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RESEARCH

Indoor Dust Levels of Polybrominated Diphenyl Ethers (PBDEs) and Polybrominated Dibenzo-*p*-Dioxins/Furans (PBDD/Fs) in Various Taiwanese Microenvironments

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Abstract

Several studies have reported contamination of polybrominated dioxins/furans (PBDD/Fs) and diphenyl ethers (PBDEs) in diverse microenvironments. However, information on their levels in nursing homes (NH) and Chinese medical clinics (CMC) remains limited. This study investigated the levels, congener patterns, and potential health risks of indoor dust PBDD/Fs and PBDEs in 17 NH, 18 CMC, normal (NR) and computer (CR) classrooms of 24 elementary schools, 10 kindergartens (K), and 20 automobiles (A) in southern Taiwan. Σ_{14}PBDEs varied substantially among environments, with the highest levels observed in CMC ($4{,}790 \pm 16{,}500$ ng g^{-1}), followed by NH, CR, and A, while significantly lower levels were found in NR and K. Principal component analysis indicated that PBDE profiles were mainly associated with commercial Deca-BDE formulations, with occasional influence from historical Penta- and Octa-BDE mixtures. Indoor dust Σ_{12}PBDD/Fs ranged from 2.51 ng g^{-1} (K) to 22.8 ng g^{-1} (A), with the highest WHO_{2005}-TEQ values in A (79.0 pg WHO_{2005}-TEQ g^{-1}). PBDFs contributed more substantially to total PBDD/F toxicity than PBDDs. Correlation analysis revealed significant associations between PBDEs and several PBDF congeners, particularly HpBDF and OctBDF, suggesting in situ debromination of PBDEs under sunlight and thermal conditions, such as those present inside automobiles. For human health risk assessment, the estimated daily intakes and both the estimated risk of cancer and non-cancer from indoor dust PBDEs and PBDD/Fs for all assessed populations, including patients and staff in NH and CMC, as well as schoolchildren and preschool children,

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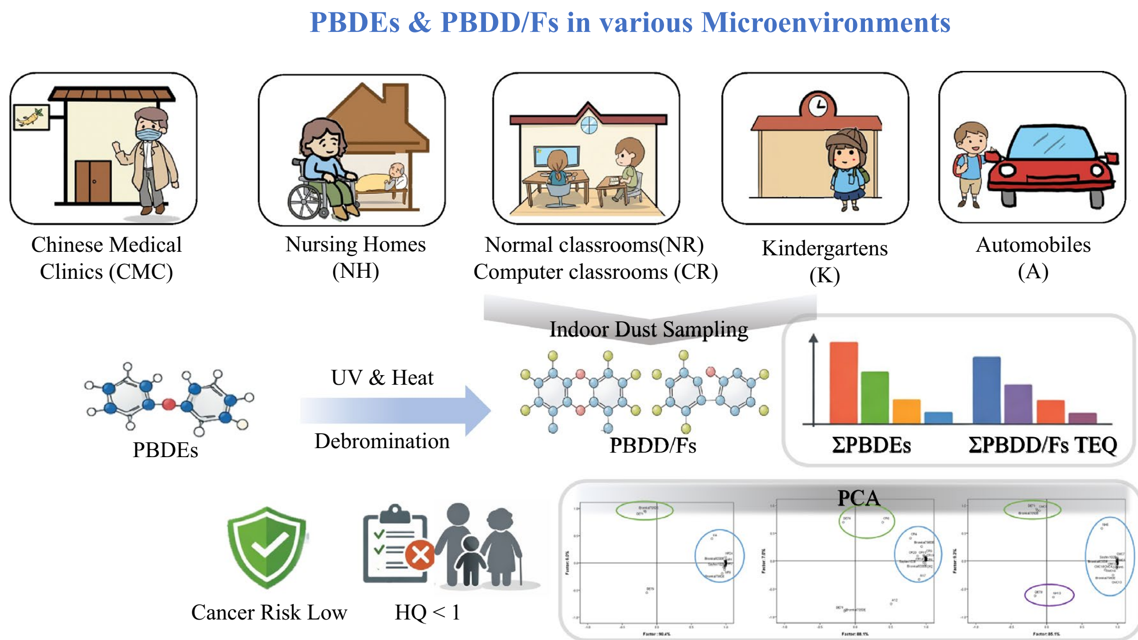
remained within acceptable safety thresholds. Overall, these findings demonstrate that although PBDEs and PBDD/Fs persist in indoor environments, especially in CMC and A, their current levels do not pose significant health risks. The present study reveals the first comprehensive assessment of PBDEs and PBDD/Fs in CMC and NHs, offering valuable insights for exposure management and indoor environmental health in Taiwan.

Graphical abstract



Keywords Nursing home · Chinese medical clinic · Polybrominated dibenzo-*p*-dioxins · Children's health · Polybrominated diphenyl ethers · Indoor dust

1 Introduction

Halogenated chemicals like polybrominated diphenyl ethers (PBDEs) and polybrominated dibenzo-*p*-dioxins/furans (PBDD/Fs) are indeed considered persistent organic pollutants (POPs), which are of public concern due to their potential environmental and health impacts (Wang et al. 2010; Lin et al. 2021; Su et al. 2022). PBDEs are only used as brominated flame retardants (BFRs) in a variety of consumer products, including electronics, furniture foam padding, vehicles, consumer products, wire insulation, and textiles (Chao et al. 2014b; Gou et al. 2016). Among PBDEs, Penta-BDEs and Octa-BDEs were phased out since 2005, while Deca-BDE was finally added to the Stockholm Convention, with a ban on its use effective from 2017. Despite the phase-outs and bans, older consumer products, furniture, automobiles, and decorations remain prevalent in various microenvironments, continuously releasing PBDEs from their surfaces and posing heightened risks to human health (Blum et al. 2019). PBDD/Fs as a group of the halogenated chemicals have a similar structure to PBDEs, which are also POPs listed in Stokholm Convention as Annex C (unintentional production) (Hayakawa et al. 2004; Chao et al. 2014a; Yang et al. 2021). PBDEs are recognized as a major contributor to the global distribution of PBDD/Fs, particularly through combustion or thermal processes, additionally, it has been suggested that PBDD/Fs may be formed from low molecular weight PBDEs during pyrolysis (Gullett et al. 2010; Liang et al. 2020).

Both brominated POPs of PBDEs and PBDD/Fs are acknowledged as toxic substances that are bioaccumulated in the environment and in living organisms, including humans (Johnson-Restrepo and Kannan 2009; Chou et al. 2016; Chen et al. 2018; Lin et al. 2021). Human exposure to these toxic substances has been related with several adverse effects, including neurotoxicity, developmental and reproductive toxicity, carcinogenicity, and cancer (Chao et al. 2014a; Domingo et al. 2025). These effects include delays in neurobehavioral and neurological development, menstrual disorders, disruptions in time to pregnancy, alterations in thyroid and growth hormone secretion, and reduced semen quality (Chao et al. 2014a). PBDEs and PBDD/Fs are ubiquitously existed in the microenvironment due to the release of PBDEs on the surface of the consumer products and inadvertent emission of PBDD/Fs from thermal degradation of PBDEs. Previous studies have shown positively significant correlations between dust levels of PBDD/Fs and PBDEs in indoor environments, indicating that these chemicals may be released from consumer or electronic products to be accumulated in indoor dust (Harrad et al. 2006; Suzuki 2006; Gou et al. 2016; Kajiwara and Takigami 2016; Su et al. 2022). PBDD/Fs and PBDEs can be accumulated in indoor dust, which poses a risk to human health (Chao et al. 2014a; Gou et al. 2016; Su et al. 2022). Understanding the sources and concentrations of PBDD/Fs and PBDEs in indoor dust is essential for formulating effective strategies to minimize exposure and protect human health (Suzuki 2006; Su et al. 2022). Several studies have investigated levels of PBDEs and PBDD/Fs in the microenvironments including indoor dust and indoor air (Tue et al. 2013; Gou et al. 2016; Su et al. 2022). Compared with PBDEs, the PBDD/F levels in the indoor dust were relatively lower in our previous reports (Gou et al. 2016; Su et al. 2022). Our previous study examined PBDEs and PBDD/Fs in gymnasium indoor environments to determine their levels in $PM_{2.5}$ and dust to further identify potentially positive correlations between these chemicals and the matrices based on our findings (Su et al. 2022). Additionally, our study conducted in Taiwan found a positive association between PBDD/Fs and PBDEs in indoor dust from elementary classrooms and gyms, suggesting that these chemicals may originate from the same sources in the indoor environment (Gou et al. 2016; Su et al. 2022).

PBDE concentrations were the most frequently investigated in house dust among the microenvironments including primary school, daycare centers, and vehicles in the previous studies (Raffy et al. 2017; Muenhor and Harrad 2018; Lin et al. 2022; Yu et al. 2023; Zhu et al. 2023). In the previous studies (Ali et al. 2011; Thuresson et al. 2012; Bu et al. 2019), the indoor air and dust of PBDEs were investigated in daycare centers to reveal higher concentrations of PBDEs compared to house dust. In the Chinese study (Lin et al. 2022), indoor dust PBDEs collected from the residence and kindergartens in the non-e-waste region were notably lower than those in the e-waste region and it also observed that PBDEs in house dust were higher than the dust from the kindergartens. High levels of PBDEs from automobile dust samples were found in the studies from South Africa (Abafe and Martincigh 2016) and China (Jin et al. 2021).

Given that the Taiwanese population is aging and life expectancy is increasing, many elderly individuals in Taiwan regularly visit traditional Chinese medicine clinics or reside long-term in nursing homes due to disabilities, functional impairments, or chronic diseases. A variety of electronic and consumer products are used in these clinics and nursing homes in Taiwan. No one knows the extent of PBDE and PBDD/F contaminants in the indoor environments of traditional Chinese medicine clinics and nursing homes, or whether their concentrations negatively impact human health. (Sun et al. 2021) reported that PBDEs are contaminated in Pheretima, a traditional Chinese medicine listed in the Chinese Pharmacopoeia. In the Chinese medical clinics, acupuncture and orthopedic equipment may release PBDEs during thermal therapies. However, it remains unclear whether PBDEs or PBDD/Fs are emitted from the surfaces of electronic devices or traditional medicine, potentially leading to deposition on the indoor floors of these clinics.

In our previous studies, PBDEs in indoor dust were investigated in the several types of Taiwanese microenvironments including the residence, offices, elementary schools, medical and dental clinics, and gymnasium. In the present study, PBDE and PBDD/F levels in indoor dust collected from the normal and computer classrooms of the elementary schools, kindergartens, automobiles, nursing homes, and Chinese medical clinics in Taiwanese microenviornments were investigated to assess the risk of Taiwanese population via non-dietary pathway.

2 Methods

2.1 Reagents and Materials

The external standards for PBDEs and PBDD/Fs consist of a native mixture of unlabeled brominated compounds including 14 PBDE congeners (BDE-28, 47, 99, 100, 153, 154, 183, 196, 197, 203, 206, 207, 208, and 209) and 12 PBDD/F congeners (2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, 2,3,4,7,8-PeBDF, 1,2,3,4,7,8-HxBDF, 1,2,3,4,6,7,8-HpBDF, OctBDF, 2,3,7,8,-TeBDD, 1,2,3,7,8-PeBDD, 1,2,3,7,8,9-HxBDD, 1,2,3,4,6,7,8-HxBDD, 1,2,3,4,6,7,8-HpBDD, and OctBDD) from Cambridge Isotope Laboratories (Andover, MA, USA). The internal brominated-$^{13}C_{12}$-labeled standards for PBDD/Fs (2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, 2,3,4,7,8-PeBDF, 1,2,3,4,7,8-HxBDF, 1,2,3,4,6,7,8-HpBDF, OctBDF, 2,3,7,8-TeBDD, 1,2,3,7,8-PeBDD, 1,2,3,7,8,9-HxBDD, 1,2,3,4,6,7,8-HxBDD, 1,2,3,4,6,7,8-HpBDD, and OctBDD) and PBDEs (BDE-28, 47, 99, 153, 154, 183, 197, 207, and 209) are supplied by Wellington Laboratories (Guelph, Canada). High-grade chemicals such as potassium oxalate, sodium sulfate, silica gel, and alumina oxide are sourced from Merck (Darmstadt, Germany). HPLC-grade solvents, including dichloromethane, toluene, acetone, and n-hexane, are obtained from Tedia (Fairfield, USA), Merck, or Sig-ma-Aldrich (St. Louis, USA).

2.2 Study Design and Collection of Samples

This study is part of a survey on PBDEs and PBDD/Fs in Taiwanese microenvironments (MOST 106-2221-E-020-001-MY3 and PTVGH-11303) (Su et al. 2022). The collection of organobromines from indoor dust, including floor dust and air-conditioner filter dust, is referenced in the reports described previously (Chou et al. 2016). This study investigates specific indoor environments, such as normal (NR) and computer (CR) classrooms of the elementary schools, kindergartens (K), automobiles (A), Chinese medical clinics (CMC) and nursing homes (NH), where activities for the sensitive population like the elderly and young children in Taiwan were conducted. More than 200 institutes including NR, CR, K, A, CMC, and NH are invited, but only 24 paired samples of NR and CR in the elementary school, 10 K, 20 A, 18 CMC, and 17 NH in Kaohsiung City and Pingtung County are agreed to participate in this project. Indoor dust, including air-conditioner filter dust and floor dust, is collected using a strict procedure with a Nilfisk Advance Euroclean UZ934 HEPA canister vacuum cleaner (Chou et al. 2016; Gou et al. 2016; Su et al. 2022).

The vacuum equipped the high efficiency particulate air filter (HEPA filter) used in the present study is recommended by the U.S. Department of Housing and Urban Development (HUD) and the Occupational Safety and Health Administration (OSHA), as it is designed to theoretically capture at least 99.97% of particulate matter, dust, mold, bacteria, and pollen as small as 0.3 µm. The HEPA filter vacuum is ideal for cleaning homes, offices, hospitals, schools, hotels, retail outlets, and restaurants. The Nilfisk Advance Euroclean UZ934 HEPA canister vacuum cleaner features three separate filtration systems: a filter bag, a pleated filter, and an exhaust filter, effectively capturing indoor dust, including dust from air conditioner filters and floors. Prior to sampling, all tools and accessories of the HEPA canister vacuum, including a six-foot suction hose, a two-piece aluminum wand, a 10.5-inch crevice tool, an optional turbo nozzle, and a dusting tool, are thoroughly cleaned using soapy water followed by rinsing with deionized distilled water (Ali et al. 2011; Shy et al. 2015). Additionally, the surface of the vacuum is cleaned using an n-hexane-impregnated disposable wipe. After sampling indoor floor dust or air conditioner filters, the dust is swiftly transported to our laboratory, where it is immediately sieved through a 100-mesh screen by shaking for at least 10 min at the National Pingtung University of Science and Technology. The homogenized dust samples are then collected in a brown glass bottle with a Teflon cap and stored at − 20 °C for further chemical analysis.

2.3 Samples Preparation

The pretreatment process, consisting of Soxhlet extraction followed by multi-column clean-up with a multilayer silica gel column, an alumina column, and an activated carbon column, was performed according to methods described in our previous studies (Chao et al. 2016; Gou et al. 2016; Lee et al. 2023), with minor modifications applied prior to chemical analysis. Briefly, labeled isotopes of PBDEs and PBDD/Fs are spiked into indoor dust samples to evaluate the loss of these organohalogenated compounds during the pretreatment processes. Before the initial extraction procedure, the internal standards of $^{13}C_{12}$ PBDD/Fs and PBDEs are added to the indoor dust samples and thoroughly homogenized with toluene. Dust samples weighing 5–10 g are mixed with 250 mL of toluene in a Soxhlet extractor and subjected to extraction for 24 h. The ex-tract is then concentrated to 2 mL using a rotary evaporator (LabTech, Taiwan) before passing through the multi-column clean-up systems. Elution is performed with 15 mL of hexane, followed by 25 mL of dichloromethane/hexane (1:24, v/v), 5 mL of toluene/methanol/ethyl acetate/hexane (1:1:2:16, v/v/v/v), and finally 40 mL of toluene. The final elute is evaporated to approximately 1 mL in a vial. The solution in the vial is then concentrated to near dryness using a gentle stream of nitrogen gas, resulting in a final volume of 0.2 mL.

2.4 Analysis of PBDD/Fs and PBDEs

Thirty-five indoor dust samples from 17 nursing homes and 18 Chinese medical clinics are analyzed to determine the congeners of 12 PBDD/Fs and 14 PBDEs using a high-resolution gas chromatograph (Hewlett–Packard 6970 Series, CA) equipped with a high-resolution mass spectrometer (Micromass Autospec Ultima, Manchester, UK) (HRGC/HRMS). The system includes a splitless injector and a DB-5HT silica capillary column (L = 60 m, i.d. = 0.25 mm, film thickness = 0.1 µm) (J&W Scientific, CA). The analytical methods, procedures, conditions, and equipment are detailed in our previous studies (Gou et al. 2016; Lee et al. 2023). The quality assurance and quality control (QA/QC) for the PBDE and PBDD/F analyses in indoor dust adhere to US EPA Method 1614 and the Taiwanese Ministry of Environment (TMOE) NIEA M803.00B. To confirm the precision and recovery (PAR), the spiked internal and surrogate standards showed recoveries of 44.3% to 103% for the labeled internal standards and 87.6% to 139% for the spiked surrogate standards. Blanks (BKs) for the analysis of PBDD/Fs and PBDEs, including field blanks, filter blanks, solvent blanks, and glassware blanks, are routinely checked in each sample batch. The detection limits, including the limits of detection (LODs, defined as signal-to-noise (S/N) > 3), limits of quantification (LOQs, S/N > 10), and method detection limits (MDLs, evaluated as 2.5–5.0 times the estimated LODs), were examined in this study. The MDLs for PBDD/Fs range from 0.0534 to 24.8 pg g^{-1} and from 4.33 to 57.6 pg g^{-1}, except for BDE-209, which has an MDL of 299 pg g^{-1}. According to previous studies (van den Berg et al. 2013; Gou et al. 2016), It is recommended to apply the same World Health Organization (WHO) toxic equivalency factor (TEF) schemes for PCDD/Fs and PBDD/Fs when evaluating human health risks. Accordingly, the toxic equivalency (TEQ) values of PBDD/Fs in this study are calculated based on the TEFs proposed by van den Berg et al. (2006).

2.5 Risk Assessment and Statistical Analysis

The risk assessments for patients and professional staff in NH and CMC, as well as for young children (schoolchildren and preschoolers) in various indoor environments (K, NR, and A), through non-dietary exposure pathways, are conducted following the approaches described in our previous reports (Chou et al. 2016; Gou et al. 2016; Su et al. 2022), with minor modifications. The equations of daily intake, chronic daily intake, non-cancer risk, and cancer risk of PBDEs and PBDD/Fs in the microenvironments for our assessed populations are presented in the supplementary materials. Measurements of PBDD/Fs below the MDLs for organobromines are recorded as zero. The levels of PBDD/Fs and PBDEs in indoor dust from our selected microenvironments do not follow a normal distribution after the Kolmogorov–Smirnov tests. The nonparametric tests, such as the Mann–Whitney *U* and Kruskal–Wallis *H* tests, are used to confirm the differences in PBDE and PBDD/F concentrations in indoor dust among the selected microenvironments in this study. Principal component analysis (PCA) is used to examine the association between the composition of BDE congeners in our samples and the PBDE commercial formulations, including Penta-BDEs, Octa-BDEs, and Deca-BDEs. Spearman's rho correlation coefficient tests are used to examine the correlations between PBDD/Fs and PBDEs in indoor dust. All statistical analyses were conducted using Statistical Product and Service Solutions (SPSS), version 12.0.

3 Results and Discussion

3.1 Levels of PBDEs and PBDD/Fs in Various Microenvironments

The present study focuses on investigating brominated POPs, such as PBDEs and PBDD/Fs, in indoor dust within specific microenvironments, including the classrooms (NR and CR) in the elementary schools, K, A, NH and CMC. Table 1 shows the levels of 14 PBDE congeners (BDE-28, 47, 100, 99, 154, 153, 183, 197, 203, 196, 208, 207, 206, 209) in indoor dust (floor dust) collected from CMC, NH, NR, CR, K, and A. Dust levels of Σ_{14}PBDEs in NR (mean ± standard deviation (SD): 440 ± 343 ng g^{-1}) and K (453 ± 288 ng g^{-1}) are notably lower than those collected from CR (2,330 ± 2,740 ng g^{-1}) with statistically significant differences ($p < 0.05$), but their levels are shown no significant difference ($p > 0.05$) compared with those in A (1,980 ± 3,600 ng g^{-1}), CMC (4,790 ± 16,500 ng g^{-1}) and NH (2,320 ± 4,130 ng g^{-1}). Although the mean Σ_{14} PBDE levels in indoor dust in NH, CR, and A are lower compared to the level found in CMC, no statistical differences are observed. BDE-209 is the predominant congener among the 14 PBDE congeners, comprising 69.1%, 59.8%, 64.5%, 59.8%, 65.7%, and 65.1% of the total PBDEs in indoor dust collected from both NR, CR, K, A, CMC, and NH, respectively.

Based on our results (Table 1), levels of Σ_{14}PBDEs in NR and K are lower than those in the other selected microenvironments (CR, A, CMC, and NH), indicating that low amounts of PBDEs are released from the consumer products in NR and K. Indoor dust Σ_{14}PBDE levels in CR, A, and NH are comparable to be lower than that that in CMC. In Taiwan, our findings (Tables 1, S1 and S2) indicate that indoor dust Σ_{14}PBDE levels in NH, A, and CR in the present study are not significantly higher than those found in residents (Chao et al. 2014b), CR in urban areas (Gou et al. 2016), and gyms (Su et al. 2022) from southern Taiwan. Σ_{14}PBDEs in indoor dust collected from NR and K in this report seem to have the similar magnitudes to those from NR in the rural and urban regions in our previous study (Gou et al. 2016). PBDE concentrations in indoor floor dust from NH, A, and CR in the present report are relatively high compared to those in NR from urban and rural schools (Gou et al. 2016) and are comparable to those in residential houses (Chao et al. 2014a), CR in both urban and rural areas (Gou et al. 2016), and gyms (Su et al. 2022). Compared to NH, A, and CR, elevated levels of indoor dust PBDEs are found in CMC, comparable to those in electronic dust from residential houses (Chao et al. 2014a, 2014b; Shy et al. 2015). Indoor dust concentrations of BDE-209 in various Taiwanese microenvironments, including NH, CMC, residences, NR, CR, K, A, and gyms, accounted for 65–90% of Σ_{14}PBDEs (Table 1 and Table S1). Compared to indoor dust collected from e-waste recycling areas or workshops (Wannomai et al. 2021; Lin et al. 2022), the levels of PBDEs in indoor dust from Taiwanese microenvironments in the present and the previous studies were relatively lower. In Figure S1, the diversity of PBDE patterns among indoor dust samples from different microenvironments likely reflects variations in the use and composition of PBDE-treated consumer products. This is the first report to announce indoor dust PBDE

Table 1 Indoor dust concentrations of PBDEs in normal classroom (NR), computer classroom (CR), kindergarten classroom (K), automobiles (A), nursing homes (NH), and Chinese medical clinics (CMC) (ng g^{-1})

	Elementary school		Kindergarten ($n=10$)	Automobiles ($n=20$)	Nursing homes ($n=17$)	Chinese medical clinics ($n=18$)
	NR ($n=24$)	CR ($n=24$)				
BDE-28	0.0249–0.754 (0.141 ± 0.181)[a]	0.0225–1.07 (0.305 ± 0.292)	0.0260–0.295 (0.115 ± 0.0862)	0.0325–3.13 (0.637 ± 0.986)	0.0741–1.150 (0.245 ± 0.298)	0.0570–9.424 (0.879 ± 2.18)
BDE-47	0.368–62.3 (4.57 ± 12.4)	0.403–48.5 (5.82 ± 9.51)	0.866–21.6 (5.95 ± 6.95)	0.749–565 (54.2 ± 133)	1.70–222 (20.1 ± 52.8)	0.645–1080 (65.2 ± 254)
BDE-100	0.0551–11.8 (1.02 ± 2.37)	0.0909–8.56 (1.19 ± 1.74)	0.245–8.28 (2.35 ± 2.81)	0.259–199 (17.6 ± 45.9)	0.327–85.6 (7.25 ± 20.5)	0.113–709 (41.00 ± 167)
BDE-99	0.389–65.4 (5.93 ± 13.0)	0.710–34.6 (6.62 ± 7.45)	1.80–53.2 (16.0 ± 18.7)	1.70–1018 (98.6 ± 242)	2.08–364 (34.9 ± 87.0)	0.991–1990 (121 ± 467)
BDE-154	0.172–4.00 (0.861 ± 0.890)	0.136–111 (5.73 ± 22.3)	0.384–7.24 (2.24 ± 2.46)	0.332–188 (18.5 ± 46.2)	0.262–55.1 (8.54 ± 16.8)	0.291–575 (33.7 ± 135)
BDE-153	0.478–7.45 (2.08 ± 1.54)	0.427–556 (29.1 ± 113)	0.970–12.2 (5.03 ± 4.37)	0.566–220 (29.0 ± 64.2)	0.515–419 (38.2 ± 101)	0.854–890 (54.8 ± 208)
BDE-183	1.21–17.6 (6.04 ± 4.21)	1.91–2020 (109 ± 410)	1.10–47.0 (11.9 ± 16.0)	0.833–31.5 (7.31 ± 7.23)	1.94–1600 (124 ± 387)	2.470–71.3 (13.1 ± 17.4)
BDE-197	1.02–12.7 (4.68 ± 3.01)	1.68–1220 (68.6 ± 247)	0.823–25.3 (7.34 ± 8.62)	1.82–75.7 (10.3 ± 16.9)	1.11–933 (73.04 ± 225)	0.783–189 (16.7 ± 43.6)
BDE-203	0.916–18.4 (6.83 ± 4.79)	2.13–598 (48.6 ± 123)	0.736–17.9 (6.73 ± 5.89)	2.55–193 (22.5 ± 42.3)	0.477–131 (18.3 ± 34.3)	0.953–394 (26.9 ± 91.8)
BDE-196	1.13–19.2 (7.43 ± 4.55)	2.30–1090 (74.4 ± 222)	0.723–12.0 (6.21 ± 4.57)	2.44–226 (22.3 ± 50.2)	0.901–396 (40.8 ± 96.8)	1.05–540 (36.8 ± 126)
BDE-208	2.61–39.5 (16.5 ± 11.0)	4.88–547 (82.1 ± 138)	3.28–40.5 (14.6 ± 12.7)	10.2–624 (63.6 ± 139)	3.20–482 (50.1 ± 113)	3.38–2380 (149 ± 556)
BDE-207	5.73–117 (35.0 ± 24.8)	10.7–950 (205 ± 294)	7.26–84.6 (34.0 ± 27.4)	20.0–1480 (166 ± 362)	10.1–1160 (181 ± 297)	11.670–7970 (500 ± 1860)
BDE-206	4.91–127 (44.8 ± 34.9)	8.97–2070 (301 ± 544)	6.35–182 (48.5 ± 53.3)	20.9–2540 (290 ± 632)	8.50–1740 (194 ± 408)	11.5–9730 (607 ± 2280)
BDE-209	90.1–1100 (304 ± 263)	105–5263 (1400 ± 1610)	49.5–606 (292 ± 170)	194–10,220 (1180 ± 2290)	37.1–13,400 (1525 ± 3207)	23.9–49,400 (3120 ± 11,600)
Σ_{14}PBDEs	114–1350 (440 ± 343)	147–8142 (2330 ± 2740)	131–1050 (453 ± 288)	291–15,300 (1980 ± 3600)	79.2–17,300 (2320 ± 4130)	73.6–70,600 (4790 ± 16,500)

[a]Expressed as range (mean ± SD)

levels in CMC and NH, particularly in CMC that use several electronic devices for diagnosing patients. PBDEs may be released from far-infrared light used in far-infrared therapy, electro-acupuncture devices, acupuncture medical beds, and dispending science traditional Chinese medicine (products of traditional Chinese medicine manufactured by a pharmaceutical factory certified with Good Manufacturing Practice (GMP) standards) in CMC. In NH, the levels and patterns of PBDEs in indoor dust are comparable to those in CR and A, indicating that the certain amounts of consumer products are existed in these three microenvironments. This is why indoor dust PBDEs in CMC is higher than that in NH, but their pattens are different.

Indoor dust PBDEs in CR, A, and NH in Taiwan are comparable to or slightly higher than those in offices (1700 ng g^{-1}, Σ_9PBDEs: BDE-47, 71, 99, 119, 183, 196, 206, 207, 209) and daycare centers (2100 ng g^{-1}, Σ_9PBDEs) in Korea (Lee et al. 2020). These levels are also similar to those in residences (2200 ng g^{-1}, Σ_8PBDEs: BDE-28, 47, 99, 110, 153, 154, 183, 209) in Beijing, China (Bu et al. 2019), and in public transport microenvironments, including hybrid buses (1200 ng g^{-1}, Σ_6PBDEs: BDE-28, 47, 99, 110, 153, 209), electric buses (1,160 ng g^{-1}. Σ_6PBDEs), and subways (926 ng g^{-1}, Σ_6PBDEs) in Hangzhou, China (Jin et al. 2022). In CMC of the present study, indoor dust PBDE levels are similar to those in offices (4500 ng g^{-1}, Σ_8PBDEs) in Beijing, China (Bu et al. 2019), and in residences (4350 ng g^{-1}, Σ_{12}PBDEs: BDE-17, 28, 47, 49, 66, 99, 110, 153, 154, 155, 183, 209) in Latvia (Pasecnaja et al. 2021). In agreement with recent global household-dust data from Korea, China, and Croatia, PBDE levels in indoor dust from NR and K were comparable (Guo et al. 2020; Lee et al. 2020; Klinčić et al. 2021; Li et al. 2022; Qi et al. 2023).

From Fig. 1a–c, the statistics of PCA with the methods of Varimax rotation and extraction based on eigenvalue > 2 are utilized to analyze PBDE congeners in indoor dust

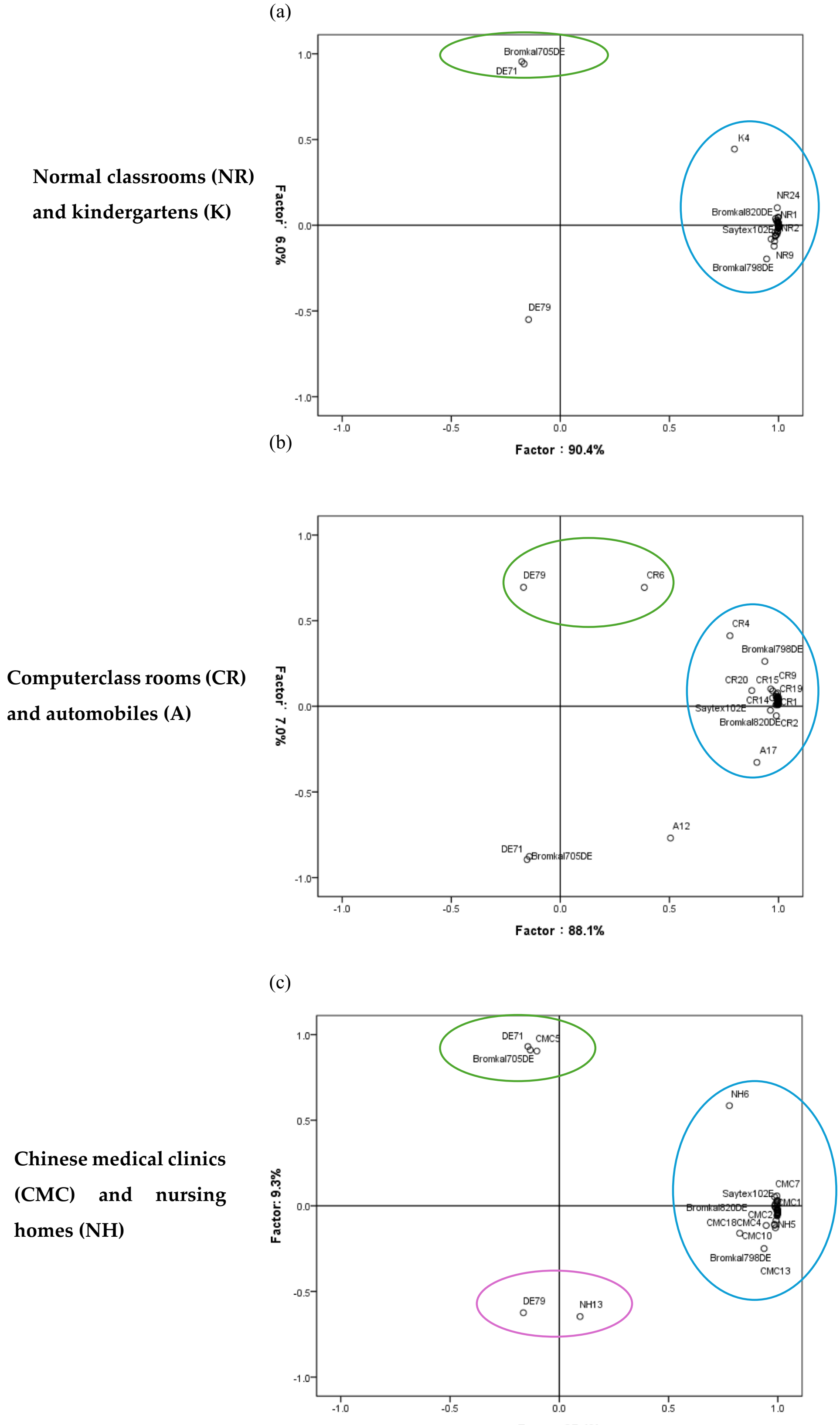


Fig. 1 The associations between PBDE homologues in indoor dust collected from the selected miroenviornments and commercial Penta-, Octa, and Deca-PBDE flame retardant mixtures, including DE-71 (Penta-BDE mixture), Bromkal 70-5DE ((Penta-BDE mixture), DE-79 (Octa-BDE mixture), Bromkal 79-8DE (Octa-BDE mixture containing 49.6% of BDE-209), Saytex 102E (Deca-BDE mixture) and Bromkal 82-0D (Deca-BDE mixture) formulation by principal component analysis (PCA). **a** Normal classrooms of the elementary schools ($n=24$) and kindergartens ($n=10$) were espressed as NR1 to NR24 and K1 to K10, respectively. **b** Computer classrooms of the elementary schools ($n=24$) and automobiles ($n=20$) were espressed as CR1 to CR24 and A1 to A20, respectively. **c** Chinese medical clinics (n = 18) and nursing homes ($n=17$) wee espressed as CMC1 to CMC18 and NH1 to NH17, respectively

samples collected from various Taiwanese microenvironments, including NR, K, CR, A, NH, and CMC and their commercial formulations (Penta-BDEs: DE-71 and Bromkal 70-5DE; Octa-BDEs: DE-79; Bromkal 79-8DE; Deca-BDEs: Saytex 102E, and Bromkal 82-0D). In Fig. 1a for NR ($n = 24$) and K ($n = 10$), indoor dust PBDE congeners and commercial Deca-BDE formulations, such as Bromkal 79-8DE, Saytex 102E, and Bromkal 82-0D, are grouped in Factor 1 to explain 90.4% of the data variances. The total variance explained (σ) by the PCA test in these microenvironments is 96.4%. Figure 1b shows that PBDE congeners in indoor dust samples from CR ($n = 24$) and A ($n = 20$) and Deca-BDE formulations are grouped together in Factor 1 to account for 88.1% of the variances ($\sigma = 96.4\%$). A computer classroom (CR6) is assigned with DE79 (Octa-BDE mixture), indicating that the different PBDE profile of CR6 compared with those collected from CR and A. In Fig. 1c, for indoor dust samples collected from NH and CMC, PBDE congeners in the microenvironments and Deca-BDE formulations are clustered, which is explained for 85.1% of the variances ($\sigma = 94.4\%$). The special cases like CMC5 and NH13, PBDE congeners in the indoor environment grouped with DE-71, Bromkal 79-8DE, and DE-79, respectively, indicating the distinct PBDE profiles of CMC5 and NH13 compared to the other samples collected from CMC and NH. Due to their persistence, toxicity, and bioaccumulation, commercial mixtures of Penta-BDEs and Octa-BDEs were phased out of production starting in 2004, while the Deca-BDE formulation was targeted by the Stockholm Convention, listed in Annex A since 2017, to gradually reduce global demand following a ban on its use (Blum et al. 2019). In accordance with the Stockholm Convention's guidelines to protect human health and the environment from POPs, the voluntary phase-out of Penta-BDEs and Octa-BDEs and the restricted use of Deca-BDE between 2009 and 2017 led to varied indoor dust PBDE levels in Taiwanese private and public microenvironments. Based on the current finding, PCA tests reveal that indoor dust PBDE congeners are mainly associated with Deca-BDE formulations, with a few samples showing distinct Penta- and Octa-BDE profiles reflecting historical usage patterns.

3.2 Levels of PBDD/Fs and their Corresponding WHO-TEQs in Various Microenvironments

Table 2 presents the dust concentrations of PBDD/Fs in indoor dust from NH, CMC, A, K, NR, and CR. The results indicate that the concentrations of Σ_{12} PBDD/Fs are $6{,}030 \pm 6{,}670$ pg g^{-1} ($n = 17$) in NH, $8{,}020 \pm 22{,}000$ pg g^{-1} in CMC ($n = 18$), $22{,}800 \pm 64{,}400$ pg g^{-1} ($n = 20$) in A, 2510 ± 1700 pg g^{-1} ($n = 10$) in K, while the indoor dust concentration is 4090 ± 3800 pg g^{-1} ($n = 24$) and $10{,}400 \pm 22{,}500$ pg g^{-1} ($n = 24$) in NR and CR, respectively. The WHO_{2005}-TEQ values for indoor dust are 39.0 ± 40.8 pg-WHO_{2005}-TEQ g^{-1} in NH, 37.8 ± 70.7 pg-WHO_{2005}-TEQ g^{-1} in CMC, 79.0 ± 136 pg-WHO_{2005}-TEQ g^{-1} in A, and 16.1 ± 10.4 pg-WHO_{2005}-TEQ g^{-1} in K, and 27.5 ± 25.6 and 58.8 ± 89.2 pg-WHO_{2005}-TEQ g^{-1} in NR and CR, respectively. Indoor dust WHO-TEQs in CR and A are higher than that in K probably due to high concentrations of 1,2,3,6,7,8-HpBDF (CR/A vs K: 32.2/51.0 vs 8.86 pg-WHO_{2005}-TEQ g^{-1}) and 1,2,3,4,7,8-HxBDD (CR/A vs K: 17.4/16.6 vs 4.72 pg-WHO_{2005}-TEQ g^{-1}) in CR and A.

In Table 2, PBDFs contribute more to PBDD/F contamination in the indoor environment than PBDDs, as indicated by their concentrations and WHO_{2005}-TEQ values. Compared to PBDD/F concentrations and WHO_{2005}-TEQ levels from our previous studies (levels vs. WHO_{2005}-TEQs: gyms: 7090 pg g^{-1} vs. 37.8 pg WHO_{2005}-TEQ g^{-1}; classrooms in elementary schools: 4720–7510 pg g^{-1} vs. 28.1–63.9 pg WHO_{2005}-TEQ g^{-1}) (Gou et al. 2016; Su et al. 2022), the indoor dust PBDD/F concentrations and WHO_{2005}-TEQ levels in the present study are of similar magnitudes. The distribution profiles of PBDD/Fs in NH significantly differ from those in CMC, A, NR, CR, and K (Fig. 2). The data indicate that the PBDD/F profile in CMC is similar to those in K and NR from the present study and CR from our previous report (Gou et al. 2016), while the profiles in NH resembled those found in Taiwanese gyms (Su et al. 2022). Indoor dust PBDD/F patterns both in NR in the present and previous (Gou et al. 2016) studies are very similar. The indoor dust PBDD/F patterns in the NR investigated in the present work are consistent with those previously reported in our earlier work (Gou et al. 2016). indoor dust PBDD/F patterns from the microenvironments in global datasets (Tue et al. 2010, 2013; Ortuño et al. 2015) differ from those found in Taiwan including the present study and the earlier reports (Gou et al. 2016; Su et al. 2022). In contrast, the PBDD/F profiles in the Taiwanese indoor environment, including those from the present study, are quite similar to each other when compared with global data from other countries (Gou et al. 2016; Su et al. 2022).

3.3 Correlations of PBDEs and PBDD/Fs in the Selected Taiwanese Microenvironments

Correlations between PBDEs and PBDD/Fs in indoor dust collected from the selected Taiwanese microenvironments, including NR ($n = 24$), CR ($n = 24$), K ($n = 10$), A ($n = 20$), CMC ($n = 18$), and NH ($n = 17$), are tested by Spearman's *rho* correlation coefficients in Table 3. In indoor dust samples from these microenvironments, PBDFs, such as 2,3,7,8-TeBDF, 1,2,3,7,8-PeBDF, 2,3,4,7,8-PeBDF, 1,2,3,4,7,8-HxBDF, 1,2,3,4,6,7,8-HpBDF, and OctBDF, are significantly correlated with PBDEs from tri to deca ($r = 0.432$–0.726, $p < 0.001$- $p = 0.06$). Higher levels of

Table 2 Indoor dust concentrations of PBDD/Fs in normal classroom (NR), computer classroom (CR), kindergarten classroom (K), automobiles (A), nursing homes (NH), and Chinese medical clinics (CMC) (pg g^{-1})

	Elementary school		Kindergarten ($n=10$)	Automobiles ($n=20$)	Nursing homes ($n=17$)	Chinese medical clinics ($n=18$)
	NR ($n=24$)	CR ($n=24$)				
PBDFs						
2,3,7,8-TeBDF	<LOD – 19.1 (2.07 ± 4.05)[a]	<LOD – 13.6 (3.51 ± 3.45)	<LOD – 2.50 (0.398 ± 0.873)	<LOD – 12.4 (2.08 ± 3.09)	<LOD – 5.68 (1.75 ± 2.13)	<LOD – 9.91 (2.87 ± 3.50)
1,2,3,7,8-PeBDF	<LOD – 38.9 (5.89 ± 8.64)	<LOD – 41.4 (10 ± 10.3)	<LOD – 8.61 (3 ± 2.65)	<LOD – 21 (5.09 ± 5.22)	<LOD – 31.9 (4.63 ± 7.88)	<LOD – 37.5 (5.11 ± 9.12)
2,3,4,7,8-PeBDF	1.72–37.6 (9.47 ± 8.71)	3.37–81.3 (20.1 ± 19)	1.57–16.1 (6.20 ± 4.18)	3.34–71 (17.9 ± 17.4)	<LOD – 66.4 (13.6 ± 14.7)	<LOD – 43.1 (11.2 ± 10.9)
1,2,3,4,7,8-HxBDF	9.94–439 (89.8 ± 102)	19.5–1140 (174 ± 231)	19.9–127 (47.2 ± 32.9)	21.6–824 (166 ± 187)	26.2–470 (109 ± 99)	25.1–650 (97.0 ± 145)
1,2,3,4,6,7,8-HpBDF	289–5990 (1440 ± 1230)	480–29100 (3220 ± 5660)	204–2090 (886 ± 564)	537–45,500 (5100 ± 9820)	412–11,700 (2250 ± 2580)	368–21,200 (2230 ± 4800)
OctBDF	379–12,200 (2420 ± 2480)	682–83,600 (6600 ± 16,500)	269–3040 (1300 ± 821)	802–247000 (17,400 ± 54,500)	567–12,700 (3610 ± 3130)	<LOD – 61,900 (4980 ± 14,300)
PBDDs						
2,3,7,8,-TeBDD	<LOD – 297 (0.124 ± 0.0607)	<LOD – 0.426 (0.0177 ± 0.0869)	<LOD	<LOD – 1.19 (0.0593 ± 0.265)	<LOD	<LOD
1,2,3,7,8-PeBDD	<LOD – 0.73 (0.0304 ± 0.149)	<LOD – 2.65 (0.228 ± 0.588)	<LOD	<LOD – 4.97 (0.28 ± 1.11)	<LOD	<LOD – 2.99 (0.166 ± 0.704)
1,2,3,7,8,9-HxBDD	<LOD	<LOD – 5.50 (0.343 ± 1.23)	<LOD	<LOD – 6.67 (0.334 ± 1.49)	<LOD	<LOD
1,2,3,4/6,7,8-HxBDD	<LOD	<LOD – 3.55 (0.262 ± 0.774)	<LOD	<LOD – 3.6 (0.212 ± 0.810)	<LOD	<LOD
1,2,3,4,6,7,8-HpBDD	<LOD – 50.8 (9.84 ± 12.4)	<LOD – 77.8 (17.2 ± 19.6)	2.30–5.67 (10.3 ± 16.5)	<LOD – 40 (7.73 ± 10.2)	<LOD – 20.0 (6.63 ± 6.45)	<LOD – 306 (19.7 ± 71.5)
OctBDD	<LOD – 733 (116 ± 156)	27.6–1800 (319 ± 418)	16.1–1810 (257 ± 553)	<LOD – 663 (92.6 ± 157)	<LOD – 153 (33.1 ± 47.4)	<LOD – 11,700 (675 ± 2740)
ΣPBDDs	<LOD – 778 (126 ± 166)	34.4–1860 (337 ± 431)	18.4–1870 (267 ± 570)	<LOD – 680 (101 ± 165)	<LOD – 160 (39.7 ± 53.9)	<LOD – 12,000 (695 ± 2810)
ΣPBDFs	697–18,300 (3970 ± 3740)	1210–114000 (10,000 ± 22,400)	495–5290 (2240 ± 1420)	1660–293000 (22,700 ± 64,400)	1010–25000 (5990 ± 5830)	730–83,800 (7320 ± 19,300)
ΣPBDD/Fs	0742–18500 (4090 ± 3800)	1300–114000 (10,400 ± 22,500)	513–5490 (2510 ± 1700)	1680–293000 (22,800 ± 64,400)	1010–25000 (6030 ± 6670)	730 -95,700 (8020 ± 22,000)
ΣPBDD/F-WHO_{2005}TEQs (pg WHO_{2005}-TEQ g^{-1})	5.73–84.3 (27.5 ± 25.6)	9.91–456 (58.8 ± 89.2)	4.68–39.9 (16.1 ± 10.4)	9.17–634 (79.0 ± 136)	7.89–189 (39.0 ± 40.8)	6.49–315 (37.8 ± 70.7)

[a]Expressed as range (mean ± SD)

1,2,3,4,6,7,8-HpBDF (r = 0.427–0.690, $p < 0.001$) and OctBDF (r = 0.427–0.690, $p < 0.001$) have strongly significant relations with individual PBDE congeners and Σ_{14}PBDEs in indoor dust compared with other PBDF congeners. For PBDDs, there are only, 1,2,3,4,6,7,8-HxBDD and OctBDD to be associated with individual PBDE congeners and Σ_{14}PBDEs (r = 0.239–0.690, $p = 0.08$-$p < 0.001$) in indoor dust samples. Two constitutional isomers of HxBDDs, such as 1,2,3,7,8,9-HxBDD and 1,2,3,4,7,8-HxBDD, had the significantly slight correlations with PBDEs from hepta to deca (r = 0.239–0.690, $p = 0.08$-$p < 0.001$) in indoor dust samples. The most toxic brominated compounds (TEF = 1.00) of 1,2,3,7,8-PeBDD and 2,3,7,8-TBDD are not significantly linked to PBDEs from octa to deca ($p < 0.005$) in indoor dust. Indoor dust 1,2,3,7,8-PeBDD is significantly correlated with BDE-28 (r = 0.283, $p = 0.002$), BDE-47 (r = 0.194, $p = 0.033$), BDE-153 (r = 0.252, $p = 0.005$), BDE-154 (r = 0.229, $p = 0.011$), BDE-183 (r = 0.277, $p = 0.002$), and Σ_{14}PBDEs (r = 0.239, $p = 0.08$). We only found that 2,3,7,8-TBDD has a significantly slight association with BDE-28 (r = 0.180, $p = 0.048$).

Table 3 shows that certain PBDEs are significantly correlated with certain PBDD/Fs mainly due to the similar sources. In Table 2, indoor dust levels of PBDD/Fs and PBDD/F-WHO_{2005}-TEQ are shown to be of similar magnitudes between NH and CMC, while these two values in

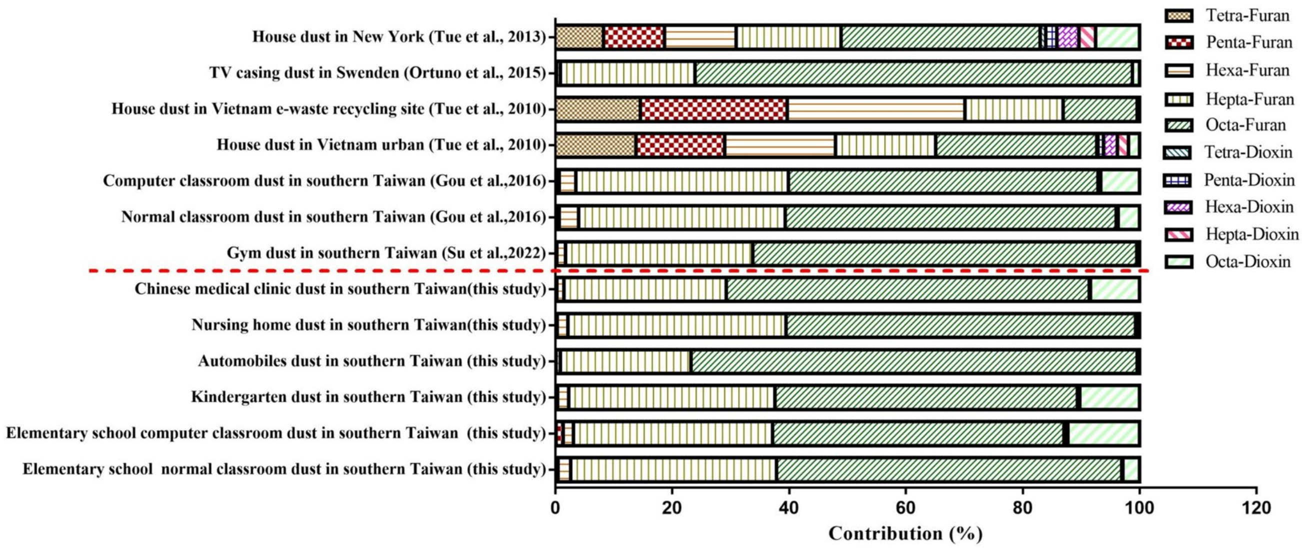


Fig. 2 The composition of ΣPBDD/Fs in the indoor dust from the present study and the currently global data

A and CR are obviously higher than that in CMC. On the contrary, although indoor PBDE levels do not differ among CR, A, NH, and CMC (Table 1), the magnitude in CMC is approximately two-folded higher than those in NH, A, and CR. Based on this finding, indoor dust PBDE levels are not strongly linked into the corresponding values of PBDD/Fs and PBDD/F-WHO$_{2005}$-TEQs in indoor dust among our selected microenvironments ($n = 113$). The frequent cleaning with the vacuum cleaner leads to be associated with lowering debromination of PBDEs to convert to PBDD/Fs and debromination of PBDEs needs time and several environmental factors can accelerate transformation of PBDEs to PBDD/Fs (i.e., UV light or thermal pyrolysis) (Ortuño et al. 2015; Wang et al. 2018). In the present report based on Table 1 to Table 3, Spearman's correlation analysis showed that certain PBDE congeners were significantly associated with specific PBDD/Fs in indoor dust, although the overall linkage between PBDE and PBDD/F levels across microenvironments was weak. The findings of the present study indicate that the persistence of PBDE contamination is most likely associated with the presence of old furniture, electronics, and other consumer products in the investigated microenvironments. In addition, the debromination of PBDEs appears to be markedly enhanced under conditions of sunlight exposure and elevated temperatures, such as those commonly found inside automobiles, thereby contributing to the continuous formation of PBDD/Fs. Finally, it was included that the predominance of PBDFs over PBDDs was recommended that indoor PBDD/Fs were primarily derived from PBDE-containing the surface of consumer products, particularly highly brominated PBDEs (i.e., BDE-209), as major sources, while minor contributions were raised from thermal and photolytic transformation pathways. Owing to the structural similarity between PBDEs and PBDD/Fs, higher-brominated congeners, especially BDE-209, might undergo debromination under elevated temperature and UV exposure, leading to the in-situ formation of PBDFs in indoor environments.

3.4 Risk Assessments of Taiwanese Exposed to PBDEs and PBDD/Fs in the Selected Microenvironments

Table 4 shows the estimated risks for the adults and children exposure to PBDEs and PBDD/Fs through ingestion of indoor dust in the selected microenvironments in Taiwan. House dust PBDE levels used in the present study are referenced from our previous report (Chao et al. 2014b). The risks of 7–9 aged school and 4–6 aged preschool children exposure to PBDD/Fs are not assessed mainly due to lack of house dust PBDD/F data in Taiwan. For the adults, the mean estimated non-dietary daily intakes of PBDEs (7.91×10^{-7} and 3.83×10^{-7} mg kg b.w.$^{-1}$ day^{-1}) and PBDD/Fs (3.78×10^{-12} and 3.90×10^{-12} mg WHO$_{2005}$-TEQ kg b.w.$^{-1}$ day^{-1}) for the staff are assessed by indoor dust levels of CMC and NH, respectively. In this study, it is assumed that the patients spend 0.0149 h day^{-1} (2.5 h week^{-1}) and 24 h day^{-1} in CMC and NH, respectively. The mean estimated non-dietary daily intakes of PBDEs (3.53×10^{-8} (CMC) and 1.15×10^{-6} (NH) mg kg b.w.$^{-1}$ day^{-1}) and PBDD/Fs (1.34×10^{-13} and 1.17×10^{-11} mg WHO$_{2005}$-TEQ kg b.w.$^{-1}$ day^{-1}) for the CMC and NH patients are estimated. In the present study, the estimated daily intakes of PBDEs for school-age and preschool-age children are 2.34×10^{-7} (NR) and 5.03×10^{-7} (K), 2.11×10^{-7} (A) and 2.76×10^{-7} (A), and 8.23×10^{-7} (H) and 1.08×10^{-6} (H)

Table 3 The Spearman's rho correlation coefficients between PBDD/Fs and PBDEs in the Chinese medical clinics (n = 18), Nursing homes (indoor dust $n = 17$, air conditioner dust $n = 8$), Normal classroom ($n = 24$), Computer classroom ($n = 24$), kindergarten (n = 10) and vehicle ($n = 20$)

	BDE28	BDE47	BDE100	BDE99	BDE154	BDE153	BDE183	BDE197	BDE203	BDE196	BDE208	BDE207	BDE206	BDE209	Σ_{14} PBDEs
2,3,7,8-TeBDF	0.401^{***,a}	0.315***	0.323***	0.324***	0.497***	0.501***	0.460***	0.370***	0.339***	0.362***	0.249**	0.333***	0.279**	0.292**	0.351***
	<0.001[b]	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.006	<0.001	0.002	0.001	<0.001
1,2,3,7,8-PeBDF	0.477***	0.370***	0.363***	0.366***	0.515***	0.523***	0.508***	0.421***	0.427***	0.424***	0.331***	0.376***	0.326***	0.337***	0.380***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2,3,4,7,8-PeBDF	0.567***	0.525***	0.452***	0.470***	0.604***	0.616***	0.611***	0.572***	0.568***	0.538***	0.569***	0.617***	0.569***	0.619***	0.647***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
1,2,3,4,7,8-HxBDF	0.556***	0.485***	0.410***	0.431***	0.623***	0.605***	0.624***	0.628***	0.620***	0.588***	0.527***	0.585***	0.527***	0.565***	0.572***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
1,2,3,4,6,7,8-HpBDF	0.514***	0.484***	0.432***	0.450***	0.533***	0.518***	0.572***	0.672***	0.726***	0.676***	0.696***	0.704***	0.693***	0.679***	0.678***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
OctBDF	0.507***	0.478***	0.427***	0.438***	0.518***	0.505***	0.556***	0.649***	0.690***	0.649***	0.681***	0.681***	0.686***	0.645***	0.645***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
2,3,7,8,-TeBDD	0.180*	0.040	0.014	0.008	0.001	0.003	0.054	-0.006	0.029	0.021	0.037	0.037	0.023	0.082	0.057
	0.048	0.664	0.879	0.932	0.995	0.970	0.560	0.945	0.752	0.820	0.686	0.689	0.799	0.369	0.538
1,2,3,7,8-PeBDD	0.283**	0.194*	0.150	0.151	0.229*	0.252**	0.277**	0.128	0.145	0.134	0.069	0.130	0.116	0.169	0.239**
	0.002	0.033	0.100	0.098	0.011	0.005	0.002	0.163	0.112	0.144	0.452	0.154	0.206	0.064	0.008
1,2,3,4,7,8-HxBDD	0.170	0.101	0.040	0.050	0.140	0.147	0.195*	0.208*	0.234**	0.208*	0.199*	0.225*	0.215*	0.220*	0.230*
	0.062	0.268	0.665	0.585	0.127	0.109	0.032	0.022	0.010	0.022	0.028	0.013	0.018	0.015	0.011
1,2,3,7,8,9-HxBDD	0.182*	0.187*	0.147	0.164	0.232*	0.251**	0.289**	0.261**	0.289**	0.242**	0.242**	0.295**	0.304**	0.295**	0.306**
	0.046	0.040	0.107	0.072	0.010	0.005	0.001	0.004	0.001	0.008	0.008	0.001	0.001	0.001	0.001
1,2,3,4,6,7,8-HpBDD	0.277**	0.249**	0.239**	0.243**	0.319***	0.376***	0.468***	0.484***	0.487***	0.499***	0.417***	0.432***	0.403***	0.440***	0.469***
	0.002	0.006	0.008	0.007	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
OctBDD	0.507***	0.478***	0.427***	0.438***	0.518***	0.505***	0.556***	0.649***	0.690***	0.649***	0.681***	0.681***	0.686***	0.645***	0.645***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Σ_{12} PBDD/Fs	0.520***	0.494***	0.444***	0.457***	0.543***	0.528***	0.582***	0.679***	0.724***	0.677***	0.696***	0.701***	0.697***	0.669***	0.673***
	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001

[a]The Spearman's rho correlation coefficient (r) = 0.401

[b] $^{*}p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$

Table 4 Risk assessment of indoor PBDEs and PBDD/Fs for the adults and children in various microenvironments via ingestion of indoor dust

Assessment of indoor dust	The adults				The young children					
	Chinese medical clinics		Nursing homes		K[a]	NR[b]	A[c]		Floor dust (House)[d]	
	Patient	Staff	Patient	Staff	Children (4–6 yr)	Children (7–9 yr)	Children (4–6 yr)	Children (7–9 yr)	Children (4–6 yr)	Children (7–9 yr)
Daily intake (mg kg b.w.$^{-1}$ day^{-1})										
BDE-47	2.35×10^{-10}	5.26×10^{-9}	4.87×10^{-9}	1.62×10^{-9}	3.36×10^{-9}	1.23×10^{-9}	3.83×10^{-9}	2.93×10^{-9}	5.28×10^{-9}	4.05×10^{-9}
BDE-99	4.35×10^{-10}	9.75×10^{-9}	8.45×10^{-9}	2.82×10^{-9}	9.02×10^{-9}	1.60×10^{-9}	$6.96\times10–^{9}$	5.33×10^{-9}	8.42×10^{-9}	6.45×10^{-9}
BDE-153	1.97×10^{-10}	4.42×10^{-9}	9.24×10^{-9}	3.08×10^{-9}	2.84×10^{-9}	5.61×10^{-10}	2.05×10^{-9}	1.57×10^{-9}	2.25×10^{-9}	1.73×10^{-9}
BDE-209	3.08×10^{-9}	6.89×10^{-8}	1.01×10^{-7}	3.36×10^{-8}	4.51×10^{-8}	2.24×10^{-8}	2.29×10^{-8}	1.75×10^{-8}	9.87×10^{-8}	7.56×10^{-8}
PBDEs	3.53×10^{-8}	7.91×10^{-7}	1.15×10^{-6}	3.83×10^{-7}	5.03×10^{-7}	2.34×10^{-7}	2.76×10^{-7}	2.11×10^{-7}	1.08×10^{-6}	8.23×10^{-7}
PBDD/Fs-WHO_{2005}-TEQ (mg WHO_{2005}-TEQ kg b.w.$^{-1}$ day^{-1})	1.34×10^{-13}	3.78×10^{-12}	1.17×10^{-11}	3.90×10^{-12}	1.12×10^{-11}	9.20×10^{-12}	6.91×10^{-12}	5.29×10^{-12}	–	–
Non-cancer risk (HQs)										
BDE-47	1.29×10^{-6}	1.58×10^{-5}	2.67×10^{-5}	4.88×10^{-6}	6.93×10^{-7}	2.55×10^{-7}	7.90×10^{-7}	6.05×10^{-7}	1.99×10^{-6}	8.36×10^{-7}
BDE-99	2.39×10^{-6}	2.93×10^{-5}	4.64×10^{-5}	8.47×10^{-6}	1.86×10^{-6}	3.30×10^{-7}	1.44×10^{-6}	1.10×10^{-6}	3.18×10^{-6}	1.33×10^{-6}
BDE-153	5.41×10^{-7}	6.64×10^{-6}	2.54×10^{-5}	4.63×10^{-6}	2.93×10^{-7}	5.79×10^{-8}	2.11×10^{-7}	1.62×10^{-7}	4.25×10^{-7}	1.78×10^{-7}
BDE-209	2.41×10^{-7}	2.96×10^{-6}	7.91×10^{-6}	1.44×10^{-6}	1.33×10^{-7}	6.62×10^{-8}	6.75×10^{-8}	5.17×10^{-8}	5.31×10^{-7}	2.23×10^{-7}
PBDD/Fs-WHO_{2005}-TEQ	1.05×10^{-4}	1.62×10^{-3}	9.17×10^{-3}	1.67×10^{-3}	3.32×10^{-4}	2.71×10^{-4}	2.04×10^{-4}	1.56×10^{-4}	–	–
Cancer risk (Rs)										
BDE-209	1.18×10^{-12}	1.45×10^{-11}	3.88×10^{-11}	7.08×10^{-12}	6.51×10^{-13}	3.24×10^{-13}	3.31×10^{-13}	2.53×10^{-13}	2.60×10^{-12}	1.09×10^{-12}
PBDD/Fs-WHO_{2005}-TEQ	7.34×10^{-9}	1.14×10^{-7}	6.42×10^{-7}	1.17×10^{-7}	2.33×10^{-8}	1.90×10^{-8}	1.42×10^{-8}	1.09×10^{-8}	–	–

[a]Classrooms of the kindergartens

[b]Normal classrooms of the elementary schools

[c]Automobiles

[d]House dust of PBDE concentrations (median level) were used based on our earlier study (Chao et al. 2014a, b)

mg kg b.w.$^{-1}$ day^{-1}, respectively. For PBDD/Fs, the daily intakes for school-age children are 9.20×10^{-12} (NR) and 5.29×10^{-12} (A) mg WHO$_{2005}$-TEQ kg b.w.$^{-1}$ day^{-1}; while those for preschool-age children are 1.12×10^{-12} (K) and 6.91×10^{-12} (A) mg WHO$_{2005}$-TEQ kg b.w.$^{-1}$ day^{-1}. All estimated daily intakes of BDE-47, BDE-99, BDE-209, and 2,3,7,8-TBDD in our selected populations were considerably lower than the oral reference doses (RfDs) provided by the U.S. EPA (2012), such as BDE-47: 0.01 mg kg b.w.$^{-1}$ day^{-1}; BDE-99: 0.001 mg kg b.w.$^{-1}$ day^{-1}; BDE-209: 0.007 mg kg b.w.$^{-1}$ day^{-1}; 2,3,7,8-TBDD: 1×10^{-8} mg kg b.w.$^{-1}$ day^{-1}, as detailed in the supplementary materials.

Non-cancer risks (HQs) of PBDEs (BDE-47, 99, 153, and 209) and PBDD/Fs from non-dietary exposure for the adults and children are also assessed in Table 4. All HQ values of certain PBDEs (BDE-47, 99, 153, and 209) and PBDD/F-WHO$_{2005}$-TEQs for the adults (2.41×10^{-7}–4.64×10^{-5}) and children (5.17×10^{-8}–3.18×10^{-6}) through ingestion of indoor dust in the present study are extremely lower than the critical value of 1.00 (U.S. EPA 2012). Cancer risks (Rs) of BDE-209 and PBDD/F-WHO$_{2005}$-TEQs for the adults (1.18×10^{-12}–6.42×10^{-7}) and children (2.53×10^{-13}–2.33×10^{-8}) are also notably lower than the critical value of 1.00×10^{-6} (U.S. EPA 2012). Levels of HQs and Rs in the present study for the adults and children are still under the safe criteria. To our knowledge, this is the first study to investigate indoor dust PBDEs and PBDD/Fs in CMC and NH and to further evaluate the associated health risks for patients. Even among elderly individuals with disabilities requiring continuous, 24-h care in NH, both non-cancer and cancer risks remain low. Compared with the previous reports (Gou et al. 2016; Su et al. 2022; Choi et al. 2023; Yu et al. 2023), the risk levels observed in the selected populations are comparable and remain within safe criteria. A recent Nigerian study (Ibeto et al. 2021) demonstrated that ingestion of house dust constitutes the primary pathway of PBDE exposure for toddlers and young children, while dermal absorption plays a comparatively greater role in adults. This study further indicates that the average daily intakes of PBDEs are substantially higher in toddlers and children than in adults (Ibeto et al. 2021). Although estimated cancer and non-cancer risks were within acceptable levels, the ubiquitous presence of PBDEs and PBDD/Fs raises public health concern and supports continued dust management and routine monitoring. The estimated daily intakes of PBDEs and PBDD/Fs for preschool children, schoolchildren, and the elderly in nursing homes were higher than those of adults, reflecting greater dust ingestion rates and exposure susceptibility; however, these intakes remained several orders of magnitude below reference doses, corresponding to negligible non-cancer ($HQ < < 1$) and cancer risks ($R < < 10^{-6}$). Despite relatively elevated exposures in these sensitive groups, particularly in computer classrooms and nursing homes, the associated health risks remained within acceptable limits, indicating low immediate concern but warranting continued monitoring due to potential long-term exposure and accumulation. These observations highlight the age-dependent differences in exposure routes and suggest that future research should focus on elucidating the mechanisms and extent of dermal absorption of brominated POPs, as well as evaluating their potential health risks in diverse populations.

4 Conclusions

This study is the first to investigate indoor dust levels of PBDEs and PBDD/Fs in NH and CMC. The findings reveal relatively high levels of PBDEs in the dust of CMC compared to other microenvironments (NR, CR, K, A, and NH), to make our amazements, elevated levels of PBDD/Fs and PBDD/F-WHO$_{2005}$-TEQs in A are found compared with those in NR, CR, K, NH, and CMC. PBDEs contamination in the selected microenvironments likely originates from consumer products, while sunlight and heat (e.g., inside automobiles) accelerate their debromination, leading to continuous formation of PBDD/Fs. Notably, the distribution pattern of PBDEs in indoor dust from these facilities aligns closely with that of Deca-BDE commercial formulations. Our findings demonstrate, based on Monte Carlo simulation, that both non-cancer and cancer risks associated with indoor dust PBDEs and PBDD/Fs in patients and staff at CMC and NH, as well as in schoolchildren and preschool children, remain within acceptable safety criteria.

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Data Availability The data used/analyzed during the present study are available from the corresponding author upon reasonable request.

Declarations

Conflict of Interest The authors declare no competing interests.

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