sekabira et al. 2010). Inauguration of the Kampala Capital City Authority in 2010 also improved city drainage as more lined canals were constructed within Kampala (KCCA 2010). Such changes allow for more waste collection but also hinder effective quantification of pollutant loading into the IMB. Figure 1 shows the streams and monitoring points in the IMB and its bathymetry.

Data

The bathymetry of the IMB for the year 2012 and measured water level variations were obtained from the Ministry of Water and Environment, Uganda. In Uganda, Lake Victoria levels are observed at two stations: Jinja and Entebbe. Murchison Bay lies between these two locations and is closer to Entebbe. The average elevation at Entebbe (1134 m a.s.l) is 0.545 m higher than Jinja. Satellite lake height variations at the lake midpoint were obtained online from USDA (2015); these measurements were already quality controlled and corrected based on the more reliable water level measurements at Jinja. They were checked for consistency against measured levels at Entebbe. The differences in elevations are accounted for by the lake bathymetry. Since IMB and Entebbe lie at similar elevations (1134 m a.s.l), the satellite levels were adjusted by a correction term of +0.545 to obtain lake level variations for input in the IMB hydrodynamic model.

The available spatial water quality measurements within the Murchison Bay were obtained from the NWSC Laboratory and the Ministry of Water and Environment, Uganda, for the period 2001–2014. These measurements were irregular and available during certain months and years at a frequency of at least 1 month. These measurements were mainly available at periods when the Lake Victoria Environment Management Project (LVEMP) was active. Examples of parameters analysed include temperature (T), dissolved oxygen (DO), total suspended solids (TSS), electrical conductivity (EC), total phosphorus (TP), ammonium (NH_4^+) and biochemical oxygen demand (BOD) at the different monitoring points within the IMB (Fig. 1a). These measurements were checked for consistency and analysed to determine the trends in water quality within the IMB. BOD and NH_4^+ have been chosen for the

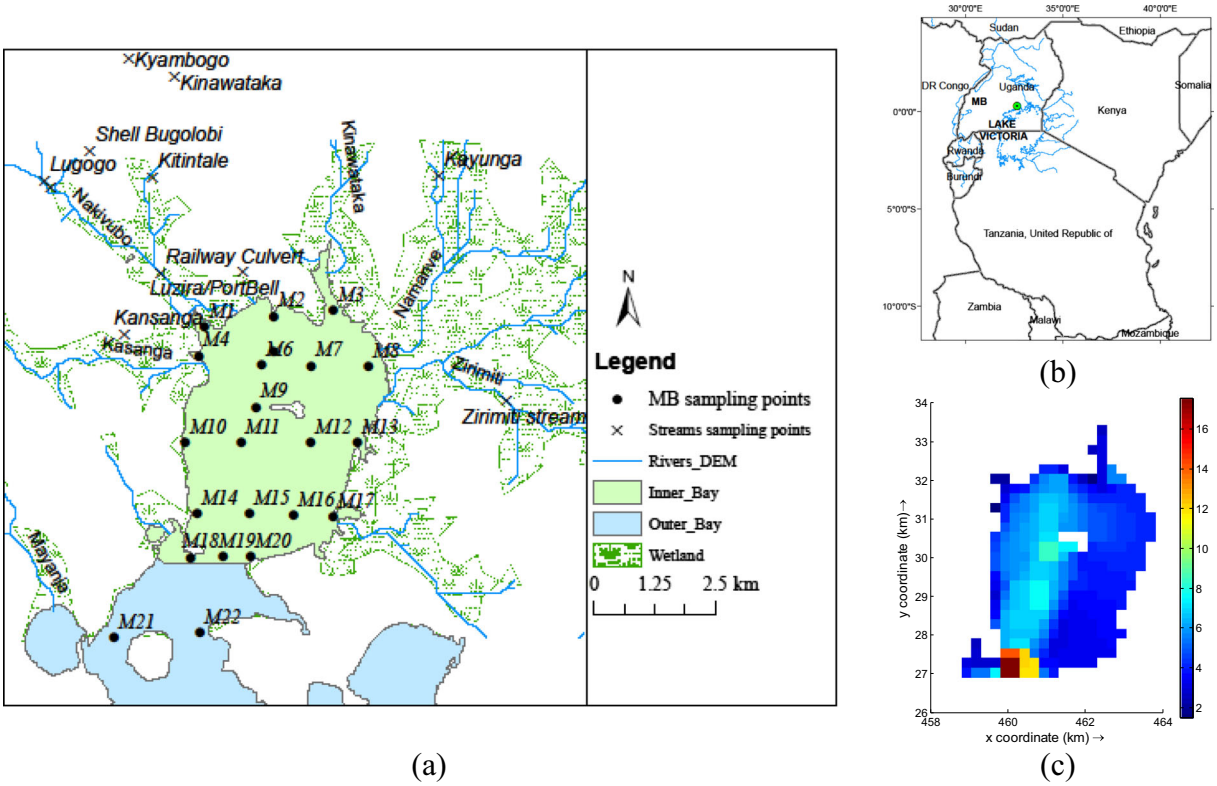

Fig. 1 **a** Streams and monitoring points in the MB. **b** Map of East Africa showing the location of MB within Lake Victoria. **c** IMB bathymetry in 2012 showing depth in metres (m)

water quality analysis as they give an indication of the largest pollution in the bay which is mainly organic matter, sewage and industrial waste disposal entering the bay through streams and channels that flow within Kampala City. DO levels are indicative of the physical, chemical and biological activities in water bodies.

The main streams into the IMB at M1, M2, M3 and M4 are Nakivubo, Kitintale, Kinawataka and Kansanga, respectively. Water quality trends and regressions have been used to study catchment behaviour based on concentration–time relationships (Naddafi et al. 2007; Tabari et al. 2011). Analysis of the water quality trends at the IMB shores and exit provides a means of understanding the impact of dilution of the main Lake. The annual statistical properties for the onshore and off-shore waters of the IMB were studied based on the Mann–Kendall test and Sen's slope estimator as applied in Tabari et al. (2011), Machiwal and Jha (2015) and Berton et al. (2016). The Mann–Kendall test is a non-parametric test used to identify trends in time series data by comparing the relative magnitudes of sample data rather than the data values themselves especially when the data does not conform to any particular distribution (Kendall 1948). Sen's slope estimator provides an indication of the true slope or change per unit time of the water quality parameter (Sen 1968).

Pollutant concentrations in bays largely depend on the inflows and internal processes; such processes may be difficult to ascertain in the ungauged IMB, hindering the estimation of pollutant loading into the IMB. Since streams flowing into the IMB are collectors of both point and non-point source pollution from Kampala and surrounding urban centres, their loadings were compared with IMB shore concentrations to determine the impact of wetland degradation and IMB behaviour over time. Our working hypothesis assumes that the measured concentrations within the IMB shores provide credible estimates of input loading. These estimates provide point source discharges for the water quality model. A water quality model was set up for the IMB and simulation results analysed against available observations to explain the behaviour of the concentrations over time. This model was calibrated based on downstream measurements towards the IMB exit at M14, M15 and M19.

Water quality model

A numerical water quality model was set up for the IMB using Delwaq module in Delft3D. Delft3D is a flexible integrated modelling suite, which simulates 2D or 3D flow, sediment transport and morphology, waves, water quality, ecology and interactions between these processes in time and space. It consists of several modules, which are linked to and integrate with one another e.g. D-Flow, D-Morphology, D-Water Quality, D-Ecology and D-Particle Tracking. In this study, D-Flow and D-Water Quality are used. D-Flow simulates non-steady flows in relatively shallow water. It incorporates the effects of tides, winds, air pressure, density differences, waves, turbulence, drying and flooding. The output of that module is used as input for the other modules in the Delft3D suite (Delft3D-Flow 2013). D-Water Quality simulates the far and mid-field water and sediment quality due to a variety of transport and water quality processes. It includes several advection–diffusion solvers and an extensive library of standardised process formulations and substances. It also provides an open system to define additional substances, processes acting on new and existing substances, additional coefficients to be used in the formulae and external forcings (D-Water Quality 2013).

D-Water Quality (2013) describes a general DO model based on the basic Streeter–Phelps formulation, which was derived for a continuous BOD load to a river (Chapra 1997). This model simulates decay of organic matter (BOD), nitrification of NH_4^+ and replenishment of DO through re-aeration. Consumption of DO is mainly due to decay of organic matter which may reach the system through discharge of sewage effluent or indirectly in eutrophic conditions through decay of algal and phytoplankton matter.

Mineralisation of carbonaceous BOD is subject to a linear and temperature- and oxygen-dependent decay process as defined in Eq. (1) below:

$$\text{Mineralisation flux} = -Rc_{\text{BOD}} \times \text{CBOD}_5 \times \vartheta_{\text{BOD}}^{(\text{T}-20)} \times \frac{\left(\text{O}_2 - \text{DO}_{\text{BOD}}^{\text{Cr}}\right)}{\text{DO}_{\text{BOD}}^{\text{Opt}} - \text{DO}_{\text{BOD}}^{\text{Cr}}} \quad (1)$$

where CBOD_5 is the concentration of carbonaceous BOD_5 [gO_2/m^3], Rc_{BOD} is the first-order rate constant at 20 °C [day^{-1}], ϑ_{BOD} is the temperature coefficient [−], T is the water temperature [°C], (O_2) is the DO concentration [$gO_2\ m^{-3}$], $\text{DO}_{\text{BOD}}^{\text{Cr}}$ is the critical DO concentration for BOD mineralization [gO_2/m^{-3}] and $\text{DO}_{\text{BOD}}^{\text{Opt}}$ is the optimal DO concentration for BOD mineralization [gO_2/m^{-3}]. Nitrification follows a similar reaction to Eq. (1) at stoichiometry of $NH_4^+/O_2/NO_3^- = -1{:}-4.571{:}1$ ($gN\ m_{-3}/gO_2\ m^{-3}/gN\ m^{-3}$]). Nitrification proceeds under aerobic conditions only and stops when the water temperature drops below a certain critical level, usually set to 3 °C.

Re-aeration depends on the wind speed, stream velocity and water depth. It is formulated as a first-order process working on the oxygen deficit as per Eq. (2):

$$\text{Reaeration flux} = Rc_{\text{rear}} \times \left(\text{DO}_{\text{sat}} - (\text{O}_2)\right) \times \vartheta_{\text{rear}}^{(\text{T}-20)} \quad (2)$$

where Rc_{rear} is the first-order re-aeration rate constant [day^{-1}], DO_{sat} is the saturation concentration of DO depending on temperature and salinity [$gO_2\ m^{-3}$] and ϑ_{rear} is the temperature coefficient for re-aeration [−]. The 'sediment oxygen demand' process, which allows to specify additional oxygen consumption by the sediments, has been neglected in this study since sediment properties are not available for the IMB. This process proceeds without limitation if the oxygen concentration is over 2 mg/l and stops completely when the concentration drops to 0 mg/l.

Water quality (WQ) models were created for the IMB based on a numerical grid of 250 × 250 m. The boundaries between the wetlands and IMB were considered as closed, meaning that discharges from/into the swamps were neglected and only dynamics caused by main lake water variations were considered. Continuity can be used to assess influence of specific inputs like lake water levels into the system (D-Water Quality 2013). Based on continuity evaluations, it is found that more than 95% of IMB volume is accounted for by the Lake Victoria flux. It is on this basis that the IMB model enables water quality modelling of the Bay. The open boundaries were set at the exit of the IMB into the OMB. Monitoring points at M14, M15 and M19 were used to check quality at the IMB exit while M1, M2, M3 and M4 provided average IMB shore behaviour. The main discharge locations were at M1, M2, M3 and M4 located at the extreme shores of the IMB while M8, M10, M13 and M17 were considered as minor discharge points. The hydrodynamic and water quality models are simulated for the period 2001–2014 based on a constant onshore wind speed of 2.5 m/s. It is assumed that the Lake Victoria epilimnion is deeper than the Murchison Bay

suggesting vertical mixing in the bay and therefore no need to account for stratification and turnover in the models. In shallow gulfs, daily mixing maintains high oxygen levels in the whole water column throughout the year (Gikuma-Njuru and Hecky 2005).

Results and discussion

Water level variations

IMB volume is determined by the runoff discharges from surrounding wetlands of Lake Victoria and from Lake Victoria. The lake levels at Entebbe are generally higher than those at Jinja owing to the bathymetry of the lake even though they follow similar fluctuation patterns. Generally, the lake levels declined by about 1 m in the period 2005–2006 (Fig. 2) and increased thereafter, partly attributed to the construction of the Kiira Dam at Jinja and climatic forcing (Awange et al. 2008; Swenson and Wahr 2009). The difference between water levels before and after the shift in 2006 is significant at the 95% confidence level based on the two-sample Student *t* test ($p = 0.03$). Water level fluctuations in Lake Victoria have been linked to climate and anthropogenic effects. Awange et al. (2008) explained that 80% of the Lake Victoria refill is predominantly rainfall compared to the 20% from basin discharge. Akurut et al. (2014a) reported that total annual precipitation is expected to increase by about 10% for the RCP4.5 mid-range greenhouse gas (GHG) mitigation scenario and less than 20% for the RCP8.5 high GHG scenario over the twenty-first century with higher (up to 40%) increase in extreme daily intensities that will affect the Lake water levels.

Pollution in the IMB

The highest pollution points/hotspots are located at the northern shores of the IMB especially around heavily industrialised areas namely M1 (Nakivubo Channel outlet), M2 (Luzira/Port Bell discharge), M3 (Kinawataka stream) and M4 (Kansanga stream), as shown in Table 1. It is generally noticed that the water quality at the IMB shores tends to be worse than that in the streams. The large range of values is attributed to the intermittent monitoring especially if sampling occurred after a heavy rainfall event or during waste discharge from neighbouring industries. Generally, BOD measurements were larger at the IMB shores compared to the streams, while NH_4^+ values were constant between the streams and the IMB shores. This indicates accumulation of wastes in the IMB shores.

The water abstraction points are located towards the OMB at M14 and M18, about 4.5 km downstream of the sewage discharge point located near M1 (Fig. 1). The points M8, M10, M13 and M17 are representative of the minor discharge points. However, wetlands around the IMB have been destroyed leading to *short-circuiting* of the wastes from the city directly into the lake. A comparison of the pollution in the streams upstream of the IMB and the concentration of the pollutants downstream inside the IMB with time has been studied. Pollution

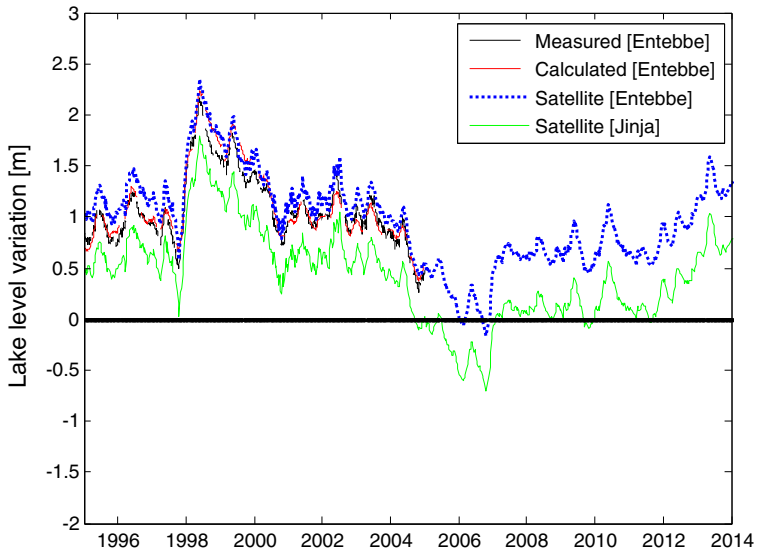

Fig. 2 Lake level variations at Entebbe and Jinja over time

Table 1 Water quality along IMB streams and shores (2001–2014)

	Nakivubo Channel	M1	Luzira/Port Bell discharge	M2	Kinawataka stream	M3	Kansanga stream	M4
BOD [mg/l]								
Mean ± SD	71.9 ± 59.1	60.2 ± 99.9	21.5 ± 20.8	47.5 ± 98.0	52.2 ± 51.7	37.1 ± 60.8	32.6 ± 43.5	62.1 ± 132.7
Min	4.9	2.1	4.8	0.3	4.2	2.0	2.2	1.6
Max	248.0	396.8	106.0	530.7	210.0	228.7	208.0	716.0
DO [mg/l]								
Mean ± SD	0.9 ± 1.3	1.1 ± 1.7	2.3 ± 1.8	3.1 ± 2.8	0.3 ± 0.9	2.7 ± 2.5	3.5 ± 1.3	2.1 ± 2.4
Min	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0
Max	4.6	6.4	5.6	8.2	2.8	7.9	4.6	6.8
NH_4^+ [mg/l]								
Mean ± SD	17.3 ± 14.2	10.2 ± 10.3	11.9 ± 7.9	4.2 ± 6.3	6.1 ± 5.8	5.0 ± 6.3	7.0 ± 6.5	6.1 ± 7.8
Min	0.0	0.1	0.1	0.0	0.1	0.0	0.0	0.0
Max	50.0	38.6	28.0	25.0	21.0	23.7	22.5	35.9

into the IMB generally increases exponentially over time despite the linear increase in pollutants from upstream channels (Fig. 3). Nakivubo stream, located upstream of M1, is the most polluted, and its influence affects larger portions of the IMB; i.e. the rapid exponential rise of BOD in the IMB at M1 is synonymous with that at M2. The rate of increase of BOD upstream of M2 at the Kitintale stream is lower than the accumulation rate in the IMB at M2 since this loading is further influenced by the high rate of loading from the Nakivubo stream upstream of M1. Furthermore, the influence of the Nakivubo wetland around the IMB diminishes tremendously after 2012 when the BOD and NH_4^+–N concentrations at the IMB shore (M1) approach the stream loading at the Nakivubo Railway culvert (Fig. 3).

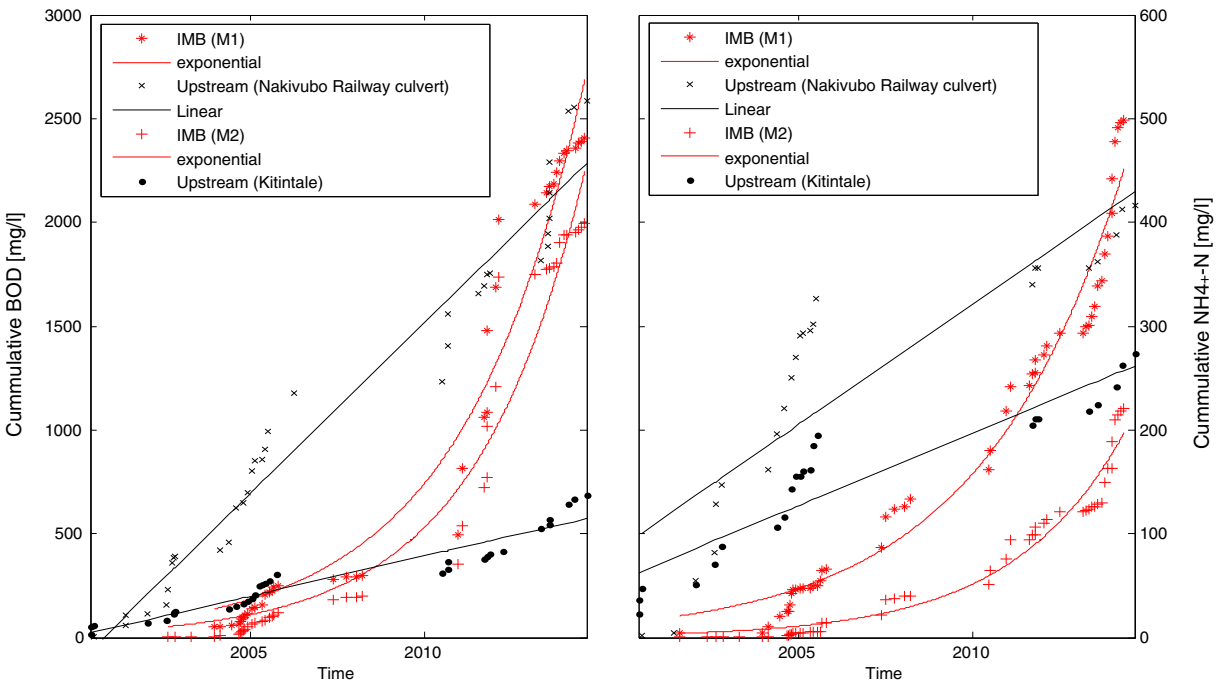

Fig. 3 BOD and NH_4^+ concentrations within the IMB and upstream of points M1 and M2

Despite the linear increase of pollutants upstream, IMB shore concentrations increase exponentially even beyond stream loadings, indicating stronger accumulation of pollutants in the IMB compared to the inputs from streams. Contrary to Luyiga et al. (2015), there is indeed accumulation of pollutants in the IMB over the past decade. Given the presence of the wetlands between the streams and IMB hotspots (onshore monitoring points in the lake), the difference between the linear stream loading at the Nakivubo railway culvert, for example, and the exponential curve of the IMB concentrations shows the impact of degradation of the Nakivubo wetland around the IMB dubbed as *wetland effect*. The bay has become a sink of pollutants entering the Lake; e.g. after 2012, the water quality towards the shores increased more than the stream loading. It is this basis that justified our working hypothesis to assume the bay shores as closed boundaries with concentrations at IMB hotspots considered as discharges in the IMB for the Delwaq model.

Trend analysis

Table 2 shows the average annual water quality along the IMB shores and exit. The annual statistical properties for the onshore (_Pol) and off-shore (_Exit) waters are shown in Fig. 4a based on the Mann–Kendall test and Sen's slope estimator. The average annual temperature in the IMB at the shores and towards the exit in the period 2001–2014 generally decreases over time. EC, TP, NH_4^+ and BOD concentrations generally increase as indicated by the positive *Z* values whereas DO decreased over time, as shown by the negative *Z* values of the Mann–Kendall test (Fig. 4a). This is an indication of an increase in pollution in the IMB.

Generally, the rate of increase of pollutants at the IMB shores is higher than that at the IMB exit as the rate of dilution and mixing varies within different parts of the Bay for the different parameters (Fig. 4b). TSS concentration at the IMB shores showed the strongest positive and significant trends increasing at a rate of 17% per annum in the period 2001–2014 (Fig. 4b). DO reduced at an average rate of 14% per year around the shores but only 0.12% at the IMB exit where more mixing with the main lake occurs. There was a significant increase at 99% confidence level in TSS, TP and NH_4^+ at the IMB shore at rates greater than 10% per annum due to an increase in stream loadings upstream of M1, M2, M3 and M4. Again, this shows that more pollutants accumulated in the IMB especially at the shores compared to the OMB.

Water levels and dilution effect

The effect of dilution was noticed for pH which increases towards the IMB exit despite the decreasing

Table 2 Average annual water quality parameters along IMB shores (_Pol) and exit during 2001–2014

Year	Temp [°C]		PH		EC [μS/cm]		TSS [mg/l]		TP [mg/l]		NH_4^+ [mg/l]		BOD [mg/l]		DO [mg/l]	
	Pol	Exit	Pol	Exit	Pol	Exit	Pol	Exit	Pol	Exit	Pol	Exit	Pol	Exit	Pol	Exit
2001	28.4	28.0	7.6	7.6	99.7	103.0	18.3	20.7	0.1	0.2	0.2	0.1	1.9	0.9	5.5	5.0
2002	28.4	28.0	8.8	8.4	99.7	103.0	18.3	20.7	0.1	0.1	0.2	0.1	1.9	0.9	5.5	4.7
2003	26.2	27.1	8.0	8.0	167.6	103.8	10.0	21.8	0.4	0.1	0.1	0.2	13.0	3.1	2.0	4.7
2004	27.6	25.9	7.2	7.6	235.5	104.7	28.7	22.9	0.6	0.2	2.6	0.2	7.5	5.2	2.6	4.5
2005	26.0	25.6	7.7	7.6	151.4	106.7	39.7	20.7	0.6	0.2	1.6	0.1	8.7	3.8	3.3	5.7
2006	25.8	25.1	7.2	7.7	295.2	108.1	79.3	18.1	1.0	0.2	6.5	0.1	15.1	3.9	1.6	5.9
2007	25.6	24.7	6.8	7.8	439.1	109.5	118.9	15.5	1.5	0.2	11.4	0.2	21.5	3.9	0.0	6.1
2008	24.0	24.7	6.7	7.6	387.7	109.3	175.5	19.0	1.2	0.2	4.2	0.2	3.0	2.0	0.0	5.7
2009	24.2	24.7	6.8	7.7	384.0	109.1	210.2	14.0	0.9	0.2	11.1	0.1	72.4	1.7	0.0	5.7
2010	24.3	24.6	6.8	7.8	380.3	109.0	244.8	9.0	0.5	0.2	18.0	0.1	141.9	1.4	0.0	5.7
2011	24.8	24.1	6.8	7.7	404.8	105.7	310.3	23.2	4.1	0.3	9.2	0.1	191.3	0.9	0.0	5.4
2012	23.8	24.3	6.8	7.5	476.9	107.9	214.9	10.1	4.7	0.2	5.3	0.0	254.8	4.2	0.0	4.7
2013	25.6	25.9	6.9	8.0	228.3	117.4	87.6	14.6	2.8	0.3	7.1	0.3	30.3	7.1	2.0	4.7
2014	26.5	25.8	7.5	8.2	222.9	119.0	42.2	56.0	3.7	0.5	11.8	0.3	24.7	5.9	4.0	4.7

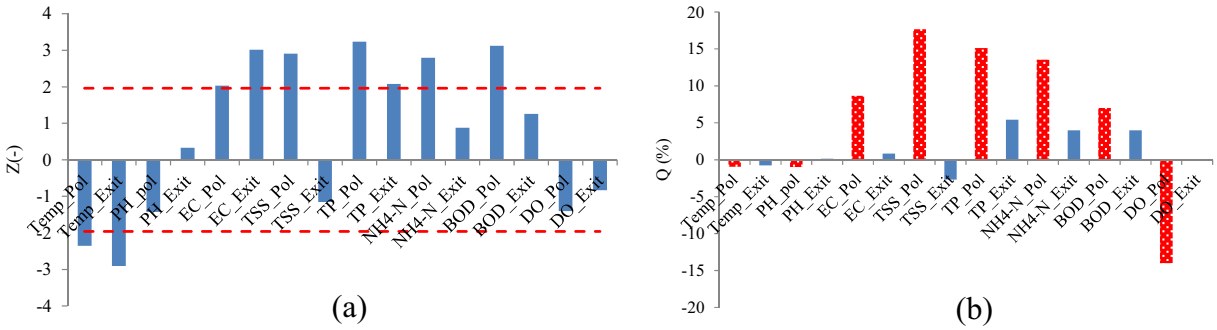

Fig. 4 **a** Mann–Kendall *Z* statistic and **b** Sen slope estimator for water quality parameters at the shores (_Pol) and towards the exit (_Exit) of the IMB

pH at the shores and TSS which decreases towards the IMB exit despite the increasing TSS at the IMB shores (Fig. 4). EC has been used in freshwater ecosystems as a tracer of exchange and mixing of water (Rueda and MacIntyre 2010; Gikuma-Njuru et al. 2013). The highest impact of dilution and flow dynamics is noticed at the IMB shores when water levels are between 0.6 and 1.2 m (Fig. 5a). Conductivity measures the ability of water to pass an electrical current and is affected by the presence of dissolved solids/ions and temperature. EC was generally low when water level changes in the IMB were lower than 0.6 m and also when the changes were greater than 1.2 m (Fig. 5a). At low water levels, water flow into the IMB is limited, encouraging precipitation and settlement of solids and consequently reduction of ion concentrations in the water column. At higher water levels, more flux from the lake dilutes the pollutants, lowering the ion concentration and consequently conductivity. However, water level variations alone cannot fully explain the changes in EC, e.g., Fig. 5b shows that between 2004 and 2006, EC increases as water level decreases but also continues to increase as water levels increase. This indicates that besides water levels, other factors such as changes in loading rates are vital in determining ion concentrations and consequently EC in the IMB. Generally, the TSS and EC values at the shores increased in the period 2001–2011 and decreased after 2012 when the water level increased further, indicating the influence of dilution of pollutants and hydrodynamics of the IMB (Fig. 5b).

Further analysis of BOD concentration at the IMB shores with water level variation shows that the BOD concentration improved with an increase in lake levels especially after 2012. This was not necessarily true for NH_4^+ concentration, highlighting the importance of dilution in the IMB and biodegradation of organic matter (Fig. 6). The reduction in BOD at the shores despite an increase in stream loadings may be attributed to the increasing water levels, which allow for more mixing and oxygenation leading to increased biodegradation of

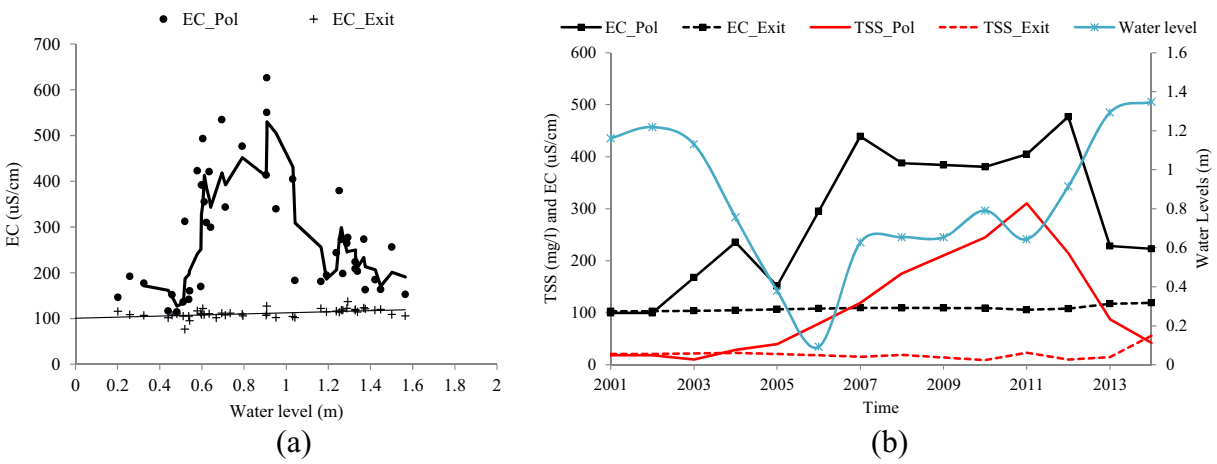

Fig. 5 **a** EC against water level variations in the IMB. **b** Annual TSS/EC (conductivity) trends at the IMB shores and exit against time

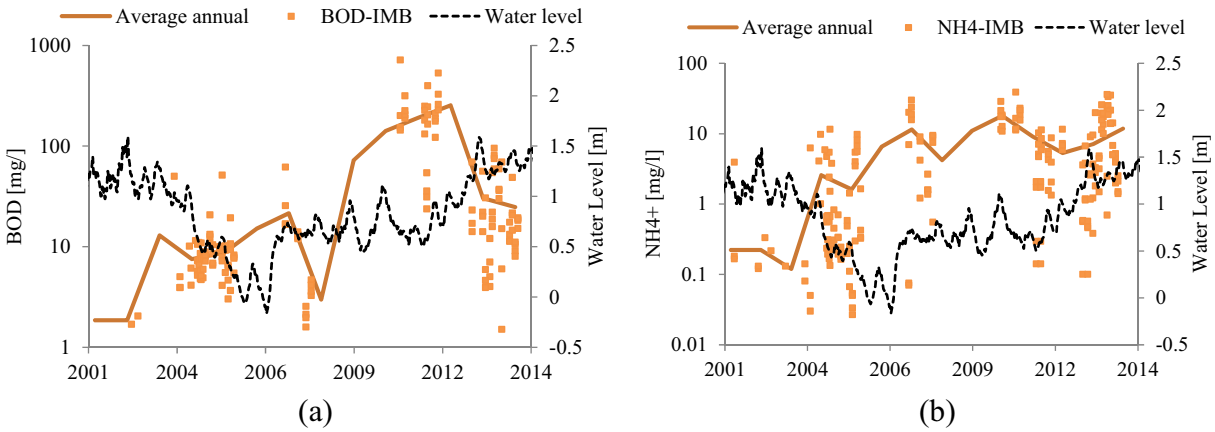

Fig. 6 BOD and NH_4^+ concentrations at the IMB shores against water level variations

BOD. The NH_4^+ concentrations are mainly affected by the stream loading and decay of organic matter in the IMB.

Temperature

The IMB hydrodynamic model was validated based on temperature measurements within the IMB at locations M14, M15 and M19 towards the IMB exit. Based on the observed and simulated temperatures, the Nash–Sutcliffe efficiency coefficient (NSE) of the IMB model was satisfactory at 0.58. It is worth noting that Radwan and Willems (2008) classified mean relative errors for temperature measurements in the range 0–20%, implying that the hydrodynamic model was acceptable for water quality modelling. Figure 7 shows a general decline in temperature in the IMB from 28.5 °C in 2001 to 24 °C in 2007 after which the temperature in the IMB is generally constant around 24.5 °C between 2007 and 2011, before increasing after 2012 when water levels increased. It was noted that the decline in temperature in 2001–2007 was consistent with the decline in water levels in the same period.

Water temperature is generally higher along the shallower areas of the IMB where regular vertical mixing occurs compared to the exit of IMB. Rainy seasons lead to an increase in water levels in Lake Victoria and hence more mixing (Semyalo et al. 2009). Earlier research in the IMB indicated that temperatures in the bay were warmer in the rainy seasons than in the dry seasons (Semyalo et al. 2009; Ssebiyonga et al. 2013). The lower temperatures in periods of lower water

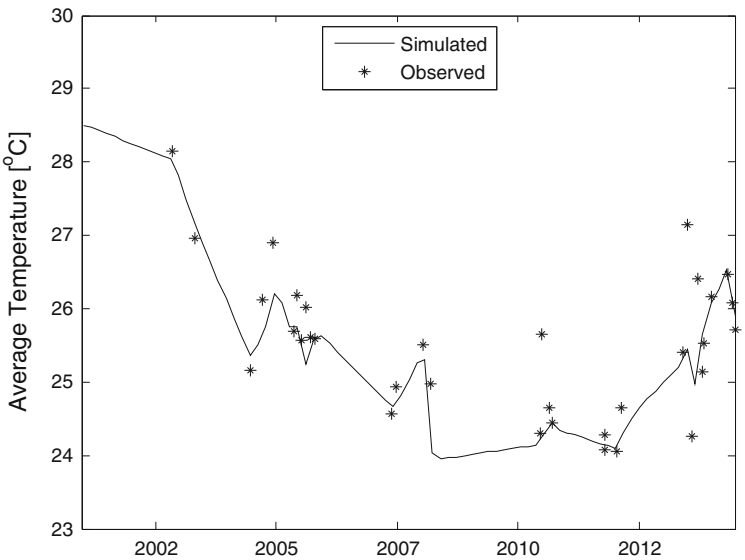

Fig. 7 Average temperature variation within the IMB

levels may be attributed to the increased eutrophication and sedimentation as discussed in Haande et al. (2011). Light has been reported as a limiting factor for algal photosynthesis in the IMB, and its attenuation is greatly determined by particulate matter in the Bay (Ssebiyonga et al. 2013). Algal and sediment layers absorb sunlight rays for photosynthetic activities, and these blankets constrain release of thermal energy back into the water, hence reducing the amount of heat available to warm the water in the bay. Alterations of water temperature may cause significant shifts in the aquatic communities and decrease of fishes. Also, higher temperatures lead to higher conductivity. Generally, thermal pollution is detrimental to the water supply industry as lower temperatures inhibit coagulation and flocculation processes during raw water treatment. It is not surprising that the Gaba III water treatment plant extension in the IMB was completed in 2006 to boost additional water supply to the greater Kampala metropolitan area.

IMB hydrodynamics

The depth averaged velocity in the IMB is very low with magnitudes not exceeding 0.06 m/s (Fig. 8), yet these velocities increase with increase in water levels. The flow of water in the IMB occurs in a circular anticlockwise motion, which partly explains the lower pollutant concentrations towards the centre of the bay and the higher concentrations along the northern shores of the IMB. From Fig. 3, M2 concentrations approach M1 loadings to levels even much greater than the loading from the Kitintale stream (upstream of M2). This could be attributed to advection of the pollutant loads at M1 as illustrated by the higher velocity in the north-western shores of the IMB (Fig. 8). This concurs with Pinardi et al. (2015) who found that the main hydrodynamic effects that influence chlorophyll-a distribution in the shallow fluvial lakes are related to the combined effect of advection due to wind force, riverine current and gyres, which induce re-circulation and stagnation regions. Ssebiyonga et al. (2013) reported that the physiological condition of algae in the Murchison Bay depends on the physical characteristics of the water column as high photochemical energy quantum conversion efficiency was obtained when water was well mixed, linking the importance of the hydrodynamics of the bay to its water quality.

The residence time within the IMB was estimated at 50–100 days even though it varied from point to point with areas around the northern wetlands at M1, M3 and M4 having very high residence times often exceeding 10 years (Akurut et al. 2014b). Similar findings in the Winam Gulf of Lake Victoria showed that nutrients entering the gulf through river inflows and municipal sources were largely retained in the gulf, with only a small fraction entering into the main lake (Gikuma-Njuru et al. 2013).

IMB water quality

Water quality models were set up after neglecting the effect of the extensive water hyacinth at the shoreline area and considering only pollutant loadings at M1, M2, M3, M4, M8, M11, M10 and M17. The concentrations at these locations were considered as discharges into the model. The water quality model was run at 15-min time

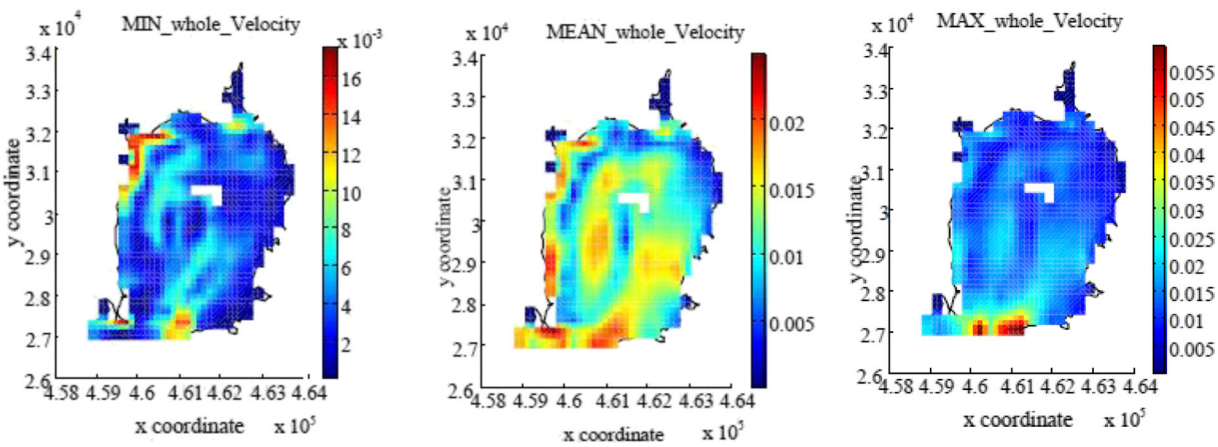

Fig. 8 Delft3D flow model results: minimum, mean and maximum velocity profiles in the IMB for the period 2001–2014

steps to ensure numerical stability in the solution of the advection diffusion equations. Acceptable results were obtained for NH_4^+, DO and BOD concentrations using the general DO model for parameters in Table 3.

The parameters in Table 3 were tuned to obtain a close fit of the model and observations towards the IMB exit. SW_{rear} is an input parameter that defines the different coefficients in the re-aeration process in Delwaq that occurs at the top water layer. This empirical relation is valid for tropical lakes and independent of stream velocity (D-Water Quality 2013). A single set of parameter values may constrain the conceptual capacity of the model required to render marked variations in ecological regime types (Nielsen et al. 2014), but provides an attempt to explain IMB characteristics over time in this case since the Bay has gradually become eutrophic.

Figure 9 shows comparison between simulated and observed water quality parameters within the IMB. The accuracy of the water quality simulation results improves towards the exit of the IMB where greater lake mixing occurs, showing the significance of changes by transport of flowing water. This is vivid for monitoring point M19. The DO simulations are generally overestimated as the sediment oxygen demand is neglected in the model. The depths of the sediment layers in the IMB have increased over time due to accumulation of pollutants, but their volumes have not been monitored. For NH_4^+ and BOD, the IMB model generally simulated their concentrations and temporal variability well considering measurement uncertainties in the range of 20–30% for NH_4^+ and 40–100% for BOD (Radwan and Willems 2008). Therefore, this model was applied in an attempt to explain the processes and water quality in the IMB.

The water quality concentrations generally followed the pollution loading pattern rather than lake level dynamics. Even the 2-m water level decline in the period 2000–2007 did not necessarily yield a significant change in the water quality simulations. By 2011, NH_4^+ levels at the IMB exit had accumulated to 0.3 mg/l from 0.015 mg/l in 2001. The average DO concentration at the IMB exit declined to about 3 mg/l by the end of 2011 when the BOD peaked at 12 mg/l. Ssebiyonga et al. (2013) reported that DO levels tend to increase in months with higher rainfall. The depth averaged DO increased with increasing water level especially after 2011. Such increase in DO may lead to increased biodegradation of organic matter in the IMB and hence reduction of BOD concentrations (to less than 5 mg/l) at the IMB exit after 2012. Water mixing has the potential to influence oxygen in the water column, water temperature changes, nutrients within the water column and disruption of bottom sediment. By comparing the average $NH4^+$ and BOD levels between the IMB shores and exit, it can be estimated that lake inflow accounted for about 40–60% of pollutant dilution in the IMB.

It is generally noticed that model simulations closer to the shoreline areas at M14 are generally lower, especially after 2010, for both NH_4^+ and BOD, probably attributed to the higher diffuse loadings from surrounding wetlands. Bracchini et al. (2007) showed that wetland-released chromophoric dissolved organic matter greatly influenced the attenuation of UV and light in the lake water. It should also be noted that upstream of M14 is the discharge point for the domestic waste from the Gaba water treatment works and headquarters. Diffuse loadings from the surrounding wetlands encourage algal blooms and water hyacinth growth along the Lake Victoria shores (Machiwa 2003; Banadda et al. 2009). However, wetlands around the lake have become degraded over time explaining the larger dispersion towards the end of the decade. For example, the Nakivubo swamp is estimated to have dwindled from 5 km^2 in

Table 3 General DO model parameters

Process	Parameter	Value	Unit
BOD mineralization	Rc_{BOD}	0.1	[day^{-1}]
	ϑ_{BOD}	1.04	[−]
	DO_{BOD}^{Cr}	1	[gO_2/m^{-3}]
	DO_{BOD}^{Opt}	5	[gO_2/m^{-3}]
Nitrification	Rc_{nit}	0.1	[day^{-1}]
	ϑ_{nit}	1.07	ϑ_{BOD} [−]
	DO_{nit}^{Cr}	2	[gO_2/m^{-3}]
	DO_{nit}^{Opt}	5	[gO_2/m^{-3}]
Re-aeration	SW_{rear}	13	[−]

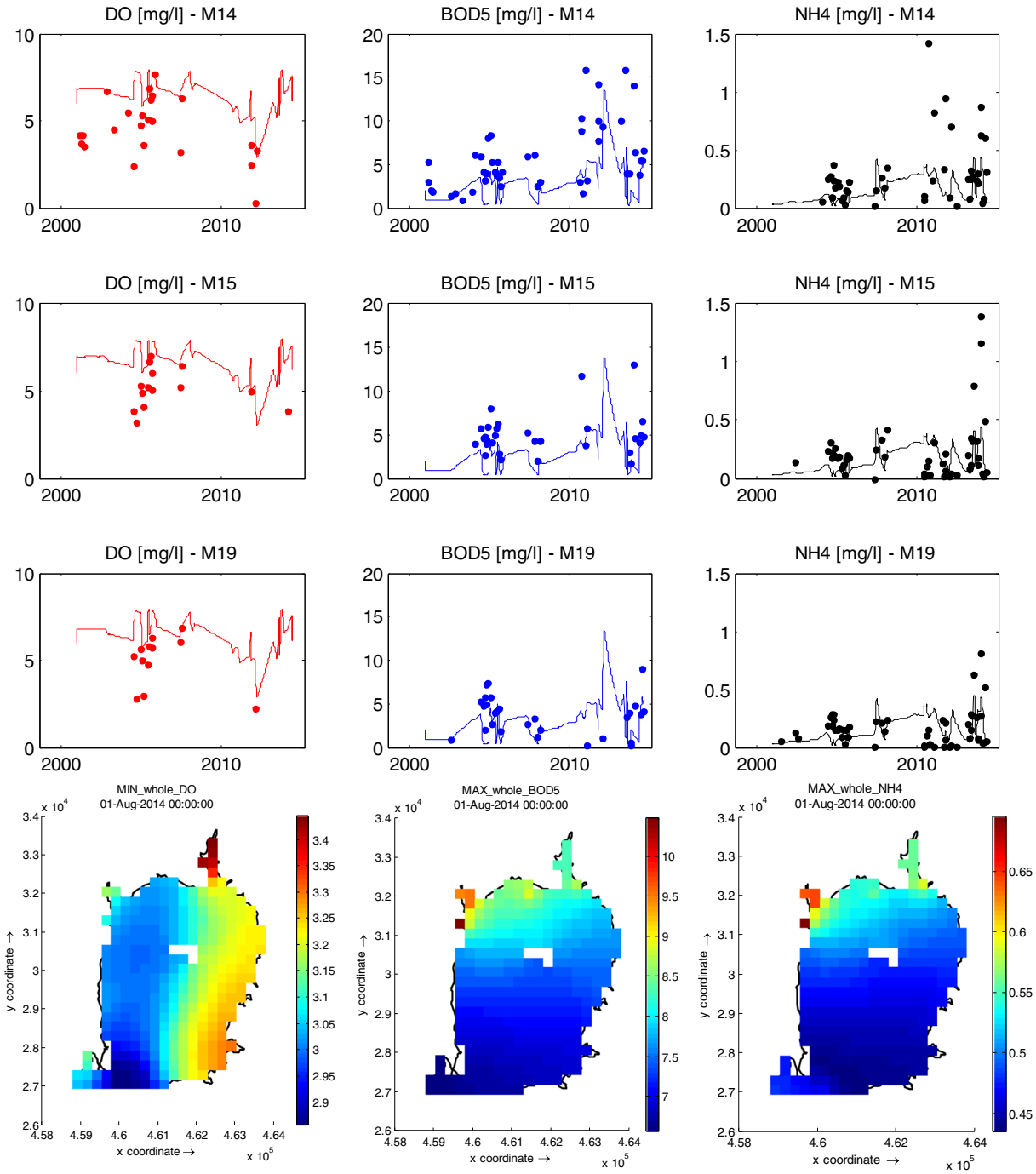

Fig. 9 Simulated NH_4^+, DO and BOD concentrations in the IMB compared to measurements (*dots*) from NWSC, and critical spatial distribution of NH_4^+, DO and BOD within the IMB during the simulation period

1951 to 2.8 km^2 in 1991 and 0.69 km^2 in 2007 (Kansiime et al. 2007).

The rapid increase in BOD after 2010 could also be linked to other anthropogenic, institutional and political factors surrounding the IMB water quality. Diffuse pollution in the model was considered to enter the IMB through streams flowing into it; yet in reality, a large proportion of both organic and inorganic wastes

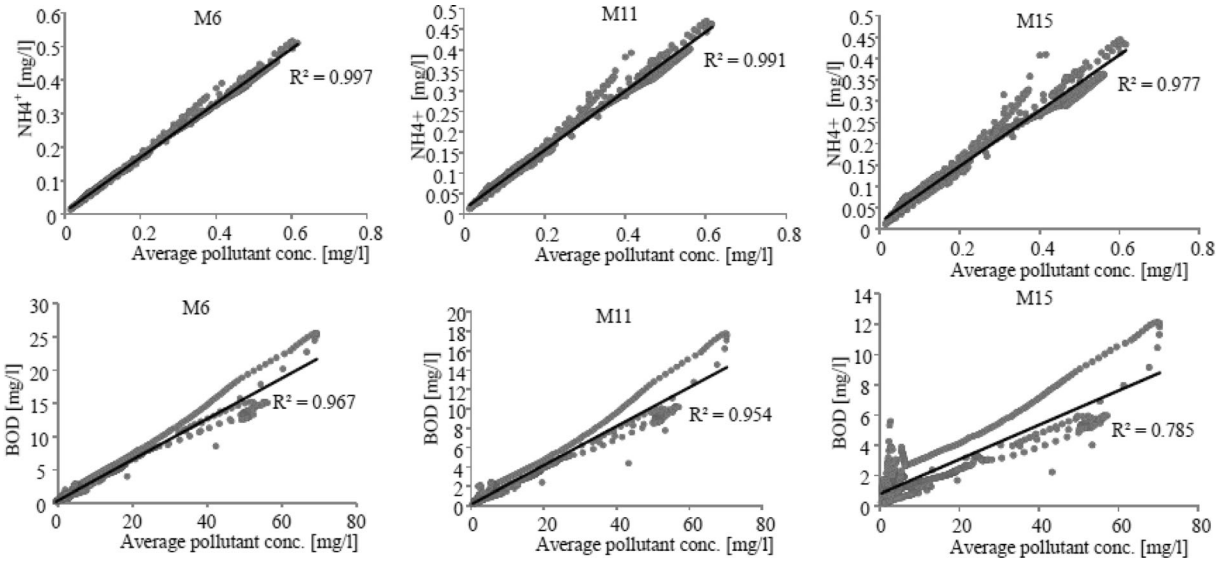

Fig. 10 Linear regression along the IMB transect for NH_4^+ and BOD at M6, M11 and M15 against average pollutant concentration into the IMB

remained uncollected. KCCA, whose mandate includes cleanliness, urban drainage and waste disposal for the Kampala City, was inaugurated in 2010 (KCCA 2010). More areas of the city became paved, and several drainage canals were constructed afterwards. This suggests that pollution in the IMB due to the ongoing wetland encroachment and increase in the pollutant loading in the streams flowing into the IMB increased more rapidly after 2010.

Water quality model robustness

To further illustrate the impact of hydrodynamic conditions and model inputs, linear regressions and goodness-of-fit analysis along a transect downstream of the IMB (M6, M9, M11, M15 and M19) were done. This analysis compared BOD and NH_4^+ concentrations with velocities, water depth, boundary conditions and average pollutant loading at the IMB shore as shown in Fig. 10 and Table 4.

More than 90% of the variance in the water quality model of the IMB is explained by the pollutant input into the IMB (Fig. 10). Model dependence on pollutant input reduces towards the IMB exit for both BOD and NH_4^+ concentrations. This decline is larger for BOD concentrations downstream, signifying accumulation of organic matter in the IMB upstream of M15. Less dispersion is observed for NH_4^+ compared to BOD concentrations along the transect, probably because nitrification occurs under aerobic conditions which is not often the case in the IMB. Nitrifying bacteria, *Nitrosomonas* and *Nitrobacter*,

Table 4 Coefficient of determination along the IMB transect for NH_4^+ and BOD based on velocity, water depth, boundary conditions and pollutants into the IMB.

Transect	Velocity		Water depth		Boundary conditions		Pollutants[1]	
	NH_4^+	BOD	NH_4^+	BOD	NH_4^+	BOD	NH_4^+	BOD
M6	0.175	0.072	0.028	0.005	0.037	0.021	*0.997*	*0.967*
M9	0.111	0.027	0.028	0.005	0.042	0.019	*0.994*	*0.964*
M11	0.152	0.054	0.027	0.005	0.047	0.016	*0.990*	*0.954*
M15	0.106	0.028	0.024	0.003	0.060	0.004	*0.977*	*0.785*
M19	0.096	0.067	0.021	0.002	0.078	0.009	*0.954*	*0.039*

[1] The highest correlation of the water quality parameters was achieved with pollutants at the IMB shores.

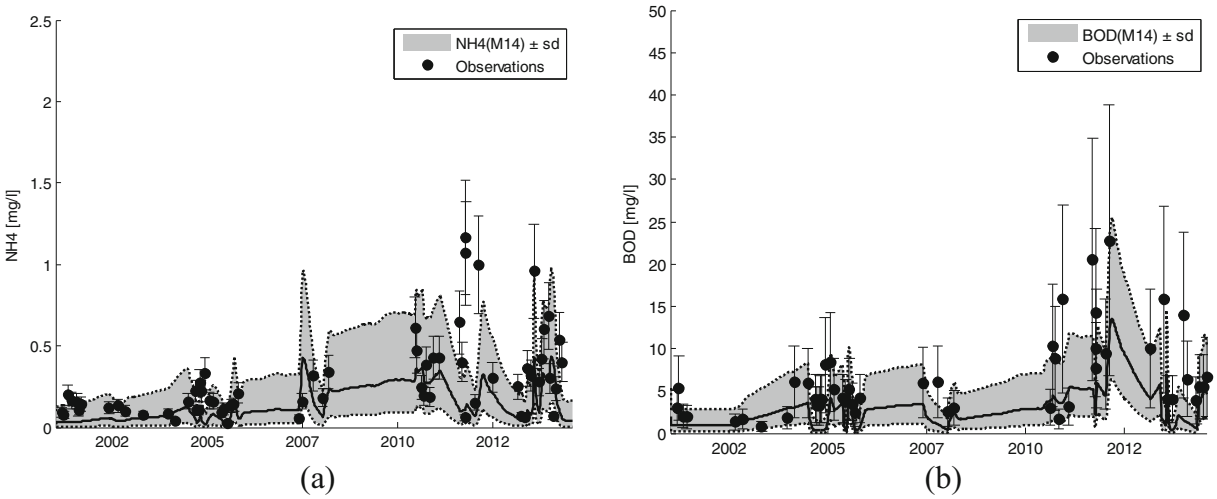

Fig. 11 Measurement uncertainty and bounds based on one times the standard deviation (after Box–Cox transformation of the concentrations) for modelled simulations for **a** NH_4^+ and **b** BOD concentrations at the IMB exit for point M14

are generally not able to compete with heterotrophic bacteria in high BOD waters. Buswell et al. (1954) showed that rate of growth of *Nitrosomonas* decreased with increase in organic matter and slowed during longer periods of storage, which is similar in our case of the IMB.

Further decomposition of the average pollutant dependency from different hotpots showed that M2 had the highest effect on pollutant concentration at the IMB exit, probably linked to the direction and flow of water at M2. Although inputs such as depth of the water column and imposed boundary conditions downstream of the IMB show very low correlation coefficients (Table 4), the effect of velocity due to water level variations cannot be neglected. Movement of water, both in magnitude and direction, is indeed an important factor in determining the quality of water in the IMB especially towards the shores.

The quantified uncertainty in the water quality modelling may be linked to a variety of errors sprouting from both the hydrodynamic and water quality models. Water quality modelling errors are attributed to uncertainty in the model structure, measurement errors, parameters, model input and missing data (Beck 1987, 2013; Radwan et al. 2004; Willems 2008). Radwan and Willems (2008) estimated measurement uncertainty for different parameters in the River Nile and classified the mean relative error range for NH_4^+ at 20–40% and BOD at 40–100%. Relying on these measurement error estimates, the total uncertainty in the water quality model results is quantified based on the model residual errors, taking the measurement error estimates into account by variance decomposition (Willems 2008). It is considered that these errors follow a normal distribution after Box–Cox transformation, hence providing confidence interval bounds to the model water quality results (Fig. 11). These uncertainty estimates provide important additional information next to the deterministic model outputs in support of risk-based water management, planning and decision making that would rely on simulations with the model developed in this study.

Conclusions

This study provided an insight in the long-term IMB water quality concentration variations and developed a plausible water quality model for the Bay. The conclusions on the attribution of the concentration variations were obtained after combining observations with simulation results from the Delwaq DO–BOD–NH_4^+ model. This model accounts for the *wetland effect* around the IMB by considering concentrations at the hotspots located downstream of the major IMB streams.

Increase in nutrient and organic concentrations (NH_4^+ and BOD) in the IMB in the period 2001–2014 is directly linked to increased pollutant loading into the IMB through streams. The impact of diffuse loadings at the IMB exit was especially higher after 2011, when the *wetland effect* became negligible and more diffuse pollution entered the Bay. The increase in diffuse pollution coupled with the long residence time in the IMB enhances pollutant accumulation and biodegradation of organic matter. The high

residence time at the shores, especially at the M1, M3 and M4 hotspots, leads to accumulation of wastes in the IMB. The water quality in the IMB shores was worse than that in the streams upstream.

The IMB model shows that dilution accounts for less than 60% of the pollutant reduction at the IMB exit. The lake level variations affect the water quality by altering IMB velocities and water depths/volumes. The decrease in BOD at the IMB exit after 2011 could be attributed to an increase in water levels in the IMB that improved the DO concentrations in the IMB and improvement of waste collection in Kampala City. Although effects of dilution by lake level changes are negligible compared to pollution loadings, advection of pollutant waste loads facilitated by the IMB flows greatly impacts on the water quality especially at the northern shores.

Further improvements of this study may include incorporating sediment oxygen demand in the model to improve water quality simulations and extending water level and wind measurements. This model can be applied as a decision tool or applied to study impacts of IMB management practices and external driving forces such as climate change. It can also be used in forecasting and hindcasting water quality in the IMB despite the sparse temporal measurements. This is vital in establishing environmental policies to abate environmental pollution and water quality management.

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References

Akurut, M., Niwagaba, C. B., & Willems, P. (2014a). Potential impacts of climate change on precipitation over Lake Victoria, East Africa, in the 21st Century. *Water, 6 (Water Resources in a Variable and Changing Climate)* (pp. 2634–2659). doi:10.3390/w6092634.

Akurut, M., Willems, P., & Niwagaba, C. B. (2014b). Assessing the influence of Lake Victoria flux on the Inner Murchison Bay water quality. In C. Brebbia (Ed.), *Water Pollution XII, Algarve, Portugal* (vol. 182, pp. 51–62). WIT Transactions on Ecology and The Environment, WIT Press. doi:10.2495/WP140051.

Awange, J. L., Ogalo, L., Bae, K.-H., Were, P., Omondi, P., Omute, P., et al. (2008). Falling Lake Victoria water levels: is climate a contributing factor? *Climatic Change, 89*, 281–297. doi:10.1007/s10584-008-9409-x.

Banadda, E. N., Kansiime, F., Kigobe, M., Kizza, M., & Nhapi, I. (2009). Landuse-based nonpoint source pollution: a threat to water quality in Murchison Bay, Uganda. *Water Policy, 11*, 93–104. doi:10.2166/wp.2009.106.

Beck, M. B. (1987). Water quality modeling: a review of the analysis of uncertainty. *Water Resources Research, 23*, 1393–1442. doi:10.1029/WR023i008p01393.

Beck, M. B. (2013). *Water quality management: a review of the development and application of mathematical models* (vol. 11). Springer Science & Business Media.

Berton, R., Driscoll, C. T., & Chandler, D. G. (2016). Changing climate increases discharge and attenuates its seasonal distribution in the northeastern United States. *Journal of Hydrology: Regional Studies, 5*, 164–178. doi:10.1016/j.ejrh.2015.12.057.

Bracchini, L., Loiselle, S. A., Tognazzi, A., Dattilo, A. M., Focardi, S., Cózar, A., et al. (2007). The optical qualities of shallow wetland lined bays in Lake Victoria. *Wetlands Ecology and Management, 15*, 509–519. doi:10.1007/s11273-007-9050-x.

Buswell, A., Shiota, T., Lawrence, N., & Van Meter, I. (1954). Laboratory studies on the kinetics of the growth of nitrosomonas with relation to the nitrification phase of the BOD test. *Applied Microbiology, 2*(1), 21.

Chapra, S. C. (1997). *Surface water-quality modeling*. Illinois: Tufts University: Waveland Press, Inc. Long Grove.

Cózar, A., Bergamino, N., Mazzuoli, S., Azza, N., Bracchini, L., Dattilo, A., et al. (2007). Relationships between wetland ecotones and inshore water quality in the Ugandan coast of Lake Victoria. *Wetlands Ecology and Management, 15*, 499–507. doi:10.1007/s11273-007-9046-6.

Delft3D-Flow (2013). *Simulation of multi-dimensional hydrodynamic flows and transport phenomena, including sediments—User Manual: Deltares, Rotterdamsesweg 185, 2600 MH - Delft*, Netherlands.

D-Water Quality (2013). *Versatile water quality modelling in 1D, 2D or 3D systems including physical, (bio)chemical and biological processes—User Manual: Deltares, Rotterdamsesweg 185, 2600 MH - Delft*, Netherlands.

Gikuma-Njuru, P. (2008). *Physical and biogeochemical gradients and exchange processes in Nyanza Gulf and main Lake Victoria (East Africa)*. Ph.D, University of Waterloo, Waterloo.

Gikuma-Njuru, P., & Hecky, R. (2005). Nutrient concentrations in Nyanza Gulf, Lake Victoria, Kenya: light limits algal demand and abundance. *Hydrobiologia, 534*(1–3), 131–140.

Gikuma-Njuru, P., Hecky, R., Guildford, S., & MacIntyre, S. (2013). Spatial variability of nutrient concentrations, fluxes, and ecosystem metabolism in Nyanza Gulf and Rusinga Channel, Lake Victoria (East Africa). *Limnology and Oceanography, 58*(3), 774–789.

Haande, S., Rohrlack, T., Semyalo, R. P., Brettum, P., Edvardsen, B., Lyche-Solheim, A., et al. (2011). Phytoplankton dynamics and cyanobacterial dominance in Murchison Bay of Lake Victoria (Uganda) in relation to environmental conditions. *Limnologica - Ecology and Management of Inland Waters, 41*, 20–29. doi:10.1016/j.limno.2010.04.001.

Hecky, R., Bugenyi, F., Ochumba, P., Talling, J., Mugidde, R., Gophen, M., et al. (1994). Deoxygenation of the deep water of Lake Victoria, East Africa. *Limnology and Oceanography, 39*(6), 1476–1481.

Hecky, R., Mugidde, R., Ramlal, P., Talbot, M., & Kling, G. (2010). Multiple stressors cause rapid ecosystem change in Lake Victoria. *Freshwater Biology, 55*(s1), 19–42.

Janse, J. H. (1997). A model of nutrient dynamics in shallow lakes in relation to multiple stable states. *Hydrobiologia, 342*, 1–8.

Kansiime, F., & Nalubega, M. (1999). *Wastewater treatment by a natural wetland: the Nakivubo Swamp, Uganda*. Ph.D, UNESCO-IHE, Institute for Water Education, Rotterdam.

Kansiime, F., Kateyo, E., Oryem-Origa, H., & Mucunguzi, P. (2007). Nutrient status and retention in pristine and disturbed wetlands in Uganda: management implications. *Wetlands Ecology and Management, 15*, 453–467. doi:10.1007/s11273-007-9054-6.

Kayima, J., Kyakula, M., Komakech, W., & Echimu, S. P. (2008). A study of the degree of pollution in Nakivubo Channel, Kampala. *Uganda. Appl. Sci. Environ. Manage., 12*, 93–98.

KCCA (2010). *The Kampala Capital City Act, 2010* (pp. 1–43). Uganda: Uganda Online Law Library.

Kendall, M. G. (1948). *Rank correlation methods*. Oxford: Griffin.

Khisa, P., Romero, J. R., Imberger, J., Ewing, T., Antenucci, J., Njuguna, H., et al. (2006). The effect of the Mbita Causeway on water currents in the region of Rusinga Channel, Winam Gulf, Lake Victoria: a 3 D modelling study with ELCOM. In *Proceedings of the 11 th World Lakes Conference—Proceedings* (vol. 2, pp. 74–78)

Luyiga, S., Haande, S., Semyalo, R. P., Kizito, Y. S., Miyingo-Kezimbira, A., Brettum, P., et al. (2015). How water exchange and seasonality affect the eutrophication of Murchison Bay, Lake Victoria. *Limnologica-Ecology and Management of Inland Waters, 53*, 60–73.

Machiwa, P. K. (2003). Water quality management and sustainability: the experience of Lake Victoria Environmental Management Project (LVEMP)—Tanzania. *Physics and Chemistry of the Earth, Parts A/B/C, 28*(20), 1111–1115.

Machiwal, D., & Jha, M. K. (2015). Identifying sources of groundwater contamination in a hard-rock aquifer system using multivariate statistical analyses and GIS-based geostatistical modeling techniques. *Journal of Hydrology: Regional Studies, 4*, 80–110.

Morana, C., Sarmento, H., Descy, J.-P., Gasol, J. M., Borges, A. V., Bouillon, S., et al. (2014). Production of dissolved organic matter by phytoplankton and its uptake by heterotrophic prokaryotes in large tropical lakes. *Limnology and Oceanography, 59*(4), 1364–1375.

Naddafi, K., Honari, H., & Ahmadi, M. (2007). Water quality trend analysis for the Karoon River in Iran. *Environmental Monitoring and Assessment, 134*(1–3), 305–312.

Nielsen, A., Trolle, D., Bjerring, R., Søndergaard, M., Olesen, J. E., Janse, J. H., et al. (2014). Effects of climate and nutrient load on the water quality of shallow lakes assessed through ensemble runs by PCLake. *Ecological Applications, 24*(8), 1926–1944.

NWSC (2015). *Kampala Water—Lake Victoria Watsan Project. (vol. package 4A: feasibility studies and water quality monitoring for the development of a new water treatment plant east of Kampala, pp. 138)*. Kampala: National Water & Sewerage Corporation.

Pinardi, M., Fenocchi, A., Giardino, C., Sibilla, S., Bartoli, M., & Bresciani, M. (2015). Assessing potential algal blooms in a shallow fluvial lake by combining hydrodynamic modelling and remote-sensed images. *Water, 7*(5), 1921–1942.

Radwan, M., & Willems, P. (2008). Uncertainty analysis for river quality measurements. *International Journal of Ecological Economics & Statistics (IJEES), 11*, 21–30.

Radwan, M., Willems, P., & Berlamont, J. (2004). Sensitivity and uncertainty analysis for river quality modelling. *Journal of Hydroinformatics*, 83–99.

Rueda, F. J., & MacIntyre, S. (2010). Modelling the fate and transport of negatively buoyant storm–river water in small multi-basin lakes. *Environmental Modelling & Software, 25*(1), 146–157.

Scheren, P., Zanting, H., & Lemmens, A. (2000). Estimation of water pollution sources in Lake Victoria, East Africa: application and elaboration of the rapid assessment methodology. *Journal of Environmental Management, 58*(4), 235–248.

Sekabira, K., Oryem, O. H., Basamba, T. A., Mutumba, G., & Kakudidi, E. (2010). Assessment of heavy metal pollution in the urban stream sediments and its tributaries. *International Journal of Environmental Science & Technology, 7*, 435–446.

Semyalo, R., Nattabi, J. K., & Larsson, P. (2009). Diel vertical migration of zooplankton in a eutrophic bay of Lake Victoria. *Hydrobiologia, 635*(1), 383–394.

Sen, P. K. (1968). Estimates of the regression coefficient based on Kendall's tau. *Journal of the American Statistical Association, 63*(324), 1379–1389.

Ssebiyonga, N., Erga, S. R., Hamre, B., Stamnes, J. J., & Frette, Ø. (2013). Light conditions and photosynthetic efficiency of phytoplankton in Murchison Bay, Lake Victoria, Uganda. *Limnologica-Ecology and Management of Inland Waters, 43*(3), 185–193.

Swenson, S., & Wahr, J. (2009). Monitoring the water balance of Lake Victoria, East Africa, from space. *Journal of Hydrology, 370*, 163–176. doi:10.1016/j.jhydrol.2009.03.008.

Tabari, H., Marofi, S., & Ahmadi, M. (2011). Long-term variations of water quality parameters in the Maroon River, Iran. *Environmental Monitoring and Assessment, 177*(1–4), 273–287.

Tamatamah, R. A., Hecky, R. E., & Duthie, H. (2005). The atmospheric deposition of phosphorus in Lake Victoria (East Africa). *Biogeochemistry, 73*, 325–344. doi:10.1007/s10533-004-0196-9.

USDA (2015). USDA. http://www.pecad.fas.usda.gov/lakes/images/lake0314.TPJO.2.txt. Accessed 12 May 2015.

van Gerven, L., Brederveld, R. J., de Klein, J. J., DeAngelis, D. L., Downing, A. S., Faber, M., et al. (2015). Advantages of concurrent use of multiple software frameworks in water quality modelling using a database approach. *Fundamental and Applied Limnology/Archiv für Hydrobiologie, 186*(1–2), 5–20.

Vermeiren, K., Van Rompaey, A., Loopmans, M., Serwajja, E., & Mukwaya, P. (2012). Urban growth of Kampala, Uganda: pattern analysis and scenario development. *Landscape and Urban Planning, 106*(2), 199–206.

Willems, P. (2008). Quantification and relative comparison of different types of uncertainties in sewer water quality modeling. *Water Research, 42*(13), 3539–3551. doi:10.1016/j.watres.2008.05.006.