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Occurrence of PBDEs in car and airplane dust in Poland – exposure assessment and health risk characterization for selected age ranges

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Abstract

Introduction and Objective. Polybrominated diphenyl ethers (PBDEs) are a class of flame-retarding synthetic compounds. They may cause a potential threat to human health due to their bio-accumulative and toxicological properties, and ubiquitous presence in the environment. Food, and ingested dust constitute principal sources of human exposure to PBDEs. The aim of this study was to assess the potential human exposure to selected polybrominated diphenyl ethers measured in dust found in cars and airplane cabins to characterize the health risk.

Materials and method. 31 samples of car dust and 14 samples of airplane dust were collected and concentrations of BDE-47, BDE-99, BDE-153 and BDE-209 congeners were determined by gas chromatography with micro-electron capture detector (GC-µECD). Exposures were estimated for infants (0–1 year), toddlers (1–3 years) and adults (>18 years). The Hazard Quotients (HQs) were calculated by comparing the estimated exposure values to reference doses (RfD) established by the US EPA.

Results. The study found that BDE-209 levels were much higher in the majority of samples than in the remaining PBDEs The estimated values of average and reasonable maximum exposure (P95) in each age group ranged from <0.001 ng kg^{-1} b.w. day^{-1} to 382 ng kg^{-1} b.w. day^{-1} and from <0.001 ng kg^{-1} b.w. day^{-1} to 1.2 µg kg^{-1} b.w. day^{-1}, respectively (considering the individual analysed PBDE congeners). Additionally, the exposure of infants and toddlers was estimated using the highest PBDE concentration reported in the study and the maximum daily dust intake values. All the HQ values were lower than 1, in the majority of cases 2 orders of magnitude lower than 1.

Conclusions. The levels of tested PBDE congeners measured both in car and aircraft dust did not indicate health risk for these selected populations resulting from ingestion of dust.

Key words

exposure, PBDEs, health risk, car dust, airplane dust

INTRODUCTION

Polybrominated diphenyl ethers (PBDEs) belong to the group of synthetic brominated flame retardants. They have been in use since the 1970s and added to products such as textiles (e.g. carpets, carpet and vinyl flooring, decorative and upholstery fabrics), and to foams used to fill furniture and electronic equipment [1, 2]. There were 3 types of commercial PBDE mixtures available: decaBDE manufactured in the largest quantities, in which decabromodiphenyl ether (BDE-209) constitutes 90% of the final product, pentaBDE containing the highest concentrations of BDE-47, BDE-99 and BDE-100, and octaBDE containing BDE-183, BDE-190, BDE-197 and BDE-196 [3–5]. PBDEs are persistent in the environment, can bio-accumulate in animal and human tissue and bio-magnify in food chains. They belong to the group of compounds referred to as persistent organic pollutants (POPs) [6–8].

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Over the years, the production and use of these compounds has gradually been limited with the help of international regulations, such as the Stockholm Convention on Persistent Organic Pollutants (POPs) [7, 8]. In 2004, EU Member States implemented a directive limiting the use of products containing mixtures of pentaBDE and octaBDE of over 0.1% [9]. Commission Regulation (EU) 2017/227 imposed a ban on the manufacture of products containing decabromodiphenyl ether, with a derogation for its use in the automobile industry (permission for its use covered cars to be manufactured by 1 March 2019) and aviation industry (permission for use by 1 March 2027) [10].

The long-term application of PBDEs has resulted in their common presence throughout the environment which may constitute a potential threat to human health based on their toxicological properties. They interfere with the homeostasis of thyroid gland hormones, i.e. triiodothyronine (T_3) and thyroxine (T_4) [11–13], are neurotoxic [14–18] and cause negative reproductive health outcomes, among others [19, 20]. Addition of PDBEs to fodder fed to laboratory animals, revealed oxidative stress-related damage to multiple organs [21–23], with the liver found to be the most affected [1].

The main sources of the emission of PBDEs into the environment are directly from PBDE-containing products and incineration or recycling of waste. These compounds can be transported in the air over large distances, adsorbed on the surface of a solid particulate (e.g. dust), or suspended in the gas phase to subsequently become deposited on soil or dust. Consequently, they are detected worldwide in environmental samples even long distances from emission sources [24–26]. The principal sources of the exposure to the general population to PBDEs is in food, especially of animal origin, and for some populations, ingested dust [1, 24, 27–30].

Given the long outgassing period, aircraft and cars can remain a potential source of release for many years. For example, in 2019, the mean age of passenger cars in Poland was 14.1 years [31], whereas the estimated service life of aircraft was approximately 30 years [32, 33]. Only a few studies are available that assess the exposure of humans to PBDEs from working on or being inside aircraft [34–36]. To address this knowledge gap, this study attempts to assess the exposure potential of chosen sub-age groups to selected PBDEs via ingested dust from inside of cars and aeroplane cabins, and to estimate the resultant risk to human health.

MATERIALS AND METHOD

Dust from cars and aircraft cabins. Between 2013–2015, 31 samples of dust were collected from the interiors of cars. The samples were collected by vacuuming the same surfaces (floor, upholstery, dashboard, headliner, and baggage rack), using the same Samsung SC61E7 vacuum cleaner, into disposable paper bags.

In 2016, 14 dust samples were collected from the interiors of aircraft at Warsaw Airport. Six samples were collected during the selected age groups planes (a mixed dust sample).

Preparation and storage of samples. On delivery to the laboratory, the samples were immediately sifted through a 150 μm steel sieve using a vibratory sieve shaker Retsch AS 200 basic. The samples were stored in closed aluminium containers under controlled conditions at the temperature of -20^{o}C until analysis.

Reference materials and reagents. This study used:
– Standards: BDE-47, BDE-99, BDE-153 and BDE-209, each in the form of a 1.2 ml ampule of 50 μg/ml concentration in the nonane (Cambridge Isotope Laboratories, Andover, U.S.A.)
– Reference material: SRM 2585 (NIST).
– Reagents: *n*-hexane for ECD and FID gas chromatography (Merck, Darmstadt, Germany), acetone for ECD and FID gas chromatography (Merck, Darmstadt, Germany), dichloromethane for the analysis of pesticides residue (Merck, Darmstadt, Germany), *n*-dodecane for synthesis, silica gel 60 extra pure (70–230 mesh ASTM) for column chromatography (Merck, Darmstadt, Germany), aluminium oxide 90 active, neutral for column chromatography (Merck, Darmstadt, Germany), Florisil and glass wool (Merck, Darmstadt, Germany), sodium sulphate anhydrous (12–60 mesh, J.T. Baker, Deventer, The Netherlands) and cellulose filter tubes (43 x 123 mm, Munktell, Barestein, Gernamy).

Extraction and purification of samples. The preparation of the dust samples was performed in a darkened laboratory to minimise the potential for photolytic debromination of PBDEs, especially the BDE-209 congener. In addition, amber laboratory glass was used throughout for the storage of all samples. Once the samples reached room temperature, they were extracted and purified according to the procedure described by Korcz et al. [37]. Briefly, the extraction was performed with a 3:1 (v:v) mixture of *n*-hexane:acetone with an automated Büchi B-811extraction system. The clean-up procedure was carried out in a glass column (60 cm x 1.5 cm i.d.) containing the following layers, from the bottom: 0.5 cm of sodium sulphate anhydrous, 10 g of silica gel, 5 g of aluminium oxide and 0.5 cm of sodium sulphate anhydrous. Next, the samples were analysed by the gas chromatography with electron-capture detection (GC-μECD) as described below.

Chromatographic analysis. Quantitative determination was performed using an Agilent Technologies 6890N gas chromatograph with a μECD detector. The instrument operating parameters are provided in Table 1. Identification of the congeners BDE-47, BDE-99 and BDE-153 was confirmed by gas chromatography coupled to a Varian 4000 ion trap mass spectrometer. The confirmatory method has been described previously by Korcz et al. [37]. BDE-209 was not confirmed on GC-MS due to its thermolability which was why the μECD detector was used for analyte concentration measurement. Identification of BDE-209 was accomplished, using a capillary column stationary phase with a different polarity (DB-XLB column) from that of the column used for quantitation (Tab. 2).

Table 1. Agilent Technologies 6890N GC-μECD settings in the method used for the PBDEs quantification in aircraft and car dust.

Parameter	
Column	DB–5MS column (15 m x 0.25 mm i.d., film thickness 0.25 μm
Oven temperature programe	100°C (1 min.) –30°C min. –1 – 320°C (5 min.)
Carrier gas flow Const Flow	3.1 ml min^{-1}
PTV injector	'solvent vent mode', ramp temperature programme: 40°C (0.3 min.) –700°C min^{-1} – 280°C (3 min.)
Injection volume	1 μl
Carrier gas	helium
Detector set point	325 °C

Retention times of the analysed PBDEs are: BDE-47 – 5.772 min., BDE-99 – 6.442 min., BDE-153 – 7.014 min. and BDE-209 – 10.218 min.

Validation. Sifted putty gypsum was selected as the matrix for the validation of the analytical method since it is physically similar to dust (dust particles diameter of up to 150 μm) and its contribution in dust composition can be significant [38]. Six samples fortified at the level of the limit of quantification and the upper quantification limit were analysed, i.e. the linear ranges were determined. For BDE-47, BDE-99, and BDE-153 the limits of quantification (LOQs) were measured at 2 ng g^{-1} of dust. The LOQ for BDE-209 was 10 ng g^{-1} of dust (Tab. 3). Recovery ranged from 70 – 110% and a %RSD of $\pm$ 20% was adopted as the acceptance criteria. The validation confirmed the method's suitability for the determination of selected PBDEs.

Table 2. Chromatographic settings (Agilent Technologies 6890N) taken from the method applied for the BDE-209's identity confirmation in airraft and house dust

Parameter	
Column	DB-XLB (15 m x 0.25 mm i.d., film thickness 0.1 µm)
Oven temperature programme	120°C (1 min.) – 30°C min^{-1} – 300°C (8 min.),
Carrier gas flow ramp programme	1.5 ml min^{-1} (7 min.) – 15 ml min^{-1} – 3 ml min^{-1}
PTV injector	'solvent vent mode', ramp temperature programme: 40°C (0.3 min.) – 70 °C min^{-1} – 285 °C (3 min.)
Injection volume	1 µl
Carrier gas	helium
Detector set point	335 °C

Retention time for BDE-209 – 13.55 min.

Table 3. Summary of validation parameters in the method used for quantification PBDEs in aircraft and car dust (n=5)

Compound	BDE-47	BDE-99	BDE-153	BDE-209
LOQ [ng g^{-1}]	2	2	2	10
Linearity range [ng g^{-1}]	2-300	2-300	2-300	10-500
R^2 (coefficient of determination)	0.9997	0.9995	0.9999	0.9991
Fortification level [ng g^{-1}]		2.9		10
Average recovery [%]	82	93	80	106
Standard deviation (SD)	0.10	0.07	0.05	0.06
RSD (relative standard deviation) [%]	13	8	6	5
Relative expanded uncertainty [%]	15	12	12	17
Fortification level [ng g^{-1}]		300		500
Average recovery [%]	82	85	84	71
Standard deviation (SD)	33	34	32	28.5
RSD (relative standard deviation) [%]	13	13	13	9
Relative expanded uncertainty [%]	14	15	15	11

Quality assurance. Certified reference material SRM 2585, reagent samples and duplicate samples were analysed in each sample set. The relative percent difference (RPD) of concentrations for duplicate samples was always below 20%. The relative bias for the PBDE congener concentrations measured in the SRM ranged from 11 – 17% of the certified value, lower than the relative uncertainty of the analytical measurement. In each case the acceptability criteria were met.

Assessment of human exposure to PBDEs ingested with dust. Exposure to PBDEs ingested with dust was assessed using the U.S. ATSDR methodology [39] defined by the equation:

$$D = C \times IR \times EF/BW \quad (1)$$

where: D – exposure, C – concentration of the examined PBDE congener, IR – daily intake rate, EF – exposure factor and BW – body weight

For statistical purposes, when the PBDE congener was not quantified (result below the limit of quantification), the LOQ value was assigned accordingly to the upper bound approach [40].

The United States Environment Protection Agency (US EPA) recommended reference dust intake values were used [41] to approximate the quantity of the ingested dust. The daily dust ingestion estimated for the central trend in the populations studied was estimated to be 40 mg (for infants of 6 months – 1 year of age), 40 mg (for toddlers 1 – 6 years of age) and 20 mg for the population over 12 years of age. The US EPA also recommends the maximum value of daily dust ingested for all age groups at 100 mg. For the purpose of this study, the following average body mass for chosen sub-groups was adopted: infants 5 kg, toddlers (up to 3 years of age) 12 kg, and adults 70 kg [1].

These calculations also took into account which part of the day the chosen age sub-groups theoretically spend in cars and on aircraft. The assumed value for a car was ⅓ of a day (8 hours of exposure). In 2017, the majority of passenger flights in the European Union were within the EU territory (EUROSTAT, 2021), therefore a ⅙ factor was adopted (average flight time from Warsaw to Madrid, ca. 3 hours 50 minutes) while for longer flights, outside the EU territory, a 5⁄12 factor (10 hours of flight) was selected.

Health risk characterization. Hazard Quotients (HQs) were calculated by comparing the calculated human exposure to the selected PBDE congener (D) with the corresponding reference dose (RfD) values established by the US Environment Protection Agency [42, 43]. RfD is defined as the daily exposure of the human population (sensitive subgroups included), which should not generate a perceptible risk of the development of harmful health effects over an entire lifetime (US EPA, 1993). The US EPA specified reference doses (RfD) for 4 PBDE congeners included in the study, i.e. BDE-47, BDE-99, BDE-153 and BDE-209. They amounted to 0.0001, 0.0001, 0.0002 and 0.007 mg kg^{-1} of body mass day^{-1}, respectively [44–47].

$$HQ = \frac{D}{RfD} \quad (2)$$

When the calculated HQ was higher than or equal to 1, it was assumed that there was a potential risk of an adverse heath outcome from the exposure of a selected populations to a given PBDE congener ingested on dust [39].

RESULTS

PBDEs levels in car and aeroplane dust. Descriptive statistics for the distribution of PBDE concentrations found in samples of car and aeroplane dust samples is presented in Table 4.

The median values for the concentration of the PBDEs analysed in both car and aeroplane dust were lower than mean values, which is characteristic of the right skewed distribution. Skewness is a measure of the distribution's symmetry. The skewness of the PBDE concentrations fluctuated from 2.2 – 5.4.

The oldest car from which a dust sample was collected was manufactured in 1990 (23 years old at the time of sample collection), while the newest car was a month old (production year 2014). BDE-209 was the only PBDE congener quantitated in all samples. The domination of BDE-209 over all other PBDE congeners, in all samples analysed was confirmed. The percent of decabromodiphenyl ether in the total PBDEs

Table 4. Descriptive statistics of PBDE concentrations measured in car and aircraft dust.

Compound	df	Mean	Median	Min	Max	P95
BDE-47 [ng g^{-1} dust]						
Cars	7/31	15	<2	<2	236	75
Aircraft	11/14	200	35	<2	1185	1093
BDE-99 [ng g^{-1} dust]						
Cars	6/31	19	<2	<2	347	89
Aircraft	11/14	298	37	<2	1963	1648
BDE-153 [ng g^{-1} dust]						
Cars	2/31	3.5	<2	<2	35	9
Aircraft	2/14	19	<2	<2	236	8
BDE-209 [µg g^{-1} dust]						
Cars	31/31	22	0.6	0.08	530	57
Aircraft	14/14	36	10	1.2	282	146

df – a numer of samples above the LOQ versus total number of samples

analysed ranged from 92.0% – 100.0%. The 3 highest BDE-209 concentrations were found in the samples originating from Japanese cars manufactured in the years 2004 – 2013. The remaining PBDE congeners: BDE-47, BDE-99, and BDE-153, were present at levels above the respective LOQs in 7, 6, and 2 samples, respectively. The percent of BDE-47 in the total PBDEs concentrations ranged from 0.0% – 3.0%, while that of BDE-99 – from 0.0% – 4.0%, and that of BDE-153 from 0.0% – 0.4%. The highest levels of BDE-99 and BDE-153 were detected in 2 samples that originated from German cars manufactured in the 1990s.

Aircraft dust also had very large ranges in the measured PBDEs concentrations. The percent of the BDE-47 congener in relation to the total of the PBDEs analysed reached 9.0% while that of BDE-99 – 3.0%. BDE-153 oscillated from 0.0 – 2.0%. The decabromodiphenyl ether dominated in all the dust samples from aircraft, ranging from 71.0 – 99.0% of the total PBDEs content.

Due to the presence and dominant nature of BDE-209 found in all dust samples, values were compared between cars and aircraft. The BDE-209 levels detected in dust from aircraft cabins were statistically significantly higher than in the car dust ($p = 0.001$). Both BDE-47 and BDE-99, were found to be above the LOQs for the majority of samples. The fraction of BDE-153 and its frequency of detection was insignificant.

Assessment of human exposure to PBDEs ingested with dust. The calculations used the mean value of the given PBDE congener concentration result and its 95% percentile (P95), as well as the values of the daily dust ingestion (central trend and maximum daily dust ingestion) according to the US EPA [41]. Additionally, the exposure of the most vulnerable sub-age groups, infants and toddlers, was estimated using the highest PBDE concentration reported in the study and the maximum daily dust ingestion. Only these 2 subpopulations were because the dust ingestion value (and subsequent level of PBDEs ingestion) per kg of body mass in infants and toddlers was considerably much higher than in adults.

The results of the estimated exposure of the selected populations to PBDEs ingested with car dust are presented in Table 5. Tables 6–7 present the values of estimated human exposure to the ingested PBDE congeners during aircraft flights of 4 and 10 hours.

Table 5. Estimated exposures through ingestion in selected populations to 4 PBDE congeners in car dust (assumed 8 hours of exposure in car interiors per day)

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
Exposure [ng kg^{-1} b.w. day^{-1}] (using recommended daily dust ingestion values – central tendency)[1]								
Infants (0-1 year)	0.04	0.2	0.05	0.2	0.009	0.02	59	153
Toddlers (1 – 3 year)	0.02	0.08	0.02	0.1	0.004	0.009	25	64
Adults (>18 years)	0.001	0.02	0.002	0.03	<0.001	0.003	2.1	5.5
Exposure [ng kg^{-1} b.w. day^{-1}] (using maximum recommended daily dust ingestion values)[2]								
Infants (0-1 year)	0.1	0.5	0.1	0.6	0.02	0.06	148	382
Toddlers (1 – 3 year)	0.04	0.2	0.05	0.2	0.01	0.02	61.5	159
Adults (>18 years)	0.007	0.04	0.009	0.04	0.002	0.004	5.5	27

[1]According to US EPA recommended daily dust ingestion values for infants, toddlers and adults are, respectively: 40, 40, and 20 mg.
[2]100 mg of dust.

Table 6. Estimated exposures through ingestion in selected populations to 4 PBDE congeners in aircraft dust (4 hours flight)

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
Exposure [ng kg^{-1} b.w. day^{-1}] (using recommended daily dust ingestion values – central tendention)[1]								
Infants (0-1 year)	0.3	1.5	0.4	2.4	0.03	0.01	48	195
Toddlers (1 – 3 year)	0.1	0.6	0.2	0.9	0.01	0.004	20	81
Adults (>18 years)	0.009	0.05	0.01	0.08	<0.001	<0.001	1.7	7
Exposure [ng kg^{-1} b.w. day^{-1}] (using maximum recommended daily dust ingestion values)[2]								
Infants (0-1 Year)	0.7	3.6	1	5.5	0.06	0.03	119	487
Toddlers (1 – 3 year)	0.3	1.5	0.4	2.3	0.03	0.01	50	203
Adults (>18 years)	0.05	0.3	0.07	0.4	0.005	0.002	8.5	35

[1]According to US EPA recommended daily dust ingestion values for infants, toddlers and adults are, respectively: 40, 40, and 20 mg
[2]100 mg of dust.

Additionally, the exposure of the most vulnerable populations was estimated based on the highest PBDE congeners levels found in this study and the maximum modelled daily dust ingestion value. For car dust, BDE-47, BDE-99, BDE-153 and BDE-209 exposure values for infants calculated in this way were: 1.6, 2.3, 0.2 ng kg^{-1} b.w.$day^{-1}$and 3.5 µg kg^{-1} b.w.day^{-1}, respectively. The corresponding exposure values calculated for toddlers to the afore-mentioned PBDE congeners were: 7.9, 11.4, 0.2 ng kg^{-1} b.w.day^{-1}, and 2.8 µg kg^{-1} b.w. day^{-1}, respectively.

Estimating the exposure of infants during a 4-hour flight to the highest levels of the analysed PBDE congeners found in aircraft dust, using the maximum daily dust ingestion value, created estimates of 4, 6.5, 0.8, and 940 ng kg^{-1} b.w. day^{-1}

Table 7. Estimated exposures through ingestion in selected populations to four PBDE congeners in airplane dust (intercontinental flight by plane)

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
Exposure [ng kg^{-1} b.w. day^{-1}] (using recommended daily dust ingestion values – central tendency)[1]								
Infants (0-1 year)	0.7	3.6	1	5.5	0.06	0.03	119	488
Toddlers (1 – 3 year)	0.3	1.5	0.4	2.3	0.03	0.01	50	203
Adults (>18 years)	0.02	0.1	0.04	0.2	0.002	<0.001	4.3	17
Exposure [ng kg^{-1} b.w. day^{-1}] (using maximum recommended daily dust ingestion values)[2]								
Infants (0-1 Year)	1.7	9.1	2.5	13.7	0.2	0.07	299	1219
Toddlers (1 – 3 year)	0.7	3.8	1	5.7	0.07	0.03	124	508
Adults (>18 years)	0.1	0.7	0.2	1	0.1	0.005	21	87

for BDE-47, BDE-99, BDE-153, and BDE-209, respectively. For toddlers, these values were: 1.7, 2.7, 0.3, and 392 ng kg^{-1} b.w. day^{-1}. For an intercontinental, 10-hour flight, the values for infants and toddlers were proportionally higher and amounted to 9.9, 16.4, 2 ng kg^{-1} b.w. day^{-1}, and 2.4 µg kg^{-1} b.w. day^{-1}, and 4.1, 6.8, 0.8 and 980 ng kg^{-1} b.w. day^{-1}, respectively.

Characterization of health risk for chosen age sub-groups associated with exposure to PBDEs ingested with dust. The associated health risk from ingestion of BDE-47, BDE-99, BDE-153 and BDE-209 in dust was estimated for mean values and P95. Table 8 presents the results of the risk characterization associated with exposure to PBDEs present in dust from a car interior. Based on these results, it was concluded that only the BDE-209 estimated exposure values were higher than 1% of the RfD.

Table 8. Health risk characterization in selected age groups resulting from their exposures to PBDEs ingested in car dust

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
HQs (for the exposures using recommended daily dust ingestion values – central tendency)[1]								
Infants (0–1 year)	<0.001	0.002	0.001	0.002	<0.001	<0.001	0.008	**0.022**
Toddlers (1–3 year)	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	0.004	0.009
Adults (>18 years)	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.007
HQs (for the exposures using maximum recommended daily dust ingestion)[2]								
Infants (0–1 year)	0.001	0.005	0.001	0.006	<0.001	<0.001	**0.016**	**0.055**
Toddlers (1–3 year)	<0.001	0.002	0.001	0.002	<0.001	<0.001	0.009	**0.023**
Adults (>18 years)	<0.001	0.001	<0.001	<0.001	<0.001	<0.001	0.002	0.004

All the HQs higher than 1% of the corresponding RfD values are in bold.
[1]According to US EPA recommended daily dust ingestion values for infants. toddlers and adults are, respectively: 40, 40, and 20 mg
[2]100 mg of dust.

The hazard quotient (HQ) results presented in Tables 9–10 were calculated to assess the potential health risk associated from exposure to the PBDEs ingested with aircraft cabin dust. In all cases, the HQ values were much lower than 1. The highest exposure value in infants, (P95) was estimated to be 17.0% of the RfD for BDE-209 for a 10-hour flight and the recommended maximum for daily dust ingestion [41]. Human exposure to any PBDE congeners ingested with aircraft cabin dust does not pose a health risk.

Table 9. Health risk characterization of selected age groups from estimated exposures to PBDEs ingested in airplane dust (4 hour flight)

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
HQs (for the exposures using recommended daily dust ingestion values – central tendency)[1]								
Infants (0-1 year)	0.003	**0.015**	0.004	**0.022**	<0.001	<0.001	0.007	**0.028**
Toddlers (1 – 3 year)	0.001	0.006	0.002	0.009	<0.001	<0.001	0.003	**0.012**
Adults (>18 years)	<0.001	0.001	<0.001	0.001	<0.001	<0.001	<0.001	0.001
HQs (for the exposures using maximum recommended daily dust ingestion)[2]								
Infants (0-1 year)	0.007	**0.036**	0.010	**0.055**	<0.001	<0.001	**0.017**	**0.070**
Toddlers (1 – 3 year)	0.003	**0.015**	0.004	**0.023**	<0.001	<0.001	0.007	**0.029**
Adults (>18 years)	<0.001	0.003	0.001	0.004	<0.001	<0.001	0.001	0.005

All the HQs higher than 1% of the corresponding RfD values are bolded.
[1]According to US EPA recommended daily dust ingestion values for infants, toddlers and adults are, respectively: 40, 40, and 20 mg
[2]100 mg of dust..

Table 10. Health risk characterization in selected age groups resulting from their exposures to PBDEs ingested in aircraft dust (intercontinental 10 hour flight)

Population	BDE-47		BDE-99		BDE-153		BDE-209	
	Mean	*P95*	*Mean*	*P95*	*Mean*	*P95*	*Mean*	*P95*
HQs (for the exposures using recommended daily dust ingestion values – central tendency)[1]								
Infants (0–1 year)	0.007	0.036	0.010	0.055	<0.001	<0,001	0.017	0.070
Toddlers (1–3 year)	0.003	**0.015**	0.004	**0.023**	<0.001	<0.001	0.007	**0.029**
Adults (>18 years)	<0.001	0.001	<0.001	0.002	<0.001	<0.001	0.001	0.003
HQs (for the exposures using maximum recommended daily dust ingestion)[2]								
Infants (0–1 year)	**0.017**	**0.091**	**0.025**	**0.137**	<0.001	<0.001	**0.043**	**0.174**
Toddlers (1–3 year)	0.007	**0.038**	0.010	0.057	<0.001	<0.001	**0.018**	**0.073**
Adults (>18 years)	0.001	0.006	0.002	0.010	<0.001	<0.001	0.003	**0.024**

All the HQs higher than 1% of the corresponding RfD values are bolded.
[1]According to US EPA recommended daily dust ingestion values for infants. toddlers and adults are, respectively: 40, 40, and 20 mg.
[2]100 mg of dust.

An unrealistic 'worst-case scenario' for infants and toddlers was also calculated by assuming exposure to the highest levels of PBDEs. For car dust, the HQs calculated for infants' exposed to BDE-47, BDE-99, BDE-153, and BDE-209 were: 0.016, 0.023, 0.001 and 0.505, respectively, while for toddlers, HQs 0.007, 0.009, <0.001 and 0.210, respectively. It can be concluded that even a highly over-estimated exposure of vulnerable populations to the PBDEs in car dust measured for this study, does not constitute a potential threat to human health.

Similar results were obtained for dust in aircraft cabins during 4- and 10-hour flights. The HQs for infants (4/10 hours flight time) were 0.040/0.098 for BDE-47, 0.065/0.164 for BDE-99, 0.004/0.010 for BDE-153, and 0.134/0.336 for decabromodiphenyl ether.

For toddlers, HQs values calculated for 4- and 10-hour flights were: 0.017/0.041 for BDE-47, 0.027/0.068 for BDE-99, 0.002/0.004 for BDE-153, and 0.055/0.139 for BDE-209.

DISCUSSION

BDE-209 dominated the samples of car dust analysed, with the highest levels found in Japanese cars. Analogous results were also reported by Gevao et al. [48]. Other studies confirming BDE-209 results consistent with this study include Ozkaleli Akcetin et al. and Bramwell et al. [49, 50]. Of note was that the median value of BDE-209 from cars in this study was lower than those reported in publications from other European countries, and China [27, 50, 51], and higher than that reported in studies from African and West Asian countries [48, 49, 52, 53]. The frequency of occurrence and levels of the remaining PBDE congeners analysed in the current study were lower than those described by other authors. This was probably due to reduction in the use of the commercial pentaBDE mixtures in automotive components. The levels of PBDE congeners in car dust obtained in this study were characterized by significant dispersion. A large spread in results was also reported by other authors [49, 50, 54]. The primary difference in these results demonstrates a considerable diversity of the materials and sub-assemblies used in cars. It can also be assumed that the profile of PBDEs detected in car dust can be affected by weather conditions, such as sunlight and temperature inside the car which trigger the process of BDE-209 debromination. Debromination may play the role in creating lower to BDE-209 congeners and in lowering the BDE-209 concentration in dust [27, 51, 53].

The distributions of the concentrations of the PBDEs analysed in dust coming from cars and aircraft were right skewed. Positive skewness of PBDEs concentrations distributions in car dust were also obtained by Muernhor and Harrad [55], which may be due to the significant accumulation of items of equipment containing BDE-209 in the individual cars and aircraft from which the dust samples were obtained (outliers).

A systematic survey of the literature with the use of the Medline Complete base found only one publication on PBDE levels in dust collected from aircraft [34]. The authors of that study measured higher PBDEs concentrations than those obtained in the current study. Allen et al. [34] also reported BDE-209 as the dominant PBDE congener.

Measurement of the levels of PBDE congeners allows for estimation of oral exposure to these compounds applied with the assumed scenarios. A conservative approach was adopted which assumed the bioavailability factor of PBDEs ingested with dust as 1 (bioavailability of 100%) [39]. An analogous approach was also adopted by Harrad et al. [56]. Given the lack of relevant European data, the values for daily dust ingestion currently recommended by the US Environment Protection Agency were adopted [41]. This approach may be considered as the most protective or a 'worst case scenario'. Since there is no uniform approach available, different authors have adopted different values of the bio-availability factor as well as different values of the daily dust ingestion [57, 58]. The PBDEs ingestion from dust in aircraft cabins reported in the current study was an order of magnitude higher than their analogous ingestion with car dust (for P95 scenario). The exposure of toddlers to PBE-209 ingested with dust from car interiors observed in this study, was higher than that reported by Hassan and Shoeib [52], but was lower than that estimated by Olukunle et al. [53].

Given the diverse ways of presenting results by different authors, i.e. for estimation of intake per person, without providing the mean body mass of the study population, presentation of the exposure to a total PBDE congeners analysed or to selected PBDE groups (e.g. pentaPBDE) instead of to individual congeners, while including data from dissimilar age groups (e.g. school children), renders it difficult or simply impossible to compare the exposure values reported in particular studies.

All the HQs calculated in this study (for the mean and P95 values) were lower than 1 (in the majority of cases by 2 orders of magnitude), despite adopting a conservative approach. This indicates the lack of health risk due to PBDEs ingestion with car and aircraft dust to the chosen sub-age populations. Results consistent with this observation for car dust have also been reported by other authors [27, 51, 53].

Due to the very large dispersion of results, the exposure of infants and toddlers was also estimated by using the highest levels of PBDEs detected in car dust and dust from aircraft cabins to characterize the associated health risk. Despite the fact that all HQs calculated for this scenario did not exceed the 'border' value of 1 (in the case of BDE-209 they ranged from 0.13 – 0.51), the results may arouse a certain amount concern – the health risk characterized in this study refers only to one source of human exposure to PBDEs. Should other sources of exposure be included, e.g. dietary PBDEs intake, the total PBDEs intake, in which dust would have a significant share, would potentially constitute a potential health risk to infants and toddlers. A high HQ of 0.7 for BDE-209 ingested with home dust was reported by Zhu et al. who studied a population of toddlers [59].

The data available in the literature seems to indicate that the share of sources of PBDEs exposure (and exposure to other organic contaminants) varies depending on the stage of human life. For infants, breast milk constitutes the principal source of PBDEs intake, and as indicated by numerous studies, it constitutes the main source of the ingestion of congeners, such as BDE-47, BDE-153 and BDE-99 [60–62]. For toddlers, the principal sources of exposure to PBDEs are the diet and dust ingestion. In the USA, dust is the main source of exposure to PBDEs via ingestion, and results in much higher estimated exposures, compared to Europe, and corresponding to higher levels of PBDEs in dust [63, 64]. Simultaneously, Fromme et al. assessed the exposure of toddlers and adults and concluded that food is the main source of PBDEs intake in Germany (populations of toddlers and adults) [65]. In Sweden, Sahlström et al. showed that BDE-209 ingestion with dust as the dominant path of exposure for mothers and their children, but for BDE-47 and BDE-153, it was food that constituted the main source of exposure [66]. Differences in the results obtained in this study, compared with those reported by other authors, might result from different assumptions made in the estimation of exposure, e.g. different time of stay in a given environment, different bioavailability factors adopted and the assumed size of the daily dust ingestion.

There is a need to continue the studies to further monitor the level of the compounds analysed in dust in Poland. Studies of this kind could answer the question of whether exposure to PBDEs in particularly sensitive age groups, such as infants and toddlers, is decreasing, as well as to discover whether the impact of mandatory restrictions in the application of polybrominated diphenyl ethers in the EU has become verifiable.

CONCLUSIONS

No risk was found to human health resulting from the intake of PBDE congeners with car dust and aircraft cabin dust. The levels of selected PBDEs in dust samples were characterized by a very high distribution resulting from the different construction and decorative materials found in the environment where the samples were collected. Given the fact that food and dust represent the largest contribution to PBDEs intake, it would be advisable to estimate the total exposure to these compounds ingested from the sources, particularly for infants and toddlers.

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