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PCDD/Fs and dioxin-like PCBs in surface sediments from Lake Victoria, East Africa

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HIGHLIGHTS

- Surface sediments from Lake Victoria were analyzed for PCDDs, PCDFs and dl-PCBs.
- The maximum concentration of PCDD/Fs and dl-PCBs was 136pgg−1 dw.
- The WHO2005-TEQs for PCDD/Fs lay in the range of 0.07 to 5.53pg TEQg−1 dw.
- The WHO2005-TEQs for dl-PCBs varied from 0.01 to 0.23pg TEQg−1 dw.
- 23.1% of samples had their PCDD/F results above the set WHOPCDD/Fs-TEQ values.

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ABSTRACT

Surface sediments (<60 cm) from the Napoleon Gulf and Thurston Bay on the northern shore of Lake Victoria were analyzed for polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and dioxin-like polychlorinated biphenyls (dl-PCBs). Fifteen PCDD/Fs and eleven dl-PCBs were found in 75.5% of the samples. The maximum concentrations of PCDD/Fs and dl-PCBs were 44.1 and 136 pg g^{-1} dry weight (dw), respectively. Octachlorodibenzo-*p*-dioxin was the predominant PCDD/F congener at the Napoleon Gulf and Thurston Bay area. Regarding the dl-PCBs, a variation in levels was observed between the mono-*ortho* PCBs and non-*ortho* PCBs, with the former having higher levels than the latter. The PCDD/F and dl-PCB levels, in the sediments of Napoleon Gulf, which is near urban centers and industrial areas were markedly higher ($\alpha \leq 0.05$) than those from the Thurston Bay, which is offshore, suggesting that human activities could be sources of the pollutants to the surrounding water resources. World Health Organization-toxic equivalency quotients (WHO-TEQs) lay in the range of 0.07–5.53 pg g^{-1} dw for PCDD/Fs and 0.01–0.23 pg g^{-1} dw for dl-PCBs. 23.1% of samples from the Napoleon Gulf had their results above the set $WHO_{PCDD/Fs}$-TEQ value.

1. Introduction

Polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and the dioxin-like polychlorinated biphenyls (dl-PCBs) are carbon-based organic substances. They possess a combination of physical and chemical properties which include being non-biodegradable (Field and Sierra-Alvarez, 2008), undergoing long-range atmospheric transport (Li et al., 2007) and accumulating in the fatty tissue of living organisms (Çok et al., 2007; Bordajandi et al., 2008). In animals, these contaminants elicit a diverse spectrum of toxic and biochemical responses including body weight loss (Fierens et al., 2003), induction of gene expression (Brouwer et al., 1995) and immunotoxicity (Sormo et al., 2009).

The major sources of PCDD/Fs are thermal processes (Lin et al., 2007), forest fires (Kim et al., 2003), fossil fuel combustion, synthesis of chloro-phenols, production of chloro-organic pesticides, pulp bleaching in the process of paper production (Quaß et al., 2004), among others. PCBs can be released into the atmosphere through leakage of old equipments like transformers and capacitors, leaching from landfills, smokestacks and weathering of asphalt, etc. Despite the extensive information on PCB inventory, use and disposal, especially in the developed nations, only the National Environment Management Authority (2007) has documented their status in Uganda, a country which became an official party of the Stockholm Convention in 2004 (Ritter et al., 2007).

In the environment, the atmosphere is a major pathway for the transport and deposition of PCBs and PCDD/Fs. Once released into aquatic systems, the pollutants tend to accumulate in the sediments for a long period of time and are subject to partitioning, degradation and transport processes. The sediments can then act as natural sinks

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for the pollutants whose re-suspension at the water–sediment interface, especially in storm events and during lake mixing, may increase accumulation and bio-availability in the food chain (Evenset et al., 2007). Relative to the water column, surface sediments are a more appropriate environmental compartment that can be related to the concentrations of pollutants in aquatic organisms (Nguyen et al., 2005).

Of recent, L. Victoria is increasingly experiencing pollution from various sources like biomass burning, industry, transport and households (Awange and Obera, 2007; Kwach and Lalah, 2009). A considerable proportion of the pollutants reach the lake, thereby contributing to deterioration in the water quality (Odada et al., 2004). As far as we know, this is the first report on PCDD/Fs and dl-PCBs in the sediments of L. Victoria. The aim of this study was to evaluate the concentration magnitude of the dioxins in the sediments of L. Victoria and to compare the results with those obtained in studies elsewhere.

2. Materials and methods

2.1. Study area and sample collection

Samples of overlying surface sediments were collected in March 2011 from the Napoleon gulf and Thurston bay (Fig. 1). The Napoleon Gulf is on the shoreline of Jinja town with a dense urban development and industrialization. A great deal of bio-mass burning occurs along the banks of the Thurston Bay where sugarcane growing is also significant. The shorelines of the Napoleon Gulf and Thurston Bay which were previously fringed by papyrus (*Cyprus papyrus*), have been extensively degraded by a rapid rising human population, mounting pressure on the lake, in terms of water quality and sedimentation (Wasswa et al., 2010). For the reasons above, the two locations were earmarked as study areas.

Within each location, samples were taken from marked stations. The stations included A (00°24′50″N, 33°14′20″E), B (00°24′60″N, 33°17′10″E) and C (00°25′70″N, 33°16′40″E) at the Napoleon Gulf, and D (00°25′50″N, 33°20′55″E) and E (00°26′50″N, 33°21′50″E) at the Thurston Bay. A sediment tube sampler was used for coring, to minimize the likelihood of cross contamination; all the utensils used were thoroughly washed and rinsed with acetone, followed by hexane before coring. A total of twenty five samples (five from each station) were collected. The number of collected samples was dictated by the high cost of analysis of PCDD/Fs and dl-PCBs. The samples were transferred into clean solvent-rinsed glass bottles. The bottles were immediately transferred to the laboratory and kept at −28 °C until extraction.

2.2. Reagents, chemicals and standards

All the organic solvents (picograde) used, ^{13}C-labeled PCDD/F and dl-PCB standards were purchased from LGC Standards (Wesel, Germany), and their purity was over 99.9%. Anhydrous sodium sulfate, alumina, C_{18}-modified silica gel, sulfuric acid and silica gel were of analytical grade. The silica gel was heated overnight at 200 °C before use, to reduce the background levels of analytes. Reversible carbon columns filled with 100 mg Carboxen 1016, were purchased from Supelco (Taufkirchen, Germany). The screening included a total of fifteen 2,3,7,8 PCDD/F congeners, four non-*ortho* PCBs (77, 81, 126 and 169) and seven mono-*ortho* PCBs (105, 114, 118, 123, 156, 157 and 167), because they are the most toxic and have a common mode of action. Analytical stock solutions were prepared in amber flasks using *n*-nonane and stored in a fridge maintained at 4 °C.

2.3. Sample extraction and purification procedures

The sediment samples were defrosted prior to extraction. Detailed descriptions of the sample extraction and purification procedures have previously been reported by El-Kady et al. (2007). Briefly, 10 g of the sample was spiked with internal standards (^{13}C-labeled PCDD/Fs and dl-PCBs) and converted to a free flowing powder using

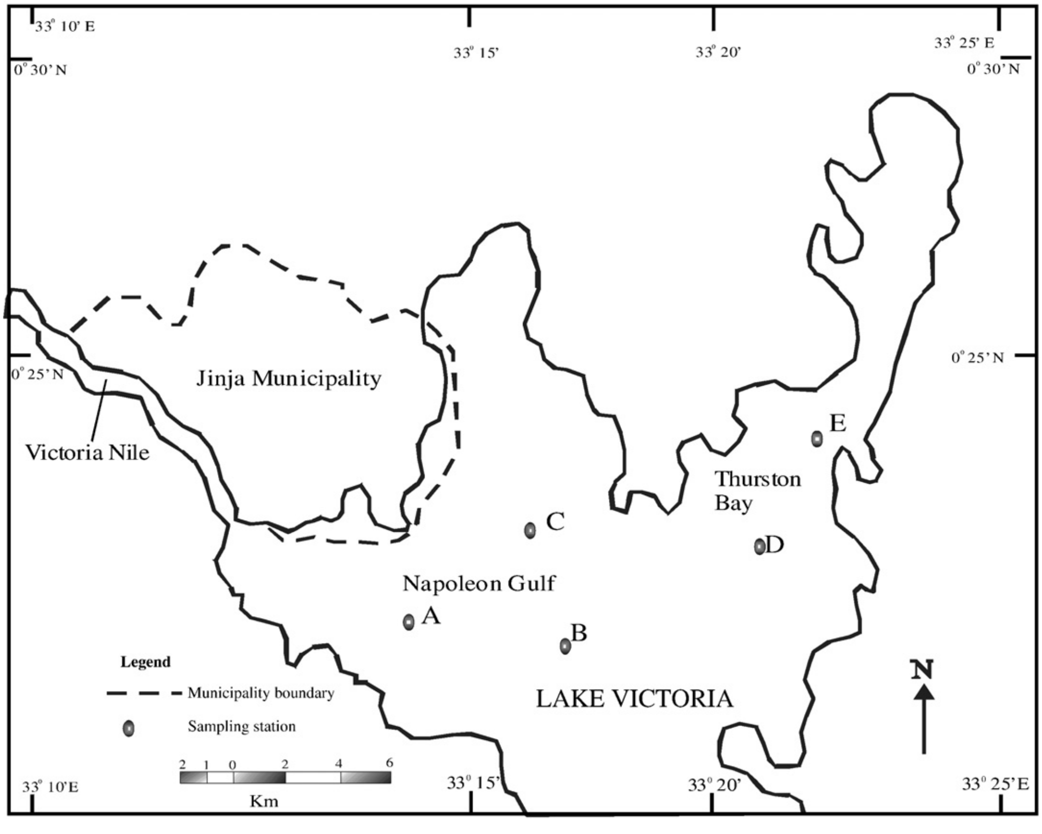

Fig. 1. Map showing the study stations.

hydromatrix™ (Varian). The samples were extracted using an accelerated solvent extractor (ASE 200 Sunnyvatem CA, USA) with two 10 min static cycles of 75/25 *n*-hexane–acetone mixture at 120 °C and a pressure of 120 bar. After extraction, the resultant volume was passed through a drying funnel of anhydrous sodium sulfate and eluted with 100 mL of *n*-hexane. The extracts were reduced to 1 mL on a rotary evaporator and kept for double step purification.

Primary purification was achieved by using a multi-layer silica gel column which was filled with 5 g of anhydrous Na_2SO_4 (bottom), 2 g of silica gel, 3 g of 30% NaOH-silica gel, 1 g of silica gel, 5 g of 44% H_2SO_4-silica gel, 10 g of 22% H_2SO_4-silica gel and 5 g of anhydrous Na_2SO_4 (top). The column was pre-washed with 60 mL of *n*-hexane before it was directly connected to a reversible carbon column (Carboxen 1016, Supelco) that had been rinsed by an equi-volume (25 mL) of toluene and *n*-hexane, respectively. The concentrated extract was added to the multi-layer column and eluted with 100 mL of *n*-hexane. The mono-*ortho* PCBs passed through both columns, while PCDD/Fs and non-*ortho* PCBs were retained in the carbon column. The multi-layer column was removed; the carbon column was washed with 30 mL of dichloromethane/*n*-hexane (1:9, v/v), then reversed and the PCDD/Fs were eluted with 100 mL of toluene. The co-extracted analytes were separately concentrated to 1 mL and kept for secondary purification.

Secondary purification was achieved as described by Çok et al. (2009). Briefly, the first fraction containing PCDD/Fs was eluted through an alumina column filled from top to bottom with 2 g of anhydrous Na_2SO_4, 5 g of alumina and 2 g anhydrous Na_2SO_4. The second fraction containing PCBs was purified using a solid phase extraction cartridge filled with 1 g of C_{18}-modified silica gel. Silica gel was first conditioned with 6 mL of acetonitrile before the extract was added and later eluted with 5 mL of acetonitrile. The resulting purified extracts from both fractions were reduced to <1 mL using a rotary evaporator, transferred to sample vials containing internal recovery standards and then kept for gas chromatographic analysis.

2.4. Instrumental analysis

The analysis of PCDD/Fs and non-*ortho* PCBs was performed with a high resolution mass spectrometer MAT 95S (Thermo Electron GmbH, Bremen, Germany) coupled with an Agilent gas chromatograph 6890 (Agilent Technologies, Palo Alto, CA, USA). Chromatographic separation was achieved by injecting 1 μL extract in the pulsed splitless mode with a period of 120 s on a Restek Rtx-Dioxin2 capillary column (40 m × 0.18 mm i.d. × 0.18 μm). The oven temperature of the gas chromatograph (GC) was held at 130 °C for 1.5 min, rising to 205 °C at 25 °C min^{-1}, then to 310 °C at 4 °C min^{-1} and finally holding it at 310 °C for 15 min. The injection temperature was 300 °C, while the transfer-line temperature was 300 °C. The analysis of mono-*ortho* PCBs was carried out using an Agilent GC 5890 Series 11 equipped with a high resolution mass spectrometer MAT 95S. The separation was achieved by injecting 1 μL on a Phenomenex MultiResidue-2 column (30 m × 0.25 mm i.d × 0.20 μm). The oven temperature was initially programmed at 90 °C. It was then held for 1.5 min, increased to 170 °C at 20 °C min^{-1}, held for 7.5 min, increased to 265 °C at 3.5 °C min^{-1}, then to 310 °C at 20 °C min^{-1} and held for 10 min. Helium was used as a carrier gas at a constant pressure of 16 psi. The mass spectrometer was operated at a resolution of over 10,000 under positive electron ionization conditions (45 eV), and data was obtained in SIM mode using the two most intense ions of the molecular ion cluster for labeled and unlabeled isomers.

2.5. Quality assurance and data analysis

To ensure quality of the data, replicate analyses for precision control, recovery standards for control of losses and internal reference materials for accuracy control were used. The results presented and discussed in this study were blank corrected. One procedural blank was performed for every 5 samples. In the absence of detected blank values the limit of detection (LOD) was defined by a signal to noise ratio (S/N) > 3 times the average baseline variation. If blank values were detected, the LOD was defined as 3 times the standard deviation of all blanks analyzed. The LOD was a function of the extracted mass and varied between analytes (0.01–0.33 pg g^{-1} for PCDD/Fs and 0.02–1.00 pg g^{-1} for dl-PCBs). The recoveries for surrogate standards ranged between 45% and 108%. All concentrations in the present study were normalized to dry sample weight (dw). Arithmetic means were calculated from only positive quantifiable samples and statistical significance was defined at $p < 0.05$.

3. Results and discussion

3.1. Concentrations of PCDD/Fs in the sediments

The present study is the first investigation on the concentrations of PCDD/Fs in sediments from the Napoleon Gulf and Thurston Bay of L. Victoria. The concentrations of sigma PCDDs varied widely from one location to another (Table 1). The maximum concentration at the study stations was 44.1 pg g^{-1} dw. Octachlorodibenzodioxin (OCDD) was the major contributor in the majority of the samples (>29.7% to sum of seven PCDD congeners) from the Gulf and Bay area. The reason for the domination of OCDD could be that it is emitted at higher rates in the atmosphere than other PCDD congeners. Octachlorodibenzodioxin is usually produced in high amounts as a by-product during combustion of domestic and fuel oil mixtures (Anderson and Fisher, 2002), in sewerage sludge and open fire of garden waste (Wevers et al., 2004). Therefore, the Napoleon Gulf being near urban centers and industrial areas could be receiving OCDD related compounds from the above sources. It was observed that 1,2,3,4,7,8-HxCDD was detected at low levels in most sediments from the stations. One of the reasons for the low 1,2,3,4,7,8-HxCDD levels found could be biotransformation processes. Muir et al. (1985) have reported 1,2,3,4,7,8-HxCDD to undergo more rapid biotransformation compared to other PCDDs.

The concentrations of PCDDs in the sediments from station B were remarkably higher compared to A and C. It was noted that the sediments from station B were clay in nature while those from A and C were sandy. This suggests that grain size of sediments could be an important factor governing the levels of PCDDs. At the Thurston Bay, station E had higher concentrations of PCDDs than D (Table 1). In the vicinity of station E is Kakira sugar plantation, where open burning of sugarcane is normally done every harvesting season. The concentrations of the $\sum$PCDDs recorded in the present study varied from 0.42 to 52.8 pg g^{-1} dw. Our results are nearly the same as those in sediments from Liaohe River in China (Zhang et al., 2010) and the Oder River in Poland (Niemirycz and Jankowska, 2011).

Polychlorinated dibenzofurans in the Napoleon Gulf sediments ranged from 0.07 to 5.61 pg g^{-1} dw. Congener 1,2,3,4,7,8,9-HpCDF was the most abundant homologue in the sediments from station B (based on mean concentrations it represented 21.3% of the $\sum$PCDFs) while 2,3,7,8-TeCDF, the reference compound of the furans group, was the least detected (it contributed less than 8.56% to the $\sum$PCDFs). The concentration ranges in pg g^{-1} dw and contribution to the $\sum$PCDFs by other furans in the sediments from station B were 2,3,4,7,8-PeCDF (1.42–2.72, 12.7%); 1,2,3,4,7,8-HxCDF (1.51–4.13, 15.0%) and 1,2,3,4,7,8,9-HpCDF (2.07–4.89, 21.3%). The PCDF concentrations at station B were 2 fold higher than those from A. The levels of PCDFs from station A were one order of magnitude higher than those at C. It was noted that for most sediments from the Napoleon Gulf, congener 2,3,4,7,8 PeCDF was predominant. Baker and Hites (2000) have reported that 2,3,4,7,8 PeCDF principally comes from technical PCBs and combustion processes. Therefore, their prevalence in the Gulf sediments can be attributed to urban waste incineration

Table 1
Mean concentrations and ranges (pg g^{-1} dw) of PCDD/Fs in sediments from the study stations.

Congeners	Napoleon Gulf			Thurston Bay	
	Station A	Station B	Station C	Station D	Station E
PCDDs					
2, 3, 7, 8-TeCDD	1.15	1.33	n.d	n.d	n.d
	1.10–1.18	1.28–1.38			
1, 2, 3, 7, 8-PeCDD	0.52	2.47	2.43	n.d	n.d
	0.49–0.55	2.33–2.61	2.18–2.70		
1, 2, 3, 4, 7, 8-HxCDD	0.74	1.50	n.d	n.d	0.30
	0.21–1.15	1.48–1.52			0.28–0.32
1, 2, 3, 6, 7, 8-HxCDD	0.70	15.7	0.42	0.33	0.56
	0.44–1.08	4.74–26.7	0.4–0.44	0.31–0.35	0.44–0.67
1, 2, 3, 7, 8, 9-HxCDD	0.66	1.17	0.39	n.d	0.54
	0.57–0.75	1.12–1.22	0.25–0.52		0.25–0.83
1, 2, 3, 4, 6, 7, 8-HpCDD	5.20	2.43	2.23	0.41	3.63
	4.92–5.47	1.83–3.02	0.67–4.16	0.16–0.65	0.37–6.88
OCDD	35.4	10.4	28.8	2.45	36.3
	35–35.5	10.4–10.5	28.7–28.9	2.41–2.49	28.9–44.1
$\sum$PCDDs	44.4	35.1	34.2	3.19	41.4
PCDFs					
2, 3, 7, 8-TeCDF	0.59	1.40	0.35	n.d	0.40
	0.55–0.63	1.38–1.42	0.18–0.51		0.28–0.51
2, 3, 4, 7, 8-PeCDF	0.89	2.07	1.21	0.08	0.47
	0.27–1.74	1.42–2.72	0.32–2.87	0.07–0.09	0.38–0.55
1, 2, 3, 4, 7, 8-HxCDF	0.61	2.45	1.24	0.09	0.38
	0.26–1.26	1.51–4.13	0.31–3.27	0.07–0.11	0.31–0.50
1, 2, 3, 6, 7, 8-HxCDF	0.57	1.81	0.71	n.d	0.59
	0.24–1.15	1.45–2.18	0.40–1.22		0.50–0.68
1, 2, 3, 7, 8, 9-HxCDF	0.23	3.71	0.39	n.d	0.53
	0.19–0.29	1.81–5.61	0.37–0.41		0.41–0.64
1, 2, 3, 4, 6, 7, 8-HpCDF	0.92	1.43	0.85	n.d	1.13
	0.82–1.02	1.41–1.45	0.78–0.92		0.92–1.34
1, 2, 3, 4, 7, 8, 9-HpCDF	0.16	3.48	0.21	n.d	0.21
	0.15–0.17	2.07–4.89	0.18–0.25		0.20–0.23
OCDF	1.14	n.d	1.17	n.d	n.d
	1.12–1.15		1.11–1.27		
$\sum$PCDFs	5.11	16.4	6.13	0.17	3.71
$\sum$PCDD/Fs	49.5	51.4	40.4	3.36	45.1
WHO$_{2005}$-TEQ (PCDD/Fs)	0.98	4.24	1.32	0.08	0.42
	0.21–2.05	1.67–5.53	0.40–1.38	0.07–0.09	0.08–0.69
Ratios ($\sum$PCDFs/$\sum$PCDDs)	0.12	0.47	0.18	0.05	0.13
	0.06–1.86	0.51–0.62	0.10–1.67	0.02–0.05	0.08–1.37

n.d—nondetectable.

and chronic PCB runoff as sources of contamination. A different PCDF congener profile was observed in the Thurston Bay sediments, with 1,2,3,4,7,8 HxCDF being the prevalent congener at station D (52.9% to the grand $\sum$PCDFs) and 1,2,3,4,6,7,8 HpCDF at E (30.5% to the $\sum$PCDFs). Octachlorodibenzofuran (OCDF) was not detected in any sample from the Thurston Bay. Bakoglu et al. (2005) have reported that the main source of OCDF into the environment is mainly chemical manufacturing. Therefore, since the Thurston Bay is located far from industrial and urban activities such a result would be expected.

The levels of $\sum$PCDD/Fs from this study (range 0.20–56.7 pg g^{-1} dw), were an order of magnitude higher than were reported by Kim et al. (2009) in the sediment of Han River, Korea (range 23.1–368 pg g^{-1} dw). The calculated ratios ($\sum$PCDFs/$\sum$PCDDs) ranged from 0.06 to 1.86 and were >1 in 26.3% of the sediments, confirming combustion as one of the possible sources of PCDD/Fs and related compounds into the Napoleon Gulf and Thurston Bay. The ratios of PCDDs/PCDFs can be used to identify the main source(s) of dioxins to the environment. Normally, if the ratio (PCDFs/PCDDs) is >1, the major source is especially combustion (Huang and Buekens, 1995).

3.2. Concentrations of the dl-PCBs in sediments

The dl-PCBs (non-*ortho* and mono-*ortho* PCBs) were detected in 76.9% of the samples. PCB 77 was the major contributor to the $\sum$non-*ortho* PCBs in most samples from the Gulf (stations A 89.3% and C 83.8%) and Bay area (stations D 88.5% and E 85.2%). The high levels of PCB 77 could be attributed to the fact that it does not undergo degradation easily (Miao et al., 1999). Although this isomer was dominant in our study, its concentrations are comparable to those in sediments from Lake Maggiore in Northern Italy (Castro-Jiménez et al., 2008) and, residential and industrial areas in central South Africa (Nieuwoudt et al., 2009). The contribution to the $\sum$non-*ortho* PCBs by PCB 77 was followed by 126 (4.57% for sediments from station A, 11.5% for D and 8.38% for E). On the contrary, the contribution by PCB 77 at station C was followed by 169 (it contributed 8.27% to the $\sum$non-*ortho* PCBs). Low levels of PCB 81 were detected from station A (mean 1.07 ± 0.04 pg g^{-1} dw). The low levels could be due to the high limits of detection observed in our laboratory. Similarly, PCB 169 exhibited low concentrations at all sampling stations (Table 2). PCB 169 tends to show limited mobility to molecular diffusion processes (Chiarenzelli et al., 2001) and is more susceptible to the dissolution loss process (Li et al., 2009). Therefore, the low concentrations in the present study could be attributed to the fact that the congener is more soluble in water. The levels of the $\sum$non-*ortho* PCBs in our study ranged from 1.24 to 39.8 pg g^{-1} dw. The levels are in the same range as those in sediments from Marmara Sea in Turkey (Telli-Karakoç et al., 2002).

With regard to the levels of mono-*ortho* PCBs, 105 and 118 were the most dominant at stations A and C, followed by 114 and 156. This kind of distribution of congeners is also reported by Zhou et al. (2012) in surface sediments from Lake Taihu in China. A variation in concentrations was observed between the mono-*ortho* PCBs and

Table 2
Mean concentrations and ranges (pg g^{-1} dw) of the dl-PCBs in sediment samples from each study station.

Congeners	Napoleon Gulf		Thurston Bay	
	Station A	Station C	Station D	Station E
Non-ortho PCBs				
PCB 77	28.2	14.0	28.2	18.2
	26.0–30.3	13.9–14.1	22.1–34.2	13.7–22.7
PCB 81	1.07	n.d	n.d	n.d
	1.04–1.10			
PCB 126	1.44	1.33	2.78	1.79
	1.09–1.63	1.30–1.35	1.74–3.82	1.32–2.25
PCB 169	0.86	1.40	n.d	1.38
	0.48–1.23	1.19–1.71		1.19–1.57
$\sum$Non-*ortho* PCBs	31.5	16.7	31.0	21.4
Mono-ortho PCBs				
PCB 105	60.1	40.8	25.2	n.d
	47.0–73.3	39.9–41.6	23.1–27.3	
PCB 114	5.99	3.28	3.77	n.d
	5.69–6.28	3.09–3.47	3.21–4.33	
PCB 118	121	81.9	66.1	n.d
	105–136	79.7–84.1	49.7–82.5	
PCB 123	6.73	4.56	n.d	n.d
	6.06–7.40	4.14–4.98		
PCB 156	16.1	23.3	31.7	n.d
	16.0–16.2	13.2–41.1	28.2–35.3	
PCB 157	5.88	6.24	9.08	n.d
	5.68–6.07	3.19–11.8	8.91–9.25	
PCB 167	n.d	12.5	13.4	n.d
		12.0–13.3	12.7–14.1	
$\sum$Mono-*ortho* PCBs	216	173	149	–
$\sum$dl-PCBs	247	189	180	21.4
WHO$_{2005}$-TEQ (dl-PCBs)	0.17	0.09	0.02	0.21
	0.13–0.20	0.01–0.18	0.01–0.03	0.18–0.23

n.d—nondetectable.

non-*ortho* PCBs, with the former having slightly higher concentrations than the latter. Mono-*ortho* PCB congeners are usually present at high concentrations in most commercial PCB mixtures and in environmental samples, and are environmentally persistent than the non-*ortho* PCBs (Hornbuckle et al., 2006). Therefore, the high levels of the mono-*ortho* PCBs are likely to be due to the fact that they undergo slow desorption kinetics. The levels of $\sum$mono-*ortho* PCBs in the present study (range 65.1–244 pg g^{-1}) are in the same range as those reported in sediments from River Nile in Egypt (El-Kady et al., 2007), and are lower compared to the ones in the Great Lakes of Canada (Li et al., 2009). In Uganda, there is no information about the use or any possible source of PCBs. Neither is there any regulation that addresses the control of PCB use nor their disposal. However, the possible sources of the dl-PCBs could be releases from damaged old electric transformers and commercial PCB products.

3.3. World Health Organization-toxic equivalency quotients for PCDD/Fs and dl-PCBs

Toxic equivalency quotients (TEQs) for each sample were calculated using the 2,3,7,8-TCDD toxic equivalent factors (TEFs) according to suggestions by World Health Organization (WHO) (van den Berg et al., 2006). The WHO$_{2005}$-TEQs lay in the range of 0.07–5.53 pg TEQ g^{-1} dw for PCDD/Fs and 0.01–0.23 pg TEQ g^{-1} dw for dl-PCBs. Concerning the WHO-TEQs for PCDD/Fs, OCDD was the most dominant and it accounted for between 36.4 and 83.7% of the fifteen congeners. PCBs 105 and 118 were the main dl-PCBs and together they contributed >61.2% to the TEQs. The TEQs for dl-PCBs in all the samples from the Napoleon Gulf and Thurston Bay were below the interim sediment quality guideline of 0.85 pg WHO-TEQ g^{-1} dw recommended by the Canadian Council of Ministers of the Environment (2002). While, the TEQs for PCDD/Fs in sediments from Thurston Bay were below 0.85 pg g^{-1}, 23.1% of samples from the Napoleon Gulf had their TEQs above the set value.

The WHO$_{PCDD/Fs}$-TEQ values at station B (range 1.67–5.53 pg g^{-1} dw) were higher compared to those at D (0.07–0.09 pg g^{-1} dw) and E (0.08–0.69 pg g^{-1} dw), but comparable to the ones of stations A (range 0.21–2.05 pg g^{-1} dw) and C (0.09–3.21 pg g^{-1} dw). The WHO$_{PCDD/Fs}$-TEQs in the present study were twice those of sediments from the Pearl River Delta in China (Zhang et al., 2009), in the same range as those of the Mondego estuary in Portugal (Nunes et al., 2011), but lower than the ones of Liaohe River in China (Zhang et al., 2010) and the trans-boundary rivers in Belgium–France (Sanctorum et al., 2011). In light of the above, it could be concluded that the contamination of the lake is within permissible limits for dl-PCBs except for PCDD/Fs in 23.1% of the samples.

4. Conclusion

In this study, detectable levels of PCDD/Fs and dl-PCBs were found in surface sediments from L. Victoria. The measured concentrations and WHO-TEQ values for PCDD/Fs and dl-PCBs were lower compared to those reported in most studies dealing with fresh water sediments. However, since sediments play a role of first supplier to benthic feeders and consequently the biomagnifications of pollutants to higher trophic animals, the presence of even trace amounts of PCDD/Fs and dl-PCBs in the sediments of L. Victoria could be of harm since they may end up being ingested by aquatic organisms like fish.

Conflict of interest

There is no conflict of interest.

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References

Anderson RD, Fisher R. Sources of dioxins in the United Kingdom: the steel industry and other sources. Chemosphere 2002;46:371–81.

Awange LJ, Obera B. Motor vehicles: are they emerging threats to Lake Victoria and its environment? Water Air Soil Pollut 2007;182(1–4):43–56.

Baker JA, Hites RA. Is combustion the major source of polychlorinated dibenzo-*p*-dioxins and dibenzofurans to the environment? A mass balance investigation. Sci Total Environ 2000;34:2879–86.

Bakoglu M, Karademir A, Durmusoglu E. Evaluation of PCDD/F levels in ambient air and soils and estimation of deposition rates in Kocaeli, Turkey. Chemosphere 2005;59:1373–85.

Bordajandi LR, Abad E, González MJ. Occurrence of PCBs, PCDD/Fs, PBDEs and DDTs in Spanish breast milk: enantiomeric fraction of chiral PCBs. Chemosphere 2008;70(4):567–75.

Brouwer A, Ahlborg U, van den Berg M, Birnbaum LS, Boersma ER, Bosveld B, et al. Functional aspects of developmental toxicity of polyhalogenated aromatic hydrocarbons in experimental animals and human infants. Environ Toxicol Pharmacol 1995;293(1):1-40.

Canadian Council of Ministers of the Environment. Canadian environmental quality guidelines for polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (PCDD/Fs). Technical supporting document. Ottawa: National Guidelines and Standards Office, Environmental Quality Branch; 2002.

Castro-Jiménez J, Mariani G, Eisenreich SJ, Christoph EH, Hanke G, Canuti E, et al. Atmospheric input of POPs into Lake Maggiore (Northern Italy): PCDD/F and dioxin-like PCB profiles and fluxes in the atmosphere and aquatic system. Chemosphere 2008;73:S122–30.

Chiarenzelli J, Alexander C, Ronalds A, Amespagano C, Ramirez W. Polychlorinated biphenyls in non accumulating, century-old sediments: sources, signatures, and mechanism of introduction. Sci Total Environ 2001;35:2903–8.

Çok İ, Donmez MK, Satıroğlu MH, Aydınuraz B, Henkelmann B, Kotalik J, et al. Concentration of polychlorinated dibenzo-*p*-dioxins, polychlorinated dibenzofurans and dioxin-like PCBs in human adipose tissue from Turkish men. Chemosphere 2007;66(10):1955–61.

Çok I, Dönmez KM, Demirkaya E, Henkelmann B, Shen H, Kotalik J, et al. Polychlorinated dibenzo-p-dioxin (PCDD), polychlorinated dibenzofuran (PCDF) and PCB levels in human milk in Turkey. Chemosphere 2009;76(11):1563–71.

El-Kady AA, Abdel-Wahhab MA, Henkelmann B, Belal MH, Morsi MKS, Galal SM, et al. Polychlorinated biphenyl, polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran residues in sediments and fish of the River Nile in the Cairo region. Chemosphere 2007;68:1660–8.

Evenset E, Christensen GN, Carrol J, Zaborska A, Berger U, Herzke D, et al. Historical trends in persistent organic pollutants and metals recorded in sediment from Lake Ellasjøen, Bjørnøya, Norwegian Arctic. Environ Pollut 2007;146:196–205.

Field JA, Sierra-Alvarez R. Microbial degradation of chlorinated dioxins. Chemosphere 2008;71(6):1005–18.

Fierens S, Mairesse H, Heilier J-f, De Burbure C, Focant J-f, Eppe G, et al. Dioxin/polychlorinated biphenyl body burden, diabetes and endometriosis: findings in a population-based study in Belgium. Biomarkers 2003;8(6):529–34.

Hornbuckle KC, Carlson DL, Swackhamer DL, Baker JE, Eisenreich SJ. PCBs in the Great Lakes. In: Hites RA, editor. persistent organic pollutants in the Great Lakes. Heidelberg: Springer-Verlag; 2006. p. 33–95.

Huang H, Buekens A. On the mechanism of dioxin formation in combustion processes. Chemosphere 1995;31(9):4099–117.

Kim E-J, Oh J-E, Chang Y-S. Effects of forest fire on the level and distribution of PCDD/Fs and PAHs in soil. Sci Total Environ 2003;311(1–3):177–89.

Kim K-S, Lee CS, Kim K-H, Shim JW, Hong HS, Choi HK, et al. Survey on organochlorine pesticides, PCDD/Fs, dioxin-like PCBs and HCB in sediments from the Han River, Korea. Chemosphere 2009;75:580–7.

Kwach OB, Lalah OJ. High concentrations of polycyclic aromatic hydrocarbons found in water and sediments of car wash and kisat areas of Winam Gulf, Lake Victoria-Kenya. Bull Environ Contam Toxicol 2009;83(5):727–33.

Li H, Yu L, Sheng G, Fu J, Peng P. Severe PCDD/F and PBDD/F pollution in air around an electronic waste dismantling area in China. Environ Sci Technol 2007;41(16):5641–6.

Li A, Rockne JK, Sturchio N, Song W, Ford CJ, Wei H. PCBs in sediments of the Great Lakes—distribution and trends, homolog and chlorine patterns, and in situ degradation. Environ Pollut 2009;157:141–7.

Lin L-F, Lee W-J, Li H-W, Wang M-S, Chang-Chien G-P. Characterization and inventory of PCDD/F emissions from coal-fired power plants and other sources in Taiwan. Chemosphere 2007;68(9):1642–9.

Miao XS, Chu SG, Xu XB. Degradation pathways of PCBs upon UV irradiation in hexane. Chemosphere 1999;39(10):1639–50.

Muir DCG, Marshall WK, Webster GRB. Bioconcentration of PCDDs by fish: effects of molecular structure and water chemistry. Chemosphere 1985;14:829–33.

National Environment Management Authority. Identification and quantification of dioxins and furans in Uganda: POPs inventory report, Kampala, Uganda; 2007.

Nguyen HL, Leermakers M, Osa´n J, Török S, Baeyens W. Heavy metals in Lake Balaton: water column, suspended matter, sediment and biota. Sci Total Environ 2005;340(1–3):213–30.

Niemirycz E, Jankowska D. Concentrations and profiles of PCDD/Fs in sediments of major Polish rivers and the Gdansk Basin-Baltic Sea. Chemosphere 2011;85(3):525–32.

Nieuwoudt C, Laura PQ, Pieters R, Jordaan I, Visser M, Kylin H, et al. Dioxin-like chemicals in soil and sediment from residential and industrial areas in central South Africa. Chemosphere 2009;76:774–83.

Nunes M, Marchand P, Vernisseau A, Bizec LB, Ramos F, Pardal AM. PCDD/Fs and dioxin-like PCBs in sediment and biota from the Mondego estuary (Portugal). Chemosphere 2011;83:1345–52.

Odada OE, Olago OD, Kulindwa K, Ntiba M, Wandiga S. Mitigation of environmental problems in Lake Victoria, East Africa: causal chain and policy options analyses. Ambio 2004;33:1–2.

Quaß U, Fermann M, Bröker G. The European dioxin air emission inventory project—final results. Chemosphere 2004;54:1319–27.

Ritter L, Solomn KR, Forget J, Stemeroff M, O'Leary C. Persistent organic pollutants: an assessment report on: DDT, aldrin, dieldrin, endrin, chlordane, heptachlor, hexachlorobenzene, mirex, toxaphene, polychlorinated biphenyls, dioxins and furans. Prepared for the international programme on chemical safety (IPICS), within the framework of the inter-organisation programme for sound management of chemicals (IOMC); 2007.

Sanctorum H, Elskens M, Leermakers M, Gao Y, Charriau A, Billon G, et al. Sources of PCDD/Fs, non-ortho PCBs and PAHs in sediments of high and low impacted transboundary rivers in Belgium–France. Chemosphere 2011;85(2):203–9.

Sormo EG, Larsen HJ, Johansen GM, Skaare JU, Jenssen BM. Immunotoxicity of polychlorinated biphenyls (PCBs) in free-ranging gray seal pups with special emphasis on dioxin-like congeners. J Toxicol Environ Health A 2009;72:266–76.

Telli-Karakoç F, Tolun L, Henkelmann B, Klimm C, Okay O, Schramm K-W. Polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) distribution in the Bay of Marmara sea: Ĭzmit Bay. Environ Pollut 2002;119:383–97.

van den Berg M, Birnbaum LS, Denison M, De Vito M, Farland W, Feeley M, et al. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. Toxicol Sci 2006;93(2):223–41.

Wasswa J, Kiremire BT, Nkedi-Kizza P, Mbabazi J, Ssebugere P. Organochlorine pesticide residues in sediments from the Ugandan side of Lake Victoria. Chemosphere 2010;82(1):130–6.

Wevers M, De Fré R, Desmedt M. Effect of backyard burning on dioxin deposition and air concentrations. Chemosphere 2004;54:1351–6.

Zhang S, Peng P, Huang W, Li X, Zhang G. PCDD/PCDF pollution in soils and sediments from the Pearl River Delta of China. Chemosphere 2009;75:1186–95.

Zhang H, Zhao X, Ni Y, Lu X, Chen J, Su F, et al. PCDD/Fs and PCBs in sediments of the Liaohe River, China: levels, distribution, and possible sources. Chemosphere 2010;79:754–62.

Zhou L, Li H, Yu Z, Ren M, Zeng X, Peng P, et al. Chlorinated and brominated dibenzo-*p*-dioxins and dibenzofurans in surface sediment from Taihu Lake, China. J Environ Monit 2012;14:1935–42.