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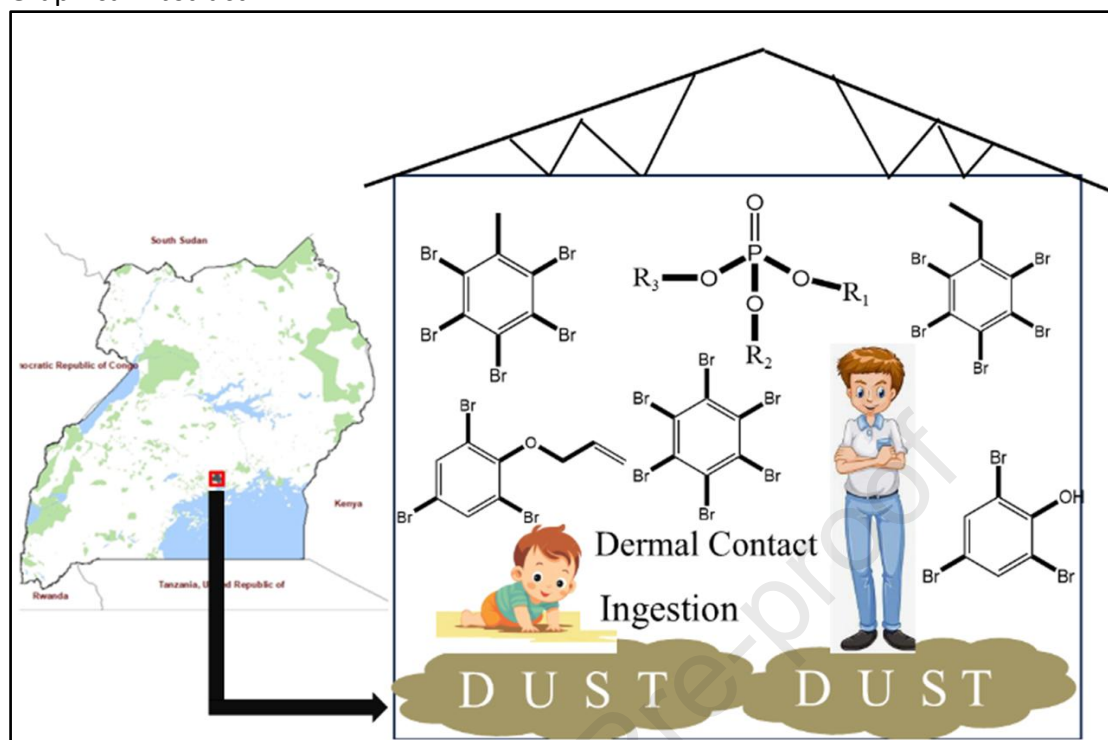
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Graphical Abstract



Exposure of urban population to organophosphate esters and novel brominated flame retardants via indoor dust: Occurrence, sources and health risks in Uganda, East Africa

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Abstract

Indoor environments are critical exposure pathways to flame retardants, yet data from Africa remain scarce. This study provides the first quantitative assessment of organophosphate esters (OPEs) and novel brominated flame retardants (NBFRs) in indoor dust from Ugandan households, establishing baseline data for East Africa. Dust samples collected from homes in Kampala were analyzed using gas chromatography–mass spectrometry (GC–MS). Concentrations of total OPEs (Σ_5 OPEs) and total NBFRs (Σ_4 NBFRs) ranged from 1,520–102,000 ng g⁻¹ and 214–6,600 ng g⁻¹, respectively. Tris(1,3-dichloro-2-propyl) phosphate (TDCIPP) and 2,4,6-tribromophenyl allyl ether (TBP-AE) dominated OPE and NBFR profiles, respectively. Chlorinated OPEs contributed most to Σ OPEs, reflecting emissions from building materials and consumer products. Principal component analysis indicated household items, electrical equipment, and floor finishes as the major emission sources. Estimated human exposure doses showed that ingestion was the dominant route, with children more exposed than adults. Hazard index (HI) values were below 1 for all compounds, indicating negligible non-carcinogenic risk. Carcinogenic risk (CR) values ranged from 10⁻⁶–10⁻¹², suggesting minimal lifetime cancer risks, though children exhibited relatively higher vulnerability. Compared with global datasets, OPE and NBFR levels in Kampala were within lower-to-median range but represent the inaugural dataset for sub-Saharan Africa. These findings provide a crucial regional reference for indoor flame-retardant exposure and highlight the need for extended studies across multiple indoor microenvironments to evaluate the long-term exposure pathways and inform safer chemical management.

Key words: OPEs; NBFRs; Household dust; Principal component analysis; Health risk assessment

1. Introduction

Organophosphate esters (OPEs) and novel brominated flame retardants (NBFRs) are a class of synthetic organic chemicals extensively applied in consumer goods and construction materials due to their flame-retardant and plasticizing properties (Marklund et al., 2003; Wei et al., 2015). OPEs are commonly used in plastics, foams, textiles, hydraulic fluids, lacquers, and coatings, while NBFRs are utilized as replacements for phased-out legacy brominated flame retardants such as octa-, penta-, and deca-brominated diphenyl ethers (BDEs) and hexabromocyclododecane (HBCDD), which are listed under the Stockholm Convention on Persistent Organic Pollutants (He et al., 2018). Unlike reactive flame retardants that form chemical bonds with polymer matrices, OPEs and NBFRs are additive in nature, allowing them to leach, volatilize, or abrade from products during manufacture, use, or disposal (Pang et al., 2017; Xiong et al., 2019). Consequently, they are emitted into the surrounding indoor environment and adsorbed onto settled dust, which serves as a major reservoir of semi-volatile organic pollutants (Rauert & Harrad, 2015).

Human exposure to flame retardants via indoor dust is now recognized as a significant pathway, particularly for children who are more susceptible due to frequent hand-to-mouth activity, higher ingestion rates, and immature metabolic systems (DeLuca & Cohen Hubal, 2024). Numerous studies in developed and developing regions have demonstrated that OPEs and NBFRs are ubiquitous in indoor dust from homes, offices, schools, and vehicles (Al-Omran et al., 2021; Gbadamosi et al., 2023; Niu et al., 2019). Although intended as safer alternatives to legacy flame retardants, increasing evidence shows that many OPEs and NBFRs exhibit persistence, bioaccumulation potential, and toxicity comparable to their predecessors (M. Li et al., 2023; Meng et al., 2025). This phenomenon—often termed “regrettable substitution”—has heightened global concern about their long-term ecological and human health implications (Zuiderveen et al., 2020). Reported health outcomes include endocrine and reproductive disruptions, neurotoxicity, immunotoxicity, and carcinogenicity (Hou et al., 2016a; Khani et al., 2023; Liu et al., 2023; Liu et al., 2022).

Indoor environments serve as stable micro-ecosystems with limited photolytic and microbial degradation, enabling prolonged retention of pollutants (Hwang et al., 2008;

Söderström et al., 2004). Dust particles therefore represent an excellent proxy for assessing human exposure to flame retardants, since about 80–90% of human time is spent indoors (Ali et al., 2021). The accumulation of these pollutants in dust reflects not only the intensity of use in consumer products but also the efficiency of local waste management systems and indoor ventilation. Understanding their occurrence and associated risks is essential for formulating mitigation strategies and guiding chemical policy interventions in developing countries. Globally, recent syntheses e.g., Liu et al. (2023) have consolidated knowledge on the occurrence, sources, and exposure risks of OPEs and NBFRs in indoor dust across multiple continents, revealing strong spatial contrasts between industrialized and less-industrialized regions. However, these global evaluations rely almost exclusively on data from Europe and Asia, leaving tropical and African settings largely unexplored.

Despite the growing global evidence base, data on OPEs and NBFRs in the African indoor environments remain limited. Previous reports have focused more on Nigeria (Akinrinade et al., 2021; Gbadamosi et al., 2023), South Africa (Abafe & Martincigh, 2019; Brits et al., 2019), and Egypt (Abdallah & Covaci, 2014; Hassan & Shoeib, 2015; Khairy & Lohmann, 2018, 2019). These studies identified multiple contamination sources, including emissions from consumer products, informal electronic waste recycling, plastic and textile manufacturing, and open dumping of solid waste (Muhwezi et al., 2025). However, no systematic investigation has yet been conducted in East Africa, particularly in Uganda, where urbanization and industrialization are intensifying pollutant emissions. Flame retardants have already been detected in Ugandan air (Arinaitwe et al., 2014; Wang et al., 2019) and surface waters (Nantaba et al., 2021), indicating potential cross-media contamination. The present study therefore, provides a pioneering comprehensive assessment of OPEs and NBFRs in indoor household dust from Kampala City, Uganda. Specifically, the study aimed to; (i) determine the concentrations of target OPEs and NBFRs in indoor dust from Kampala City, (ii) identify their possible emission sources, and (iii) evaluate human health risks associated with dust exposure. Findings from this work provide baseline data for the region and contribute to the broader understanding of indoor exposure to emerging flame retardants in tropical urban settings.

2. Materials and methods

2.1 Study area

This study was conducted in Kampala City, the capital of Uganda (0°18'49" N, 32°34'52" E), covering approximately 189 km² (**Fig. 1**). Kampala represents a rapidly urbanizing tropical metropolis with a population exceeding 1.8 million people and a density of about 9,500 persons km⁻² (UBOS, 2024). The city experiences a tropical climate with two wet seasons (March–May and September–November) and relatively stable annual temperatures (21–28 °C). These climatic conditions favor the accumulation and persistence of semi-volatile organic pollutants in indoor environments with limited ventilation. Sampling sites were strategically selected to represent different socio-economic settings and potential contamination hotspots, including residential areas near the industrial zones, solid waste collection points, and wastewater discharge zones (Kawempe II, Kisenyi II, Mutundwe, Kabowa, Bugolobi, and Ggaba). This design provided spatially representative coverage of households influenced by both industrial and domestic emission sources.

[Insert Fig. 1]

2.2 Chemicals and standards

Target analytes included seven OPEs—tri-n-butyl phosphate (TNBP), tris(2-butoxyethyl) phosphate (TBOEP), tris(2-chloroisopropyl) phosphate (TCIPP), tris(2-chloroethyl) phosphate (TCEP), tris(1,3-dichloro-2-propyl) phosphate (TDCIPP), triphenyl phosphate (TPHP), and ethylhexyl diphenyl phosphate (EHDPP)—and five NBFRs—hexabromobenzene (HBB), 2,3,4,5,6-pentabromotoluene (PBT), 2,3,4,5,6-pentabromoethylbenzene (PBEB), 2,4,6-tribromophenol (TBP), and 2,4,6-tribromophenyl allyl ether (TBP-AE) which were obtained from AccuStandard[®], Inc (New Haven, CT, USA). Isotope-labelled internal standard, ¹³C₁₂ labelled 2,2',3,3',4,4'-Hexabromodiphenyl ether (¹³C₁₂-BDE 128), was also obtained from AccuStandard[®], Inc while Pentadeuterated tribromophenol allyl ether (d₅-TBP-AE), tri-n-butyl phosphate-d₂₇ (d₂₇-TNBP), and Phenol-2,3,4,5,6-d₅ (d₅-TBP) were purchased from Sigma-Aldrich (Germany). All solvents used for extraction and clean-up were pesticide-residue grade and of high purity (*Detailed*

information on target analytes, and their physical chemical properties is provided in **Table S1**).

2.3 Sampling

Indoor dust samples ($n = 43$) were collected during morning hours from the floors of living rooms of the residential houses in November–December 2023, a period representing the transition from the second wet season to the early dry season. Moreover, the selected period coincides with the end-of-year festive period, which is usually characterized by intense indoor activity, higher human occupancy, and frequent cleaning routines. This period was purposely selected to capture realistic exposure conditions and peak household dust accumulation under limited ventilation and elevated social activity. Dust samples were collected using a household vacuum cleaner fitted with 25 μm nylon sampling socks (Powley et al., 2005). Each sock was pre-cleaned with isopropyl alcohol, sealed after sampling, and wrapped in aluminum foil. All equipment were rinsed with distilled water and hexane-impregnated wipes between the households to minimize cross-contamination. The samples were then transported to the Pesticide Residue Laboratory, Makerere University, and stored at $-4\text{ }^{\circ}\text{C}$ until extraction.

To provide contextual information on possible emission sources, a structured household questionnaire was administered to 47 consenting participants, capturing data on flooring type, furniture materials, electrical and electronic devices, proximity to industrial areas or major roads, and renovation-history. Only 43 fully completed responses were included for analysis (*A summary of the questionnaire results is provided in Table S2*).

2.4 Sample preparation, extraction, and clean-up

Dust samples were sieved ($<250\text{ }\mu\text{m}$) to remove coarse particles, hair, and debris, then homogenized and equilibrated at room temperature for 12–24 h. Approximately 0.30 g of the dust was spiked with labelled internal standards (10 ng) and extracted three times using n-hexane/acetone (1:1, v/v) by vortex agitation (1 min), ultrasonication (20 min), and centrifugation (4000 rpm, 20 min). Extracts were combined and concentrated under ultra-pure nitrogen gas to near dryness, followed by Florisil and column clean-up to fractionate

OPEs and NBFRs, respectively, following a method reported by Dirtu et al. (2012) with minor modifications. The NBFR fraction was further purified using a 44% sulfuric acid–silica gel column to eliminate co-extracted matrix components. The final extracts were reconstituted in 100 μ L isooctane for GC–MS analysis (*Details of the clean-up procedure have been provided in Text S1*).

2.5 Instrumental analysis and quantitation

Quantification of target analytes was carried out using a Shimadzu QP2020NX GC–MS system, operated in electron-impact ionization (EI) and selected-ion monitoring (SIM) modes. Compound identification was based on the comparison of the retention times and diagnostic ions with chemical standards. A five-point calibration curve (100–900 ng mL^{-1} for OPEs and 250–1500 ng mL^{-1} for NBFRs) was constructed with R^2 value > 0.99 obtained for the standard calibration curve of each target compound. *A summary of the instrumental conditions, temperature programs, qualifier and quantifier ions are found in Text S2 and Table S3.*

2.6 Quality assurance and control (QA/QC) protocol

Field and procedural blanks were included in every batch of five samples to assess any background contamination. No analyte was detected above the limit of detection (LOD) in the blanks. Mean recoveries except for TCEP ranged from 70–123% for both OPEs and NBFRs. LODs and limits of quantification (LOQs) were calculated from signal-to-noise ratio of 3 and 10, respectively. LODs ranged from 0.1 to 11.0 ng g^{-1} . One analyte (TCEP) showed slightly lower recovery ($\sim 51\%$), and recovery correction was applied as recommended by Thompson et al. (1999). These QA/QC results confirmed the analytical reliability and reproducibility of the data (*full QA/QC details including calibration, recoveries, and LOQs are summarized in Table S3*).

2.7 Health risk assessment

Human exposure to OPEs and NBFRs via indoor dust was evaluated using standard U.S. EPA risk assessment models (USEPA, 2024a, 2024b). Both the non-carcinogenic and

carcinogenic risks were quantified through dust ingestion and dermal contact, which represent the dominant exposure pathways in indoor environments (Wu et al., 2023). The average daily dose (ADD, $\text{mg kg}^{-1} \text{ day}^{-1}$) for ingestion and dermal contact was computed using Eqs. (1) and (2):

$$\text{ADD}_{\text{ing}} = \frac{C \times I_{\text{Ring}} \times EF \times ED \times CF}{BW \times AT} \quad (1)$$

$$\text{ADD}_{\text{der}} = \frac{C \times SA \times AF \times ABS \times EF \times ED \times CF}{BW \times AT} \quad (2)$$

Where C (mg g^{-1}) is the concentration of the target compound in dust; I_{Ring} (mg day^{-1}) is the dust ingestion rate; EF (days year^{-1}) is exposure frequency; ED (years) is exposure duration; CF ($10^{-3} \text{ g mg}^{-1}$) is the conversion factor; BW (kg) is body weight; AT (days) is the averaging time; SA ($\text{cm}^2 \text{ day}^{-1}$) is the exposed skin area; AF (mg cm^{-2}) is the skin adherence factor; and ABS is the dermal absorption fraction.

The non-carcinogenic risk was assessed using the hazard quotient (HQ) (Eq. 3), derived from the ratio of total exposure to the reference dose (RfD). The hazard index (HI) (Eq. 5) was obtained by summing HQ values for all individual compounds.

$$\text{HQ} = \frac{\text{ADD}_{\text{total}}}{\text{RfD}} \quad (3)$$

$$\text{ADD}_{\text{total}} = \text{ADD}_{\text{ing}} + \text{ADD}_{\text{der}} \quad (4)$$

Where RfD measured in $\text{mg kg}^{-1} \text{ day}^{-1}$, represents the value of the reference dose of individual OPEs or NBFRs. The total HQ for the OPEs and NBFRs investigated in this study were independently calculated using Eq. 5.

$$\text{HI} = \text{HQ}_1 + \text{HQ}_2 + \text{HQ}_3 + \dots + \text{HQ}_n \quad (5)$$

An $\text{HI} < 1$ indicates no expected adverse non-carcinogenic effects, while $\text{HI} > 1$ suggests potential health risk (Shi et al., 2023).

The carcinogenic risk (CR) was estimated for compounds with available cancer slope factors (CSF) using Eq. (6):

$$CR = CR_{ing} + CR_{der} = (ADD_{ing} + ADD_{der}) \times CSF \quad (6)$$

Where CSF ($\text{mg kg}^{-1}\text{day}^{-1}$) represent the cancer slope factors for each of the studied compounds. ADD_{ing} and ADD_{der} are averaged daily dose via ingestion and dermal routes, respectively. According to USEPA guidelines, $CR < 1 \times 10^{-6}$ indicates negligible risk, values between 1×10^{-6} – 1×10^{-4} represent low to moderate risk, and $CR > 1 \times 10^{-4}$ denotes elevated carcinogenic risk (Li et al., 2014; Xing et al., 2018).

Median concentrations were applied to represent average exposure scenarios, while 95th percentile values reflected high-exposure conditions. Parameter details, toxicological constants, and RfD/CSF values used in the health risk calculations are summarized in **Tables S4 and S5** (Supplementary Material).

2.8 Statistical data analysis

All statistical analyses were performed using OriginPro 2024 (OriginLab Corp., USA) and Microsoft Excel 2016. Values below the LOD were replaced with $\frac{1}{2}$ LOD. The Shapiro–Wilk test indicated non-normal data distribution, and thus Spearman correlations were used to assess the associations among the individual OPEs and NBFRs. Principal Component Analysis (PCA) was employed to identify possible common sources and grouping patterns. Statistical significance was considered at $p < 0.05$.

3. Results and discussion

3.1 Levels of OPEs in the dust samples

Levels of OPEs in the investigated household dust samples are shown in **Table 1** and **Table S6**. TNBP, TBOEP, TCEP, TCIPP, and TDCIPP were detected in all the dust samples (100 % detection), while EHDPP was present in only 20.9 % of the samples, and TPHP was not detected. The total concentrations ($\sum_5\text{OPEs}$) ranged from 1,520 – 102,000 ng g^{-1} , with a median of 21,900 ng g^{-1} . Urban households in Kampala exhibited relatively higher OPE levels, likely linked to intensive industrial emissions, traffic density, and widespread usage of consumer products containing flame retardants (Chen et al., 2019; Li et al., 2018). The

relative contribution of individual OPEs followed the sequence: TDCIPP > TCIPP > TBOEP > TNBP > TCEP > EHDPP.

[Insert Table 1]

The median concentrations of OPEs in indoor dust from Kampala, Uganda showed notable regional and global contrasts as shown in **Table S7**. The TNBP levels (173 ng g⁻¹) were comparable to South Africa (212 ng g⁻¹) (Brits et al., 2019), Belgium (130 ng g⁻¹) (Van den Eede et al., 2011) and Latvia (190 ng g⁻¹) (Pasecnaja et al., 2021) but significantly exceeded those in Egypt (17 ng g⁻¹) (Abdallah & Covaci, 2014) and Brazil (12.3 ng g⁻¹) (Cristale et al., 2018), and were much lower than those reported in Japan (1,400 ng g⁻¹) (Kanazawa et al., 2010), potentially suggesting moderately high usage of TNBP-containing materials in Uganda. TBOEP concentrations on the other hand (2,120 ng g⁻¹) were moderate compared to those reported from Sweden (79,000 ng g⁻¹) (Larsson et al., 2018) and Japan (1,570,000 ng g⁻¹) (Kanazawa et al., 2010), yet substantially higher than the levels from Egypt (18 ng g⁻¹) (Abdallah & Covaci, 2014) and China (14.2 ng g⁻¹) (Du et al., 2020), potentially reflecting divergent flame retardant applications. TCIPP levels (2,470 ng g⁻¹) were comparable with those from South Africa (3,545 ng g⁻¹) (Abafe & Martincigh, 2019) and USA (2,200 ng g⁻¹) (Dodson et al., 2012), but were lower than those reported by Kanazawa et al. (2010) (18,700 ng g⁻¹) in Japan. TCEP concentrations (135 ng g⁻¹) were lower than those reported in South Africa (7,390 ng g⁻¹) (Abafe & Martincigh, 2019) and China (1,630 ng g⁻¹) (Du et al., 2020), but higher than Egypt (22 ng g⁻¹) (Abdallah & Covaci, 2014), pointing to variable industrial and domestic sources across countries. Uganda's TDCIPP levels (15,700 ng g⁻¹) rank among the highest globally, surpassing South Africa (7,695 ng g⁻¹) (Abafe & Martincigh, 2019), Japan (4,000 ng g⁻¹) (Kanazawa et al., 2010), and the USA (2,100 ng g⁻¹) (Dodson et al., 2012), possibly due to unregulated imports.

These findings underscore the need for region-specific exposure assessments and regulatory oversight, particularly in low- and middle-income countries. The pattern observed reveals that while Uganda's indoor environment shows moderately high OPE burdens, its levels remain below those in highly industrialized countries, possibly reflecting variations in domestic production, building materials, and product regulation. The presence of imported electronics, foams, and furniture from Asian markets-particularly China-could

contribute substantially to the observed contamination (United Nations COMTRADE, 2024).

Overall, this study provides the first quantitative evidence of OPE contamination in indoor dust from Uganda, thereby extending the spatial coverage of available datasets that have largely concentrated on South Africa and North Africa. The predominance of chlorinated OPEs (TDCIPP and TCIPP) in the Ugandan households aligns with patterns observed in most European and Asian studies (Marklund et al., 2003; Meeker & Stapleton, 2010), suggesting a shared chemical usage profile in flame-retardant applications despite regional differences. Their high contribution is mainly attributed to their extensive use in polyurethane foams, insulation panels, and flooring adhesives (Zhou et al., 2017). Chlorinated OPEs exhibit higher thermal stability and slower degradation than aryl- or alkyl-OPEs, favoring their persistence in indoor dust (Wu et al., 2023). Consequently, emissions from consumer materials and imported furniture treated with these compounds likely contribute to their accumulation. The current dominance of Cl-OPEs also reflects the global substitution trend away from legacy brominated flame retardants toward OPE-based alternatives, particularly in Africa where second-hand imports remain dominant.

3.2 Levels of NBFRs in dust samples

All target NBFRs (HBB, PBT, PBEB, TBP, and TBP-AE) were detected in indoor dust samples, with varying detection frequencies (**Table 1** and **Table S8**). The $\sum_4$ NBFRs concentrations ranged from 214 to 6,600 ng g⁻¹, with a mean $\pm$ standard deviation of 908 $\pm$ 985 ng g⁻¹ and median concentration of 712 ng g⁻¹. The distribution of individual NBFRs followed the order TBP-AE > HBB > PBT > TBP > PBEB, with TBP-AE showing the highest median and mean levels of (238 and 449 ng g⁻¹, respectively) and PBEB the lowest median and mean levels of (5.8 and 7.3 ng g⁻¹, respectively). The relatively lower concentrations of PBEB, TBP, and PBT in indoor dust may be attributed to their higher vapor pressures and lower octanol–air partition coefficients (K_{oa}), which reduce their affinity for particulate matter and limit their partitioning into settled dust (Baskaran & Wania, 2023). In contrast, although TBP-AE exhibits comparable vapor pressure to these NBFRs, its consistently elevated concentrations in indoor household in this study, likely

reflect its broader application in consumer products and materials commonly found indoors. This suggests that usage patterns, rather than physicochemical properties alone, may play a dominant role in determining indoor dust burdens for certain flame retardants.

The median concentration of PBT in this study (38.4 ng g^{-1}) was significantly higher than the values reported in South China (10.10 ng g^{-1}) (Tang et al., 2019), UK (5.90 ng g^{-1}) (Ma et al., 2024), Beijing, China (0.72 ng g^{-1}) (Wang et al., 2018), and Nigeria ($<0.59 \text{ ng g}^{-1}$) (Akinrinade et al., 2021), likely reflecting regional differences in product use and regulation. PBT's low volatility, high lipophilicity ($\log K_{ow}$), and strong dust sorption contribute to its persistence in indoor dust (Liu, 2022). PBEB levels (5.8 ng g^{-1}) also exceeded those in Nigeria ($<0.71 \text{ ng g}^{-1}$) (Akinrinade et al., 2021), China (0.20 ng g^{-1}) (Niu et al., 2019), and Iraq (0.1 ng g^{-1}) (Al-Omran & Harrad, 2016), possibly due to use of PBEB-containing materials and legacy imports. HBB median level obtained in this study (341 ng g^{-1}) was lower than that obtained in Vietnam (530 ng g^{-1}) (Hoang et al., 2022) but markedly higher than that obtained in other studies (Ma et al., 2024; McGrath et al., 2018; Niu et al., 2019; Wang et al., 2018). In contrast, TBP ranged from $<\text{LOD}$ to 928 ng g^{-1} (median $<\text{LOD}$), lower than China's reported 389 to 2000 ng g^{-1} (Median 803 ng g^{-1}) (Lan et al., 2023). TBP's higher water solubility, moderate $\log K_{ow}$, and lower dust affinity may limit its accumulation. TBP-AE median value (238 ng g^{-1}) obtained in this study was several orders of magnitude higher than that obtained in Canada, (0.4 ng g^{-1}) (Shoeib et al., 2012). There is a paucity of data on TBP and its derivative, TBP-AE in the indoor dust, complicating comparisons.

The variations are likely due to the regional differences in consumer product composition, building materials, and consumer product import history. The Ugandan household dust exhibited a range of NBFRs, with TBP-AE as the major compound among novel flame retardants, implying a shift in consumer product composition and potential emerging exposure risks associated with newer, less-regulated flame retardant chemicals. The comparatively high TBP-AE levels in this study suggest its use in expanded polystyrene insulation and electronic components, which are increasingly imported into East Africa. This points to an evolving substitution trend similar to that reported in the developed economies but occurring through imported finished goods rather than local manufacturing.

A paired sample t-test confirmed that OPE concentrations were significantly higher than NBFR concentrations ($t(41) = -7.57, p < 0.001$). This difference can be explained by their physicochemical and functional properties. Additionally, OPEs have widespread use including dual functional roles as flame retardants and plasticizers which further amplifies their prevalence, making them dominant contaminants in indoor exposure pathways (Song et al., 2024). To contextualize our findings, we compared them with legacy flame retardants-polybrominated diphenyl ethers (PBDEs)-in indoor dust. Studies from mostly Nigeria, South Africa and Egypt (see **Table S9**) report median concentrations of commonly studied PBDE congeners (BDE-28, -47, -99, -100, -153, -154, -183, -209) ranging from <0.04 to 390 ng g^{-1} , with maximum values reaching 8560 ng g^{-1} (Adeyi et al., 2020; Akinrinade et al., 2021; Harrad et al., 2016; Hassan & Shoeib, 2015; Ibeto et al., 2021; Kefeni et al., 2014; Olukunle et al., 2015). These levels are comparable to those of NBFRs, such as TBP-AE (median: 238 ng g^{-1} ; range: $26.1\text{--}6230 \text{ ng g}^{-1}$), but are substantially lower than those of OPEs, which in this study reached median concentrations up to $15,700 \text{ ng g}^{-1}$ (TDCIPP). This disparity highlights a shift in indoor contamination from legacy PBDEs to emerging flame retardants, underscoring the need for targeted monitoring, regulatory oversight, and toxicological evaluation. It also calls for national inventories of flame retardants in imported consumer products and assessment of indoor exposure risks.

3.3 Sources of OPEs and NBFRs in the dust samples

Positive and statistically significant correlations ($p < 0.05$) were observed between TNBP and other OPEs (TBOEP, TCIPP, and TDCIPP) with r_s values ranging from 0.344 to 0.426. TBOEP was significantly ($p < 0.05$) correlated with TCIPP ($r_s=0.961$) and TDCIPP ($r_s=0.441$) (r_s -Spearman rank correlation). Additionally, significant correlations ($p < 0.05$) occurred between TCIPP and TDCIPP ($r_s = 0.526$) and between TCEP and TDCIPP ($r_s = 0.854$) (**Table S10**). These relationships suggest common indoor sources or similar partitioning behaviors, typical of chlorinated and alkyl OPEs co-used in foams, coatings, and flooring adhesives. Detection of TBOEP and TNBP, linked to floor finishing and hydraulic fluids, further suggests emissions from maintenance products and construction materials typical of urban households in East Africa. For the NBFRs, Spearman's rank correlations were weak and non-significant, implying multiple or independent sources with

differing emission behaviors (Yadav et al., 2019). Such variability aligns with NBFR use in diverse products-electronics, textiles, and polymers-that degrade or off-gas at distinct rates depending on product type and age.

PCA confirmed these associations. For OPEs, three principal components (PCs) with eigenvalues > 1 explained 99.0 % of total variance (**Fig. 2a**). PC1 (42.7 %) showed high loadings (> 0.4) for TBOEP, TCIPP, TCEP, and TDCIPP, whereas PC2 (35.8 %) loaded positively (> 0.5) on TCEP and TDCIPP and negatively (< -0.4) on TBOEP and TCIPP. These groupings indicate similar inputs from consumer goods, foams, and electronic casings were OPEs serve as flame retardants and plasticizers (Wei et al., 2015). PC3 (20.4 %) was dominated by TNBP (> 0.9), reflecting its use in hydraulic fluids and industrial lubricants, suggesting secondary inputs from domestic maintenance materials.

For NBFRs, two PCs with eigenvalues > 1 accounted for 64.4 % of variance (**Fig. 2b**). PC1 (37.4 %) had strong loadings (> 0.4) for HBB, PBT, and PBEB, indicating co-emission from plastics, rubber, and textiles incorporating these additive flame retardants (de Wit et al., 2010). PC2 (27.0 %) exhibited high loading (> 0.8) for TBP-AE and negative loading (< -0.4) for PBEB, reflecting different usage patterns-TBP-AE being common in expanded polystyrene and insulation boards as a substitute for phased-out PBDEs.

[Insert Fig. 2]

No significant relationships were observed between OPE or NBFR concentrations and household characteristics such as building age, distance from major roads, floor type, or number of electronics—except for TCEP, which showed a negative correlation ($p < 0.05$) with both house age and number of devices (**Table S11**). This likely reflects newer homes containing recently manufactured materials with OPE formulations, whereas older homes may emit fewer chemicals due to reduced synthetic furnishing content. Weak correlations may also result from limited sample size and uncontrolled factors such as temperature, humidity, and ventilation, which influence indoor redistribution (Abdallah & Harrad, 2010). In general, the correlation and PCA findings demonstrate that dominant OPEs in Kampala households arise mainly from modern consumer materials and building products, whereas NBFRs originate from mixed polymeric and electrical components. These results

provide the first evidence of region-specific emission pathways for indoor flame retardants in East Africa.

3.4. Health risk assessment

Human exposure to OPEs and NBFRs via dust ingestion and dermal contact was evaluated for adults and children, the two major indoor exposure pathways (Yadav et al., 2019). The assessment followed the U.S. EPA human health risk model but was contextualized for East African households, where prolonged indoor occupancy and limited ventilation may enhance exposure. Estimated average and high (95th percentile) doses are summarized in **Table 2**. Assuming 100 % absorption (Jones-Otazo et al., 2005), average daily doses (ADD_{total}) for 5 OPEs were in the range $(0.039 - 71) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, and for 4 NBFRs were in the range $(0.0017 - 1.5) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$. Under high-exposure conditions, ADD_{total} ranged $(0.12 - 180) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ for OPEs and $(0.0048 - 3.8) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ for NBFRs-values well below reference doses, indicating low non-carcinogenic risk even under conservative scenarios.

[Insert Table 2]

Under typical exposure scenarios, dust ingestion was identified as the primary pathway for human intake of flame retardants, with the estimated average daily exposures in the range $(0.038 - 8.2) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for OPEs and $(0.0017 - 1.1) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, weight/day for the NBFRs. These values increased substantially under high-exposure conditions, reaching up to $44 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for OPEs and $3.5 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for the NBFRs. In contrast, dermal contact with dust contributed comparatively lower exposures in the range $(0.00077 - 2.8) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for OPEs and $(0.000033 - 0.38) \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for NBFRs under average conditions. Even under elevated exposure scenarios, dermal uptake remained modest, with maximum estimates of $15 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for the OPEs and $0.24 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$, for the NBFRs. These findings underscore the predominance of ingestion as the key exposure route for flame retardants in indoor environments. These results are consistent with studies from China and Europe where ingestion dominated total exposure (Luongo & Östman, 2016; Wu et al., 2016).

TDCIPP and TCIPP contributed most to total OPE exposure, followed by TBOEP and TNBP, while TCEP contributed least due to its lower concentrations and adherence potential. For NBFRs, HBB and TBP-AE predominated, indicating a potentially elevated health risk associated with their presence. Children exhibited higher exposures than adults via both pathways, consistent with greater hand-to-mouth activity and dust ingestion (Annesi-Maesano et al., 2013; Hou et al., 2016b; Quirós-Alcalá et al., 2016). At high exposure, total ADDs were $0.2 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ (adults) and $310 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ (children) for OPEs, and 0.59×10^{-6} (children) and $7.1 \times 10^{-6} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ (adults) for NBFRs.

Compared with international datasets, Kampala households exhibited ingestion-based ADDs within the global range-similar to Egypt (Abdallah & Covaci, 2014) and Shanghai (Wu et al., 2016), but below Sweden (Luongo & Östman, 2016)-suggesting moderate contamination levels typical of developing urban settings.

HI-values under high exposure were 170×10^{-3} and $11 \times 10^{-3} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ for OPEs and 34×10^{-3} and $2.8 \times 10^{-3} \text{ mg kg}^{-1} \text{ bw day}^{-1}$ for NBFRs (children and adults, respectively), all below unity, confirming negligible non-carcinogenic risks (**Table 3**). Nonetheless, combined exposure to mixed FR classes may yield additive toxicity that remains poorly characterized in sub-Saharan contexts (Kojima et al., 2013). In addition, the CR values for OPEs were 2.2×10^{-6} and 5.7×10^{-6} for children, and 0.15×10^{-6} and 0.37×10^{-6} for adults at average and high exposure scenarios, respectively (**Table 3**). All the values were below the U.S. EPA's threshold (10^{-6}), indicating very low lifetime cancer risk, though slightly elevated CRs in children highlight their vulnerability.

[Insert Table 3]

Although CR values ($10^{-6} - 10^{-12}$) indicate negligible immediate concern, chronic exposure to mixed OPEs and other pollutants-particularly in children-warrants continued biomonitoring.

4. Conclusions

This study provided a comprehensive investigation of OPEs and NBFRs in indoor dust from Ugandan households, offering novel insights into their occurrence, emission sources, and potential health implications. The concentrations of Σ_5 OPEs (1,520–102,000 ng g⁻¹) were higher than Σ_4 NBFRs (214–6,600 ng g⁻¹), consistent with their broader use in the consumer products and building materials. Among the OPEs, TDCIPP and TCIPP were the predominant congeners, while HBB and TBP-AE dominated the NBFR profiles. Source apportionment suggested that emissions originated mainly from floor polishes, foams, electricals, plastics, and construction materials, reflecting product composition and household usage patterns.

Human health risk assessment revealed that dust ingestion contributed more significantly to exposure than dermal contact. The estimated hazard indices (HIs) for both adults and children were <1, indicating low non-carcinogenic risk. Carcinogenic risk estimates were within safe limits, though children exhibited relatively higher exposures due to behavioral and physiological factors. These results demonstrate that, while indoor exposure to OPEs and NBFRs in Ugandan households currently poses negligible health risk, continuous monitoring is warranted as consumer product turnover and importation patterns evolve.

This study provides baseline data for indoor flame-retardant exposure in East Africa, contributing valuable regional data to global monitoring frameworks. Future work should incorporate additional indoor microenvironments-such as offices, vehicles, and schools-and consider temporal and seasonal dynamics. Integrating dust, air, and biomonitoring data will enhance understanding of long-term exposure and inform regional policy development toward safer material use and chemical regulation.

CRediT authorship contribution statement

Godfrey Muhwezi: Writing - original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Christine Kyarimpa:** Writing - review & editing, Supervision, Funding acquisition. **Ivan Gumula:** Writing - review & editing, Supervision. **Silver Odongo:** Writing - review & editing, Methodology. **Henry Matovu:** Writing -

review & editing. **Julius Matsiko:** Writing - review & editing. **Christine Betty Nagawa:** Writing - review & editing, Funding acquisition. **Charles Drago Kato:** Writing - review & Editing. **Ivan Špánik:** Writing - review & editing, Funding acquisition. **Mika Sillanpää:** Writing - review & editing. **Adewale M. Taiwo:** Writing - review & editing. **Ashirafu Miir:** Writing - review & editing. **Patrick Ssebugere:** Writing - review & editing, Supervision, Funding acquisition, Conceptualization.

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Table 1: Levels of OPEs and NBFRs (ng g⁻¹) in household dust samples.

Analytes	Indoor dust (N=43)				
	DF (%)	Mean $\pm$ SD	Median	Min – Max	95 th Percentile
TNBP	100.0	678 $\pm$ 1,040	173	40–3,670	3,100
TBOEP	100.0	3,830 $\pm$ 4,790	2,120	4–21,200	13,200
TCIPP	100.0	3,880 $\pm$ 4,590	2,470	31–19,600	13,300
TCEP	100.0	184 $\pm$ 204	135	7–1,260	409
TDCIPP	100.0	17,400 $\pm$ 17,800	15,700	1,390–99,200	39,900
EHDPP*	18.0	97 $\pm$ 79	<LOD	<LOD –210	165
TPHP*	0.0	<LOD	<LOD	<LOD	<LOD
Σ_5 OPEs		25,900	21,900	1,520–102,000	62,400
HBB	100.0	398.0 $\pm$ 263	341.0	87.7–1440.0	780.0
PBT	93.0	52.8 $\pm$ 41.0	38.4	<LOD –176.0	154.0
PBEB	100.0	7.3 $\pm$ 5.90	5.8	0.9–29.4	16.3
TBP*	4.70	<LOD	<LOD	<LOD –928.0	<LOD
TBP-AE	100.0	449.0 $\pm$ 950	238.0	26.1–6,230.0	1050.0
Σ_4 NBFRs		908 $\pm$ 985	712	214–6,600	1852

DF (%) - Detection frequency (percentage), SD - Standard deviation, Min - Minimum, Max – Maximum, <LOD-Below limit of detection, * not part of the sum because of low DF.

Table 2: Average daily dose ($\text{mg kg}^{-1} \text{bw day}^{-1}$) for OPEs and NBFRs in adults and children.

Analytes	Category	ADD _{der} ($\times 10^{-6}$)		ADD _{ing} ($\times 10^{-6}$)		ADD _{total} ($\times 10^{-6}$)	
		Average ^a	High ^b	Average ^a	High ^b	Average ^a	High ^b
TNBP	Children	0.19	3.50	0.58	10.0	0.77	14.0
	Adult	0.0099	0.018	0.049	0.89	0.05	0.9
TBOEP	Children	2.40	15.0	7.00	44.0	9.40	59.0
	Adult	0.012	0.075	0.60	3.80	0.62	3.9
TCIPP	Children	2.80	15.0	8.20	44.0	11.0	59.0
	Adult	0.014	0.076	0.71	3.80	0.72	3.90
TCEP	Children	0.15	0.46	0.45	1.40	0.60	1.80
	Adult	0.00077	0.0023	0.038	0.12	0.039	0.12
TDCIPP	Children	18.0	45.0	53.0	130	71.0	180
	Adult	0.091	0.23	4.60	11.0	4.60	120
$\Sigma_5\text{OPEs}$	Children	23.0	78.0	69.0	230	93.0	310
	Adults	0.12	0.40	6.00	20.0	6.10	20.0
HBB	Children	0.38	0.18	1.10	2.60	1.50	2.80
	Adult	0.0019	0.0052	0.097	0.22	0.099	0.23
PBT	Children	0.043	0.034	0.13	0.51	0.17	0.55
	Adult	0.00022	0.001	0.011	0.044	0.011	0.045
PBEB	Children	0.0066	0.0037	0.019	0.055	0.026	0.058
	Adult	0.000033	0.00011	0.0017	0.0047	0.0017	0.0048
TBP-AE	Children	0.21	0.24	0.79	3.50	1.10	3.80
	Adult	0.0014	0.0071	0.068	0.30	0.069	0.31
$\Sigma_4\text{NBFR}$	Children	0.70	0.45	2.10	6.70	2.80	7.10
	Adult	0.0035	0.013	0.18	0.57	0.18	0.59

ADD_{der} -Average daily dose via dermal contact; ADD_{ing} -Average daily dose via ingestion; ^a Estimated by median concentration in indoor dust, ^b Estimated by 95th percentile concentration in indoor dust, TPHP was not detected in any of the samples and TBP and EHDPP had low DF and was not used, in exposure calculations.

Table 3: Estimated non-carcinogenic risk and carcinogenic risk from exposure to OPEs and NBFRs.

Analyte	Non-carcinogenic risks (HQ)				Carcinogenic risks			
	Children ($\times 10^{-3}$)		Adults ($\times 10^{-3}$)		Children ($\times 10^{-6}$)		Adults ($\times 10^{-6}$)	
	Average	High	Average	High	Average	High	Average	High
TNBP	0.32	5.70	0.021	0.38	0.0069	0.11	0.00045	0.0081
TBOEP	6.30	39.0	0.41	2.60	NA	NA	NA	NA
TCIPP	1.40	7.40	0.09	0.49	NA	NA	NA	NA
TCEP	0.27	0.83	0.018	0.054	0.012	0.042	0.00078	0.0024
TDCIPP	47.0	120	3.10	7.80	2.20	6.20	0.14	0.36
Σ OPEs	56.0	170	3.60	11.0	2.20	5.70	0.15	0.37
HBB	15.0	28.0	0.99	2.30	NA	NA	NA	NA
PBT	1.70	5.50	0.11	0.45	NA	NA	NA	NA
PBEB	0.26	0.58	0.017	0.048	NA	NA	NA	NA
Σ NBFRs	17.0	34.0	1.10	2.80				

NA - Not available due to absence of CSF.

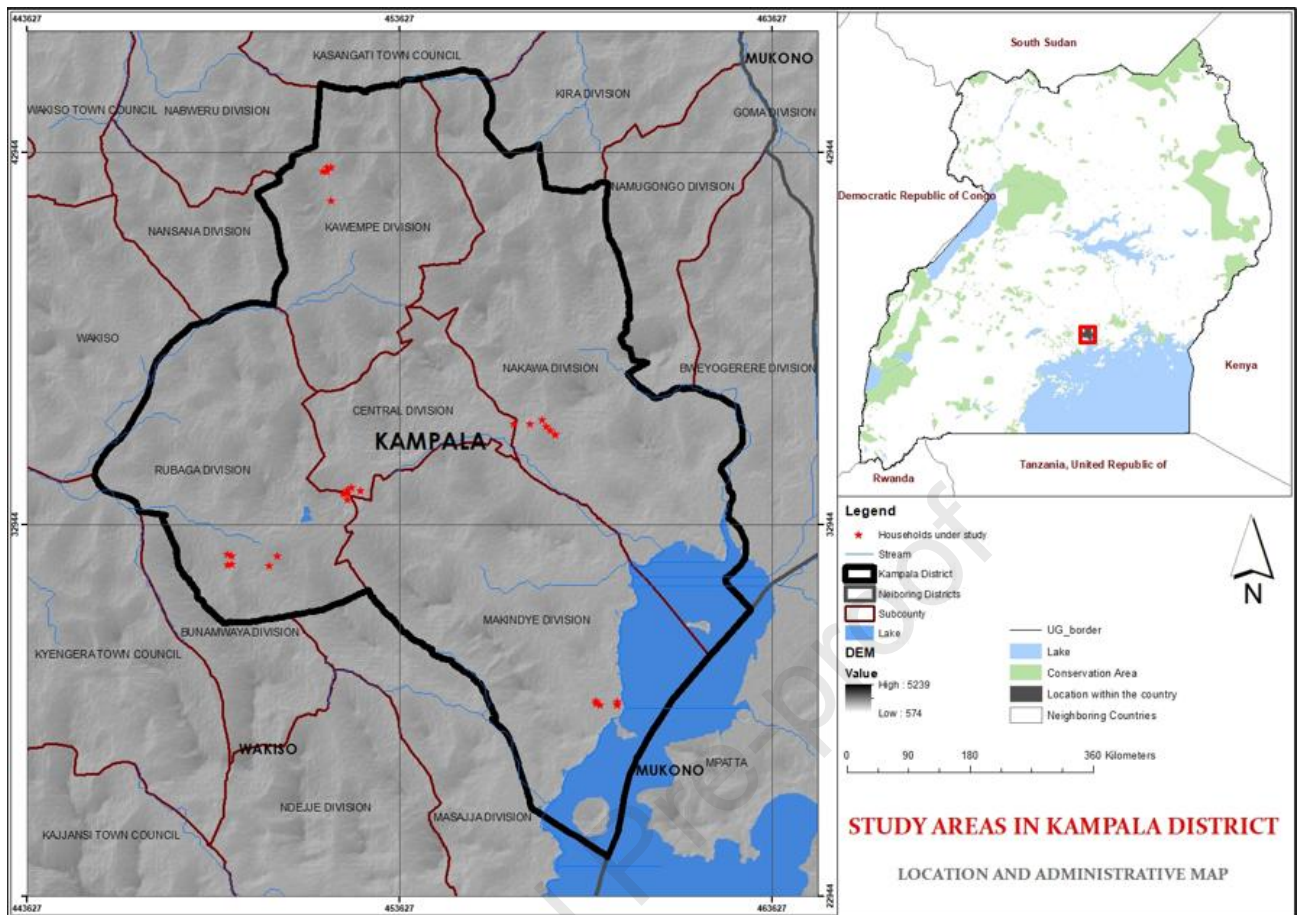


Fig. 1: A map of Kampala showing the sampling sites.

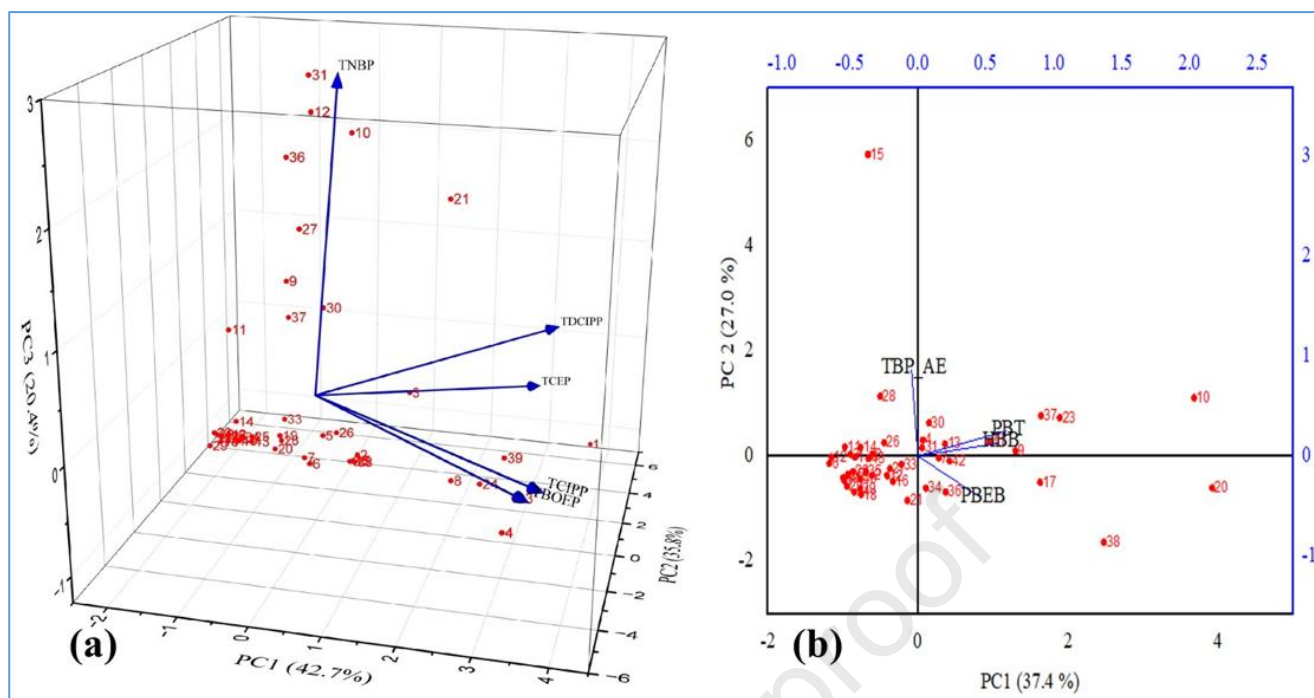


Fig. 2: Principal component analysis of: (a) OPEs and (b) NBFRs in indoor floor dust. Red numbered spots represent sample points.

Highlights

- OPEs and NBFRs were quantified in indoor dust from Uganda for the first time.
- TDCIPP dominated OPEs, while HBB and TBP-AE were the main NBFRs detected.
- Major emission sources were building products and electrical components.
- Estimated hazard indices (< 1) indicated negligible non-carcinogenic risk.
- Carcinogenic risk assessment showed relatively low risk to children and adults.

Declaration of interests

☒ 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.

☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: