



Polybrominated diphenyl ethers in car dust in Nigeria: Concentrations and implications for non-dietary human exposure



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ARTICLE INFO

Article history:

Received 12 March 2015

Received in revised form 28 May 2015

Accepted 28 May 2015

Available online 3 June 2015

Keywords:

PBDE

Concentrations

Car dust

Non-dietary exposure estimates

Nigeria

ABSTRACT

In the present study, the levels of polybrominated diphenyl ethers in car dust samples collected from 12 randomly selected cars in Nigeria were measured, and the daily intake via dust ingestion were estimated for four categories of individuals. PBDEs were extracted and analyzed using ultra-sonication extraction and gas chromatography electron impact ionization mass spectrometry (GC-EIMS), respectively. The concentrations of \sum_7 PBDEs ranged from 159 to 736 ng g⁻¹ dry weight. BDE-209 was found to be the most prevalent congener with a total of 2130 ng g⁻¹, contributing up to 47% of the total congener values (4554 ng g⁻¹) and the other congeners contributed 53%. The mean and median concentrations of detected congeners ranged from 65 to 137 and 14 to 127 ng g⁻¹ dry weight, respectively. BDE-47 was detected in all the dust samples at a detection frequency of 100%, and this was followed by BDE-209 with 92%. The daily intake (DI) was estimated using concentrations corresponding to 5th percentile, median, and 95th percentile. The estimated DI was highest for professional drivers at 0.12 ng kg⁻¹ bw day⁻¹ and this was followed by toddlers (0–3 years) at 0.09 ng kg⁻¹ bw day⁻¹ while DI estimate of 0.01 ng kg⁻¹ bw day⁻¹ was determined for children (4–18 years) and adults (>19 years). PBDEs daily intake rates from car dust estimated in the present study are way below the USEPA reference dose (RfD) for oral exposure to PBDEs 47, 99, 153, and 209 of 100, 100, 200, and 7000 ng kg bw day⁻¹.

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1. Introduction

Polybrominated diphenyl ethers are a group of organic compounds known as brominated flame retardants. Of all the brominated flame retardants, polybrominated diphenyl ethers have been used extensively to inhibit flame ignition and reduce propagation of flames, which has led to the low rate of deaths and loss of valuable properties through fire outbreaks in homes and offices [1]. PBDEs are additive flame retardants used in a wide variety of consumer and industrial products such as polyurethane foams, textiles, and various plastic products. The three major commercial formulations of PBDEs produced globally in the last two decades include Penta-BDE, Octa-BDE, and Deca-BDE [2,3]. Penta mixture has been mainly used in printed circuit boards, cable sheets, furniture, and textiles, while Octa mixture has been used in television and computer housings, fax machines, household appliances, and other small electronics [4,5]. Deca mixture, on the other hand, is applied in high-impact polystyrene (HIPS) which is used in the television industry for cabinet backs. Deca-BDE is also used for polypropylene drapery and upholstery fabric [4,5]. Other applications of Deca-BDE are in the building sector. The Penta and Octa-BDE derivatives

were banned by the European Union (EU) in August 2004 [6] since their identification as endocrine disruptors, persistent and bio-accumulative substance. Consequently, they have been listed as new persistent organic pollutants in the Stockholm Convention [7]. However, Deca-BDE is still being used with partial restrictions in some countries. Since PBDEs are used as additives in polymer products, and not covalently bonded to the matrix, they have the capacity to leach out of the products into the environment [8].

The following exposure pathways have been identified as the main routes via which PBDEs can enter the human body: diet [9–14], dust [15–18], and air [19–24]. Of all the pathways, dust appears to be the most common. According to Butte and Heinzow [25], dust is a complex heterogeneous mixture of semi-volatile organic compounds and particle-bound matter derived from biological materials (skin cells, plant pollen, human and animal hairs, textile fibers, fungal spores, and others), which could be found in diverse human environments such as houses, offices, and cars. Therefore, dust presents an ideal matrix for the study of such contaminants due to its omnipresence in the environment, intrinsic characteristics, particularly its contribution to the total human exposure through ingestion, inhalation, and dermal absorption [26].

There is currently scarcity of information on the levels of PBDEs in cars and the indoor environment in most developing countries, particularly in the continent of Africa. The observed gap of information on

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PBDEs can be attributed to (1) lack of awareness on the health implications of PBDEs, (2) inadequate funding for research on emerging contaminants, and (3) low attention given to environmental issues. Considering the fact that Nigeria is the largest growing economy within the African continent and it imports a wide range of finished products, which may be flame retarded with PBDEs from various parts of the world, PBDEs are expected to be present within the Nigerian environment. Therefore, the main objective of this work was to determine the levels of common PBDE congeners in car dust in selected parts of Nigeria and compare the levels obtained in this study to similar studies reported in other regions and subsequently assess the implications for non-dietary human exposure.

2. Materials and methods

2.1. Materials

This section describes the materials, apparatus, and methods developed for the determination of selected polybrominated diphenyl ethers in car dust samples collected in Benue State. About 1.2 mL ($50 \mu\text{g mL}^{-1}$) of certified standard solutions of 12 PBDE (BDE-47, -99, -100, -118, -153, -154, -128, -183, -209 and ^{13}C -BDE-139, -77, and -209) congeners were purchased from Wellington Laboratories (Guelph, Ontario, Canada). Copper powder (purity 99.98%, from Saarchem (Pty) Ltd., Muldersdrift, South Africa), silicagel (100–200 mesh), sodium sulphate (purity 99.9%), glass wool and HPLC grade acetone, hexane, dichloromethane, methanol, isooctane, and toluene (products of Sigma Aldrich) (Chemie GmbH, Steinheim, Germany), and Pesticarb from Separations (Pty) Ltd., South Africa, were used in the experimental aspect of the study.

All solvents were of analytical grade, therefore, were used without further purification. Other materials such as silica gel, sodium sulphate, and copper powder were activated before use at the recommended conditions. Copper powder was activated by soaking in 6 M HCl for 3 min and then rinsed thoroughly with ultrapure water followed by methanol and then kept under toluene until use. Preparation of standards and serial dilution of working standards were carried out in the fume hood.

2.2. Sampling

Twelve dust samples were collected between April and June 2014 from selected cars of different makes from Gboko, Otukpo, and MaKurdi in Benue state, Nigeria. Samples were collected using a bag-less car hand-held vacuum cleaner (Hoover, HSV40). Details of the make and characterization of the interior of the cars were recorded. The seat surfaces of the cars were vacuumed as described by Jones-Otazo et al. [27] and Abdallah et al. [28]. The vacuum cleaner was thoroughly cleaned with acetone between each sample collection in order to avoid cross contamination. Samples were transferred from the vacuum cleaner into solvent rinsed foil, wrapped and taken to the laboratory where they were filtered and stored in amber glass bottles at -20°C prior to extraction.

2.3. Extraction of PBDEs

Extraction efficiencies of *n*-hexane, toluene, and dichloromethane were tested individually and in some cases combined by extracting 10 g of activated sodium sulphate spiked with $50 \mu\text{L}$ of $2.1 \text{ ng } \mu\text{L}^{-1}$ mixed standards. A solvent combination of toluene/dichloromethane (1:1, v/v) gave the best recovery for most of the target analytes. From each of the dried and finely sieved ($150 \mu\text{m}$) dust samples, a mass of 100 mg was weighed and extracted using ultra-sonication. Prior to extraction, dust samples were spiked with $3 \mu\text{L}$ of surrogate BDE-139 and BDE-209 standards to monitor recoveries. Briefly, 100 mg dust was transferred into a 4 ml vial, and then spiked with surrogate standards and left overnight to equilibrate. Thereafter, samples were

extracted with 3 mL solvent for 15 min at 55°C using an ultra-sonic bath (Elmasonic S 40H, Germany) with a maximum power of 340 watts. The extracts were centrifuged at 300 rpm for 3 min. Three times extracts were then transferred into clean tubes and reduced under a gentle flow of N_2 to about 1 mL before subjecting it to cleaning by column chromatography to remove unwanted co-eluent.

2.4. Cleanup of crude extracts

The cleanup column used was a Pasteur pipette (230 mm) plugged with glass wool at the bottom and thereafter packed in layers from the bottom with about 0.16 g of silica, 0.06 g Pesticarb, 0.16 g silica, and finally topped with 0.5 g sodium sulphate. Glass wool was used to separate each layer of materials to enhance the cleaning. Before introduction of 1.0 mL reduced extract, the packed column was eluted to saturation with 12 mL toluene/dichloromethane (1:1, v/v). Crude samples were introduced into the column before the solvent reached the bed of the sodium sulphate plugged with glass wool and was further eluted with 4 mL combined solvent. Thereafter, nitrogen gas was bubbled into the eluent to reduce the sample volume to $100 \mu\text{L}$. Ten microliter of $2.5 \text{ ng } \mu\text{L}^{-1}$ internal standard (BDE-118) was added to monitor change in instrument analytical condition. Thereafter, $1.0 \mu\text{L}$ of the extract was injected into the GC-MS under the optimized instrumental conditions.

The purified extracts were analyzed by Shimadzu model 2010 plus gas chromatograph coupled with a model QP 2010 ultra-mass spectrometer (Shimadzu, Japan) using electron ionization and injected automatically by a Shimadzu A0C-20i auto sampler. Operation mode was in the selected ion-monitoring (SIM). A 15 m column; DB 5 (0.25 mm ID, $0.1 \mu\text{m } d_f$) was used for separation. The oven temperature program was set at 90°C (1 min), $30^\circ\text{C}/\text{min}$ to 300°C (5 min) and $10^\circ\text{C}/\text{min}$ to 310°C (10 min). Carrier gas used was helium (purity 99.999%) and set at a constant flow of 1.5 mL min^{-1} . The injector, transfer line, and ion source temperatures were set at 290, 300, and 250°C , respectively.

2.5. Quality assurance/quality control

Glass wool, silica gel, and sodium sulphate were baked at 450°C for 12 h to remove impurities. Silica gel and sodium sulphate were stored in glass jar which was pre-cleaned and rinsed with solvent (*n*-hexane/acetone), then sealed to prevent absorption of moisture and contamination. The samples were collected and stored in amber sample bottles. Glass wool was wrapped with aluminium foil and stored in the desiccator to avoid absorption of moisture and contamination. Furthermore, the method performance and validation was evaluated by extracting organic contaminants in house dust "SRM-2585" (Table 1). Prior to sample analysis and after, an initial solvent blank, a laboratory performance standard check (linearity of the calibration curve), was performed using individual and mixtures of the most common brominated flame retardants. This was to ensure proper performance of the GC-MS. Check standard was run after every five samples to monitor retention time deviation.

Table 1

Certified and measured concentrations (ng g^{-1}) of certified reference material (NIST SRM-2585-Organic contaminants in house dust).

Congener	Measured	Certified	Recovery (%)
BDE-17	104 ± 4.1	115 ± 1.2	90
BDE-28	40 ± 1.0	46.9 ± 4.4	85
BDE-47	463.3 ± 9.0	497 ± 46	93
BDE-100	146 ± 3.5	145 ± 11	101
BDE-99	627.6 ± 0.08	892 ± 53	70
BDE-154	73.3 ± 0.7	83.5 ± 2	88
BDE-153	85.3 ± 0.8	119 ± 1	72
BDE-183	47.6 ± 4.5	43 ± 3.5	111
BDE-209	2081.6 ± 11	2510 ± 190	83

Table 2
Results of measured PBDEs (ng g^{-1}) and recovery (%) from sodium sulphate using ultra-sonication extraction method.

PBDE Congener	A		B		C		D	
	Mean conc.	Hex:Ace, 3:1	Mean conc.	Hex:DCM, 1:1	Mean conc.	Tol:DCM, 1:1	Mean conc	Tol:DCM, 3:1
BDE-47	0.54	51.3	0.99	94.3	1.18	112	1.22	116
BDE-77	0.57	54.2	1.55	148	1.25	119	1.26	120
BDE-100	0.60	57.1	1.62	154	1.18	113	1.08	103
BDE-99	1.55	147	1.55	147	1.15	109	1.06	101
BDE-118	1.16	110	1.56	149	1.14	108	1.02	97.2
BDE-154	0.52	49.3	1.76	168	1.14	109	0.95	90.8
BDE-153	0.80	76.4	1.91	182	1.21	116	0.98	93.7
BDE-139 ^a	1.49	142	0.73	69.6	1.33	126	1.57	149
BDE-128	1.30	124	1.48	141	1.11	106	1.21	115
BDE-183	0.47	44.6	0.76	71.9	1.31	125	0.99	93.8
BDE-209L ^a	0.42	40.3	0.69	65.5	1.19	114	0.47	45.0
BDE-209	0.41	39.2	0.62	59.0	1.09	104	0.47	45.0

^a Labeled standard

The use of surrogate standards during extraction and after cleanup was to ensure accuracy. Retention times of the unknown analytes were matched with those of the standards and quantification was carried out by monitoring the molecular and reference ions using external methods. External calibration of the GC-MS was performed using eight level calibration points. The limit of detection was taken as 3 times the signal-to-noise ratio and limit of quantification as 10 times signal-to-noise ratio of the lowest detectable concentration of mixed standard. Observed instrumental limit of detection ranged between 0.009 and 0.025 $\text{ng } \mu\text{L}^{-1}$. Furthermore, statistical analysis was performed with Statistica software (version 2010).

3. Results and discussion

3.1. Recovery experiments

The results of extraction efficiencies of some selected solvents described in section 2.4 are as shown in Table 2. Toluene/dichloromethane (1:1, v/v) gave the best recovery for most of the target analytes than the combination of other solvents. The observed values for a mixture of *n*-hexane/acetone (3:1, v/v) ranged from 39 to 147%, *n*-hexane/DCM (1:1, v/v) (59–181%), toluene/DCM (1:1, v/v) (104–126%), and toluene/DCM (3:1, v/v) at 75–101%.

A total of seven PBDEs were analyzed and quantified in car dust samples. The selection was based on the frequency of detection as reported in the literature and because they have been found to be dominant congeners in indoor and outdoor dust samples. The final concentration of each determined contaminant was calculated by multiplying the amounts with the final extract volume and dividing by actual weight of dust. The concentration of \sum PBDEs for a collection of twelve different car makes ranged from 159 to 736 ng g^{-1} dry weight. The geometric mean, median, 5th percentile, and 95th percentiles are shown in Table 3. Also shown in Table 3 are the frequency of detection of each congener, the minimum and maximum of determined concentrations. BDE-47 was detected in all dust samples at a detection frequency of 100% followed by BDE-209 with 92%, exhibiting maximum and minimum values of 746 and 347, and 4.9 and 0.01 ng g^{-1} , respectively.

Table 3
Descriptive statistics of measured PBDE congeners (ng g^{-1}) in car dust.

Congener	A Mean	G Mean	Median	5%	95%	Min	Max	%DF
BDE-47	99.1	34.1	67.8	5.95	740	4.9	746	100
BDE-100	64.5	24.9	17.1	4.8	450	4	454	75
BDE-99	213	19.4	13.9	0.01	2500	0.01	2562	75
BDE-154	69.5	28.0	19.3	6.7	370	6.25	378	75
BDE-153	91.7	29.3	16.4	4.6	580	378	585	75
BDE-183	47.3	18.4	25.0	0.01	110	0.01	110	67
BDE-209	137	49.8	122	0.01	346	0.01	347	92

The major congener was BDE-209 with a total concentration of 2130 ng g^{-1} thereby contributing 47%, of the total congener values, while other congeners' contribution amounted to a total of 2424 ng g^{-1} which is about 53%.

The average levels of BDE-209 (137 ng g^{-1}) reported in this study was lower than the average values of 340 000 ng g^{-1} reported by Harrad et al. [29] and 272 119 ng g^{-1} by Lagalante et al. [30] for car dust samples from the UK and USA, respectively. Cunha et al. [26] also reported a range of 98–17 122 ng g^{-1} compared to 148–1832 ng g^{-1} recorded for house dust which may suggest that dust from cars are more contaminated than dust from homes. This is not surprising since homes are cleaned more often than cars. Other factors that may have contributed to the observed levels of PBDEs in car dust include, but not limited to, make of car, age, and type of materials used for the car seats. This assertion was also backed by the report by Lagalante et al. [30] which showed that the BDE-209 levels varied by almost three orders of magnitude between cars from the same manufacturer indicating that there are other factors besides vehicle make that may be responsible for the variance in BDE-209 levels. Studies have shown that highly brominated congeners are susceptible to debromination to lower congeners under ultraviolet lights [17]. This may explain the relative lower levels of BDE-209 determined in car dust in the present study. Nigeria experiences tropical climate with high temperatures virtually all year round compared to the UK and USA. Therefore, debromination by ultraviolet light is most likely to be a common occurrence with PBDE, although this was not investigated in the present study. According to Yu et al. [31], higher temperatures can lead to higher emission rates of PBDEs from household products. Available information from Bromine Science and Environmental Forum pointed out that automobile interiors are likely to contain

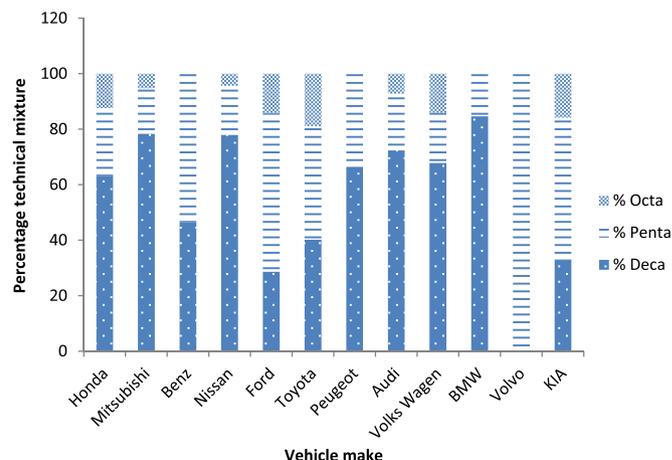


Fig. 1. Contribution of various commercial mixtures to the sum PBDE concentration measured in cars in Nigeria.

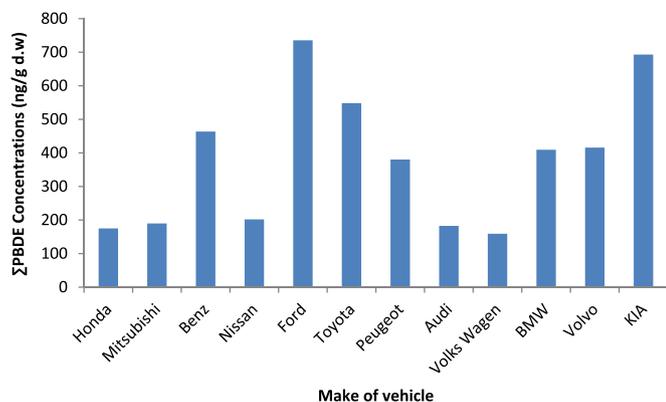


Fig. 2. Total PBDE concentrations (ng g^{-1}) measured in individual cars.

PBDEs in the acrylonitrile butadiene styrene (ABS) parts, nylon connectors, polypropylene-molded parts in the instrument panel and interior trim, in the textiles and polyurethane foams used in interior upholstery and trim, and vehicle electronics [6]. In addition, luxury packages including digital media systems (stereos, DVD players, LCD screens, and on-board computers), navigation systems, and related luxury electronics are commonly encountered in higher-end vehicles [30]. According to Tullo [32], the average content of plastics in North American vehicles has increased from 130 kg (7.3% vehicle weight) in 2000 to 152 kg (8.3% vehicle weight) in 2004, primarily motivated by the ease of molding plastic automotive parts and the desire to increase fuel efficiency by decreasing the vehicle weight.

A one-way ANOVA was performed to test the level of significance in the different mean concentrations recorded for the seven congeners considering conditions mentioned earlier. The test results showed a significant difference in the levels of PBDEs determined in each of the samples. Further analysis using Duncan test showed BDE-209 and BDE-47 to be significantly different from the rest of the congeners, while BDE-99, BDE-100, and BDE-153 concentrations are not significantly different. The contribution to various technical mixtures was as shown in Fig. 1.

The twelve different makes of cars exhibited a varying degree of \sum PBDE concentrations ranging from 159 to 735 ng g^{-1} as shown in Fig. 2. Comparative PBDE levels were observed for five makes of cars between 100 and 200 ng g^{-1} while Kia and Ford cars gave higher concentrations of 693 and 735 ng g^{-1} , respectively.

Table 4 shows the mean and median concentrations of \sum PBDEs measured in this study and other selected studies for relative comparison. As can be seen in Table 4, the highest levels of PBDEs were measured in cars from the USA followed by cars from the UK. Among all the listed studies in the table, BDE-209 reported in this study is by

far the lowest with several orders of magnitude compared to that of the USA and the UK but only close to Pakistan and Kuwait.

4.1. Exposure to PBDEs via dust ingestion

In order to assess the risk that PBDE in cars poses to the health of drivers and commuters, an attempt was made to calculate the estimated dose via dust ingestion. There is, however, great uncertainty in an automobile ingestion calculation when based on the same method used for indoor dust ingestion rate and the notion is supported by Lagalante et al. [30]. The estimation rates were based on soil matrix, owing to the fact that elevated ingestion rates in children are due to increased floor contact through crawling and may not be representative of vehicle ingestion rates. Secondly, the time spent in automobiles is significantly less than the time spent indoors and thus the exposure potential for PBDE congeners is minimized. Furthermore, the rate of dust inhalation in vehicles is influenced by the presence or absence of a vehicle cabin filter or whether vehicle occupants have the windows open or closed [30]. Different methods have been applied to estimate ingestion dose of PBDEs in humans, and most have been based on the indoor dust method, which heavily relies on the EPA Exposure Factors Handbook. Household dust exposure studies use average (20 and 50 mg day^{-1}) and high (100 and 200 mg day^{-1}) exposure rates for adults and children, respectively [37]. However, in the absence of reliable automobile ingestion rate estimates, we used the method applied by Guo and Kannan and Qi et al. [38,39] stated as follows:

$$DI = \frac{C_{\text{dust}} f_1 f_2}{M1}$$

Where DI is daily intake, C_{dust} is median dust concentration of \sum_7 PBDEs, f_1 is the fraction of time spent in vehicles, f_2 is the dust ingestion rate, and M1 is the body weight (kg). According to the EPA Exposure Factors Handbook [37], the estimated time spent in cars is about 78 min for adults and 55 min for children. In this study, therefore, we estimated DIs for humans within four different categories; toddlers (0–3 years), young children (4–18 years), adults (≥ 19 years), and professional drivers (≥ 19 years), assuming the latter spend longer time in cars (up to 10 h). The dust ingestion rate used was 0.1 g day^{-1} for toddlers and 0.05 g day^{-1} for all other groups [37]. Daily intake was, therefore, estimated using concentrations corresponding to the 5th percentile, median, and the 95th percentile. As can be seen in Table 5, DI estimated was highest for professional drivers at 0.12 $\text{ng kg}^{-1} \text{bw day}^{-1}$ and this was followed by toddlers at 0.09 $\text{ng kg}^{-1} \text{bw day}^{-1}$, while DI estimate of 0.01 $\text{ng kg}^{-1} \text{bw day}^{-1}$ was determined for children (4–18 years) and adults (> 19 years). The reference dose for oral exposure for PBDEs 47, 99, 153, and 209 are 100, 100, 200, and 7000 ng kg bw day^{-1} as suggested by the US-EPA [30]. But as shown in Table 5,

Table 4
Concentrations (ng g^{-1}) of PBDEs in car dust samples in the present study and selected other studies.

Country	Statistical parameter	BDE-47	BDE-100	BDE-99	BDE-154	BDE-153	BDE-183	BDE-209	Reference
Portugal (n = 9)	Mean	-	-	-	-	-	-	3271	[26]
UK (n = 20)	Mean	720	220	990	160	150	19	410 000	[29]
USA (n = 60)	Median	54	17	100	-	-	7.8	100 000	[30]
	Mean	1233	345	1989	194	552	238	272 119	
USA (n = 66)	Median	-	-	-	-	-	-	8120	[33]
Egypt (n = 5)	Median	5.7	4.8	23	-	-	5.8	1540	[34]
Kuwait (n = 15)	Median	5.8	1.5	8.5	-	-	1	665	[35]
Pakistan (n = 15)	Median	1.2	0.3	1.7	-	-	12	625	
Czech Rep. (n = 27)	Mean	17.5	1	21.9	1.4	1.8	2.2	3523	[36]
	Median	2.2	<0.1	<0.1	<0.3	<0.3	<0.8	-	
UK (n = 14)	Median	54	17	100	11	11	7.8	100 000	[17]
US (n = 14)	Median	880	211	1130	105	163	73	48 100	
Nigeria (n = 12)	Mean	99	4.5	213	69.5	91.7	47.3	137	This study
	Median	67.8	17.1	13.9	19.3	16.4	25	122	

Table 5Estimated human exposure in $\text{ng kg}^{-1} \text{bw day}^{-1}$ of PBDEs via dust ingestion from car dust in this and other similar studies.

Location	Matrix	Toddlers	Children	Adult	Drivers	Reference
Nigeria (Benue)						
5th percentile	Car dust (\sum_7 PBDEs)	0.001	0.0002	0.0002	0.002	This study
Median		0.008	0.001	0.001	0.01	
95th percentile		0.09	0.01	0.01	0.12	
Egypt (Cairo)						
5th percentile	House/car/office (\sum_{14} PBDEs)	0.07 (0.3) ^a	-	0.01 (0.02)	-	[34]
Median		0.5 (1.9)	-	0.06 (0.14)	-	
95th percentile		8.1 (32.6)	-	4.21 (10.5)	-	
South Africa (Pretoria)						
Median	Home dust (BDE-47 and BDE-99)	-	1.37 (1.37)	0.08 (0.08)	-	[18]
Kuwait						
5th percentile	Car dust (n = 15)	0.65	-	0.04	0.05	[35]
Median		1.54	-	0.11	0.13	
95th percentile		33.1	-	2.25	5.06	
Pakistan						
5th percentile		0.17	-	0.01	-	[35]
Median		0.69	-	0.05	-	
95th percentile		27.8	-	1.9	-	

^a High dust ingestion estimates.

DI rates for car dust estimation from the present study is way below the USEPA reference dose (RfD).

4. Conclusion

The study confirms the presence of the most important congeners of polybrominated diphenyl ethers in car dust in Nigeria. BDE-209 was the most prevalent congener in all the samples. This observation is similar to other studies reported for BDE-209 in dust samples from cars in the UK and USA. This may be attributed to a number of influencing factors such as climate, car make, age of car, and the type of materials used in car upholstery. The daily intake of PBDEs was calculated for four different categories of individuals using an equation recently applied by other researchers in the absence of car-specific estimation formula. Estimates were based on observed sum PBDEs. Low DIs were recorded which suggested that ingestion of PBDEs via dust presently poses no significant risk to affected individuals. However, as more secondhand cars are still being imported into Nigeria, the observed scenario may change sooner than later. Furthermore, the disposal of the cars once they have outlived their usefulness will present another environmental problem. To our knowledge, this is the first study to generate data on levels of BDEs in car dust and ingestion rates estimation in Nigeria. However, further studies are required to illustrate human exposure via other routes including dermal absorption, inhalation, and through diets.

Acknowledgements

The authors wish to acknowledge Tshwane University of Technology for the bursary granted to Olukunle, O.I. for his doctoral studies and the Organisation for the Prohibition of Chemical Weapon (OPCW) for funding the consumables used in this project.

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