



Anthropogenic footprint and ecological risk assessment of organochlorine pesticides and polychlorinated biphenyls in sediments from Lake Victoria, East Africa

Afuwa Kagoya^{a,b}, Kenneth Arinaitwe^c, Silver Odongo^d, Douglas Sifuna^e, Henry Matovu^f, Julius Matsiko^g, Godfrey Muhwezi^g, Ivan Špánik^h, Charles Drago Katoⁱ, Mika Sillanpää^{j,k,l}, Patrick Ssebugere^{a,*}

^a Department of Chemistry, Makerere University, P.O. Box 7062, Kampala, Uganda

^b Directorate of Research and Grants, Kabale University, P.O. Box 317, Kabale, Uganda

^c Thünen Institute of Fisheries Ecology, Bremerhaven, Germany

^d Department of Science and Vocational Education, Lira University, P.O. Box 1035, Lira, Uganda

^e Department of Physical Sciences, Jaramogi Oginga Odinga University of Science and Technology, P.O. Box 210 – 40601, Bondo, Kenya

^f School of Chemistry, University of Sydney, Sydney, NSW, 2006, Australia

^g Department of Chemistry, Muni University, P.O. Box 725, Arua, Uganda

^h Institute of Analytical Chemistry, Faculty of Chemical and Food Technology, Slovak University of Technology in Bratislava, Radlinského 9, 812 37, Bratislava, Slovakia

ⁱ School of Biosecurity, Biotechnical & Laboratory Sciences, College of Veterinary Medicine, Animal Resources & Biosecurity, Makerere University, P.O. Box 7062, Kampala, Uganda

^j Department of Chemical Engineering, School of Mining, Metallurgy and Chemical Engineering, University of Johannesburg, P. O. Box 17011, Doornfontein 2028, South Africa

^k Saveetha School of Engineering, Saveetha Institute of Medical and Technical Sciences, Saveetha University, Chennai, Tamil Nadu, 602105, India

^l Centre of Research Impact and Outcome, Chitkara University Institute of Engineering and Technology, Chitkara University, Rajpura, 140401, Punjab, India

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ABSTRACT

Lake Victoria, the world's second-largest freshwater lake, continues to face pressure from anthropogenic activities in its catchment area, releasing pollutants, which are ultimately trapped in the sediment core, thereby posing threats to aquatic organisms. This study quantified thirteen organochlorine pesticides (OCPs) and ten polychlorinated biphenyls (PCBs) in sediments from the Uganda, Kenya, and Tanzania sides of Lake Victoria using soxhlet method, fractionation column and GC-MS/MS analysis. Total ($\sum_{13}$) OCPs levels were up to 412, 148, and 522 $\mu\text{g kg}^{-1}$ dry weight (d.w), and 12.1, 8.69 and 9.87 $\mu\text{g kg}^{-1}$ dw for total ($\sum_{10}$) PCBs for sediments from Uganda, Kenya, and Tanzania, respectively. Diagnostic ratios suggested past and ongoing use of OCPs while principal component analysis confirmed that OCP profiles were mainly due to their application in controlling pests in agriculture and public health programs, and PCB congeners were largely due to volatilization, degradation of higher PCBs into lighter PCBs, improper waste disposal of old transformers, hydraulic fluids, plasticizers, and capacitors. Ecological risk assessment highlighted that the PCB levels in sediments were below threshold effect and probable effect levels, but *p,p'*-DDD, and lindane were likely to pose adverse effects to sediment-dwelling organisms in Lake Victoria. These results imply persistent pollutant loads in Lake Victoria, and consequently, a need for its enhanced management.

1. Introduction

Organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs) are part of the wider persistent organic pollutants (POPs), which are semi-volatile in nature, have a bio-accumulation potential in fatty

tissues of living organisms, can undergo long-range transport and do not degrade easily (Merhaby et al., 2019). Globally, there are 209 known PCB congeners (Kumari and Swamy, 2024). Due to their toxic and persistent nature, OCPs and PCBs were banned globally in 2004 by the Stockholm Convention on POPs (FAO, 2024). However, they have been

* Corresponding author.

E-mail address: patrick.ssebugere@mak.ac.ug (P. Ssebugere).

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broadly detected in various environmental matrices due to their long half-lives and past usage in several anthropogenic activities (Osoro et al., 2016; Ssebugere et al., 2010; Wenaty, 2019b).

In East Africa, some potential sources of OCPs contamination originate from their application in agriculture and health programs. For example, Uganda adopted the use of OCPs in the 1950s and 1960s to boost food production and curb mosquitoes and tsetse flies (Mukiibi et al., 2021; Kasozi et al., 2006). Dichlorodiphenyltrichloroethane (DDT) was widely used in Uganda until the mid-1980s for pest management in agriculture, and in the control of malaria-causing mosquitoes and tsetse flies (Ssebugere et al., 2010), and was reintroduced in 2008 for indoor residual spraying, where about 24 t were used nationwide (Van Den Berg et al., 2017). A recent report by the Food and Agricultural Organization (FAO) highlighted a substantial increase in Uganda's pesticide imports from about 2990 tons in 2010 to 6009 tons in 2022, highlighting the country's dependence on pesticides to improve agricultural production (FAO, 2024). On the other hand, PCBs have for long been used in industrial applications such as capacitors, hydraulic fluids in electronic equipment, coolants, lubricants in transformers, and as additives in paints because they are heat resistant, chemically inert, non-flammable, have a low vapour pressure, and high dielectric constant (Lundin et al., 2013), and they can find their way into the atmosphere as unintentional byproducts of waste incineration and combustion (Mao et al., 2020). A study by Arinaitwe et al. (2018) reported the occurrence of PCBs in the atmospheric environment of Lake Victoria. Generally, the environmental contamination of PCBs is well documented in East Africa (Arinaitwe et al., 2018; Ndunda and Wandiga, 2020; Polder et al., 2016; Ssebugere et al., 2014a).

Once released in the environment, OCPs and PCBs can reach the aquatic environment via direct discharges, surface runoffs, and atmospheric deposition (Werimo et al., 2009), where they can adsorb onto sediments and organic matter due to their low solubility in water (Barakat et al., 2012). Since sediments are the ultimate sink for POPs, they can provide insights into the accumulation of POPs over a long period of time, and human-driven activities leading to environmental changes (Arinaitwe et al., 2016; Tian et al., 2025). Some of the health effects due to the exposure of living organisms to OCPs and PCBs include carcinogenicity, reproductive impairment, immunotoxicity, and endocrine disruption (Helmfrid et al., 2012; Mrema et al., 2013). In birds, these chemicals are known to interfere with their endocrine, immune, and neural systems as well as their reproduction and growth (Hao et al., 2021). Aquatic organisms get exposed to these toxic chemicals via sediments, and because of their potential for bioaccumulation, they tend to travel via the food web, thereby exposing humans as well (Mrema et al., 2013).

The Lake Victoria basin and its tributary rivers are a main biodiversity hotspot, constituting about 500 fish species (with at least 234 native fish species), 135 native aquatic plant species, and 50 native freshwater mollusc species (Van Soesbergen et al., 2019). However, the Lake Victoria basin is increasingly under threat from urbanisation, unsustainable land conversion, and intensification of agriculture, which are driven by the high population growth (Van Soesbergen et al., 2019). Previous studies have reported OCPs contamination in surface water and fish species from Lake Victoria (Kasozi et al., 2006; Osoro et al., 2016; Wenaty, 2019b; Wenaty et al., 2019a) while others have documented environmental contamination of PCBs in East Africa (Arinaitwe et al., 2018; Ndunda and Wandiga, 2020; Polder et al., 2016; Ssebugere et al., 2014a). Some of the reported levels of the legacy OCPs and PCBs were below the recommended thresholds (Threshold effect level, probable effect level, effects range media and effects range level), whereas other concentrations were elevated, and often associated with historical and current usage. However, these studies had many limitations, for instance, they were mainly area specific, used different sampling periods, had different sediment characteristics and analytical methods, thereby limiting crucial regional comparison. To date, no single study has assessed these pollutants at a tri-national scale across Uganda, Kenya

and Tanzania.

This study, therefore, addresses this gap as it reports the first tri-national assessment of OCPs and PCBs on the Lake Victoria sediments, thereby presenting a comprehensive regional comparison of these pollutants and providing insights into their environmental fate and potential risks. The specific objectives of this study were to; (i) determine the concentrations and assess the spatial profiles, (ii) assess ecological risks, and (iii) identify possible pollution sources using diagnostic ratios and principal component analysis of 13 OCPs and 10 PCBs in 41 surface sediments from the selected parts of Lake Victoria.

2. Materials and methods

2.1. Study area and sample collection

Lake Victoria (Fig. 1), the world's second-largest freshwater lake, has a catchment area of about 193,000 sq. km and a surface area of 68,800 sq. km, which exposes it to wet and dry atmospheric deposition of chemical pollutants. In addition, the lake's inlets have vast drainage basins, which are contaminated with industrial chemicals and pesticides (Arinaitwe et al., 2016), yet millions of people depend on this lake for drinking water and fish for both local consumption and export. Furthermore, the lake's catchment area has undergone rapid urbanization, and increased agricultural activities due to the rapidly increasing population (Juma et al., 2014; Kiggundu et al., 2018; Njagi et al., 2022). Several agricultural farms surround the Lake Victoria basin due to its fertile soils, and surface runoffs during rainy periods could be emitting OCPs into the lake. In addition, industries such as sugar manufacturing, food processing, leather and tanning industries, breweries, and textile manufacturing might all be contributing to the deposition of PCBs into the lake (Ssebugere et al., 2014b, 2014a; Wasswa et al., 2011). Current broad monitoring on wastewater flow is limited, however, available approximates show that these industries altogether release significant effluents into the lake environment. For instance, past assessments reported that major breweries discharged approximately 1000–5000 m³/day, textile industries about 2000 m³/day or more, approximately 500–2000 m³/day for sugar factories, and leather tanneries were estimated at about 400 m³/day of wastewater (Ministry of Natural Resources, 1995). These discharges contribute to annual reported industrial loads of about 1487 t/yr in Uganda, 860 t/yr in Kenya and 3259 t/yr BOD in Tanzania, indicating the magnitude of industrial pollution on Lake Victoria (Kayombo and Jorgensen, 2005). Despite the limited data on exact effluent volumetric discharges from industries, the available data evidently show that industries around the lake's catchment release substantial amounts of effluents, which could include PCBs.

In total, forty-one surface sediments (<10 cm depth) were collected from 41 sampling sites (one sample per site) using a sediment corer from different parts of the Uganda, Kenya, and Tanzania sides of the lake (Table S1). The sampling sites were selected due to their closeness to urban centres, agricultural farms and busy industrial centres. Samples were collected during the wet season (September–October 2019) due to the increased surface runoffs from the agricultural fields and urban centres that could possibly lead to higher pollutant loads in the lake. The collected sediment samples were wrapped in aluminium foil, put in labelled zip-lock bags and transported to the Department of Chemistry, Makerere University, Kampala-Uganda and freeze-dried.

2.2. Standards, chemicals and reagents

Reference standard mixtures containing thirteen OCPs (aldrin, dieldrin, endrin aldehyde, heptachlor, heptachlor exo-epoxide, α -HCH, lindane (γ -HCH), α -endosulfan, β -endosulfan, endosulfan sulfate, p,p' -DDD, p,p' -DDE, and p,p' -DDT), and ten PCB congeners (PCB-28, -53, -104, -112, -115, -119, -140, -154, -184, and -198) were purchased from AccuStandard (New Haven, CT, USA).

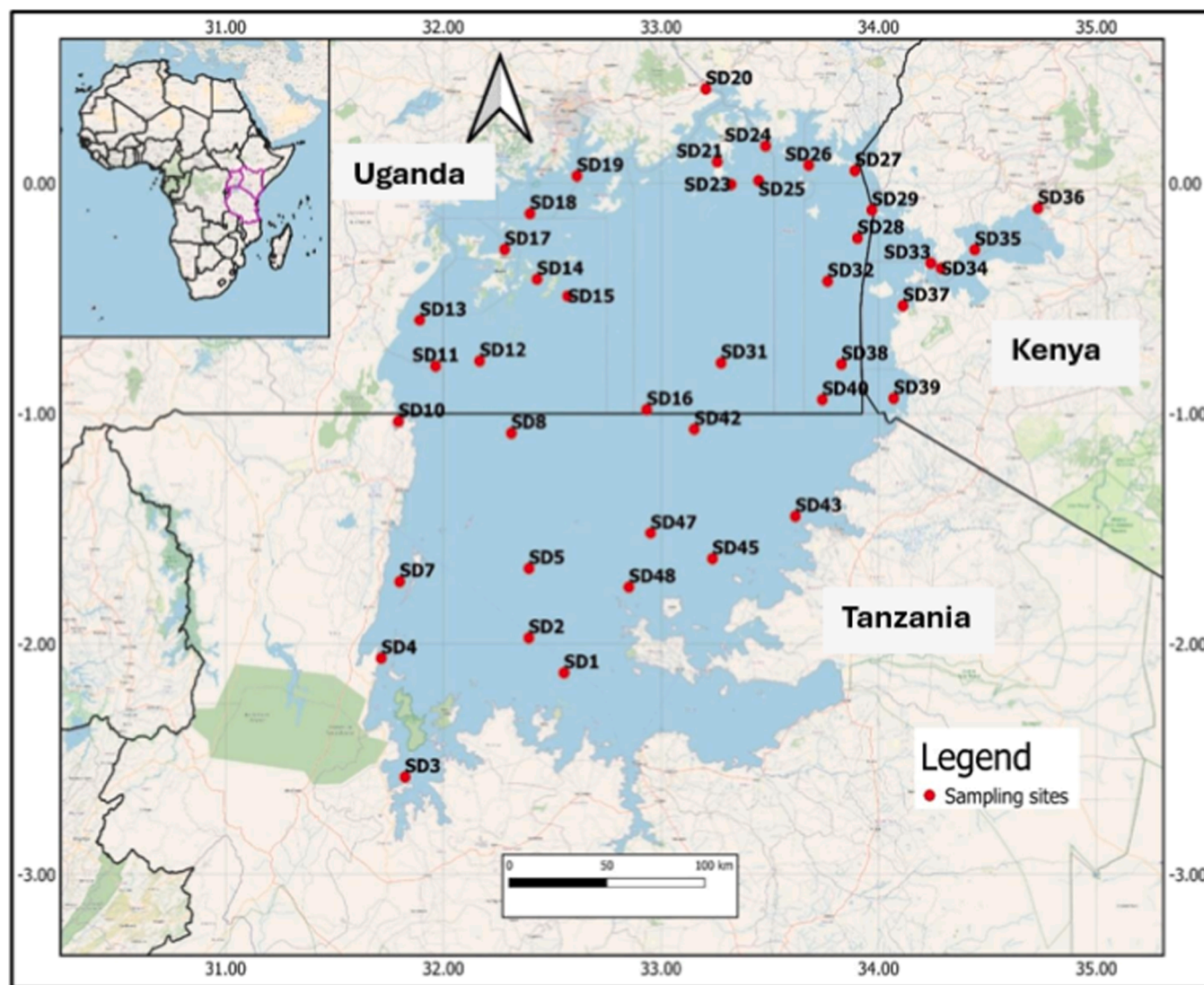


Fig. 1. A map of Lake Victoria showing the sampling sites.

Pentachloronitrobenzene (PCNB) and tetrachloro-*m*-xylene were used as internal standards and were also bought from AccuStandard. Working solutions were prepared by serial dilution in *n*-hexane. Pesticide residue grade solvents (acetone, diethyl ether, dichloromethane (DCM), *n*-hexane, and isooctane); and analytical grade silica gel, and anhydrous sodium sulfate were bought from Sigma Aldrich (St. Louis, MO).

2.3. Analytical work on sediment samples

2.3.1. Extraction and clean-up for OCPs and PCBs,

Extraction of individual sediment samples for OCPs and PCBs was done following the United States Environmental Protection Agency (U.S EPA) Method 8080 (Lopez-Avila et al., 1988), but with slight modifications. Briefly, a freeze-dried sediment sample (5 g) was homogenized, and a mixture of PCNB and tetrachloro-*m*-xylene (50 μ L) internal standards was spiked onto it, followed by the addition of activated copper powder to remove elemental sulfur. Extraction was done using a Soxhlet apparatus at 50 $^{\circ}$ C. A mixture (175 mL) of *n*-hexane and acetone (3:1, v/v) was added to the apparatus, followed by boiling stones. The extraction was allowed to proceed overnight for about 16 h. Isooctane (2 mL) was added to the extract as a keeper and then evaporated to 3 mL using a rotary evaporator. The extract was then transferred to a clean tube and rinsed thrice with *n*-hexane (1 mL).

Clean-up was done using a column packed with aluminium oxide. The column was conditioned first with *n*-hexane (15 mL), then followed by rinsing with *n*-hexane (1 mL) thrice. The extract was then loaded onto the column and eluted with *n*-hexane (170 mL). Isooctane (2 mL) was added to the clean extract in the flask, evaporated to near dryness using a gentle stream of nitrogen and then reconstituted to 1 mL with *n*-hexane.

2.3.2. Fractionation of OCPs and PCBs

Fractionation of the extract was performed using a fractionation column containing 1.8 g of 1.5 % deactivated silica gel and baked sodium sulfate on top. The column was conditioned with *n*-hexane (4 mL) before loading the extract. Elution was repeated twice with *n*-hexane (10 mL) into a centrifuge tube (fraction 1). The second elution was carried out using a 10 mL mixture of *n*-hexane and diethyl ether (85:15, v/v) (fraction 2). Fraction 1 contained PCBs while fraction 2 contained OCPs. Isooctane (1 mL) was added to the two fractions and each evaporated to 0.5 mL under a gentle stream of nitrogen. The two fractions were then separately transferred into labelled vials for GC-MS/MS analysis.

2.3.3. GC-MS/MS analysis

Extracts were analyzed for PCBs and OCPs using Agilent Intuvo 9000 Gas Chromatograph system interfaced with a triple quadrupole mass

spectrometry, using an electron impact ion source under 70 eV operated in a multiple reaction monitoring mode. Chromatographic separation was achieved on a DB-5MS fused silica capillary column (30 m $\times$ 0.25 mm i.d. $\times$ 0.25 μ m film thickness, Agilent Technologies). Pure helium (99.999 %) was used as a carrier gas with a flow rate of 1.2 mL min⁻¹. The column temperature was initially set at 40 °C held for 2 min, and then increased at a rate of 20 °C min⁻¹ to 230 °C held for 3 min, and then further raised at 5 °C min⁻¹ to 260 °C held for 4 min, then finally to 310 °C for 2 min. The injector and detector temperatures were maintained at 290 and 310 °C, respectively. The GC was operated in a split less mode with the injection volume of 5 μ L per injection of clean extracts, with a source temperature of 280 °C, 3 mL min⁻¹ septum purge flow, and a pressure of 11.339 psi. Agilent MassHunter Quantitative Analysis software was used for data acquisition and processing. Identification of the OCPs and PCBs was performed by comparing the retention times and fragmentation patterns of peaks in the chromatograms of the measured sample extracts with those in the calibration standards. Quantitative analysis was done by the internal standard method.

2.4. Quality assurance and quality control (QA/QC)

Procedural blanks were analyzed for every set of 5 samples to check for any cross-contamination. All the target analytes were below LOD in the blanks, and thus, the generated datasets were not blank-corrected. Seven-point calibration curves were constructed with the concentrations ranging from 1 to 100 μ g kg⁻¹. Linearity was evaluated based on the coefficients of correlation (r^2) of the calibration curves for each analyte, and were all greater than 0.99. Recovery studies were done to confirm the efficiency of the method by spiking QC sediment samples with 50 μ L of a standard solution of a mixed standard (5 and 100 μ g kg⁻¹) of OCPs and PCBs in triplicates. The mean recoveries of PCBs ranged from 65 $\pm$ 2.8 % to 112 $\pm$ 5.7 % while those for OCPs varied from 60.6 $\pm$ 3.8 % to 130 $\pm$ 4.0 %. The recoveries obtained in this study were within the range of values (60 %–120 %) reported in the U.S EPA Method 8080D, and values (70 %–130 %) recommended by the U.S EPA method 8000D (De Araujo et al., 2020). Therefore, the data obtained was not corrected for recovery. The limit of detection (LOD) was calculated as 3 times the signal-to-noise ratio (S/N), while the limit of quantification (LOQ) was calculated as 10 times S/N. The LODs for OCPs ranged from 0.005 to 0.05 μ g kg⁻¹dw, and those for PCBs ranged from 0.004 to 0.036 μ g kg⁻¹dw. While the LOQ values for OCPs ranged from 0.015 to 0.150 μ g kg⁻¹ dw, and those for the PCBs ranged from 0.012 to 0.108 μ g kg⁻¹dw (Table S2 of the Supplementary Material).

2.5. Statistical data analysis

Statistical analysis was done using IBM SPSS statistical software version 23 (IBM, Chicago IL, USA), OriginPro®2024 software (Origin-Lab Corp., Northampton, MA, USA) and Microsoft Excel version 21 (Microsoft Corporation, Redmond, WA USA). Analytes were considered quantifiable when their concentrations were above LOD, while non-detects were taken to be ½ LOD. Normality of the data was tested using the Shapiro-Wilk test. Since the data were not normally distributed, a non-parametric Kruskal-Wallis test was used to assess differences in the levels of OCPs and PCBs among the sampling sites. Statistical significance was considered when the *p*-value was < 0.05.

2.6. Assessment of ecotoxicological risks of OCPs and PCBs to aquatic organisms

The Canadian sediment quality guidelines (SQGs) for the protection of aquatic life issued by the Canadian Council of Ministers of the Environment (CCME) were used to assess the ecotoxicological risks of OCPs and PCBs to the aquatic organisms (MacDonald et al., 2000). In addition, the consensus-based sediment quality guidelines were also used. The guidelines entail threshold effect level (TEL), a level below which

adverse effects are not expected to occur more often, and probable effect level (PEL), a level above which adverse effects are expected to occur more often (Hellar-Kihampa et al., 2013). POPs whose concentration exceed these set guidelines adversely affect aquatic organisms and consequently affect the food web, such as fish, and humans, as the cycle continues. The SQGs used for the comparison of the levels of OCPs and PCBs in the investigated sediments are presented in Table S4 of the Supplementary Material.

2.7. Identification of possible sources of OCPs and PCBs

For OCPs (DDTs, endosulfans, and HCHs), diagnostic ratios (DRs) were used as qualitative source identifiers to distinguish between the past and recent usage of pesticides in the environment and to specify whether there was significant degradation of parent compounds. A *p,p'*-DDE/*p,p'*-DDT ratio > 1 generally indicates past use of technical DDT whereas the ratio < 1 suggests recent use (Xuan et al., 2010). For endosulfans, a ratio of α -/ β -endosulfan < 2.3 indicates past input, and that >2.3 suggests recent input of technical endosulfan (Zhonghua et al., 2016). For HCHs, α -/ γ -HCH ratios ranging from 4–7 indicate fresh input, while ratios <3 highlight past usage (Yang et al., 2019). Principal component analysis (PCA) was conducted using OriginPro and SPSS software, using the correlation matrix method. The direct oblimin method with Kaiser Normalization was used and a sampling adequacy greater than 0.5 was considered. The Bartlett's test of sphericity was used to demonstrate if the correlations between the analytes were large enough for PCA. The pattern matrix table was used to explain the loadings of analytes onto the extracted components. Loadings were considered significant when the pattern matrix values were >0.5. Principal components were retained when their eigen values were greater than 1, following the Kaiser criterion.

3. Results and discussion

3.1. Levels of OCPs in the sediment samples

The levels of the investigated OCPs in the sediments are presented in Table 1. The levels of total ($\sum_{13}$) OCPs in the sediment samples from Uganda ranged from 17.9 to 412 μ g kg⁻¹ dry weight (d.w) (with a mean level of 141 $\pm$ 106 μ g kg⁻¹ dw), while for Kenya levels varied from 19.7 to 148 μ g kg⁻¹ dw (with a mean level of 81.4 $\pm$ 48.1 μ g kg⁻¹ dw), and for Tanzania from 14.1 to 522 μ g kg⁻¹ dw (with a mean level of 132 $\pm$ 155 μ g kg⁻¹ dw). *p,p'*-DDD was the most abundant OCP residue detected in all the sediments from the three countries. Precisely, *p,p'*-DDD contributed 88.2 %, 82.9 %, and 72.9 % to the $\sum_{13}$ OCPs in sediments from Uganda, Kenya, and Tanzania, respectively. The mean levels of OCPs varied across the lake, with α -endosulfan and *p,p'*-DDD showing high standard deviation (SD) values (Table 1). The high SD values indicate significant spatial variability, likely attributed to the differences in the sampling sites, sediment characteristics, pollution sources, and environmental conditions such as spatial and temporal changes (Hu and Tao, 2022). The Kruskal-Wallis test revealed significant differences in OCP concentrations among sampling sites (*p* < 0.05), indicating spatial variability in OCP distribution across the lake. The variations in the levels of the OCPs identified in this study are shown on the spatial map (Fig. 2).

The mean levels of total DDTs were 128 $\pm$ 101, 71.9 $\pm$ 46.5, and 99.3 $\pm$ 78.3 μ g kg⁻¹ dw in sediments from Uganda, Kenya, and Tanzania, respectively. *p,p'*-DDD was the most dominant metabolite quantified in the sediments (contributing 95.9 %, 93.0 %, and 95.2 % to $\sum$ DDTs for samples from Uganda, Kenya, and Tanzania, respectively). The predominance of *p,p'*-DDD could be attributed to the past technical DDT use in the lake's catchment area which could have degraded into its metabolites. Furthermore, the predominance of DDT compared to other OCPs in this study could be attributed to its hydrophobic nature with a strong affinity for sediments and its chemical stability leading to low degradation in the environment. In addition, DDT has a high partition

Table 1
Levels of OCP residues ($\mu\text{g kg}^{-1}$ dw) in sediments from Lake Victoria.

OCP residues	Uganda (N = 21)		Kenya (N = 7)		Tanzania (N = 13)	
	Min – Max	Mean $\pm$ SD	Min – Max	Mean $\pm$ SD	Min – Max	Mean $\pm$ SD
Aldrin	n.d – 1.03	0.26 $\pm$ 0.22	0.16 – 0.19	0.17 $\pm$ 0.01	n.d – 0.25	0.18 $\pm$ 0.03
Dieldrin	n.d – 2.89	1.81 $\pm$ 0.83	n.d	n.d	n.d – 1.60	1.49 $\pm$ 0.09
Endrin aldehyde	n.d – 1.64	0.84 $\pm$ 0.29	n.d – 0.78	0.74 $\pm$ 0.39	n.d – 0.75	0.73 $\pm$ 0.01
Σ DRINs	n.d – 5.56	2.91 $\pm$ 1.34	n.d – 0.97	0.91 $\pm$ 0.41	n.d – 2.60	2.39 $\pm$ 0.13
Heptachlor	n.d – 1.24	1.20 $\pm$ 0.36	n.d – 1.19	1.19 $\pm$ 0.58	n.d – 1.20	1.18 $\pm$ 0.33
Heptachlor exo-epoxide	n.d – 1.63	1.10 $\pm$ 0.28	n.d – 1.04	1.04 $\pm$ 0.39	n.d – 1.25	1.07 $\pm$ 0.30
Σ CHLs	n.d – 2.87	2.30 $\pm$ 0.64	n.d – 2.23	2.23 $\pm$ 0.97	n.d – 2.45	2.25 $\pm$ 0.63
α -HCH	n.d – 1.44	1.38 $\pm$ 0.30	1.36 – 1.38	1.37 $\pm$ 0.01	1.36 – 1.43	1.38 $\pm$ 0.02
Lindane	1.78 – 2.58	1.92 $\pm$ 0.21	1.78 – 1.82	1.80 $\pm$ 0.02	1.75 – 1.82	1.79 $\pm$ 0.02
Σ HCHs	1.78 – 4.02	3.30 $\pm$ 0.51	3.14 – 3.50	3.17 $\pm$ 0.03	3.11 – 3.25	3.19 $\pm$ 0.04
α -endosulfan	n.d – 7.67	2.29 $\pm$ 1.71	1.37 – 1.84	1.52 $\pm$ 0.18	1.34 – 217	18.5 $\pm$ 59.8
β -endosulfan	0.79 – 5.01	1.13 $\pm$ 0.92	0.80 – 0.90	0.83 $\pm$ 0.04	0.79 – 56.0	5.10 $\pm$ 15.3
Endosulfan sulfate	0.83 – 1.92	1.00 $\pm$ 0.33	0.84 – 0.89	0.85 $\pm$ 0.02	0.83 – 2.43	0.97 $\pm$ 0.44
Σ Endosulfans	1.62 – 14.6	4.42 $\pm$ 2.96	3.00 – 3.63	3.19 $\pm$ 0.23	2.97 – 275	24.6 $\pm$ 75.5
<i>p,p'</i> -DDD	11.3 – 376	123 $\pm$ 99.0	10.4 – 131	66.8 $\pm$ 41.4	5.73 – 228	94.8 $\pm$ 75.8
<i>p,p'</i> -DDE	2.48 – 7.49	4.16 $\pm$ 1.19	2.33 – 5.60	4.11 $\pm$ 1.03	1.56 – 6.40	3.14 $\pm$ 1.35
<i>p,p'</i> -DDT	0.73 – 1.97	1.13 $\pm$ 0.31	0.82 – 1.18	0.95 $\pm$ 0.14	0.73 – 5.01	1.32 $\pm$ 1.12
Σ DDTs	14.5 – 385	128 $\pm$ 101	13.6 – 138	71.9 $\pm$ 46.5	8.02 – 239	99.3 $\pm$ 78.3
Σ OCPs	17.9 – 412	141 $\pm$ 106	19.7 – 148	81.4 $\pm$ 48.1	14.1 – 522	132 $\pm$ 155

SD – Standard deviation, Min – Minimum, Max – Maximum, n.d – non-detectable.

coefficient which facilitates its adsorption onto the sediments. The patterns of DDT and its metabolites (*p,p'*-DDD and *p,p'*-DDDE) observed in the sediments are also strongly influenced by in-situ degradation and transformation processes (Tcaciuc et al., 2018). In the environment, DDT is known to undergo two major degradation pathways depending on the prevailing sediments conditions. Under oxidizing conditions, *p,p'*-DDT primarily breaks down into *p,p'*-DDE, whereas under reducing environments, reductive dechlorination promotes technical DDT transformation to *p,p'*-DDD (Huang et al., 2001; Tcaciuc et al., 2018). Therefore, sediments with higher concentrations of *p,p'*-DDE are indicative of oxidative degradation or longer environmental aging of historical DDT inputs, while dominance of *p,p'*-DDD implies that the sediments are more anoxic and favour reductive transformation processes (Li et al., 2010).

The mean level of Σ endosulfans in sediments from Uganda was $4.42 \pm 2.96 \mu\text{g kg}^{-1}$ dw, with α -endosulfan quantified as the most dominant (50.6 %), then followed by β -endosulfan (26.2 %). The mean level of the Σ endosulfans in sediments from Kenya was $3.19 \pm 0.23 \mu\text{g kg}^{-1}$ dw, with α -endosulfan as the most dominant (47.5 %), then followed by endosulfan sulfate (26.6 %), while the mean level of the Σ endosulfans in sediments from Tanzania was $24.6 \pm 75.5 \mu\text{g kg}^{-1}$ dw, with α -endosulfan as the most dominant (78.8 %), followed by β -endosulfan (20.3 %). The predominance of α -endosulfan could be attributed to its high percentage (70 %) in the technical endosulfan formulation (Wan et al., 2005), a result of recent usage implying that most of the technical

endosulfan in the environment had not been fully degraded. In the environment, endosulfan isomers can undergo oxidation to a more stable and environmentally persistent metabolite, endosulfan sulfate (Peterson and Batley, 1993). Due to its low stability in the environment, the α -isomer degrades more rapidly than β -endosulfan (Peterson and Batley, 1993). Although anaerobic conditions slow the overall transformation, aerobic sediments promote faster conversion of endosulfan isomers to endosulfan sulfate (Ghadiri, 2001). Therefore, the dominance of endosulfan sulfate in the several sampling sites could indicate both historical use of endosulfan and strong persistence of its metabolite/natural oxidation.

The mean levels of Σ HCHs in sediments were 3.30 ± 0.51 , 3.17 ± 0.03 , and $3.19 \pm 0.04 \mu\text{g kg}^{-1}$ dw for Uganda, Kenya, and Tanzania, respectively. γ -HCH was the most dominant HCH isomer contributing 59.3 % for Uganda, 56.8 % for Kenya, and 56.6 % for Tanzania to Σ HCHs loads. The predominance of γ -HCH could be attributed to its high percentage in technical HCH formulation. The observed low levels of α -HCH in the sediments could be due to its high vapor pressure and Henry's law constant, and therefore readily lost via volatilization (Ndunda et al., 2018). Regarding the sum of dieldrin, aldrin and endrin aldehyde (DRINs), for Uganda, the mean level of Σ DRINs in the sediments was $2.91 \pm 1.34 \mu\text{g kg}^{-1}$ dw. The detection frequency of aldrin (80 %) was higher than that of dieldrin metabolite (23.8 %). The mean level of total heptachlor and heptachlor exo epoxide (Σ CHLs) in the sediments was $2.30 \pm 0.64 \mu\text{g kg}^{-1}$ dw for Uganda. For Kenya, the mean levels of Σ CHLs and Σ DRINs were 2.23 ± 0.97 , and $0.91 \pm 0.41 \mu\text{g kg}^{-1}$ dw, respectively, while for sediments from Tanzania, the mean levels of Σ CHLs and Σ DRINs were 2.25 ± 0.63 , and $2.39 \pm 0.13 \mu\text{g kg}^{-1}$ dw, respectively. Dieldrin was not detected at any site in Kenya, possibly suggesting a recent input of aldrin in the area, and could imply that aldrin had not been converted to its metabolite (dieldrin) in the environment. However, a few sites in Uganda and Tanzania had detectable residues of dieldrin, suggesting past usage of aldrin, with higher levels recorded on the Uganda side of Lake Victoria.

In comparison to related studies elsewhere (Table S5 of the Supplementary Material), the levels of OCPs quantified in the sediments from this study were generally higher than those reported in sediments from the Awash river, Ethiopia (Dirbaba et al., 2018), mangrove reserves, China (Qiu et al., 2019), Huaihe river, China (Zhonghua et al., 2016), Pearl river delta, China (Wang et al., 2017), Volturno river, Italy (Montuori et al., 2020), and Someș Mic river, Romania (Barhoumi et al., 2019). However, they were lower than those found in Richards Bay, South Africa (Mehlhorn et al., 2023), and in Pearl river delta, China (Wang et al., 2017). The OCP levels were comparable to some levels in Pearl river delta, China (Wang et al., 2017), Nairobi river, Kenya (Ndunda et al., 2018), and Kuja-Migori river, Kenya (Kiyuka Nyaundi et al., 2021), reflecting limited recent use of most OCPs and more advanced degradation. The observed OCP values from the Kenya side of Lake Victoria in this study were close to those reported in Lake Chaohu, Eastern China (Li et al., 2015). Although these comparisons highlight that Lake Victoria is moderately contaminated, they remain indicative because the differences in sediment properties, sampling periods, and analytical methods among studies influence absolute concentrations.

3.2. Levels of PCBs in the sediment samples

As seen in Table 2, the levels of Σ_{10} PCBs in sediment samples from Uganda varied from 3.64 to $12.1 \mu\text{g kg}^{-1}$ dw (with a mean level of $8.16 \pm 2.86 \mu\text{g kg}^{-1}$ dw), while for Kenya varied from 3.10 to $8.69 \mu\text{g kg}^{-1}$ dw (with a mean level of $7.74 \pm 2.17 \mu\text{g kg}^{-1}$ dw), and that for Tanzania varied from 2.46 to $9.87 \mu\text{g kg}^{-1}$ dw (with a mean level of $7.80 \pm 2.19 \mu\text{g kg}^{-1}$ dw). The levels of higher chlorinated PCBs were highest at sites close to busy industrial centers such as in Musoma, Mwanza port, a fast growing town in Tanzania, and the second largest urban settlement after Dar es Salaam, with many industries (Haule and Kilonzo, 2024), and Kisii, Nyanza region, Kenya, which dispose off industrial wastes through

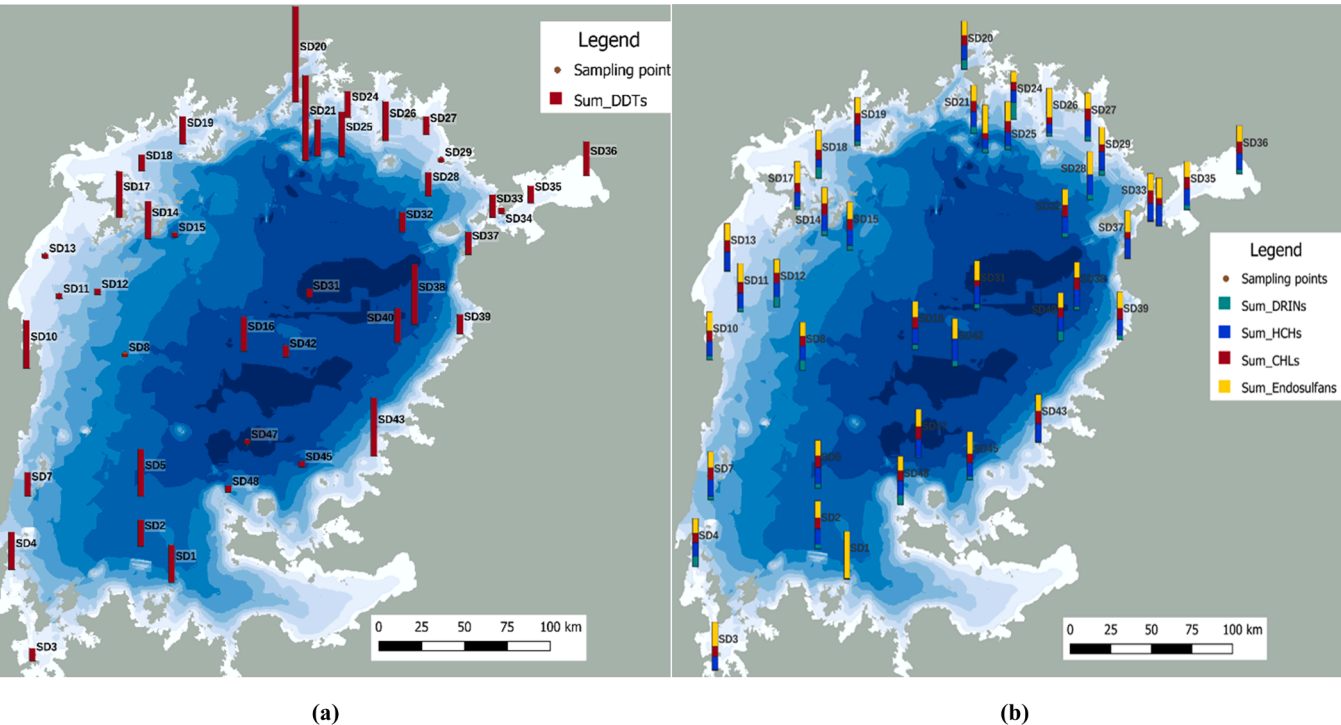


Fig. 2. A map of Lake Victoria showing spatial distributions of (a) Σ DDTs, and (b) Σ DRINs, Σ HCHs, Σ CHLs, and Σ Endosulfans in sediments.

Table 2
Levels of PCBs ($\mu\text{g kg}^{-1}$ dw) in sediments from Lake Victoria.

Analyte	Uganda (N = 21)		Kenya (N = 7)		Tanzania (N = 13)	
	Min – Max	Mean $\pm$ SD	Min – Max	Mean $\pm$ SD	Min – Max	Mean $\pm$ SD
PCB 28	0.66 – 1.34	0.86 $\pm$ 0.19	0.64 – 0.86	0.73 $\pm$ 0.07	0.63 – 0.84	0.72 $\pm$ 0.07
PCB 53	0.55 – 0.74	0.62 $\pm$ 0.05	0.56 – 0.60	0.58 $\pm$ 0.01	0.55 – 0.61	0.58 $\pm$ 0.02
PCB 104	n.d – 0.92	0.79 $\pm$ 0.18	n.d – 1.01	0.82 $\pm$ 0.32	0.72 – 0.84	0.76 $\pm$ 0.04
PCB 112	0.56 – 3.30	0.81 $\pm$ 0.62	n.d – 0.77	0.62 $\pm$ 0.25	n.d – 1.44	0.74 $\pm$ 0.33
PCB 115	0.56 – 0.77	0.63 $\pm$ 0.05	0.56 – 0.85	0.62 $\pm$ 0.10	0.56 – 0.65	0.59 $\pm$ 0.03
PCB 119	n.d – 0.58	0.51 $\pm$ 0.11	0.50 – 0.58	0.52 $\pm$ 0.03	n.d – 0.59	0.51 $\pm$ 0.15
PCB 140	n.d – 0.36	0.26 $\pm$ 0.09	0.18 – 0.27	0.22 $\pm$ 0.03	n.d – 0.43	0.23 $\pm$ 0.11
PCB 154	n.d – 0.87	0.73 $\pm$ 0.23	0.66 – 0.76	0.70 $\pm$ 0.04	n.d – 0.79	0.66 $\pm$ 0.28
PCB 184	0.66 – 1.11	1.02 $\pm$ 0.37	n.d – 1.07	1.02 $\pm$ 0.39	n.d – 1.19	1.04 $\pm$ 0.30
PCB 198	0.66 – 2.13	1.93 $\pm$ 0.96	n.d – 1.93	1.91 $\pm$ 0.93	n.d – 2.50	1.97 $\pm$ 0.88
Σ PCBs	3.64 – 12.1	8.16 $\pm$ 2.86	3.10 – 8.69	7.74 $\pm$ 2.17	2.46 – 9.87	7.80 $\pm$ 2.19

SD - Standard deviation, Min – Minimum, Max – Maximum, n.d – non-detectable.

the various channels into the site. Moreover, these industries could also be using old transformers, which might possibly be emitting PCBs into the lake basin. In addition, the presence of higher chlorinated PCBs could be attributed to their low water solubility, high adsorption rate for sediments, and very low degradability rate compared to the lower chlorinated PCB congeners (Erickson and Kaley, 2011). Dechlorination also affects the distribution of PCB concentrations in the sediments. In fact, higher-chlorinated PCB congeners can undergo reductive dechlorination to form lower-chlorinated PCBs (Jing et al., 2018). However,

more oxidizing environments enhance the persistence of the original higher-chlorinated congeners, since aerobic biodegradation of PCBs is always slower (Field and Sierra-Alvarez, 2008). Therefore, the observed PCB levels across the sampling sites could reflect the extent of these dechlorination processes in addition to their sources.

The levels of PCBs in the sediments were in the order: octa-CB > hepta-CB > hexa-CB > tri-CB > penta-CBs > tetra-CB on the Uganda side of the lake. PCB-198 was the most dominant congener (16.8 %), followed by PCB-184 (12.3 %), while PCB-140 (3.2 %) had the lowest contribution to the Σ PCBs. A similar trend was observed in sediments from Kenya where PCB-198 was the most dominant congener (19.9 %), followed by PCB-184 (12.8 %), while PCB-140 had the lowest contribution (3.3 %) (Fig. 3). The predominance of PCB-198 followed by PCB-184 could be attributed to their high hydrophobicity, water partition coefficient (K_{ow}) value and sorption to the organic matter in sediments (Menéndez-Pedriz and Jaumot, 2020). The results showed that lower chlorinated PCBs, such as the di- and tri-CBs, were more dominant at some sites than the higher chlorinated congeners, an indicator that there were no significant point sources of PCBs, but also indicated their ease of transport to longer distances from their points of contamination. The fact that PCBs are semi-volatile persistent pollutants, their solubility and volatility tend to decrease as the number of chlorine atoms increases, thus, the lighter chlorinated congeners easily volatilize in air and accumulate in sediments by atmospheric deposition, whereas the highly chlorinated congeners are poorly diffused and always found near their source of origin (Yuan et al., 2022). The Kruskal-Wallis test revealed significant differences in the PCB levels among sampling sites ($p < 0.05$), indicating spatial variability in PCB distribution across the lake. The spatial distribution of PCBs in Lake Victoria is shown in Fig. 3.

In comparison to studies reporting PCBs in sediments from freshwater ecosystems in other parts of the world (Table S6, Supplementary Material), the PCBs levels in sediments in this study were similar to those reported in Lake Chaohu, China (He et al., 2016), Lake Victoria, Tanzania (Wenaty et al., 2019a), Awash river, Ethiopia (Dirbaba et al., 2018), Somesu Mic river, Romania (Barhoumi et al., 2019), Suzhou industrial park, China (Yuan et al., 2022), and Lake Anhui, China (Yang et al., 2020). The levels were lower than the values reported in

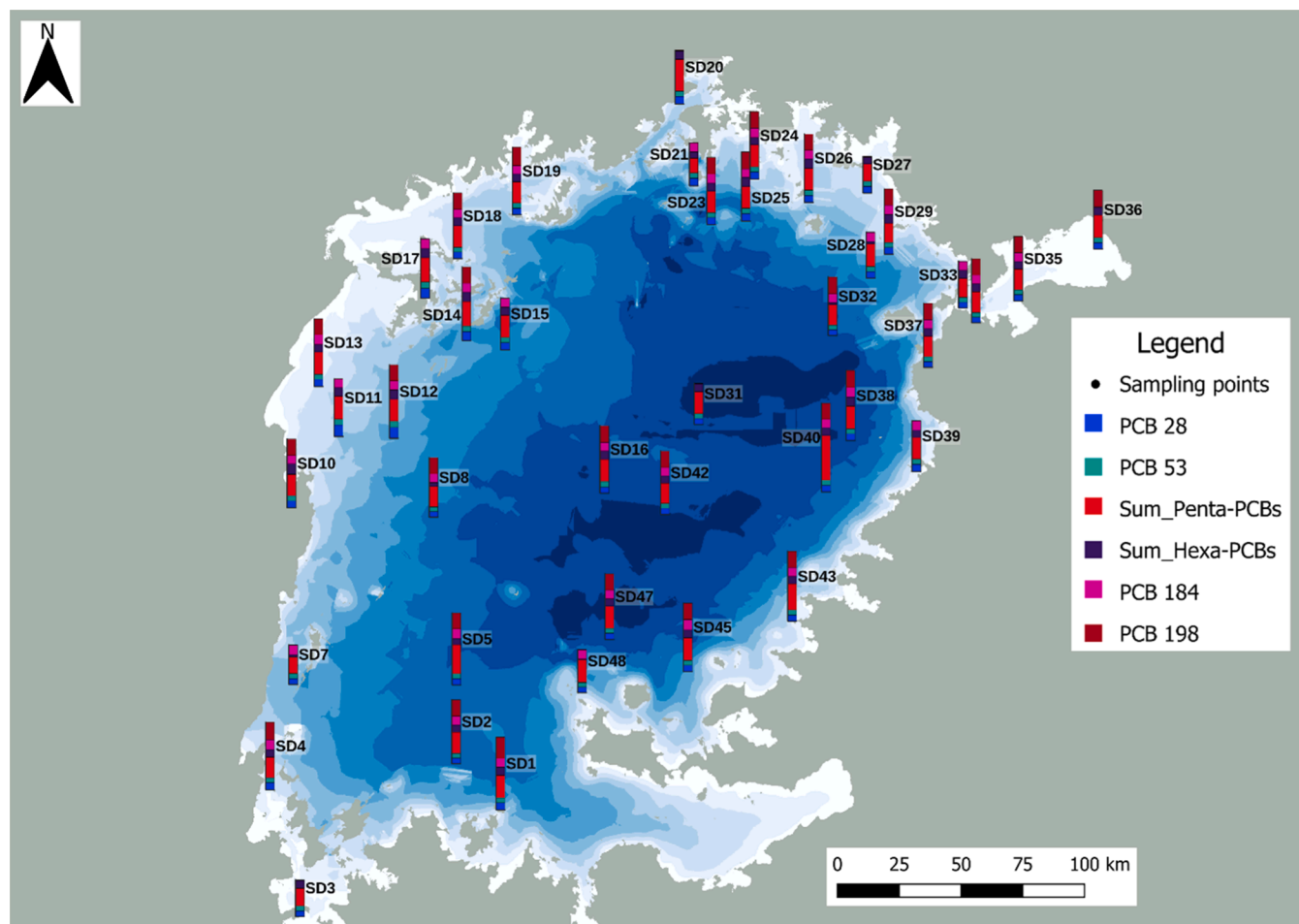


Fig. 3. A map showing the spatial distribution of PCBs in sediments of Lake Victoria.

sediments from Lake Chaohu, China (He et al., 2016), Msunduzi river, South Africa (Adeyinka et al., 2018), Nador lagoon, Morocco (Giuliani et al., 2015), North End Lake, South Africa (Kampire et al., 2017), Lake Manzala, Egypt (El-Kady et al., 2017), Lake Victoria, Tanzania (Wenaty et al., 2019a), Afiesere river, Edor river and Okpare river, Niger delta, Nigeria (Iwegbue et al., 2022), Awash river, Ethiopia (Dirbaba et al., 2018), Volturno river, Southern Italy (Montuori et al., 2020), and Somesu Mic river, Romania (Barhouni et al., 2019), and Shuangtaizi Estuary, China (Yuan et al., 2015). The similarity in the PCB levels with other similar studies elsewhere, combined with lower concentrations compared to the industrial and urban centres suggested that PCB contamination in Lake Victoria could primarily be due to legacy inputs, with no strong evidence of significant current releases, indicating that PCB inputs in the lake Victoria's catchment are mainly legacy driven, with limited evidence of ongoing high-intensity releases. However, some PCB concentrations in our study were higher than those documented in North End Lake, South Africa (Kampire et al., 2017), Napoleon gulf of Lake Victoria, Uganda (Ssebugere et al., 2014a), Murchison Bay of Lake Victoria, Uganda (Ssebugere et al., 2014b), Awash River, Ethiopia (Dirbaba et al., 2018), Yellow river, China (Gao et al., 2015), Yangtze river, China (Gao et al., 2015), Lake Chaohu, China (He et al., 2016), Suzhou industrial park, China (Yuan et al., 2022), and Lake Anhui, China (Yang et al., 2020). Consequently, these levels imply localized variations in sedimentation, waste management, and hydrodynamic redistribution (National Research Council, 2001). Therefore, the variations among studies are not directly comparable but rather indicative because of the differences in the analytical methods, PCB congener lists, sampling periods, and sediment characteristics.

3.3. Ecological risks of OCPs and PCBs to aquatic organisms

We evaluated the ecological risks associated with OCPs and PCBs contamination by comparing the data in this study with the established sediment quality guidelines using TEL and PEL values (MacDonald et al., 2000) (Table S4, Supplementary Material). Regarding DDTs, the p , p' -DDT, and p,p' -DDE mean levels in sediments from the Uganda, Kenya, and Tanzania portions of Lake Victoria were all below PEL values. On the contrary, the mean levels of p,p' -DDD, and lindane were all above their PEL and TEL values, which suggests that they would likely pose adverse effects on the sediment-dwelling organisms, but these results should be interpreted with caution due to the small sample size in this study, and we therefore recommend further monitoring studies with bigger sample sizes to confirm our findings. Moreover, risk assessment for other OCPs (aldrin, endrin aldehyde, heptachlor, α -HCH, α -endosulfan, β -endosulfan, and endosulfan sulfate) was inconclusive due to missing TEL and/or PEL values. For PCBs, the mean levels of Σ PCBs obtained in all the investigated sediment samples did not exceed TEL and PEL values, indicating that PCB congeners posed no ecological risks to the benthic organisms of Lake Victoria.

However, even below recommended threshold concentrations, the detection of these pollutants in the lake environment might lead to long-term ecological risks, such as bioaccumulation and biomagnification in the aquatic organisms, consequently affecting the aquatic life and humans (Liu et al., 2016). Contaminated sediments can negatively affect benthic organisms, fish species, and the whole ecosystem, with severe human health issues for the local people who rely on the lake resources for livelihoods. Conversely, the use of TEL and PEL guidelines to assess

the ecological risk in this study may have some limitations, and therefore, these results should be interpreted with caution. Firstly, they do not consider site-specific sediment properties such as total organic carbon and grain size, which ultimately change the bioavailability and true risk of any contaminant (Becker and Ginn, 2008). Secondly, the framework neglects species-specific sensitivity as it utilizes general thresholds, which may not accurately predict the effects of the pollutants on the local and indigenous species (Becker and Ginn, 2008). Ultimately, the TEL/PEL risk assessment approach ignores mixture effects since it evaluates individual pollutants, yet organisms are exposed to several contaminants in the environment at once. Therefore, these results must be interpreted as a cautious preliminary evaluation, and not a comprehensive ecological risk assessment.

3.4. Sources of OCPs and PCBs

Diagnostic ratios were used to identify possible sources of OCPs (DDTs, endosulfans, and HCHs) in the Uganda, Kenya, and Tanzania sides of Lake Victoria (Table S7, Supplementary Material). The ratio of *p,p'*-DDE/*p,p'*-DDT in the investigated sediments ranged from 2.58 to 6.51 (Uganda), 2.83 to 5.20 (Kenya), and 1.28 to 5.12 (Tanzania), which indicated past input of technical DDT in the catchment areas of the Lake Victoria basin. The ratio of α -/ γ -HCH in sediments ranged from 0.55 to 0.80 (for Uganda), 0.75 to 0.77 (Kenya), and 0.76 to 0.79 (Tanzania), which suggested past input of the technical HCH around the catchment areas. The ratio of α -/ β -endosulfan in sediments ranged from 0.87 to 5.79 (for Uganda), 1.55 to 2.21 (Kenya), and 1.65 to 4.43 (Tanzania). These results suggested that endosulfan profiles in sediments from the Kenyan portion of Lake Victoria were due to the past application of technical endosulfan. However, the ratio of α -/ β -endosulfan in seven (7) sediment samples from Uganda (33.3 %), and 5 sediment samples from Tanzania (38.5 %) were above 2.3, which suggests an ongoing use of technical endosulfan in the areas around Lake Victoria, which may partly be due to adulteration. Endosulfan was used in Uganda under thiodan and thionex trade names, but with an abusive past use, especially in fishing for export (Wasswa et al., 2011). Before the ban of endosulfan in 2011 in Kenya, it was widely used as a broad-based insecticide against agricultural pests such as cotton bollworm and maize stalk-borer (Akenga et al., 2020; Tefera et al., 2016). According to a report by the Kenya National Integrated Pest Management (KNIP), about 7480 tons of endosulfan were imported in Kenya from 2006 to

2010, and 10 firms were formulating and distributing various pesticides (Loha et al., 2018), while some companies imported large volumes of the pesticides and repacked them locally.

Furthermore, principal component analysis (PCA) was used to confirm the sources of OCPs, and 3 principal components (PC1–3) were generated, which accounted for 58.9 % of the variance in the dataset. Only components with eigen values >1 were extracted. Principal components 1, 2, and 3 accounted for 28.9 %, 17.5 %, and 12.5 % of the total variance, respectively. Fig. 4 shows the PCA loading plots of the components (PC1, PC2, and PC3) showing the distribution of OCPs. PC1 had high positive loadings of α -endosulfan (0.983), β -endosulfan (0.983), endosulfan sulfate (0.766), and *p,p'*-DDT (0.926). PC2 had a high positive loading of aldrin (0.749), lindane (0.733), and endrin aldehyde (0.616). The loadings of α -HCH (0.70) and dieldrin (−0.778) were more significant in PC3 indicating the same source of contamination, while PC4 had negative loadings for *p,p'*-DDD (−0.560), *p,p'*-DDE (−0.436), heptachlor (−0.770), and heptachlor exo-epoxide (−0.745), most of which are degradation products of their parent compounds. Technical DDT is easily degraded into DDD under anaerobic conditions, and DDE under aerobic conditions (Aislabie et al., 1997; Foght et al., 2001; Yu et al., 2011), which could explain the presence of *p,p'*-DDE, and *p,p'*-DDD in PC4. Therefore, all principal components (PC1, PC2, and PC3) reflected a historical use of the OCPs around the lake's catchment area, which possibly led to the degradation products or metabolites of the OCPs in the environment.

For PCBs, the extracted components PC1, PC2, and PC3 accounted for 62.9 % of the variance in the dataset. Principal component 1 (PC1) explained a variance of 28.2 %, PC2 accounted for 19.0 %, while PC3 accounted for 15.7 % of the data variance. Fig. 4 shows the PCA loading plots for PCBs in this study. PC1 had high positive loadings of tri-CB 28 (0.900), tetra-CB 53 (0.906), and penta-CB 115 (0.627). PC3 had loadings of penta-CB 104 (0.827), penta-CB 112 (0.817), hexa-CB 140 (0.565), and hexa-CB 154 (0.640). Since PC1 and PC3 comprised of tri-, tetra-, penta- and hexa-CBs, confirming that both components are from both primary and secondary sources of PCBs such as waste incineration, and waste discharges from industries like textile and paint manufacturing, degradation of higher PCBs, volatilization, poor waste disposal of electrical equipment such as electric transformers, capacitors, hydraulic fluids from metal workshops and ship repair activities and plasticizers (Emoyan et al., 2022). Principal component 2 (PC2) was dominated by high positive loadings of penta-CB 119 (0.865), hepta-CB

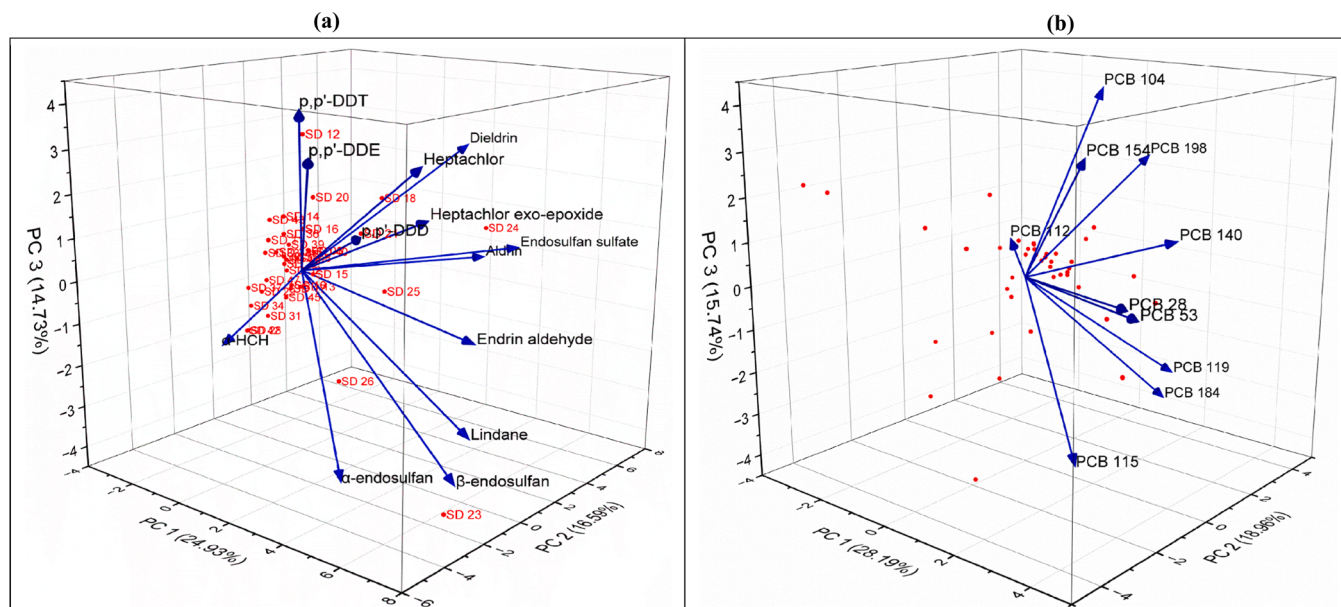


Fig. 4. PCA plots showing the distribution of (a) OCPs, and (b) PCBs in the Lake Victoria basin.

184 (0.875), and octa-CB 198 (0.545). The presence of higher chlorinated congeners (PCB-184 and PCB-198) agrees with previous literature confirming point sources from industrial activities such as waste discharges from sewage treatment plants and leather tanning industries (Balasubramani et al., 2014; Needham and Ghosh, 2019; Sánchez-Avila et al., 2009) while the presence of penta-CB 119 confirmed secondary sources of PCBs into the lake, for instance, degradation of highly chlorinated PCBs into low chlorinated PCBs as well as volatilization, whereby lower PCBs can travel long distances from their point sources to areas of no contamination (Ren et al., 2007).

4. Conclusions

The results of this study revealed that sediments from Lake Victoria were contaminated with OCPs and PCBs, with *p,p'*-DDD and PCB-198 as the most dominant OCP residue and PCB congener, respectively. Diagnostic ratios for DDTs and HCHs suggested past use of these pollutants in the three countries. However, they revealed recent input of endosulfan on the Uganda and Tanzania sides of Lake Victoria. Principal component analysis confirmed anthropogenic activities such as agricultural activities and public health management as the major sources of OCPs, while PCB sources were attributed to discharges from industrial processes, waste incineration, and sewage treatment plants. Although the levels of Σ PCBs in sediments were below the TEL and PEL values, *p,p'*-DDD and lindane posed potential risks to benthic organisms. However, risk assessment for the rest of the investigated OCPs (except dieldrin and heptachlor exo epoxide) was inconclusive because of their missing TEL and/or PEL values, highlighting the need for continuous monitoring and stronger pollution management efforts in the Lake Victoria basin.

5. Study strengths and limitations

To the best of our knowledge, this was the first tri-national study to assess the spatial distributions and concentrations, ecotoxicological risks and sources of OCPs and PCBs in sediments from Lake Victoria across Uganda, Kenya, and Tanzania. The majority of the previous related studies were national and focused on one given portion of the lake. Nonetheless, this study had some limitations, for example, it focused on a single-season assessment, and it did not analyze OCPs and PCBs in the aquatic organisms, such as fish and phytoplankton, for a more robust and comprehensive understanding of their biomagnification and bioaccumulation. Furthermore, the study relied only on TEL/PEL for risk assessment, yet it is inconclusive.

6. Recommendations

We recommend future studies on Lake Victoria to quantify OCPs and PCBs in the water columns, and biota alongside sediments sampled together to establish any direct effects of OCPs and PCBs on aquatic organisms vs. their levels of bioaccumulation and biomagnification. Organic matter and sediment particle size should be assessed concurrently to provide mechanistic insights between pollutant levels and the sediment characteristics. Seasonal variations need to be evaluated to properly understand the trends of OCPs and PCBs contamination in the lake. While performing risk assessment for the pollutants, risk quotients (RQ) and SQGQ approaches should be carried out for a comprehensive risk assessment of these pollutants. Future studies need to incorporate a broader list of pollutants for a better understanding of the lake's pollution burden. In addition, future studies should also monitor air and groundwater for a more comprehensive understanding of the environmental pathways and fate of these semi-volatile compounds, since this would provide insightful complementary data. To comprehensively understand the potential interactions and to enhance ecological risk estimates due to synergistic effects, future research should consider assessing the mixture toxicity of OCPs and PCBs since combined exposures can lead to increased ecological risks. In addition, we also

recommend that key stakeholders, especially government Agencies, should improve policy formulation on waste management and toxic emissions into the Lake Victoria basin to enable the development of more effective and sustainable strategies to address the shared regional environmental challenges.

CRedit authorship contribution statement

Afuwa Kagoya: Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Kenneth Arinaitwe:** Writing – review & editing, Supervision, Funding acquisition. **Silver Odongo:** Writing – review & editing. **Douglas Sifuna:** Writing – review & editing. **Henry Matovu:** Writing – review & editing. **Julius Matsiko:** Writing – review & editing. **Godfrey Muhwezi:** Writing – review & editing. **Ivan Špánik:** Writing – review & editing, Funding acquisition. **Charles Drago Kato:** Writing – review & editing. **Mika Sillanpää:** Writing – review & editing. **Patrick Ssebugere:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.hazadv.2025.100979](https://doi.org/10.1016/j.hazadv.2025.100979).

Data availability

No data was used for the research described in the article.

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