Published in final edited form as:
Environ Sci Technol. 2010 April 1; 44(7): 2648–2653. doi:10.1021/es903240g.

PBDEs in 2–5 Year-Old Children from California and Associations with Diet and Indoor Environment

Melissa Rose[†], **Deborah H. Bennett**[†,*], **Åke Bergman**[§], **Britta Fängström**[§], **Isaac N. Pessah**[‡], and **Irva Hertz-Picciotto**[†]

[†]Department of Public Health Sciences, University of California, Davis, California 95616

[‡]Department of Molecular Biosciences, School of Veterinary Medicine, University of California, Davis, California 95616

[§]Department of Environmental Chemistry, Stockholm University, SE-10691, Stockholm, Sweden

Abstract

This study determined the body burden of PBDEs in 100 California children, and evaluated associations with sociodemographic, household, and dietary factors. In national and international comparisons, California dust, breast milk, and human serum samples contain higher concentrations of PBDEs. Higher levels in children suggest exposure pathways depend upon age.

Plasma samples were analyzed for PBDEs using GC/MS. Circulating levels of PBDEs were ten-to 1000-fold higher than similar aged populations in Mexico and Europe, 5-times higher than similar aged children across the U.S., and 2- to 10-fold higher than U.S. adults. Increased levels of higher-brominated congeners were associated with the recent purchase of new upholstered furniture or mattress and consumption of pork. Concentrations of lower-brominated congeners increased with frequency of poultry consumption. Lower maternal education was independently and significantly associated with higher levels of most congeners in the children.

Introduction

Evidence has accumulated from experimental animal studies showing adverse neurodevelopmental consequences following prenatal and early life exposure to PBDEs [1]. Both animal and human studies indicate disruption of thyroid homeostasis after either prenatal or postnatal exposure [2–7]. PBDEs have also been demonstrated to mediate the activity of key signaling proteins in the brain during a critical period of development [8].

Numerous studies have reported higher PBDE levels in North American human biospecimens, wildlife, and environmental samples compared with Europe and elsewhere [9–11]. Similarly, PBDEs in the indoor environment are markedly higher in North America than elsewhere. In an international comparison of house dust samples, U.S. dust contained concentrations of Σtri-hexa-BDEs an order of magnitude higher than New Zealand and U.K. dust samples [12]. Within the U.S., it has been hypothesized that Californians are disproportionately exposed to PBDEs as a result of stringent state-level flammability standards. Zota and colleagues examined biomonitoring data from the National Health and

*Corresponding author phone: (530) 754-8282.

Supporting Information Available

Table S1 Characteristics of the study population, Table S2 PBDEs and associations with maternal birthplace, Table S3 Multivariate model of PBDEs and breastfeeding, Figure S1 Boxplots of PBDE concentrations by infant feeding type, Figure S2 Scatterplot of PBDE concentrations and age by infant feeding type

Nutrition Examination Survey (NHANES), and dust samples collected from residences in Massachusetts and California. Their analysis revealed significantly higher PBDE concentrations in California residents compared to non-California residents. California dust samples were 4–10 times higher in congeners found in the penta-PBDE formulation, than dust samples from Canada, Massachusetts and Washington, D.C. [13].

Moreover, increasing evidence suggests an age gradient in the distribution of PBDEs in populations, with younger age conferring higher risk for exposure. In a case study of a northern California family, the parents' levels approximated the median for U.S. adults, while the children's levels approached the 95th percentile for U.S. adults [14]. A study of 20 U.S. mother-child pairs found children had on average 3.2 times the levels of PBDEs as their mothers [15]. Using pooled serum from 8,132 Australia residents, stratified by age, Toms and colleagues observed levels in 0–4 year-olds to be 4 times the levels measured in pooled samples from individuals older than 16 years [16].

Higher levels in children are likely attributable to increased exposure via breastfeeding, and increased ingestion of dust due to frequent hand to mouth contact. Johnson-Restrepo and Kannan estimated total daily exposure to PBDEs from diet, indoor air, and dust for five age groups: infants, toddlers, children, teenagers and adults. They estimate breastfed infants have the highest daily exposure, with 91% of their exposure coming from breast milk [17]. A Spanish study of 244 children found breastfed infants had significantly higher concentrations of BDE-47 and BDE-99 compared to formula-fed children when measured at four years of age [18]. Moreover, a recent study showed that concentrations of PBDEs in breast milk do not predictably decrease over time for the duration of breastfeeding, and in the case of BDE-153, can actually increase in some women [19, 20].

For toddlers, it is hypothesized that greater mobility and interaction with the environment shifts the primary source from diet to household dust, and greater exposure occurs through dermal contact and incidental dust ingestion from mouthing behavior. Stapleton and colleagues found children's exposure via hand-to-mouth behavior is nine times that of adults [21]. Johnson-Restrepo and Kannan estimate daily exposure from all sources for children 1–5 years of age to be 13.3 ng/kg-bw/day, with approximately 77% attributable to dust. For children 6–11 years of age, the estimated total daily dose decreases to 5.3 ng/kg-bw-day, with 58% attributable to dust [17].

Here we report PBDE concentrations in our study population, and associations between PBDEs and demographic, household, and dietary factors. A manuscript reporting associations with neurodevelopmental outcomes is being submitted separately.

Materials and Methods

This pilot study was designed as an add-on to the CHARGE (CHildhood Autism Risks from Genetics and the Environment) Study, an ongoing investigation of environmental influences on neurodevelopment and the etiology of autism [22]. Children are 24–60 months, born in California, and living with a biological parent who speaks either English or Spanish. Potential participants are identified from State databases, including birth files and records of those eligible for state-funded services. Children in this pilot were selected from all three CHARGE subject groups—autism (AU), developmental delay without autism (DD), and children from the general population group (GP). CHARGE GP children are randomly selected from California birth records and frequency matched to cases on age, gender and very broad geographic area. Children for the AU and DD groups are identified from State agency records or referred to the study by service providers and advocacy groups. We

identified all children in the study population with blood samples of sufficient volume for analysis. We then randomly selected 50 AU, 25 DD, and 25 GP.

Questionnaire Administration

We used two questionnaires. A general environmental exposure questionnaire (EEQ) collected information pertaining to a broad range of exposures from pre-pregnancy to the time of study participation, as well as demographic information, and a second questionnaire focused on specific sources and pathways of exposure for PBDEs. The EEQ was administered in a telephone interview within three months of the clinic visit and blood sample collection. The EEQ is approximately 1–2 hours in length and includes socioeconomic factors, breastfeeding information, residential, occupational and vehicle histories, and household and personal care product use. For those participants who did not complete an EEQ, we obtained sociodemographic information from the child's birth record.

The PBDE Questionnaire takes approximately 30 minutes, and was administered in a subsequent telephone interview between two and three years post-sample collection. Our PBDE Questionnaire was based on one developed by Wu and colleagues, and contained three sections: characteristics of the child's primary environments, time-activity patterns, and food frequency [23]. Home characteristics included the age and size of the home. For participants who did not complete the PBDE questionnaire, we obtained age and size of some residences from publicly available records.

We asked about the presence of carpeting and upholstered furniture in the main play area, and the number of electronic items in the house and hours in use per day. Parents estimated how many hours per day the child spent in the main playroom at home and how many hours per week they spent in a car. We collected information about the age of the car, or cars in which the child rode during the six months prior to sample collection, and if the interior or child's car seat had exposed foam. A study to determine PBDE concentrations in the interior of cars found the highest levels in newer cars, with concentrations decreasing over time [24].

Dietary factors were assessed for the six months prior to collection of the blood sample through parental report. Parents were asked for the number of times per week children consumed various foods that have been reported in the literature to contain PBDEs, and that have been correlated with human serum concentrations (meats, fish, dairy, eggs) [23, 25, 26]. Visual aids were sent electronically or via postal service, to facilitate estimates of typical serving sizes.

The Child Medical History provided additional breastfeeding data, and height and weight were measured at the physical examination, from which we calculated BMI and weight-for-age z-scores.

Laboratory Methods for PBDE Quantification

Vials of 2 ml of plasma were extracted and analyzed using gas chromatography with mass spectrometry according to the protocol described in Hovander and summarized here [27].

Surrogate standards (SS), BDE-77 (0.5 ng) and 13C-BDE-209 (1 ng) were added. Samples were extracted using a multistep process and evaporated. Neutral and phenolic substances were separated with potassium hydroxide (0.5 M in 50% EtOH) and hexane partitioning. Neutral fraction lipids were removed with concentrated sulfuric acid treatment. Additional cleanup was then preformed on a silica:sulfuric acid column (0.9 g). The neutral fraction was fractionated on a column of activated silica gel (0.7 g). PBDEs were eluted with dichloromethane (8 ml) followed by solvent exchange to hexane. Samples were protected from daylight during handling and storage.

Chemical standards for the individual PBDE congeners BDE-28, BDE-47, BDE-66, BDE-85, BDE-99, BDE-100, BDE-153, BDE-154, BDE-183, BDE-197 and BDE-207 were synthesized in-house while BDE-209 (Fluka Chemie Buchs, Switzerland) and 13C-labeled BDE-209 (Cambridge Isotope Laboratories, Andover, MA) were purchased [28, 29]. Laboratory Reference Material (LRM) used in the present study was plasma material obtained from a Stockholm hospital, as previously described [14, 15, 30].

The PBDE analysis was performed by gas chromatography/mass spectrometry (GC/MS) utilising a Finnigan SSQ 700 instrument (ThermoFinnigan, Bremen, Germany) connected to a Varian 3400 gas chromatograph equipped with a CTC A200S autosampler. GC separation of PBDE congeners was performed with a DB-5 HT capillary column (15 m × 0.25 mm i.d., 0.1 μm film thickness (J&W Scientific, Folsom, CA, USA) using instrumentation settings reported elsewhere [30]. The PBDE congeners, except BDE 209, were analyzed with selected ion monitoring (SIM) with BDE 209 analyzed through isotopic dilution in MS/ECNI, details presented elsewhere [31]. Eleven PBDE congeners, BDE-28, BDE-47, BDE-66, BDE-85, BDE-99, BDE-100, BDE-153, BDE-183, BDE-197, BDE-207 and 209 were analyzed with GC/MS (ECNI), as specified above, and quantified with the surrogate standards, BDE-77 and 13C-BDE-209. BDE-154 could not be quantified alone due to co-elution of the 2,2’,4,4’,5,5’-hexabromobiphenyl (BB-153) known to be present in human serum samples [32]. Procedure solvent blank samples representing every seventh sample were analyzed in the same way as the serum extracts. Limits of quantification (LOQs) were defined in direct relation to the amount of interference of PBDEs in the blank samples. The PBDEs in the samples had to be 3 times the concentration of the PBDEs in the blank to be considered for quantification. The average blank sample amount has been subtracted from the results. Laboratory reference material (LRM) was run in parallel to the analyzed samples.

In addition to precision analysis of these low-level contaminants, measurements of lipids by the enzymatic method were obtained. Trigylcerides and cholesterol were measured and an estimate of total serum lipids was calculated according to the formula of Phillips [33]. Due to limited sample volume, 94 of the 100 specimens have lipid values and data are only reported for these 94 children.

Data Analysis

Laboratory data were merged with questionnaire and covariate information. All data were cleaned, verified and reviewed for plausibility and consistency. A log-transformation was applied to all individual BDE congeners and their sums due to the skewed distributions, as typically occurs with environmental contaminants. The sum of lower brominated congeners includes BDE-28, -47, -66, -85, -99, -100, -153. The sum of higher brominated congeners includes BDE-197, -207, -209. Results are reported in units of pmol/g lipid, with the exception of Table 1, where we report in ng/g lipid to facilitate cross-study comparison. Stata version 11.0 (StataCorp LP, College Station, Texas) was used for all statistical analyses.

We conducted bivariate analyses using food frequencies, consumer products contained in the home, and vehicle characteristics, each in relation to PBDE concentrations. We constructed multivariate models to assess the effect of breastfeeding on children’s PBDE concentrations. Candidate confounders included: maternal education and age at delivery, parity, child’s age, sex, ethnicity, and weight-for-age. Those covariates showing associations with both breastfeeding and PBDEs were considered for inclusion in the multivariate analysis. Redundancy was assessed to reduce the number of variables for the final model.

Results

Population Characteristics and Associations with PBDEs

The mean age of children with lipid adjusted PBDE measures was 3.7 years and 81% were male, with 44% Hispanic and 40% non-Hispanic white. Thirty-eight percent of mothers had a college or advanced degree and 24% were foreign -born. Study population characteristics are presented in Supporting Information Table S1. We had data to determine BMI for 79 children. Of these 78.5% were normal weight, 16.5% were overweight, and 5% were obese.

All congeners, with the exception of BDE-28, BDE-183 and BDE-207, were found at higher concentrations in children whose mothers had lower education ($p<.05$), with the strongest correlation between maternal education and BDE-209 (Spearman rho −0.38 $p<.001$). Hispanic children had higher levels of lower brominated congeners than non-Hispanic children, but the difference was not statistically significant ($p=0.63$). In contrast, Hispanic children had lower levels of the higher brominated congeners, 8.2 versus 11.4 pmol/g lipid ($p=0.07$). Interestingly, children of foreign-born mothers, (52% of whom were also Hispanic), had significantly lower levels of higher brominated congeners, 6.3 v. 11.4 pmol/g lipid. $P=0.02$), with children of Mexican born mothers having the lowest concentrations (Table S2). The length of time in the U.S. varied from as short as two years to as long as 31 years, and the average length of time was16 years. Length of time in the U.S. was not associated with PBDE levels in the children.

Sex was not associated with PBDE levels, but power was low given the small number of females (based on the sex ratio in autism and matching of controls to cases). Age was not independently a significant predictor of PBDEs in this population. Total PBDEs (pmol/ g lipid) were 449.3 (age 2 to ≤ 3 years), 322.7 (>3 to ≤ 4 years), and 426.6 (>4 to ≤ 5 years). Neither BMI nor weight-for-age was associated with PBDE concentrations.

Distributions of PBDEs in the Study Population

Distributions for each of the congeners are displayed as boxplots in Supporting Information Figure S1. The most abundant congener was BDE-47, followed by BDE-99, -100, and -153, which were present at similar levels. The least abundant were BDE-183, and -66. BDE-183 is considered a marker for the commercial Octa-BDE mixture, which was produced in smaller quantities compared to the Deca and Penta mixtures. In 1999, the global market demand for Deca, Penta and Octa, was 54,800, 8,500, and 3,825 tonnes respectively [34]. The relatively small concentrations of BDE-183 may be the result of lower rates of use in commercial products, in particular, household products that are subject to California flammability standards. Table 1 presents arithmetic mean, geometric mean and median values for all measured congeners, and provides a cross-study comparison of PBDE concentrations, which includes studies of similar-aged U.S. and international populations, as well as studies of adults from California and the National Health and Nutrition Examination Survey (NHANES).

As we would expect, TetraBDE (-47 and -66) and PentaBDE (-85, -99, and -100) congeners were highly correlated with each other (Spearman rho: 0.84 – 0.94, $p < .001$). The NonaBDE (-197 and -207) congeners were correlated with DecaBDE, (Spearman rho: 0.59 – 0.67, $p < 0.001$). The HexaBDE (-153), was moderately correlated with higher brominated compounds, and more strongly correlated with lower brominated and total PBDEs, (Spearman rho: 0.73 and 0.74, respectively $p < 0.001$).

Characteristics of the Home Environment and PBDEs

Children living in larger homes had significantly lower BDE-209 concentrations, a result also found by others [35]. The average home size of the child's primary residence was 158m2 (1700 square feet), with a range of 46.5m2–400m2. The size of the home was not associated with any lower brominated congeners. Newer homes tend to have lower air exchange rates than older homes which can increase concentrations of indoor air pollutants [36]. In our sample, the age of the home is positively correlated with home size (Spearman's rho=0.53 p <.001). This may have mitigated the expected impact of home size as newer homes tended to be larger. Higher maternal education was also significantly associated with larger homes.

Concentrations of higher brominated congeners (-197, -207, -209) were associated with the purchase of new upholstered furniture or mattress in the six months prior to blood sample collection, with 70% of respondents reporting that the new mattress was for the child (Table 2). We saw no relationship between electronics in the home (number and hours in use) and PBDEs in the children. We did not collect information relating to the age of televisions and computers, which might contribute to the amount of PBDEs emitted from these products [36].

PBDEs and Vehicles

On average, children spent approximately 7.2 hours per week in the car (range 0–20). There was no association between hours spent in a car and serum PBDE levels. We also investigated whether the age of the car was associated with PBDEs and found no relationship.

Dietary Factors and PBDEs

Breastfeeding information was available for 93 children with lipid-adjusted PBDE measurements. Of these, ten mothers reported never breastfeeding, and of the remaining 83, 30 reported exclusively breastfeeding (never fed infant formula). Average duration for breastfeeding was approximately 7 months, and 40% of mothers reported breastfeeding their child for 12 months or longer.

Because few mothers did not breastfeed at all, we combined into one category children who were exclusively formula-fed and children who were breastfed for less than one month (hereafter referred to as the "non/limited-breastfed" group). BDE 47 and the sum of lower brominated congeners were significantly lower in breastfed children less than 3.7 years of age (mean age of this group) at the time of blood collection, but significantly higher in breastfed children 3.7 years or older at time of blood collection. Among younger children, the mean concentration for the sum of lower brominated congeners was 537.8 pmol/g lipid for non/limited breastfed children versus 305.0 pmol/g lipid for breastfed children (p-value = 0.04). Among older children, the mean concentration for the sum of lower brominated congeners was 202.1 pmol/g lipid for non/limited breastfed children versus 440 pmol/g lipid for breastfed children (p-value = 0.07). We observed a significant interaction between breastfeeding and child's age for the individual congeners BDE-28 through BDE-153. For the group of children who had no/limited breastfeeding, the slope for age was negative ($\beta = -0.73$, p-value = 0.003), while for the breastfed group, the slope was positive ($\beta = 0.20$, p-value = 0.14), (Supporting Information Table S3, Figure S1 and Figure S2). Duration of breastfeeding was not significantly associated with any individual congeners or sums of congeners. Finally, we observed differences in the individual congener contributions to the sum of lower brominated congeners, with the contribution from BDE-153 higher in the breastfed than in the non/limited breastfed children (16% versus 11% p-value = 0.06), and the contribution from BDE-47 lower (46% versus 50% p-value = 0.03).

Food frequencies (number of eating occasions per week) were reported for the period six months prior to sample collection. Increased consumption of processed meat was associated with small increases in concentrations of the sum of lower brominated congeners, but did not reach statistical significance at 0.05. However, pork consumption was positively associated with the ΣPBDE197,207,209 ($\beta = .27$, $p = 0.01$) (Table 2). The individual congeners, BDE-197, BDE-207 and BDE-209 were also significantly associated with pork consumption. Poultry consumption was associated with higher concentrations of the sum of tri through hexa-BDEs ($\beta = .10$, $p = .05$). PBDEs were not associated with consumption of dairy products or fish.

Discussion

In comparison to similar-aged populations (portions of Table 1), children in our sample had PBDE concentrations two to nine times the levels recently reported for 20 mother-child pairs across the U.S. Levels in our northern California sample were four to nine times the concentrations in children from Mexico, and one to two orders of magnitude higher than concentrations measured in children from Spain and the Faroe Islands. Remarkably, PBDE levels found in our sample of children, approach or exceed levels measured in occupationally exposed U.S. adults working as polyurethane foam recyclers and carpet installers [37].

In 244 children from Menorca, Spain, levels of BDE-47 and BDE-99 in four-year-old children were higher among breast-fed compared to formula-fed children [18]. We found significantly lower levels of BDE-47 in breastfed versus formula-fed children in our sample, but only among younger children, with the relationship reversed for older children. PBDE concentrations decreased with age among the non/limited-breastfed group. This is consistent with the recent report by Toms and colleagues, who used pooled blood samples to show that when children ages 0–4 are stratified by 6-month age intervals, PBDE concentrations peak at around 2.6 to 3 years of age, and then begin to decline [38]. For the breastfed group however, we saw a positive slope, such that concentrations increased with age, suggesting that breastfed children may reach peak PBDE concentrations at a later age than children who are not breastfed.

We observed a larger contribution to total PBDE concentrations from BDE-153 in children who were breastfed, compared with children who were primarily formula-fed. A recent French study found that BDE-153 was the predominant congener in maternal adipose tissue (47% compared to 25% for BDE-47), a much larger contribution from BDE-153 than is found in the commercial Penta mixture, which is typically between 5–8% [39]. In rodents, BDE-153 is resistant to metabolism, which, if also applicable to humans, may explain proportionally higher BDE-153 in human matrices compared to commercial mixtures [40]. In the French study, PBDE concentrations in maternal adipose tissue were strongly correlated with breast milk concentrations [41]. Taken together, these findings may explain the larger proportion of BDE-153 in the breastfed children compared to limited/non-breastfed children, who would have a smaller contribution from maternal adipose stores. However, we did not have PBDE concentrations for mothers. Therefore, while the literature cited suggests that maternal transfer may be a significant source of BDE-153 for infants, we were not able to study correlations between maternal and infant levels in this study population.

In spite of visual aids, parents had difficulty estimating portion sizes. Studies have demonstrated that portion size estimation is highly prone to error, even with visual aids, and estimating toddler intakes is particularly problematic due to spillage and leftovers [42–44]. To avoid reducing our sample size by dropping those who could not recall portion size, we

opted to use the number of eating occasions per week for each food item, which has shown to be an accurate method for evaluating relative intake [45]. Our findings of increased concentrations of lower brominated PBDEs with more frequent poultry consumption, and the lack of association with dairy and fish consumption are concordant with a recent NHANES report [46].

Our finding of an association between newly purchased furniture or mattress and higher levels of Nona and Deca BDEs suggests that these items are a significant source for congeners found in the commercial Deca mixture commonly used in upholstery fabric. This finding supports the hypothesis that stringent flammability standards may have a role in the elevated PBDE levels seen in California human and environmental samples.

Our analysis had several limitations: a small sample size, lack of corresponding maternal and/or house dust samples, and the significant lag time between sample collection and administration of the PBDE questionnaire. Additionally, this pilot was a retrospective study using randomly selected banked samples. As a result, some participants were lost to follow up, while others declined to participate in additional data collection. As we would expect, respondents of the PBDE questionnaire tended to be older and more educated than non-respondents and therefore selection bias is a further limitation of this study.

Recall bias was a potential concern since all reporting was retrospective and this pilot was part of a larger case-control study. However, we found no differences in reporting rates of any significant predictors of PBDEs between cases and controls when we conducted t-tests. For example, *p*-values for group differences in reported food frequencies were 0.34, 0.36, and 0.67 for pork, processed meat, and chicken respectively. In addition, equal proportions of cases and controls reported purchasing a new mattress, and the proportion of college educated mothers was also not different between cases and controls in this group. Finally, this analysis was to assess PBDE exposures and parents were unaware of their child's levels at the time of data collection. Therefore, while errors in reporting are always possible, we believe it is very unlikely that significant recall bias was introduced.

In spite of limitations with this pilot study, this appears to be the first report to evaluate children's body burdens in relation to their diet, indoor environment, and socio-economic characteristics. Poultry, pork and processed meat, as well as new furniture and mattresses, appear to contribute to circulating flame retardants in preschoolers. Future exposure assessments would benefit from the inclusion of matched maternal and dust samples.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

This study was supported by NIEHS (grant numbers 2P01ES001269 and R01ES015359), U.S. EPA (grant numbers R833292 and R829388), the UC Davis M.I.N.D. Institute, and Autism Speaks. We thank the families who volunteered to participate in the CHARGE study.

References

1. Costa LG, Giordano G. Developmental neurotoxicity of polybrominated diphenyl ether (PBDE) flame retardants. Neurotoxicology. 2007; 28(6):1047–1067. [PubMed: 17904639]
2. Abdelouahab N, Suvorov A, Pasquier JC, Langlois MF, Praud JP, Takser L. Thyroid Disruption by Low-Dose BDE-47 in Prenatally Exposed Lambs. Neonatology. 2009; 96(2):120–124. [PubMed: 19321978]

3. Darnerud PO, Aune M, Larsson L, Hallgren S. Plasma PBDE and thyroxine levels in rats exposed to Bromkal or BDE-47. Chemosphere. 2007; 67(9):S386–S392. [PubMed: 17257644]
4. Ellis-Hutchings RG, Cherr GN, Hanna LA, Keen CL. Polybrominated diphenyl ether (PBDE)-induced alterations in vitamin A and thyroid hormone concentrations in the rat during lactation and early postnatal development. Toxicol Appl Pharmacol. 2006; 215(2):135–145. [PubMed: 16580039]
5. Hallgren S, Sinjari T, Hakansson H, Darnerud PO. Effects of polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyls (PCBs) on thyroid hormone and vitamin A levels in rats and mice. Arch Toxicol. 2001; 75(4):200–208. [PubMed: 11482517]
6. Lema SC, Dickey JT, Schultz IR, Swanson P. Dietary exposure to 2,2',4,4'-tetrabromodiphenyl ether (PBDE-47) alters thyroid status and thyroid hormone-regulated gene transcription in the pituitary and brain. Environ Health Perspect. 2008; 116(12):1694–1699. [PubMed: 19079722]
7. Turyk ME, Persky VW, Imm P, Knobeloch L, Chatterton R, Anderson HA. Hormone disruption by PBDEs in adult male sport fish consumers. Environ Health Perspect. 2008; 116(12):1635–1641. [PubMed: 19079713]
8. Viberg H, Mundy W, Eriksson P. Neonatal exposure to decabrominated diphenyl ether (PBDE 209) results in changes in BDNF, CaMKII and GAP-43, biochemical substrates of neuronal survival, growth, and synaptogenesis. Neurotoxicology. 2008; 29(1):152–159. [PubMed: 18061678]
9. Hites RA. Polybrominated Diphenyl Ethers in the Environment and in People: A Meta-Analysis of Concentrations. Environ Sci Technol. 2004; 38(4):12.
10. Frederiksen M, Vorkamp K, Thomsen M, Knudsen LE. Human internal and external exposure to PBDEs--a review of levels and sources. Int J Hyg Environ Health. 2009; 212(2):109–134. [PubMed: 18554980]
11. Harrad S, Ibarra C, Robson M, Melymuk L, Zhang X, Diamond M, Douwes J. Polychlorinated biphenyls in domestic dust from Canada, New Zealand, United Kingdom and United States: implications for human exposure. Chemosphere. 2009; 76(2):232–238. [PubMed: 19356786]
12. Harrad S, Hazrati S, Ibarra C. Concentrations of polychlorinated biphenyls in indoor air and polybrominated diphenyl ethers in indoor air and dust in Birmingham, United Kingdom: implications for human exposure. Environ Sci Technol. 2006; 40(15):4633–4638. [PubMed: 16913117]
13. Zota AR, Rudel RA, Morello-Frosch RA, Brody JG. Elevated house dust and serum concentrations of PBDEs in California: unintended consequences of furniture flammability standards? Environ Sci Technol. 2008; 42(21):8158–8164. [PubMed: 19031918]
14. Fischer D, Hooper K, Athanasiadou M, Athanassiadis I, Bergman A. Children show highest levels of polybrominated diphenyl ethers in a California family of four: a case study. Environ Health Perspect. 2006; 114(10):1581–1584. [PubMed: 17035146]
15. Lunder S, Jacob A. Fire Retardants in Toddlers and Their Mothers. http://www.ewg.org/node/26900.
16. Toms LM, Harden F, Paepke O, Hobson P, Ryan JJ, Mueller JF. Higher accumulation of polybrominated diphenyl ethers in infants than in adults. Environ Sci Technol. 2008; 42(19):7510–7515. [PubMed: 18939594]
17. Johnson-Restrepo B, Kannan K. An assessment of sources and pathways of human exposure to polybrominated diphenyl ethers in the United States. Chemosphere. 2009
18. Carrizo D, Grimalt JO, Ribas-Fito N, Sunyer J, Torrent M. Influence of breastfeeding in the accumulation of polybromodiphenyl ethers during the first years of child growth. Environ Sci Technol. 2007; 41(14):4907–4912. [PubMed: 17711201]
19. LaKind JS, Berlin ChestonM Jr, Sjodin Andreas, Turner Wayman, Wang RichardY, Needham LarryL, Paul IanM, Stokes JenniferL, Naiman DanielQ, Patterson DonaldG Jr. Do Human Milk Concentrations of Persistent Organic Chemicals Really Decline During Lactation? Chemical Concentrations During Lactation and Milk/Serum Partitioning. Environ Health Perspect. 2009
20. Schecter A, Papke O, Lis A, Ball M, Ryan JJ, Olson JR, Li L, Kessler H. Decrease in milk and blood dioxin levels over two years in a mother nursing twins: estimates of decreased maternal and increased infant dioxin body burden from nursing. Chemosphere. 1996; 32(3):543–549. [PubMed: 8907231]

21. Stapleton HM, Kelly SM, Allen JG, McClean MD, Webster TF. Measurement of polybrominated diphenyl ethers on hand wipes: estimating exposure from hand-to-mouth contact. Environ Sci Technol. 2008; 42(9):3329–3334. [PubMed: 18522114]
22. Hertz-Picciotto I, Croen LA, Hansen R, Jones CR, van de Water J, Pessah IN. The CHARGE study: an epidemiologic investigation of genetic and environmental factors contributing to autism. Environ Health Perspect. 2006; 114(7):1119–1125. [PubMed: 16835068]
23. Wu N, Herrmann T, Paepke O, Tickner J, Hale R, Harvey LE, La Guardia M, McClean MD, Webster TF. Human exposure to PBDEs: associations of PBDE body burdens with food consumption and house dust concentrations. Environ Sci Technol. 2007; 41(5):1584–1589. [PubMed: 17396645]
24. Mandalakis M, Stephanou EG, Horii Y, Kannan K. Emerging contaminants in car interiors: evaluating the impact of airborne PBDEs and PBDD/Fs. Environ Sci Technol. 2008; 42(17):6431–6436. [PubMed: 18800511]
25. Lorber M. Exposure of Americans to polybrominated diphenyl ethers. J Expo Sci Environ Epidemiol. 2008; 18(1):2–19. [PubMed: 17426733]
26. Schecter A, Papke O, Harris TR, Tung KC, Musumba A, Olson J, Birnbaum L. Polybrominated diphenyl ether (PBDE) levels in an expanded market basket survey of U.S. food and estimated PBDE dietary intake by age and sex. Environ Health Perspect. 2006; 114(10):1515–1520. [PubMed: 17035135]
27. Hovander L, Athanasiadou M, Asplund L, Jensen S, Wehler EK. Extraction and cleanup methods for analysis of phenolic and neutral organohalogens in plasma. J Anal Toxicol. 2000; 24(8):696–703. [PubMed: 11110024]
28. Christiansson A, Teclechiel D, Eriksson J, Bergman A, Marsh G. Methods for synthesis of nonabromodiphenyl ethers and a chloro-nonabromodiphenyl ether. Chemosphere. 2006; 63(4): 562–569. [PubMed: 16297962]
29. Marsh G, Hu JW, Jakobsson E, Rahm S, Bergman A. Synthesis and characterization of 32 polybrominated diphenyl ethers. Environ Sci Technol. 1999; 33(17):3033–3037.
30. Athanasiadou M, Cuadra SN, Marsh G, Bergman A, Jakobsson K. Polybrominated diphenyl ethers (PBDEs) and bioaccumulative hydroxylated PBDE metabolites in young humans from Managua, Nicaragua. Environ Health Perspect. 2008; 116(3):400–408. [PubMed: 18335110]
31. Bjorklund J, Tollback P, Ostman C. Large volume injection GC-MS in electron capture negative ion mode utilizing isotopic dilution for the determination of polybrominated diphenyl ethers in air. Journal of Separation Science. 2003; 26(12–13):1104–1110.
32. Sjodin A, Wong LY, Jones RS, Park A, Zhang Y, Hodge C, Dipietro E, McClure C, Turner W, Needham LL, Patterson DG Jr. Serum concentrations of polybrominated diphenyl ethers (PBDEs) and polybrominated biphenyl (PBB) in the United States population: 2003–2004. Environ Sci Technol. 2008; 42(4):1377–1384. [PubMed: 18351120]
33. Phillips DL, Pirkle JL, Burse VW, Bernert JT Jr, Henderson LO, Needham LL. Chlorinated hydrocarbon levels in human serum: effects of fasting and feeding. Arch Environ Contam Toxicol. 1989; 18(4):495–500. [PubMed: 2505694]
34. Alaee M, Arias Pedro, Sjodin Andreas, Bergman Ake. An overview of commercially used brominated flame retardants, their applications, their use patterns in different countries/regions and possible modes of release. Environment International. 2003; 29:683–689. [PubMed: 12850087]
35. Stapleton HM, Dodder NG, Offenberg JH, Schantz MM, Wise SA. Polybrominated diphenyl ethers in house dust and clothes dryer lint. Environ Sci Technol. 2005; 39(4):925–931. [PubMed: 15773463]
36. Zhang X, Diamond MiriamL, Ibarra Catalina, Harrad Stuart. Multimedia Modeling of Polybrominated Diphenyl Ether Emissions and Fate Indoords. Environ Sci Technol. 2009; 48(8): 2845–2850. [PubMed: 19475960]
37. Stapleton HM, Sjodin A, Jones RS, Niehuser S, Zhang Y, Patterson DG Jr. Serum levels of polybrominated diphenyl ethers (PBDEs) in foam recyclers and carpet installers working in the United States. Environ Sci Technol. 2008; 42(9):3453–3458. [PubMed: 18522133]

38. Toms L-ML, Sjodin Andreas, Harden Fiona, Hobson Peter, Jones Richard, Edenfield Emily, Mueller JochenF. Serum Polybrominated Diphenyl Ether (PBDE) levels are higher in children (ages 2 to 5 years) than in infants and adults. Environ Health Perspect. 2009

39. La Guardia M, Hale RobertC, Harvey Ellen. Detailed Polybrominated Diphenyl Ether (PBDE) Congener Composition of the Widely Used Penta-, Octa-, and Deca-PBDE Technical Mixtures. Environ Sci Technol. 2006; 40:6247–6254. [PubMed: 17120549]

40. Sanders JM, Lebetkin EH, Chen LJ, Burka LT. Disposition of 2,2',4,4',5,5'-hexabromodiphenyl ether (BDE153) and its interaction with other polybrominated diphenyl ethers (PBDEs) in rodents. Xenobiotica. 2006; 36(9):824–837. [PubMed: 16971346]

41. Antignac JP, Cariou R, Zalko D, Berrebi A, Cravedi JP, Maume D, Marchand P, Monteau F, Riu A, Andre F, Le Bizec B. Exposure assessment of French women and their newborn to brominated flame retardants: determination of tri- to deca- polybromodiphenylethers (PBDE) in maternal adipose tissue, serum, breast milk and cord serum. Environ Pollut. 2009; 157(1):164–173. [PubMed: 18804904]

42. Godwin SL, Chambers Et, Cleveland L. Accuracy of reporting dietary intake using various portion-size aids in-person and via telephone. J Am Diet Assoc. 2004; 104(4):585–594. [PubMed: 15054344]

43. Ovaskainen ML, Paturi M, Reinivuo H, Hannila ML, Sinkko H, Lehtisalo J, Pynnonen-Polari O, Mannisto S. Accuracy in the estimation of food servings against the portions in food photographs. Eur J Clin Nutr. 2008; 62(5):674–681. [PubMed: 17440523]

44. Fisher JO, Butte NF, Mendoza PM, Wilson TA, Hodges EA, Reidy KC, Deming D. Overestimation of infant and toddler energy intake by 24-h recall compared with weighed food records. Am J Clin Nutr. 2008; 88(2):407–415. [PubMed: 18689377]

45. Schlundt DG, Buchowski MS, Hargreaves MK, Hankin JH, Signorello LB, Blot WJ. Separate estimates of portion size were not essential for energy and nutrient estimation: results from the Southern Community Cohort food-frequency questionnaire pilot study. Public Health Nutr. 2007; 10(3):245–251. [PubMed: 17288621]

46. Fraser AJ, Webster TF, McClean MD. Diet Contributes Significantly to the Body Burden of PBDEs in the General U.S. Population. Environ Health Perspect. 2009; 117(10):1520–1525. [PubMed: 20019900]

47. Thuresson K, Hoglund P, Hagmar L, Sjodin A, Bergman A, Jakobsson K. Apparent half-lives of hepta- to decabrominated diphenyl ethers in human serum as determined in occupationally exposed workers. Environ Health Perspect. 2006; 114(2):176–181. [PubMed: 16451851]

Table 1

Cross-study Comparison of PBDE concentrations in ng/g lipid

	Children						Children and Adolescents	Adolescents and Adults		Adults	
	This Study California 2–5 yrs (n=94)			Lunder U.S.* 1.5–4 yrs (n=20)	Carrizo Spain 4 yrs (n=244)	Fangstrom Faroe Islands 7 yrs (n=42)	Perez-Maldonado Mexico Age 6–13 (n=173)	Sjodin U.S.* Age 12->60 (n=2062)	Zota California Age 12-≥ 60 (n=276)	Bradman California Adults (n=24)	Lunder U.S.* Adults (n=20)
Time of Sample Collection	2003–2005			2006–2007	2001–2002	2000–2001	2006	2003–2004	2003–2004	1999–2001	2006–2007
	Mean +SD ng/g lipid	Median ng/g lipid	GM +GSD ng/g lipid	Median ng/g lipid	Mean ng/g lipid	Median ng/g lipid	GM ng/g lipid	GM ng/g lipid	GM ng/g lipid	Median ng/g lipid	Median ng/g lipid
BDE-28	1.5 ± 1.2	1.2	1.1 ± 2.3	1.0	ND			1.2	2.1		0.4
BDE-47	89.8 ± 78.0	69.5	61.9 ± 2.6	30.6	2.9	0.87	7.12	20.5	36.2	11	8.8
BDE-66	0.73 ± 0.69	0.50	0.49 ± 2.6		ND						
BDE-85	5.3 ± 5.8	3.5	3.2 ± 2.9	0.4	ND						ND
BDE-99	41.4 ± 44.9	28.1	25.0 ± 2.9	6.2	1.2	0.47	6.0	5.0	7.4	2.9	1.5
BDE-100	32.3 ± 30.3	25.8	21.4 ± 2.7	6.2	0.18	0.14	3.79	3.9	6.0	1.8	1.2
BDE-153	30.2 ± 26.7	20.8	20.4 ± 2.6	12.5	0.021	2.5	4.46	5.7	6.8	1.5	5.8
BDE-183	1.1 ± 3.7	0.63	0.66 ± 1.96	ND	ND		ND			<0.1	ND
BDE-197	1.9± 2.6	1.3	1.3 ± 2.4	0.5							0.3
BDE-207	2.6 ± 2.4	2.0	2.1 ± 1.8								
BDE-209	4.4 ± 6.2	2.6	2.9 ± 2.4	1.7		1.0					ND

Table 2

Factors associated with PBDE concentrations in children. Regression coefficients are reported in log transformed pmol/g lipid. Results are unadjusted and adjusted for child's age, breastfeeding, and maternal education (significant associations are in bold).

		Unadjusted			Adjusted			
PBDE pmol/g lipid	**Predictor**	**β Estimate**	**Standard Error**	**p-value**	**β Estimate**	**Standard Error**	**p-value**	**N**
ΣBDE197-209								
	Pork[a]	**0.27**	0.10	0.01	**0.25**	0.11	0.02	50
	New Mattress/Furniture[b]	**0.50**	0.20	0.02	**0.20**	0.29	0.05	50
	Home size[c]	−0.20	0.11	0.08	−0.18	0.11	0.11	71
ΣBDE28-153								
	Processed meat[a]	0.12	0.06	0.06	0.06	0.07	0.39	50
	Poultry[a]	**0.10**	0.05	0.05	0.10	0.05	0.06	50

[a] measured in average number of eating occasions per week in the six months prior to blood sample collection.

[b] Purchased either a new mattress or upholstered furniture in the six months prior to blood sample collection.

[c] measured as total square meters/100