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# Polychlorinated biphenyls in sediments and fish species from the Murchison Bay of Lake Victoria, Uganda

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## HIGHLIGHTS

- High concentrations of PCBs were found at sites near wastewater discharges.
- The PCB concentrations were low to moderate compared to other locations worldwide.
- Based on the European Commission set TEQ, the fish were fit for human consumption.

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## ABSTRACT

Polychlorinated biphenyls (PCBs) were determined in sediments and two fish species collected from the Murchison Bay in Lake Victoria, using high resolution gas chromatography coupled to a high resolution mass spectrometer. Total PCB concentrations ($\Sigma_{18}$PCBs) varied widely with mean values ranging from 777 to 4325 pg $g^{-1}$ dry weight (dw) for sediments and 80 to 779 pg $g^{-1}$ wet weight (ww) for fish. The PCB levels in the sediments were significantly higher at the station closest to Nakivubo channel, presumably due to effluents discharged by the channel, which may contain domestically produced commercial PCB mixtures. For fish, the concentrations in Nile perch (*Lates niloticus*) were significantly greater than those in Nile tilapia (*Oreochromis niloticus*) at all study stations, possibly due to dietary differences among species. World Health Organization-toxic equivalents ($WHO_{2005}$-TEQs) for the dioxin-like PCBs were 0.04–0.64 pg $g^{-1}$ dw and 0.01–0.39 pg $g^{-1}$ ww for sediments and fish, respectively. The non-*ortho* PCBs exhibited the highest contribution to the $\Sigma_{12}$TEQs (>75%) compared to the mono-*ortho* PCBs in both fish species. The TEQs in the present study were lower than many reported worldwide in literature for fish and were within the permissible level recommended by the European Commission, implying that the fish did not pose health hazards related to PCBs to the consumers.



## 1. Introduction

Lake Victoria is a vital natural aquatic resource in Uganda, but its ecological health is being threatened, mainly by rapid urbanization and industrialization. These developments coupled with a population of over one million people in its catchment area have resulted in increased pollution (Banadda et al., 2009, 2011a, 2011b; Wasswa et al., 2010). Polychlorinated biphenyls (PCBs) have previously been documented in the lake (Ssebugere et al. 2013a, b; Ssebugere et al., 2014). The PCBs are released into the environment as a result of industrial applications such as capacitors, hydraulic fluids in electronic equipment, and coolants and lubricants in transformers (Gidarakos et al., 2009; Moon et al., 2012; Lundin et al., 2013; Miller et al., 2013).

Exposure to PCBs has been associated with diseases like diabetes mellitus, cancer and hypertension (Lee et al., 2007a). Other adverse effects caused by the pollutants include endocrine disruption, neurobehavioral deficits and fetal toxicity (Lee et al., 2007b; Ha et al., 2007; 2009; Uemura et al., 2009). The PCBs undergo long-range transport causing global pollution (Wiberg et al., 2009; Sundqvist et al., 2009a, b). Furthermore, the contaminants can accumulate in sediments close to point sources and in benthic organisms and fish due to their hydrophobic nature, low metabolic transformation and long half-lives (Suedel et al., 1994; Jönsson et al., 2003; Åberg et al., 2008; 2010; Lavandier et al., 2013). Fish can be an important monitoring tool, because they concentrate PCBs directly from water through their diet, and indicate hazards to piscivorous predators and humans (Van

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der Oost et al., 2003; Huang et al., 2006; Gustavson et al., 2008; Parnell et al., 2008).

To our knowledge whereas PCBs have been reported in fish and sediments from the Napoleon Gulf and Thurston Bay of Lake Victoria (Ssebugere et al., 2013a, b; Ssebugere et al., 2014) and from other water bodies in the world (Manirakiza et al., 2002; Adu-Kumi et al., 2010; Darko et al., 2008; Helm et al., 2008; Eqani et al., 2013; Verhaert et al., 2013), literature shows no work about these pollutants in the Murchison Bay. The Bay is a major hotspot for discharge of industrial outflow, urban runoff and municipal effluents from Kampala, Uganda's capital city (Banadda et al., 2009). The objectives of this study were to investigate the occurrence and contamination levels of PCBs in sediments and fish from the Murchison Bay.

## 2. Materials and methods

### 2.1. Study area

The area of study was the Murchison Bay located in the northwestern part of Lake Victoria and southeast of Kampala city (Fig. 1). The Bay covers an area of 60 $km^2$ and includes semi-enclosed inner and outer parts. The Murchison Bay is the main recipient of sewage effluents, industrial- and municipal-waste from the central district of Kampala City, via the Nakivubo channel. The channel passes through dense residential settlements and commercial areas. In the past, the channel ended in wetland areas, allowing wastewater to be drained into papyrus swamps before entering the Bay. However, in recent years the wetlands which played the role of secondary treatment have been degraded due to settlement and commercial activities (Wasswa, 2009). The water turbidity in the Bay has increased while the water transparency has declined due to excessive waste input.

### 2.2. Sample collection

A total of 24 surface sediments (<25 cm depth) were taken using a sediment corer from four stations namely 1 and 2 in the inner Bay, and 3 and 4 in the outer Bay (Fig. 1) in March, 2013. Six sediments were randomly taken from each station at distances of approximately 200 m from one another. The geographical coordinates of the inner and outer parts of the Murchison Bay are 00°15′72″N, 32°38′74″E and 00°08′71″N, 32°37′58″E, respectively. Ninety six fish of different species namely: Nile Perch (*Lates niloticus*) a carnivorous top-predator and Nile Tilapia (*Oreochromis niloticus*) a detritivorous species were collected using gill nets within the same locality as the sediments. These two species exhibit different feeding habitats and thus may be exposed differently to contamination through trophic transfer. The weights of *O. niloticus* ranged from 289 to 652 g (mean 389 g) while those of *L. niloticus* varied from 1628 to 2979 g (mean 2378 g). The mean lengths (± standard deviation) were 29 ± 3 cm, and 68 ± 5 cm, respectively. The muscle tissues were dissected between the pectoral fin and vent of the fish, minced into pieces and subsamples taken. Subsamples of 4 fish from the same location and, of similar length and species were pooled and homogenized. The sediments and homogenized fish were transferred into acetone rinsed glass bottles and shipped to the Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences where they were freeze-dried.

### 2.3. Analytical method

Sample extraction and cleanup procedures followed the method reported by Wang et al. (2013). Briefly, 2 g of the freeze-dried sample (fish or sediment) was ground with 10 g of anhydrous sodium sulfate to a free floating powder using a mortar and pestle. The mixture was spiked with $^{13}C$-labeled surrogate standards (US EPA defined 68A-LCS) and extracted on an accelerated solvent extractor device (ASE 300, Dionex, USA) using 1:1 v/v mixture of *n*-hexane/dichloromethane (DCM). The resultant extract was concentrated to 2 mL on a rotary evaporator. The concentrated extract was divided into two subsamples for fish (0.3 and 1.7 mL). The 0.3 mL was used for gravimetric determination of the lipid content while the 1.7 mL was kept for clean-up. For the sediments, activated copper granules were added to the extract to remove elemental sulfur before purification with column chromatography. The extracts were then cleaned up with a multilayered silica column (packed from bottom to top with 1 g silica gel, 4 g basic silica gel, 1 g silica gel, 8 g acid silica gel, 2 g silica gel and 2 g anhydrous sodium sulfate), an alumina column (6 g basic alumina and 3 g anhydrous sodium sulfate

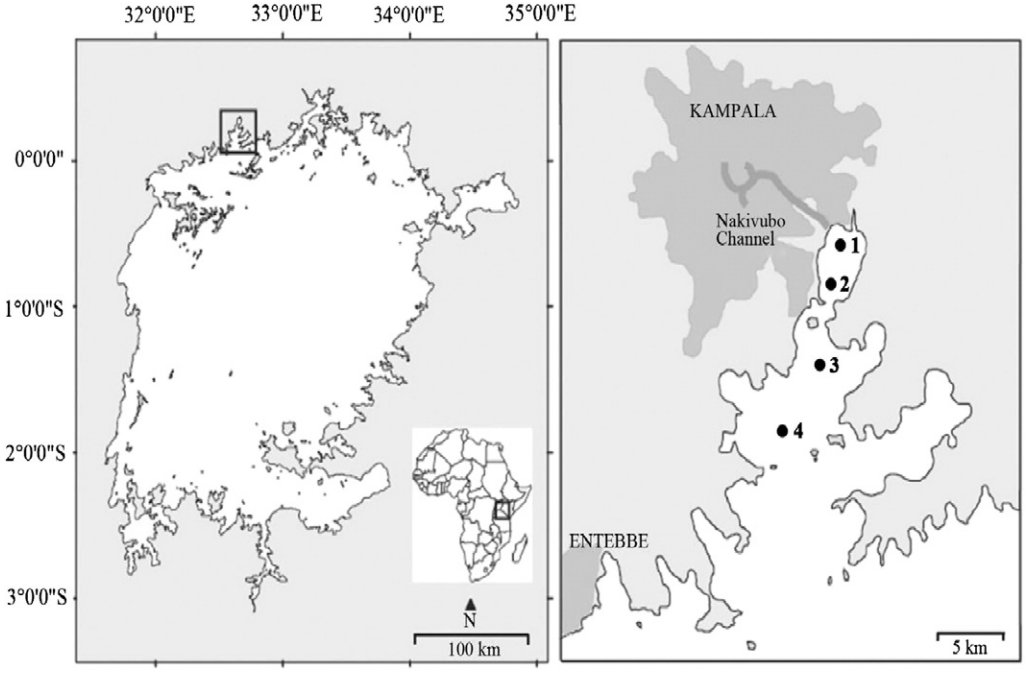


**Fig. 1.** Map of the Murchison Bay showing the study stations.
Adapted from Haande et al. (2011).

**Table 1**
Mean concentrations and ranges (pg $g^{-1}$ dry weight) of PCBs in sediments from the Murchison Bay.

| Congener | Station 1 | Station 2 | Station 3 | Station 4 |
|---|---|---|---|---|
| CB 28 | 375 ± 25 | 318 ± 30 | 103 ± 54 | 63 ± 21 |
| | 353–402 | 284–340 | 51–158 | 45–85 |
| CB 52 | 507 ± 99 | 293 ± 31 | 169 ± 14 | 108 ± 44 |
| | 397–589 | 259–319 | 153–179 | 58–137 |
| CB 101 | 585 ± 234 | 343 ± 176 | 158 ± 68 | 122 ± 41 |
| | 420–853 | 229–546 | 105–235 | 75–150 |
| CB 138 | 933 ± 446 | 553 ± 396 | 240 ± 131 | 153 ± 87 |
| | 631–1445 | 230–1009 | 151–390 | 79–249 |
| CB 153 | 898 ± 240 | 585 ± 465 | 227 ± 90 | 177 ± 85 |
| | 721–1171 | 251–1115 | 173–331 | 110–272 |
| CB 180 | 218 ± 63 | 117 ± 54 | 49 ± 23 | 29 ± 6.1 |
| | 152–276 | 61–169 | 26–72 | 23–35 |
| Σ indicator-PCBs | 3516 | 2209 | 946 | 652 |
| | 2749–4652 | 1464–3488 | 719–1282 | 409–921 |
| CB 77 | 82 ± 20 | 56 ± 6.3 | 12 ± 4.5 | 8.8 ± 1.2 |
| | 60–99 | 52–64 | 6.3–14 | 8.1–10 |
| CB 81 | 1.9 ± 1.1 | 2.5 ± 0.9 | 1.0 ± 0.6 | 0.8 ± 0.3 |
| | 0.7–2.8 | 1.6–3.3 | 0.6–1.4 | 0.6–1.0 |
| CB 126 | 4.9 ± 1.4 | 2.2 ± 0.7 | 1.2 | 0.9 ± 0.5 |
| | 3.3–5.8 | 1.7–3.0 | n.d.–1.2 | n.d.–1.2 |
| CB 169 | n.d. | 0.3 | n.d. | 0.1 |
| | | n.d.–0.3 | | n.d.–0.1 |
| Σ non-*ortho*-PCBs | 89 | 61 | 14 | 11 ± 2.0 |
| | 66–105 | 55–70 | 6.8–17 | 8.9–13 |
| CB 105 | 140 ± 70 | 59 ± 18 | 26 ± 10 | 22 ± 13 |
| | 99–221 | 47–79 | 19–38 | 11–36 |
| CB 114 | 6.7 ± 3.3 | 2.9 ± 0.6 | 1.9 ± 0.6 | 1.2 ± 0.6 |
| | 4.4–11 | 2.3–3.4 | 1.5–2.6 | 0.7–1.9 |
| CB 118 | 434 ± 214 | 174 ± 74 | 110 ± 59 | 72 ± 45 |
| | 302–682 | 139–228 | 71–177 | 32–120 |
| CB 123 | 53 ± 19 | 24 ± 7.4 | 13 ± 5.3 | 7.4 ± 6.2 |
| | 42–76 | 18–32 | 8.7–19 | 1.0–13 |
| CB 156 | 54 ± 28 | 28 ± 14 | 14 ± 5.1 | 7.5 ± 3.3 |
| | 36–86 | 16–43 | 9.2–19 | 4.9–11 |
| CB 157 | 7.7 ± 5.2 | 3.4 ± 1.3 | 2.0 ± 1.0 | 1.3 ± 0.7 |
| | 4.2–14 | 2.2–4.8 | 1.3–3.1 | 0.8–2.1 |
| CB 167 | 16 ± 7.9 | 8.9 ± 5.0 | 4.4 ± 1.1 | 1.9 ± 0.4 |
| | 11–26 | 4.6–14 | 3.1–5.2 | 1.5–2.2 |
| CB 189 | 8.0 ± 1.9 | 5.5 ± 2.7 | 2.7 ± 1.3 | 1.1 ± 0.4 |
| | 6.6–10 | 3.0–8.4 | 1.3–3.9 | 0.8–1.6 |
| Σ mono-*ortho*-PCBs | 720 | 306 | 174 | 114 |
| | 508–1124 | 231–413 | 124–266 | 53–189 |
| $\Sigma_{12}$dl-PCBs | 809 | 367 | 188 | 125 |
| | 613–1190 | 289–482 | 131–283 | 62–202 |
| $\Sigma_{18}$PCBs | 4325 | 2576 | 1134 | 777 |
| | 3762–5842 | 1753–3970 | 850–1365 | 471–1123 |
| $WHO_{2005}$-TEQs | 0.55 | 0.25 | 0.11 | 0.10 |
| | 0.37–0.64 | 0.19–0.33 | 0.04–0.12 | 0.07–0.14 |

The number of sediment samples per station (N) = 6; n.d. — non-detectable.

at the top) and carbon column (1.5 g of 18% carbon dispersed in cilite, 3 g anhydrous sodium sulfate at the top), respectively. The clean eluate was concentrated to 1 mL using a rotary evaporator and transferred into vials containing 20 μL nonane. The contents in the vials were further dried to 25 μL using a gentle stream of nitrogen and later spiked with injection standards (US EPA defined 68A-IS) prior to analysis.

### 2.4. Instrumental analysis

The analysis of 18 PCB congeners including 6 indicator PCBs (IUPAC numbers 28, 52, 101, 138, 153 and 180) and twelve dioxin-like PCBs (IUPAC numbers 77, 81, 126, 169, 105, 114, 118, 123, 156, 157, 167 and 189) was done using high-resolution gas chromatography coupled with high-resolution mass spectrometry (HRGC/HRMS) (AutoSpec Ultima, Waters, USA). Chromatographic separation was achieved by injecting 1 μL of sample on a fused silica capillary column (DB5MS, 60 m × 0.25 mm i.d. × 0.25 μm film thicknesses). The oven temperature of the GC was held at 120 °C for 1 min, then increased to 150 °C at a rate of 30 °C $min^{-1}$ and finally to 300 °C at 2.5 °C $min^{-1}$. Helium was used as a carrier gas at a flow rate of 1.0 mL $min^{-1}$. The HRMS was operated in VSIR mode at a resolution of ≥10000 under positive electron ionization conditions (35 eV) and a source temperature of 270 °C.

### 2.5. Quality assurance and quality control

To ensure the quality of data, surrogate PCB standards, blanks, replicates and a certified reference material (CRM) were included in the analysis. Recoveries for majority of the surrogate standards met the requirements of US EPA methods 1668A (were in the range of 75–127%) except for CBs 28, 118, and 138 whose recoveries were <75% (range: 56–74%). The recoveries resulting from triplicate determinations (N = 3) of the certified reference material were between 82% and 106%. The limit of detection (LOD) which was calculated as three times the signal-to-noise ratio, varied from 0.04 to 0.96 pg $g^{-1}$ dry weight (dw) for sediments and 0.02 to 0.80 pg $g^{-1}$ wet weight (ww) for fish. PCBs in the field and procedural blanks were detected at <15% of the concentration in each batch of 12 samples, so the reported data were not blank corrected.

### 2.6. Statistical data analysis

Statistical analysis was performed using Statistica® 7 for Windows. Kolmogorov–Smirnov test was used for verifying the normality of data. Since concentration values were not normally distributed, non-parametric statistics was applied. The Mann–Whitney *U*-test was used for pair wise comparison and the Kruskal–Wallis test when more than two groups were considered. In these two non-parametric tests, statistical significance was set at $p < 0.05$. In the present study, toxic

**Table 2**
Comparison of sediment PCB concentrations (pg $g^{-1}$ dry weight) in this study with those from other locations.

| Area | PCBs (pg $g^{-1}$ dry weight) | References |
|---|---|---|
| Napoleon Gulf and Thurston Bay of Lake Victoria, Uganda | 64–1262 (18 congeners) | Ssebugere et al. (2013b) and Ssebugere et al. (2014) |
| Nile River, Egypt | 1461–2244 (18 congeners) | El-Kady et al. (2007) |
| Pangani River basin, Tanzania | 357–11000 (28 congeners) | Hellar-Kihampa et al. (2013) |
| Congo River basin, Democratic Republic of Congo | n.d.–1400 (33 congeners) | Verhaert et al. (2013) |
| Ghar El Melh lagoon, Tunisia | n.d.–3987 (20 congeners) | Ameur et al. (2011) |
| Mekong River delta, Vietnam | 110–2000 (13 congeners) | Carvalho et al. (2008) |
| Riverine and coastal waters of Surabaya, Indonesia | n.d.–420000 (62 congeners) | Ilyas et al. (2011) |
| Scheldt River, Belgium | n.d.–200000 (27 congeners) | Covaci et al. (2005) |
| Remote lakes and coastal areas, Norwegian Arctic | 180–13000 (15 congeners) | Jiao et al. (2009) |
| Mersey Estuary, United Kingdom | 36000–1409000 (7 congeners) | Vane et al. (2007) |
| Lake Maggiore, Italy and Switzerland | n.d.–3000 (12 congeners) | Vives et al. (2007) |
| Besòs River, Barcelona | 6–14 (30 congeners) | Castells et al. (2008) |
| Indiana Harbor and Ship Canal, Lake Michigan, United States of America | 53000–35000000 (163 congeners) | Martinez et al. (2010) |
| Sea Lots, Port of Spain, Trinidad and Tobago | 62000–601000 (136 congeners) | Mohammed et al. (2011) |

n.d. — non-detectable.

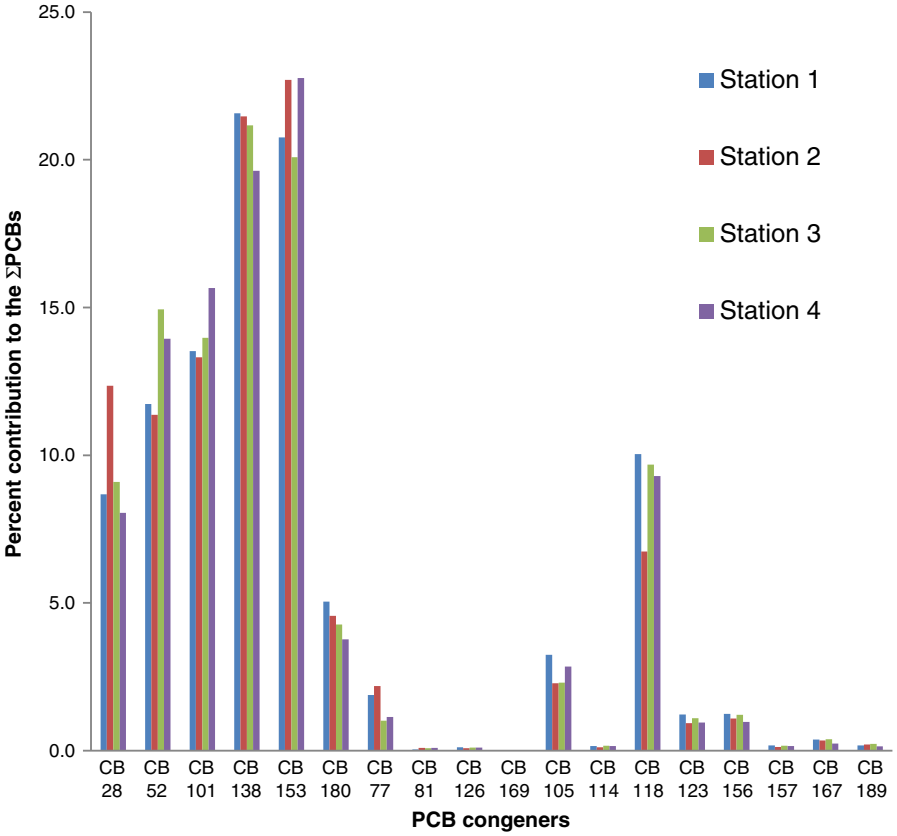


**Fig. 2.** Percent contribution of the congeners to the $\Sigma_{18}$PCBs per station.

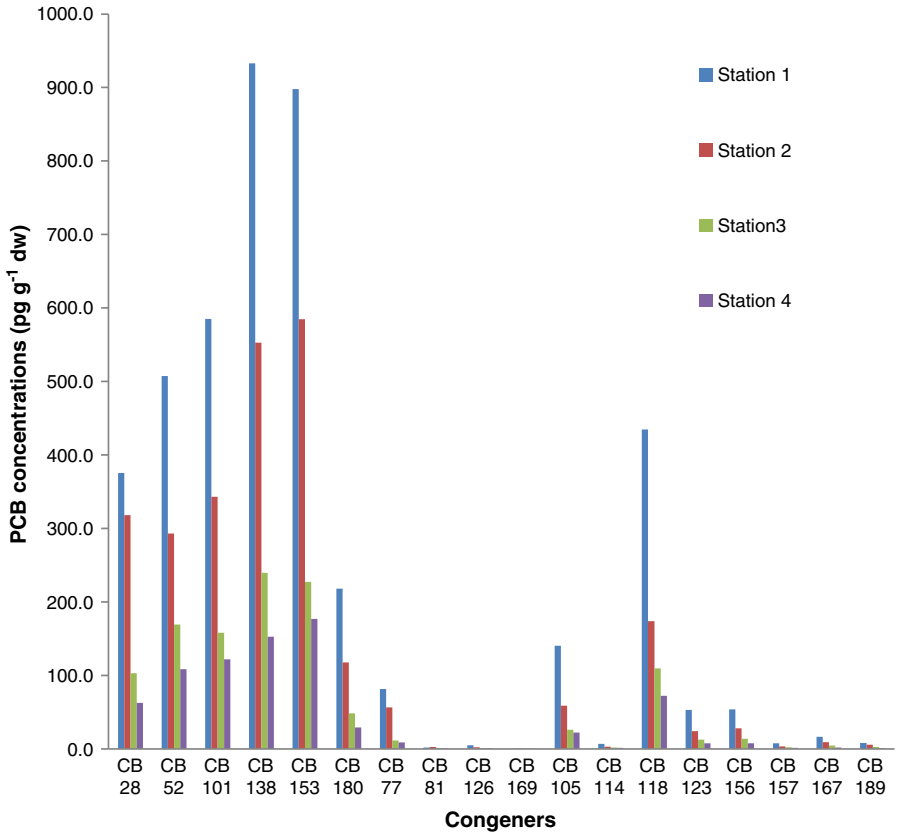


**Fig. 3.** Mean concentrations of the PCB congeners in the sediments per station.

equivalents (TEQs) for the 12 dioxin-like (dl) PCBs were calculated using individual congener concentrations and the 2005 toxic equivalency factors (TEFs) published by the World Health Organization (WHO) (van den Berg et al., 2006).

## 3. Results and discussion

### 3.1. Concentrations of PCBs in sediments from the Murchison Bay

Total PCB concentration ($\Sigma_{18}$PCBs) in Murchison Bay sediments varied widely with mean values ranging from 777 to 4325 pg $g^{-1}$ dw (Table 1). They were 3-fold higher than those reported in sediments from the Napoleon Gulf and Thurston Bay of Lake Victoria (Ssebugere et al., 2013b; Ssebugere et al., 2014). The PCB concentrations in the present study were comparable to data from other fresh water bodies in Africa: the Nile River in Egypt (El-Kady et al., 2007), Ghar El Melh lagoon in Tunisia (Ameur et al., 2011), Pangani River basin in Tanzania (Hellar-Kihampa et al., 2013) and Congo River basin in the Democratic Republic of Congo (Verhaert et al., 2013). The results were also in the range of data found in sediments from tropical areas in Asia: the Mekong River delta in Vietnam (Carvalho et al., 2008) and riverine and coastal waters of Surabaya in Indonesia (Ilyas et al., 2011). The levels were lower compared to concentrations reported for industrialized areas in Europe and United States of America (Covaci et al., 2005; Vane et al., 2007; Vives et al., 2007; Castells et al., 2008; Jiao et al., 2009; Martinez et al., 2010; Mohammed et al., 2011) (Table 2).

The proportion of the 6 indicator PCBs ($\Sigma_6$PCBs) to $\Sigma_{18}$PCBs in this study were between 80 and 88% (with a mean value of 84%). CB 138 and 153 were the predominant congeners, collectively accounting for 21% of the $\Sigma_{18}$PCBs on average (Fig. 2). Barakat et al. (2012) also reported the predominance of CB 138 and CB 153 in surface sediments from Lake Maryut in Egypt. The accumulation of PCBs in the environment is directly linked to their degree of chlorination, stereochemistry and lipophilicity (Mazet et al., 2005). PCBs with a high degree of chlorination

**Table 3**
Mean concentrations and ranges (pg $g^{-1}$ wet weight) of PCBs in fish species from the Murchison Bay.

| | Station 1 | | Station 2 | | Station 3 | | Station 4 | |
|---|---|---|---|---|---|---|---|---|
| | *L. niloticus* | *O. niloticus* | *L. niloticus* | *O. niloticus* | *L. niloticus* | *O. niloticus* | *L. niloticus* | *O. niloticus* |
| Lipid content (%) | 2.8 | 1.5 | 2.3 | 1.4 | 2.2 | 1.1 | 1.6 | 1.2 |
| *Congener* | | | | | | | | |
| CB 28 | 38 ± 22 | 14 ± 7.0 | 62 ± 21 | 13 ± 3.6 | 23 ± 7.5 | 32 ± 14 | 26 ± 15 | 9.3 ± 0.7 |
| | 13–53 | 6.0–19 | 40–83 | 8.4–15 | 18–31 | 12–68 | 10–41 | 8.6–10 |
| CB 52 | 39 ± 23 | 30 ± 24 | 60 ± 52 | 22 ± 18 | 19 ± 8.2 | 13 ± 4.6 | 12 ± 3.2 | 8.1 ± 2.1 |
| | 23–65 | 7.8–55 | 10–128 | 9.0–43 | 10–27 | 10–18 | 9.6–16 | 5.6–9.5 |
| CB 101 | 58 ± 11 | 23 ± 18 | 86 ± 39 | 17 ± 4.9 | 50 ± 25 | 13 ± 2.6 | 21 ± 6.5 | 9.8 |
| | 45–66 | 2.1–37 | 50–128 | 12–22 | 22–65 | 10–15 | 16–28 | 4.9–14 |
| CB 138 | 183 ± 98 | 43 ± 9.9 | 84 ± 38 | 17 ± 13 | 61 ± 31 | 10 ± 1.3 | 62 ± 32 | 18 ± 13 |
| | 72–259 | 32–51 | 52–127 | 7.7–32 | 25–81 | 9.8–12 | 33–96 | 5.9–31 |
| CB 153 | 220 ± 58 | 37 ± 13 | 98 ± 55 | 22 ± 14 | 76 ± 43 | 12 ± 2.1 | 72 ± 15 | 18 ± 12 |
| | 75–386 | 22–49 | 51–158 | 13–39 | 30–112 | 10–14 | 37–133 | 4.9–30 |
| CB 180 | 98 ± 57 | 12 ± 10 | 8.3 ± 2.2 | 3.8 ± 2.5 | 13 ± 6.7 | 3.2 ± 1.4 | 11 ± 6.7 | 4.9 ± 3.0 |
| | 37–151 | 5.1–24 | n.d.–9.8 | 1.1–7.7 | 8–21 | 1.7–4.5 | 6.0–18 | 1.5–7.3 |
| Σ indicator-PCBs | 636 | 159 | 398 | 95 | 242 | 83 | 204 | 68 |
| | 363–904 | 75–228 | 288–573 | 57–124 | 144–307 | 60–122 | 112–322 | 32–100 |
| CB 77 | 5.5 ± 1.7 | 3.1 ± 2.5 | n.d. | 1.0 ± 0.4 | 2.2 ± 0.9 | 1.3 ± 0.2 | 1.3 ± 0.1 | 0.7 |
| | 3.8–7.1 | 1.2–5.9 | | 0.6–1.4 | 1.3–3.1 | 1.0–1.4 | n.d.–1.4 | n.d.–0.7 |
| CB 81 | 0.9 | 1.1 ± 0.6 | n.d. | n.d. | 0.9 ± 0.2 | n.d. | 0.5 ± 0.4 | 1.1 ± 0.1 |
| | n.d.–0.9 | n.d.–1.5 | | | n.d.–1.5 | | n.d.–0.8 | n.d.–1.1 |
| CB 126 | 1.0 ± 0.7 | n.d. | 0.2 | n.d. | n.d. | n.d. | 0.8 | n.d. |
| | 0.5–1.5 | | n.d.–0.2 | | | | n.d.–0.8 | |
| CB 169 | n.d. | 0.2 | n.d. | n.d | n.d. | n.d. | 0.2 | 0.1 |
| | | n.d.–0.2 | | | | | n.d.–0.2 | n.d.–0.1 |
| Σ non-*ortho* PCBs | 7.4 | 4.4 | 0.2 | 1.0 | 3.1 | 1.3 | 2.8 | 1.9 |
| | 5.7–7.6 | 2.4–6.6 | n.d.–0.2 | 0.6–1.4 | 2.4–3.1 | 1.0–1.4 | n.d.–3.2 | 0.7–1.1 |
| CB 105 | 23 ± 11 | 7.8 ± 1.8 | 1.8 ± 0.4 | 1.9 ± 0.3 | 3.6 ± 0.1 | 1.8 ± 0.8 | 6.7 ± 5.3 | 1.8 ± 0.3 |
| | 13–34 | 5.8–8.9 | 1.4–2.2 | 1.7–2.2 | 3.5–3.7 | 1.0–2.6 | 1.5–12 | 1.4–2.1 |
| CB 114 | 1.5 ± 0.8 | 1.6 ± 0.7 | 4.2 | n.d. | 0.2 ± 0.1 | 0.2 | 0.7 ± 0.2 | 0.3 |
| | 0.8–2.4 | 0.5–3.6 | n.d.–4.2 | | n.d.–0.2 | n.d.–0.2 | n.d.–0.9 | n.d.–0.3 |
| CB 118 | 80 ± 47 | 47 ± 13 | 23 ± 16 | 6.5 ± 1.2 | 17 ± 3.1 | 6.0 ± 1.6 | 25 ± 19 | 5.5 ± 1.9 |
| | 35–128 | 19–97 | 14–41 | 5.5–7.8 | 13–19 | 4.2–7.2 | 7.5–46 | 3.3–6.8 |
| CB 123 | 14 ± 8.7 | 5.9 ± 2.2 | 3.3 ± 2.3 | 0.8 ± 0.3 | 2.3 ± 0.5 | 0.7 ± 0.3 | 4.3 ± 3.9 | 0.7 |
| | 4.4–22 | 2.7–12 | 1.4–5.8 | n.d.–1.0 | 1.8–2.7 | 0.3–0.9 | 0.8–8.5 | 0.4–1.0 |
| CB 156 | 8.5 ± 4.7 | 2.3 ± 0.2 | 2.8 ± 2.1 | 0.8 ± 0.5 | 2.3 ± 0.8 | 0.6 ± 0.1 | 3.3 ± 2.4 | 0.7 ± 0.3 |
| | 3.3–12 | 2.1–2.5 | 1.0–5.4 | 0.5–1.4 | 1.5–3.0 | n.d.–0.6 | 1.9–6.1 | n.d.–1.0 |
| CB 157 | 1.7 ± 1.3 | 0.4 ± 0.1 | n.d | 0.1 | 1.8 ± 0.4 | 0.1 | 1.0 ± 0.6 | 0.6 |
| | 0.5–3.0 | 0.3–0.4 | | n.d.–0.1 | 0.3–2.9 | n.d.–0.1 | n.d.–1.4 | n.d.–1.1 |
| CB 167 | 5.1 ± 3.7 | 0.7 ± 0.2 | 2.8 ± 1.7 | 0.3 ± 0.2 | 1.2 ± 0.4 | 0.3 ± 0.1 | 1.6 ± 1.2 | 0.6 |
| | 1.3–8.6 | 0.6–0.9 | 1.2–5.9 | n.d.–0.5 | 0.5–2.0 | n.d.–0.4 | 0.4–2.8 | n.d.–0.6 |
| CB 189 | 2.5 ± 0.3 | 1.0 ± 0.7 | 3.1 | n.d. | 0.5 | 0.1 | 1.4 | n.d. |
| | n.d.–2.8 | n.d.–1.5 | n.d.–3.1 | | n.d.–0.5 | n.d.–0.1 | n.d.–1.4 | |
| Σ mono-*ortho* PCBs | 136 | 66 | 41 | 10 | 29 | 9.7 | 44 | 10 |
| | 58–213 | 34–123 | 20–67 | 7.8–12 | 21–34 | 6.1–12 | 12–80 | 5.9–12 |
| $\Sigma_{12}$dl-PCBs | 143 | 71 | 41 | 11 | 32 | 11 | 47 | 12 |
| | 66–219 | 37–126 | 20–67 | 9.0–13 | 24–37 | 7.1–11 | 12–83 | 7.0–13 |
| $\Sigma_{18}$PCBs | 779 | 229 | 439 | 106 | 274 | 93 | 251 | 80 |
| | 429–1123 | 201–275 | 308–641 | 66–137 | 168–343 | 72–135 | 124–404 | 39–111 |
| $WHO_{2005}$-TEQs | 0.16 | 0.03 | 0.15 | 0.06 | 0.10 | 0.04 | 0.07 | 0.06 |
| | 0.07–0.22 | 0.02–0.03 | 0.01–0.39 | 0.01–0.15 | 0.03–0.20 | 0.02–0.07 | 0.04–0.09 | 0.02–0.11 |

The number of fish muscle homogenates per species per station (N) = 3; n.d. — non-detectable.

(such as CB 138 and CB 153) have lower chemical degradation rates than the less chlorinated PCBs and thus are retained in aquatic systems to a greater degree, where they may bioaccumulate in organisms like fish (Lavandier et al., 2013).

The PCB levels at station 1 were significantly higher ($p < 0.05$, Mann–Whitney *U*-test) than those from stations 3 and 4. However, statistical analysis between stations 1 and 2 showed no significant difference between the two locations. Station 1 is located near the mouth of the Nakivubo channel which brings in effluents from a number of chemical manufacturing factories, industrial waste treatment plants and municipal solid waste incinerators. Furthermore, in close vicinity of the station 1 are industrial facilities and Port bell where ships anchor on their way to and from Ports like Mwanza in Tanzania, Kisumu in Kenya and Jinja Port in Uganda. Although, the shipping industry promotes economic development in the area, it inevitably brings pollution related problems.

High concentrations of PCBs were also observed at station 2 (mean value of the $\Sigma_{18}$PCBs in sediments was 2576, range 1753–3970 pg $g^{-1}$ dw). Station 2 is situated close to the Ggaba landing site and a local market. Besides the market a lot of commercial, domestic and industrial activities occur near the station and could be releasing domestically-produced commercial PCB mixtures into the lake. The PCB levels in sediments from the outer part of the Murchison Bay (stations 3 and 4), both far from industrial complexes, were an order of magnitude lower compared to those in the inner Bay (Fig. 3). The low PCB levels in the outer Murchison Bay are likely due to the far offshore distance from the pollution sources.

The $WHO_{2005}$-TEQs for the 12 dl-PCBs in sediments of the present study varied from 0.04 to 0.64 pg TEQ $g^{-1}$. Station 1 had the highest TEQ values ranging from 0.37 to 0.64 pg TEQ $g^{-1}$ (mean 0.55 pg TEQ $g^{-1}$) while station 4 had the lowest (range 0.07 to 0.14, mean 0.10 pg TEQ $g^{-1}$). CB 126 was the highest contributor to the $\Sigma_{12}$TEQs (70–89%) compared to other congeners at all locations. The $WHO_{2005}$-TEQs in most samples from the Murchison Bay were below the interim sediment quality guideline of 0.85 pg TEQ $g^{-1}$ dw recommended by the Canadian Council of Ministers of the Environment (2002). Thus the dl-PCB concentrations in the present work are unlikely to pose a threat to benthic organisms.

### 3.2. Concentrations of PCBs in fish from the Murchison Bay

Table 3 presents the PCB concentrations in fish muscle homogenates of *L. niloticus* and *O. niloticus*. Average concentrations of the $\Sigma_{18}$PCBs ranged from 80 to 779 pg $g^{-1}$ wet weight (ww). The concentrations were in the same range as those reported in similar fish species from the Napoleon Gulf of Lake Victoria (94–716 pg $g^{-1}$ ww: Ssebugere et al., 2013a; Ssebugere et al., 2014) and River Nile in Egypt (695–853 pg $^{-1}$ ww; El-Kady et al., 2007). In neighboring Burundi, Manirakiza et al. (2002) reported PCB concentrations between 44900 and 166700 pg $g^{-1}$ lw in *O. niloticus* collected from Lake Tanganyika.

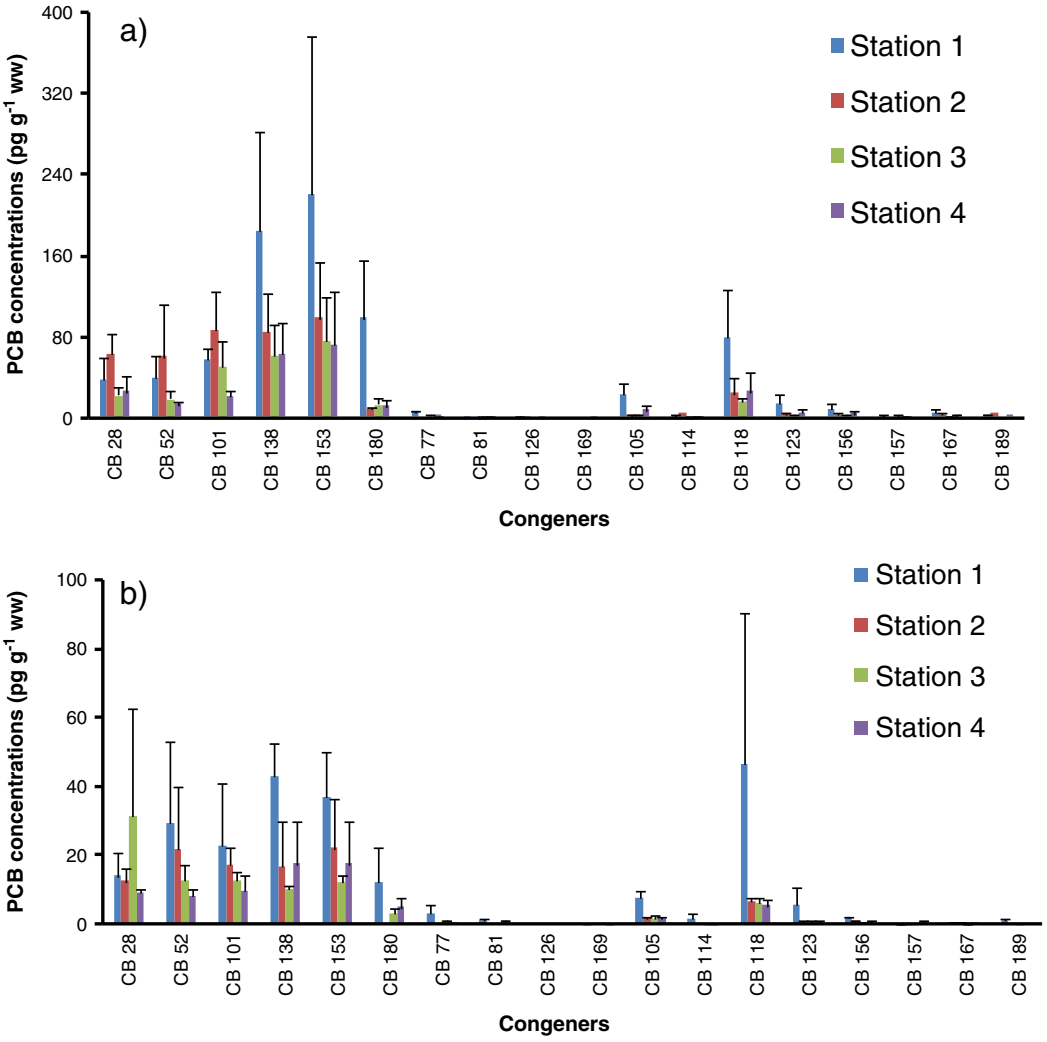


**Fig. 4.** Mean PCB concentrations in fish per station. a) Mean PCB concentrations in *L. niloticus* per station. b) Mean PCB concentrations in *O. niloticus* per station.

These levels were at least an order of magnitude higher compared to the present results. In Egypt, Said et al. (2008) reported PCB concentrations varying from 3320 to 72 060 pg $g^{-1}$ lw in fish (*O. niloticus* and *Clarias gariepinus*) from Lake Burullus. The concentrations were again higher than the data in the study herein. Studies outside Africa have also reported PCBs in fish. In Hawai'i, PCB levels ranging from 51 900 to 89 420 pg $g^{-1}$ lw were reported in tilapia (*Oreochromis mossambicus*) collected from the Manoa stream and Ala Wai Canal of O'ahu (Yang et al., 2008). The PCB levels were also higher than those reported in this study. In South China, Nie et al. (2006) reported PCB concentrations ranging from 5150 to 226 000 pg $g^{-1}$ lw in tilapia (*Tilapia mossambica*) from Pearl River Delta. The concentrations are still higher compared to the data in the current study.

The $\Sigma_{18}$PCBs in *L. niloticus* at station 1 was one-fold higher than that at station 2 and about three times higher than that at station 4. The $\Sigma_{18}$PCBs at stations 3 and 4 were quite similar. For *O. niloticus*, still station 1 showed the highest levels of the $\Sigma_{18}$PCBs (mean value 229 pg $g^{-1}$) compared to other stations. Station 1 was followed by station 2 and then station 3, while station 4 had the lowest concentration. The concentrations of the different PCB congeners at the different stations for the 2 fish species are shown in Fig. 4.

The PCB concentrations in *L. niloticus* were markedly higher ($p < 0.05$, Mann–Whitney *U*-test) than those in *O. niloticus* at all study sites. The nature of the diet could be a possible explanation for such an inter-specific differences in concentrations. *L. niloticus* are exclusively piscivorous, consuming all available fish species including its own siblings whereas *O. niloticus* relies on zooplankton and macroinvertebrates (Ogari, 1984; Njiru et al., 2004). Earlier studies established that PCBs can have great variations among different fish, as well as within species depending on their feeding habits and trophic levels (Suedel et al., 1994; Vander Zanden and Rasmussen, 1996; Davis et al., 2002; van der Oost et al., 2003).

On the whole, the $\Sigma_6$ indicator PCBs in fish of the present study were an order of magnitude higher than the $\Sigma_7$ mono-*ortho*-PCBs and $\Sigma_4$ non-*ortho*-PCBs. However, the $\Sigma_6$ indicator PCB levels (83–636 pg $g^{-1}$) in this study were within the limit of 75 000 pg $g^{-1}$ ww set for fish by the European Commission (EC, 2011), implying that the study fish were fit for human consumption in regard to indicator PCBs. Indicator PCB congeners such as CB 138 and CB 153 which were predominant in the sediments were also major contributors in *L. niloticus* (Fig. 5). However, variations in contribution of the indicator PCBs to the $\Sigma_{18}$PCBs were observed for *L. niloticus* at the different stations. For the coplanar PCBs, the mono-*ortho* PCBs were the predominant congeners in most fish (their contribution to the $\Sigma_{12}$PCBs ranged from 84 to 100%). CB 118 was the most abundant congener, accounting for 46–69% of the $\Sigma_{12}$PCBs, in comparison to other coplanar compounds. The high levels of CB 118 in the two fish species could be due to the inability of the species to metabolize the congener after accumulation. The presence of CB 118 in our samples suggests paint additives, municipal waste plants, iron ore sintering plants or medical waste incinerators as potential sources of PCBs (Abad et al., 2006; Aries et al., 2006; Shin et al., 2006; Chen et al., 2009; Jartun et al., 2009). Furthermore, it is likely that the

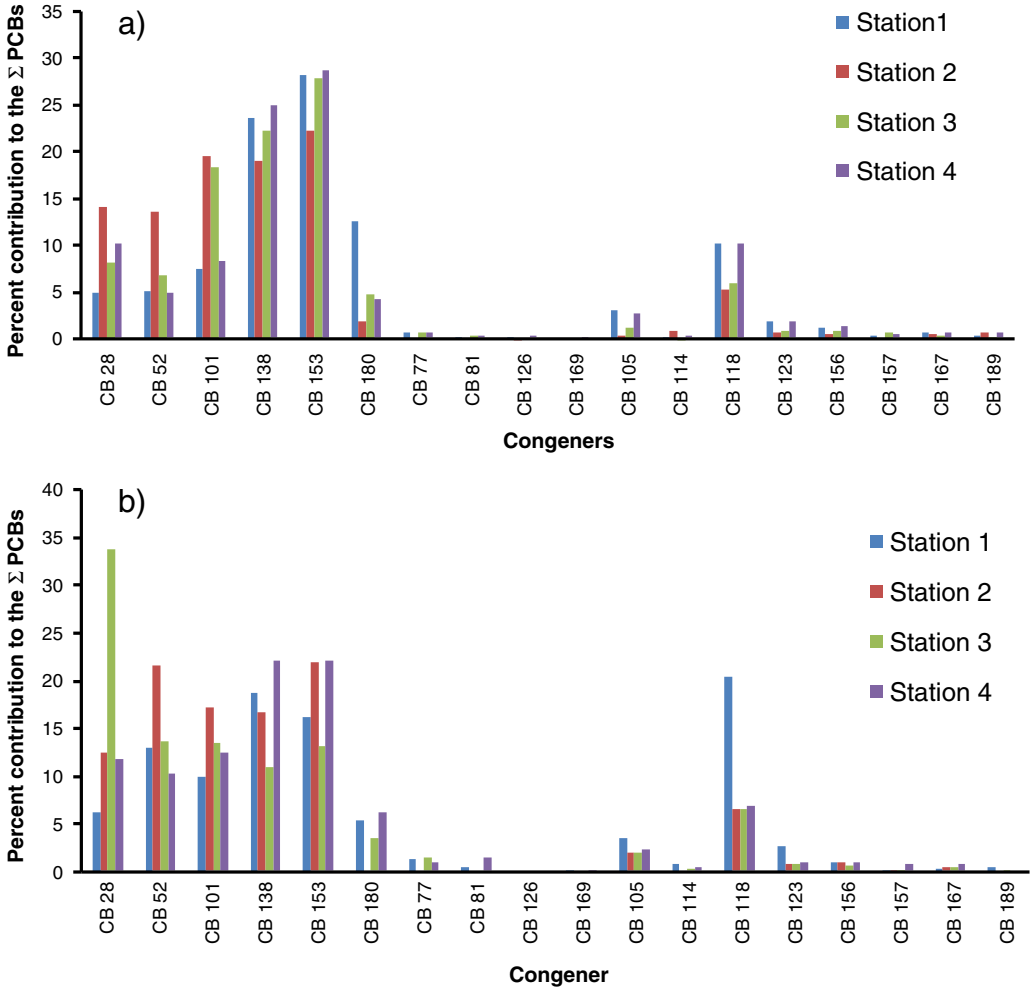


**Fig. 5.** Percent contribution of the congeners to the ΣPCBs in the fish. a) Contribution of the different congeners to the $\Sigma_{18}$PCBs in *L. niloticus*. b) Contribution of the different congeners to the $\Sigma_{18}$PCBs in *O. niloticus*.

PCBs came from the residues of former applications like transformers, electrical equipment and other industrial uses. The use of these applications containing PCBs was common in Uganda before their commercial production was banned worldwide in 1979 (National Environment Management Authority, 2007).

The $WHO_{2005}$-TEQs for dl-PCBs in the study herein were 0.01–0.39 and 0.01–0.11 pg $g^{-1}$ ww for *L. niloticus* and *O. niloticus*, respectively. The non-*ortho* PCBs exhibited the highest contribution to the $\Sigma_{12}$PCBs (>75%) compared to the mono-*ortho* PCBs in both fish species. A similar observation was also reported by Ssebugere et al. (2013a) in fish from the Napoleon Gulf and Thurston Bay of Lake Victoria. The $WHO_{2005}$-TEQs in the present study were within the permissible level of 3.5 pg $g^{-1}$ ww set by the European Commission (EC, 2011), suggesting that the fish from the Murchison Bay were safe for consumption with respect to dl-PCBs.

## 4. Conclusions

To our knowledge this is the first study to record PCB concentrations in sediments and fish from the Murchison Bay. The station closest to Nakivubo channel had the highest PCB concentrations compared to other locations. Land based activities such as urban centers, commercial and industrial establishments are likely the major sources of PCB contamination. The PCB concentrations in the present study were generally comparable to those of most studies in Africa but lower than the data examined elsewhere in the world. The WHO-TEQs were below the recommended values set for sediments and fish, indicating that the risk posed by PCBs is limited, compared to other health and environmental issues, such as the high disease burden due to diarrhea and other infectious diseases, associated with a limited availability of sanitation and treated water (Yeka et al., 2012; Tumwebaze et al., 2013; McElligott et al., 2013; Bwire et al. 2013; Katukiza et al., 2014). However, regulatory controls for monitoring and mitigating wastewater emissions into Lake Victoria need to be implemented and emphasized by the relevant governmental agencies in Uganda.

## Conflict of interest

There is no conflict of interest.

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