Polybrominated diphenyl ethers (PBDEs) have been increasingly used as additive brominated flame retardants (BFRs) ever since the 1970s (Alaee et al. 2003; Bergman 2005; World Health Organization 1997) up to the point that two of the commercial BFR products—pentaBDE and octaBDE—were banned or withdrawn from the market, as was the case for these products within the European Union (Cox and Efthymiou 2003) and in North America (Great Lakes Flame Retardants 2005). PBDEs have been used and are still used in textiles and in flexible polyurethane foams, as well as in electric appliances and electronic devices. Although used textiles and foams are discarded directly without any restrictions, electronic waste may either be discarded or recycled for valuable metals (Cui and Forssberg 2003). Uncontrolled discharges of material containing BFRs may lead to unintentionally high environmental exposure to these chemicals.

Humans are exposed to PBDEs via ingestion and inhalation. A large number of PBDE congeners have been reported to be present in food as reported by, for example, Schecter et al. (2006), with seafood being an important source. PBDEs are present in ambient air both in industrialized regions (Butt et al. 2004; ter Schure et al. 2004) and in the Arctic (de Wit et al. 2006) as well as in household dust (Schecter et al. 2005a; Stapleton et al. 2005; Wu et al. 2007). Thus the dominant PBDE exposure routes differ from those of traditional persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs). This difference appears to be attributable to numerous indoor sources for the former, but much less for the PCBs.

Most data on human PBDE exposure originate in Europe and in North America, as reviewed a few years ago (Gill et al. 2004; Hites 2004; Sjödin et al. 2003). Several more recent publications on PBDEs in humans have appeared from the United States (Bradman et al. 2007; Fischer et al. 2006; Lunder and Sharp 2004; Morland et al. 2005; Schecter et al. 2005b; She et al. 2007) and from Europe, including two-time trend studies (Covaci and Voorrips 2005; Fångström et al. 2005, 2008; Furst 2006; Harrad et al. 2004; Ingelido et al. 2007; Jaraczewska et al. 2006; Schuhmacher et al. 2007; Thomas et al. 2006; Thomesen et al. 2007; Thureson et al. 2005).

Several reports on human PBDE exposure outside of both Europe and the United States have also been published recently. Such data have been obtained from China (Bi et al. 2006), Taiwan (Chao et al. 2007), Korea (Kim et al. 2005; Lee et al. 2007), Japan (Esami et al. 2006; Inoue et al. 2006), Asia in general (Sudaryanto et al. 2005), Australia (Toms et al. 2007), New Zealand (Harrad and Porter 2007), and the Republic of Buryatia (Russia) (Tsydenova et al. 2007). It is clear that the PBDE contamination in North America is higher than in Europe and other parts of the world. Still, it is notable to observe concentrations of BDE-47 in the range of 2.8–9.6 ng/g fat, and a median of the sum of six individual PBDEs (Σ6PBDEs) = 10.2 ng/g fat in Australian human milk (Toms et al. 2007), and individuals with clearly elevated levels in New Zealand, with Σ6PBDE concentrations in blood serum of 2.8–20.1 ng/g fat (Harrad and Porter 2007). The Σ6PBDE concentrations in Canada of 2.4–22 ng/g mother’s milk fat (Hites 2004) and up to 38 ng/g plasma fat in humans in Mexico (Lopez et al. 2004) indicate possible similarity to the United States (Perreaud et al. 2003) in the use and distribution of PBDEs in these neighboring countries. Information from Latin America is still lacking, except for one report from Mexico (Lopez et al. 2004). There are no data from Africa, except one small breast milk study (Darnaud et al. 2006), and rarely data from most parts of Asia.

Data on exposure levels in children are extremely scarce. Notably, the few studies published on PBDE exposure indicate higher concentrations in children than in adults (Fischer et al. 2006; Thomsen et al. 2002).
Exposure to POPs is also a matter of exposure to their metabolites. Experimentally it has been shown that PBDEs are transformed into hydroxylated metabolites (OH-PBDEs) (Hakk and Letcher 2003) via routes similar to those for PCB congeners (Letcher et al. 2000). The metabolism of BDE-47 in mice and rats (Orr and Klasson Wehler 1998) generates tri- and tetrabrominated hydroxylated metabolites, of which six OH-tetraBDEs and three OH-triBDEs have been identified structurally in rats (Marsh et al. 2006). BDE-99 has been found to be metabolized to two pentabrominated and two tetrabrominated hydroxylated diphenyl ethers in the rat (Hakk et al. 2002), and BDE-100 has generated five OH-pentaBDEs and six OH-tetraBDEs in the rat (Hakk et al. 2006). A mixture of seven environmentally relevant PBDEs has been shown to form up to 16 OH-PBDEs that are retained in rat blood (Malmberg et al. 2005). Hydroxylated metabolites of PBDEs have also been reported to be retained in mouse plasma after exposure to a commercial pentaBDE product (Qiu et al. 2007). OH-PBDEs have been identified as metabolites of BDE-47, 99, 100, and 153 in urine and feces from female mice (Chen et al. 2006; Sraskal et al. 2006). It is likely that OH-PBDEs, which have structural similarities with thyroid hormones, also are retained in human blood.

Hydroxylated PBDEs substituted with the hydroxyl group in an ortho position to the ether bridge appear to be primarily of natural origin (Malmvärn et al. 2005; Marsh et al. 2004), and have been identified structurally in biota from marine environments. There are thus two sources of OH-PBDEs in the environment: natural production in marine ecosystems, and anthropogenic formation via the metabolism of PBDEs.

The aim of the present study was to assess human exposure to PBDEs in Nicaragua and, if high concentrations were detected, also to search for OH-PBDE metabolites in their blood. The study groups included potentially highly exposed children working as scrap scavengers at a large municipal waste disposal site. Also, we investigated women from the urban and rural areas of Managua with varying consumption of potentially contaminated fish from the nearby lake. The study, which was performed in 2002, was part of a program investigating exposure to organohalogenics (Cuadra et al. 2006), heavy metals (Cuadra 2005), and respiratory irritants, as well as work-related injuries and respiratory disease in child workers at the waste disposal site.

### Material and Methods

**Setting.** Managua, the capital of Nicaragua, is situated on the shore of Lake Managua (Figure 1). The lake, the second largest in Nicaragua, serves as the recipient of domestic and industrial wastewater from the city, and receives the superficial run-off from its drainage basin, which is intensively cultivated. Fish from the lake are an important part of the diet, not only for the population living in the rural fishing villages, but also for parts of the Managua urban population.

The main municipal domestic and industrial waste disposal site in Managua, La Chureca, which covers an area of 7 km², is located directly on the south shore of the lake. Approximately 1,000 persons work regularly at the waste disposal site, collecting recyclable waste for selling. More than 50% of those workers are children < 18 years of age. A thick cloud of smoke covers the area as the waste is burned to retrieve iron and other materials. Electronic waste is rarely found at the dump site. The waste is not compressed, the sunlight is intense, and a constant breeze from the lakeside sweeps the area. Thus, substantial amounts of airborne dust are generated.

**Study groups.** The ethics committees at Lund University and The National Autonomous University of Nicaragua–Managua (UNAN–Managua) approved the study protocol, and written informed consent was obtained from the participants, and, for children, guardians.

All children currently working at the waste disposal site were identified with the help of a local nongovernmental organization, Centro Dos Generaciones, which maintains an updated register of child workers and their families within the framework of a special program to prevent and eradicate child labor, which has existed for many years. The study of organohalogen exposure was restricted to those in the age range of 11–15 years, in all 64 children working at the waste disposal site and living there or in a nearby area, Acahualinca. With the help of the local school, we selected 80 referent children from Acahualinca. We also recruited a remote reference group, 18 children living 10–20 km away from the waste disposal site in the south and central areas of Managua, by help of the Chatelteles project, a local nongovernmental organization. None of the referent children had a history of current or previous work at the waste disposal site. All children included shared the same poor socioeconomic conditions. The distribution of age and sex was also similar in the different groups. There was a participation rate of 90% for those working at the waste disposal site, 70% for referents living nearby, and 100% for the remote reference group. The blood sampling took place in May and June 2002.

Detailed information on work history and dietary habits, especially fish consumption, was obtained by a structured interview. Blood was drawn from the cubital vein into evacuated plain tubes (Vacutainer, Rutherford, NJ,

![Image](https://example.com/map.png)

**Figure 1.** Map of the Managua area, Nicaragua. The locations of Mateare, San Francisco Libre, urban Managua, and the waste disposal site “La Chureca” are shown.
Abbreviations: CAS, currently attending school; F, female; M, male. Serum levels of PCBs are also given [data from present study and Cuadra et al. (2006)].

According to criteria described below, children were selected and stratified in terms of their work experience at the waste disposal site, the area where they were living, and their fish consumption. Five distinct groups for pooling of serum were identified (Table 1):

- **Pool 1**: children living at the waste disposal site, who had worked there for 4–10 years (median, 6 years); half of the children had been living at the dump all their life, the other half for 5–11 years; pool 2: children living in the nearby area, Acahualinca, who had worked at the waste disposal site for 4–12 years or more (median, 6 years); pools 3 and 4: children living in Acahualinca but not working at the dump; pool 5: children living in a remote urban area.

Women with markedly different patterns of fish consumption, residing in poor urban and rural districts in the Managua area, were also included in the study with the help of local authorities, local health centers, and fishermen’s cooperatives—in all, 32 women. There were sufficient quantities of serum to assemble another four distinct pools for analysis: pool A: women 15–17 years of age, living in fishermen’s families in San Francisco Libre, a fishing village on the rural northeast side of the lake; pool B: women 20–29 years of age living in fishermen’s families in Mateare, another fishing village 25 km from urban Managua; pool C: women from urban Managua 18–25 years of age; pool D: women from urban Managua 42–44 years of age. The urban women consumed either no fish or only a negligible amount. All the subjects lived under similar underprivileged socioeconomic conditions. Demographic data are presented in Table 1. The blood sampling was performed in July 2002.

Table 1. Sociodemographic characteristics for children working at a waste disposal site (WDS) in Managua (pools 1–2) and referents (pools 3–5), and for females with varying consumption of fish from Lake Managua (pools A–D).

| Group | No. (CAS) | Sex no. (M/F) | Age* (years) | Fish consumption (meals/month)* | Domicile: years duration in dwelling* | Location | Work at WDS: age of onset* | Work at WDS: years worked* | Work at WDS: Work at WDS: Work at WDS: Work at WDS: | ΣPCBs (ng/g l.w.) |
|-------|-----------|---------------|--------------|---------------------------------|----------------------------------|----------|------------------------|-------------------|-------------------|-------------------|-------------------|-------------------|
| Pool 1a | 8 (7) | 4 (4) | 14 (13–15) | 1.5 (0–8) | 12 (5–16) | WDS | 7.5 (6–10) | 6 (4–10) | 3.5 (2–12) | 5.5 (2–7) | — |
| Pool 1b | 11 (8) | 6 (5) | 14 (13–15) | 2 (8–8) | 13 (5–16) | WDS | 7 (6–10) | 6 (4–10) | 4 (2–12) | 6 (2–7) | 540 |
| Pool 2a | 21 (15) | 15 (6) | 14 (12–15) | 2 (8–8) | 13 (7–15) | Acahualinca | 8 (2–10) | 6 (4–12) | 5 (3–12) | 3 (1–7) | — |
| Pool 2b | 23 (17) | 16 (7) | 14 (12–15) | 2 (8–8) | 13 (7–15) | Acahualinca | 8 (2–11) | 6 (4–12) | 5 (3–12) | 3 (1–7) | 530 |
| Pool 3a | 15 (15) | 5 (10) | 14 (13–15) | 2 (8–8) | 13 (6–15) | Acahualinca | 8 (2–10) | 6 (4–12) | 5 (3–12) | 3 (1–7) | — |
| Pool 3b | 16 (16) | 16 (10) | 14 (11–15) | 2 (8–8) | 13 (6–15) | Acahualinca | 8 (2–10) | 6 (4–12) | 5 (3–12) | 3 (1–7) | — |
| Pool 4a | 8 (8) | 3 (5) | 13 (13–15) | 0 | 13 (6–14) | Acahualinca | — | — | — | — | 390 |
| Pool 4b | 10 (10) | 5 (5) | 14 (13–15) | 0 | 13 (6–14) | Acahualinca | — | — | — | — | 230 |
| Pool 5a | 8 (8) | 4 (4) | 13 (12–14) | 0 | 11.5 (10–14) | Urban Managua | — | — | — | — | — |
| Pool 5b | 11 (11) | 5 (6) | 13 (12–14) | 0 | 12 (10–14) | Urban Managua | — | — | — | — | 160 |
| Pool A | 5 | 5 | 15–17 | 4 (4–8) | 8 (7–17) | Fishing village | — | — | — | — | 286 |
| Pool B | 3 (3) | 20–20 | 8 (8–16) | 2 (2–8) | 476 | Fishing village | — | — | — | — | 476 |
| Pool C | 3 | 18–25 | 0 (0–0) | 10 (9–11) | Urban Managua | — | — | — | — | 176 |

Abbreviations: CAS, currently attending school; F, female; M, male. Serum levels of PCBs are also given [data from present study and Cuadra et al. (2006)].

*Median (range). †Samples with serum enough for duplicate analyses; all subjects also included in pool b. ‡Pool b used by Cuadra et al. (2006).
release the lipids and the organohalogen compounds. The analytes were extracted by cyclohexane and methyl tert-butyl ether (1:1). The organic phase was washed with a solution of potassium chloride (1%). The lipids were determined gravimetrically by constant weight determination.

Separation of neutral and phenolic compounds. The analytes and lipids were dissolved in cyclohexane. Potassium hydroxide (0.5 M) was added for partitioning the neutral and acidic analytes. The isolated alkaline solution was acidified with hydrochloric acid and the protonated phenols were extracted with cyclohexane:methyl tert-butyl ether (9:1). The solvent volume was reduced to approximately 0.5 mL before derivatization of the halogenated phenols with diazomethane.

Derivatization-rate test. Seven OH-PBDEs were used for determining the rate of methylation. Diazomethane was added to five test tubes containing exactly the same amount of each of the seven compounds. Methylation was interrupted after 30 min, 1 hr, 3 hr, and 5 hr, the additional test tube being left overnight. The methylation was stopped by reducing the excess of diazomethane under a gentle stream of nitrogen. An injection standard was added before analysis on GC/MS (ECNI). It was shown that 30 min was sufficient for quantitative methylation.

Lipid removal. The neutral compounds and the methoxylated derivatives of the phenolic compounds were isolated free from the lipids after the samples, containing the neutral and the methylated phenols dissolved in cyclohexane, had been treated with concentrated sulfuric acid.

Silica/sulfuric acid clean up. Pasteur pipette columns were packed with silica gel (0.1 g) and silica/sulfuric acid gel (2:1; 1 g) for cleanup of both the neutral and phenolic fraction. The columns were pre-washed and the analytes similarly eluted with cyclohexane:DCM (dichloromethane) (1:1, 6 mL) and DCM (10 mL).

Silica column. Because the sample volumes were significantly reduced (50 µL–200 µL) before GC/MS analysis, an additional silica column was used to remove any additional impurities in the neutral and the methylated phenol fractions. Two fractions were collected from the silica column: a first fraction in cyclohexane (3 mL) and a second fraction in hexane and methyl tert-butyl ether (1:1).

**Table 2.** Concentrations (pmol/g l.w.) of individual PBDE congeners (molecular weight) in children working at a waste disposal site in Managua (pools 1–2) and referents (pools 3–5), and for females with varying consumption of fish from Lake Managua (pools A–D).

| Lipid (%) | BDE-28 | BDE-47 | BDE-66 | BDE-100 | BDE-154 | BDE-153 | BDE-183 | BDE-209 | ΣPBDEa |
|-----------|--------|--------|--------|---------|---------|---------|---------|---------|--------|
| Pool 1ab  | 0.41/0.41 | 24/22 | 680/600 | 12/11 | 111/108 | 309/308 | 35/35 | 22/20 | 47/45 |
| Pool 1bc  | 0.43 | 16 | 450 | 7.5 | 71 | 210 | 20 | 18 | 21 |
| Pool 2ab  | 0.38/0.37 | 2.4/1.7 | 73/66 | 0.66/0.66 | 17/19 | 19/20 | 1.8/2.2 | 5.1/10 | 118/135 |
| Pool 2bc  | 0.40 | 1.7 | 57 | 0.51 | 16 | 18 | 1.8 | 9.2 | 15 |
| Pool 3ab  | 0.39/0.41 | 1.4/1.1 | 30/28 | 0.63/0.61 | 8.4/6.2 | 12/8.9 | 1.1/0.69 | 4.3/3.0 | 5.3/3.7 |
| Pool 3bc  | 0.41 | 0.87 | 29 | 0.66 | 6.6 | 9.5 | 1.1 | 2.8 | 4.5 |
| Pool 4ab  | 0.44/0.42 | 0.6/0.6 | 11/11 | 0.29/0.30 | 2.0/2.1 | 4.2/5.0 | 0.33/0.26 | 1.8/3.5 | 2.1/2.3 |
| Pool 4bc  | 0.41 | 0.73 | 18 | 0.60 | 5.9 | 1.5 | 0.56 | 2.2 | 3.2 |
| Pool 5ab  | 0.41/0.41 | 0.8/0.8 | 14/15 | 0.40/0.41 | 3.3/3.6 | 6.1/8.9 | 0.51/0.52 | 2.9/3.1 | 2.6/2.7 |
| Pool 5bc  | 0.22 | 0.69 | 14 | 0.49 | 3.1 | 6.0 | 0.56 | 2.2 | 2.5 |
| Pool A    | 0.35 | 0.83 | 20 | 0.47 | 4.2 | 11 | 0.70 | 2.0 | 2.2 |
| Pool B    | 0.51 | 0.38 | 14 | < LOQ | 3.3 | 3.7 | 0.28 | 1.8 | 1.6 |
| Pool C    | 0.41 | 3.6 | 86 | 0.67 | 16 | 14 | 1.7 | 2.0 | 12 |
| Pool D    | 0.71/0.66 | 1.9/1.9 | 63/73 | 1.1/1.3 | 10/12 | 25/29 | 2.1/3.3 | 2.5/3.1 | 4.6/6.2 |
| LOQa      | 0.56 | 0.36 | 0.46 | 0.40 | 0.48 | 0.40 | 0.46 | 0.88 | 0.84 |

**Table 3.** Concentrations (pmol/g l.w.) of methyl derivatives of individual OH-PBDEs (molecular weight) in children working at a waste disposal site (WDS) in Managua (pools 1–2) and referents (pools 3–5), and for females with varying consumption of fish from Lake Managua (pools A–D).

| Lipids (%) | 4’-OH-BDE-17 (436) | 6-OH-BDE-47 (516) | 3-OH-BDE-47 (516) | 4’-OH-BDE-49 (516) | 4-OH-BDE-42 (516) | 4-OH-BDE-90 (594) | ΣOH-PBDEsa |
|-----------|-------------------|-------------------|-------------------|-------------------|-------------------|-------------------|--------|
| Pool 1ab  | 0.41/0.41 | 17/16 | 12/13 | 8.5/4.5 | 18/14 | 9.6/6.4 | 54/45 |
| Pool 1bc  | 0.43 | 2.5 | 11 | 2.2 | 4.4 | 4.4 |
| Pool 2ab  | 0.38 | 1.3 | 4.7 | 0.83 | 0.56 | 0.56 | 0.48 |
| Pool 2bc  | 0.40 | 0.58 | 4.5 | 0.33 | 0.76 | 0.84 | 1.1 |
| Pool 3ab  | 0.39 | 0.76 | 2.0 | 0.32 | 0.96 | 0.32 | 0.70 |
| Pool 3bc  | 0.41 | 0.27 | 2.4 | 0.21 | 0.28 | < LOQ | 0.50 |
| Pool 4ab  | 0.44 | 0.44 | 1.5 | < LOQ | 0.37 | < LOQ | 3.1 |
| Pool 4bc  | 0.41 | 0.31 | 2.2 | < LOQ | 0.31 | < LOQ | 3.7 |
| Pool 5ab  | 0.41 | 0.55 | 1.6 | < LOQ | 0.47 | < LOQ | 2.7 |
| Pool 5bc  | 0.41 | 0.50 | 1.6 | < LOQ | 0.53 | < LOQ | 3.4 |
| Pool A    | 0.51 | 0.36 | 1.6 | < LOQ | 0.92 | 0.31 | 0.93 |
| Pool B    | 0.35 | < LOQ | 2.5 | < LOQ | 0.66 | < LOQ | 0.66 |
| Pool C    | 0.41 | < LOQ | 4.0 | 1.3 | 0.79 | 0.79 | 0.96 |
| Pool D    | 0.71/0.66 | 0.97/0.43 | 3.8/3.7 | 0.44/0.67 | 1.3/0.97 | 0.33/0.42 | 1.5/1.5 |
| LOQd      | 8 | 7 | 7 | 9 | 10 | 7 |
| LOQa      | 0.27 | 0.20 | 0.20 | 0.26 | 0.29 | 0.18 |

Only pool 1a and pool D were analyzed in duplicate.

*Sum of the OH-PBDEs presented in the table. **All subjects also included in pool b. ***Pool b as in Cuadra et al. (2006). **ILD provided with corresponding reference compound 3 times the signal-to-noise ratio expressed in femtograms (fg). ***A concentration of 3 times ILOD in a total sample volume of 100 µL (prior analysis) and 20 mg lipids (pmol/g l.w.).
pool compositions (a versus b in Tables 1–3) between samples involving slightly differing agreement with each other, the differences duplicate samples were generally in close agreement. Nine were quantifiable (Table 2). The results for duplicate samples were in close agreement. Nine OH-PBDE congeners were somewhat higher in those eating fish from the lake than in those who did not eat fish. However, the urban women with little or no fish consumption clearly had higher PBDE levels than the women living in rural fishing villages in the Managua area, who consumed large amounts of fish from the lake.

The children working and living at the city dump (pools 1a and 1b) had the highest levels of PBDEs, approximately 20–50 times as high as those of the referent children in urban Managua (pools 3, 4, and 5) when one considers PBDEs with up to six bromine atoms. In contrast, the difference between the groups was minimal for BDE-183 and BDE-209 (Table 2).

In the urban children who did not work at the waste disposal site, the levels of the sum of the PBDEs as well as of several individual congeners were somewhat higher in those eating fish from the lake than in those who did not eat fish. However, the urban women with little or no fish consumption clearly had higher PBDE levels than the women living in rural fishing villages in the Managua area, who consumed large amounts of fish from the lake.

The OH-PBDEs were analyzed as methyl derivatives (MeO-PBDEs). A chromatogram of methyl ethers of the OH-PBDEs in the children working and living at the dump is shown in Figure 3, confirming the presence of 19 OH-PBDEs in human plasma. Six of the OH-PBDEs were tentatively identified in terms of their retention times compared with those of authentic reference compounds that were available, and in terms of their signal for m/z = 79/81 in bromide ion trace analysis (ECNI). The chromatographic peak patterns were also compared with those found in a chromatogram, analyzed in parallel, of a plasma sample obtained from rats that had been exposed to a mixture of seven equimolar PBDE congeners (BDEs 47, 99, 100, 153, 154, 183, and 209) (Malmberg et al. 2005). The position of the hydroxyl group could be identified in 17 mono-hydroxylated PBDEs (10 meta, 6 para, and 1 ortho hydroxylated PBDEs) according to OH-PBDE mass fragmentation (electron ionization) (Athanasiadou et al. 2006). The six OH-PBDEs that were tentatively identified were quantified (Table 3). Results for duplicate samples were in close agreement. Nine OH-PBDEs (each marked with an asterisk (*)) were identified as being the same metabolites that were formed in the experimental study where rats were exposed to an artificial PBDE mixture (Malmberg et al. 2005), but not yet structurally identified due to a lack of appropriate standards.

Of the metabolites having three to four bromines, it appeared that 4-OH-BDE-17 and 4’-OH-BDE-49 were the dominant phenolic metabolites. The peak marked 4-OH-BDE-90 is a double peak determined by two pentabrominated hydroxylated diphenyl ethers. The peak was quantified relative to the 4-OH-BDE-90 standard.

The pools obtained from children living and working at the dump showed a markedly higher serum level of hydroxylated metabolites than found in any of the other pools (Table 3).

**Discussion**

The occupational exposure to PBDEs in children working at a waste disposal site, and a substantial background level of exposure to...
PBDEs in the urban population, are clearly shown. The PBDE concentrations in the serum of children working and living at the waste disposal site are among the highest ever reported. Most interesting, we found that hydroxylated PBDE metabolites are retained in human serum, just like OH-PVCs.

**Design considerations.** We used pooled samples to keep the amount of blood sampled as low as possible in this population of malnourished children but also for economic reasons not allowing us to analyze individual samples for PBDEs. The result from a pooled sample is equivalent to the mean for the group in question, but its weakness is that there is no information concerning the variation between individuals. To partially overcome this limitation, we selected the subjects to be included very carefully, so that the separate pools would clearly differ with regard to the factors we wished to investigate (the effects of work at the waste disposal site, the area of living involved, and fish consumption), but as homogeneous as possible within each pool. The children at the waste disposal site who were enrolled represent most of the child workers of their age. As referred, we enrolled children not involved in scrap scavenging, but living in the same area, and children from low-income families living far away from the waste disposal site. The body mass index, blood iron content, and blood lipid concentration found in the child workers and the referents were comparable (Cuadra et al. 2005). A previous study of 103 scrap scavengers and 103 referent children from Acahualinca 6–15 years of age (the children selected in our study being a subgroup of these children) had indicated similar prevalence of chronic malnutrition (height for age below the 50th percentile, obtained from Epi-Info: Nut-stat 2nd revision; Centers for Disease Control and Prevention 2007) and acute malnutrition (weight per age below the 50th percentile, obtained from Epi-Info: Nut-stat 2nd revision) in child workers and referents—around 85% prevalence for chronic and around 65% for acute malnutrition (Hernández Romero D, personal communication). Thus, there are indications that the desired socioeconomic similarity of the groups, including nutritional status, was achieved.

**Exposure to polybrominated diphenyl ethers.** The present data from Nicaragua should be put in context of PBDE exposures elsewhere, as illustrated in Figures 4–6, in which observations from different areas worldwide are presented. Unexpectedly, the children living and working at the waste disposal site had very high levels of medium BDEs, with the group mean being almost an order of magnitude higher than hitherto reported. In all the pools, BDE-47 was the dominant PBDE congener, followed by BDE-99, BDE-100, and BDE-153. Thus, the congener profile suggests exposure mainly to the technical pentabDE product (La Guardia et al. 2006). This in accordance with the absence of electronic waste at the waste disposal site.

Also, quite unexpectedly, the levels of the medium BDEs observed among referent children and young and middle-age women living in an unindustrialized urban area in the second poorest country in the Americas were comparable to contemporary observations in the United States (Bradman et al. 2007; Fischer et al. 2006; Hites 2004; Lunder and Sharp 2004; Morland et al. 2005; Schecter et al. 2005b; She et al. 2007), and much higher than the concentrations reported in the United States (Bradman et al. 2007; Fischer et al. 2006; Hites 2004; Lunder and Sharp 2004; Morland et al. 2005; Schecter et al. 2005b; She et al. 2007), and much higher than recent exposure only, because the congener has a short half-life in serum, approximately 15 days (Thuresson et al. 2006). The PBDE-209 concentrations were rather homogeneous between the different pools (Table 2). A similar pattern was also indicated for BDE-183, which has an apparent half-life of approximately 3 months.

Our data clearly support the hypothesis that human exposure to PBDE is strongly influenced by dust inhalation and ingestion, rather than by contamination of food. At the waste disposal site, high levels of dust are generated as the waste is burned. In 2005, the mean levels of particulate matter (PM <2.5 µm aerodynamic diameter) at the waste disposal site were 700 µg/m3, compared with 100 µg/m3 in Acahualinca nearby (Hernandez Romero D, personal communication). We hypothesize that the uncontrolled burning of waste also plays a major role as a source of PBDE exposure in the general population. La Chureca is the major waste disposal site in Managua, but due to deficiences in the local waste management system, many informal small open dumps where wastes

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**Figure 4.** Serum levels of BDE-47 (pmol/g LW) in different regions. Data are from: 1: Thomasen et al. (2007); 2: Thomas et al. (2006); 3: Fängström et al. (2005); 4: Covaci and Voorspoels (2005); 5: Harrad and Porter (2007); 6: Inoue et al. (2005); 7: Kim et al. (2005); 8: Bradman et al. (2007); 9: Schecter et al. (2005b); 10: Lopez et al. (2004). For comparison, reported concentrations were transformed into serum concentrations in pmol/g LW when needed. X̅ = mean; X̂ = median; GM, geometric mean; GSD, standard deviation of GM; WDS, waste disposal site.
are burned are found in the city, and daily backyard burning is common. This also happens in the rural areas, but the burden of waste is far less than in the urban areas. Thus, the situation for PBDEs may be similar to that of dioxins, with uncontrolled burning of waste being a major source of exposure (Hedman et al. 2005).

Elevated levels of PBDEs in soil due to open burning of electronic waste has been reported from China (Leung et al. 2007), and PBDEs have been found in leachates from open waste disposal sites in Canada (Danon-Schaffer et al. 2006) and in Japan (Osako et al. 2004). Also, elevated levels of PBDEs have been reported in breast milk from women living near an open waste site in India (Kunisue et al. 2006).

That the findings for the women differ markedly in their fish consumption indicates that factors linked to urban dwelling were more important for the levels of medium brominated PBDEs than high fish consumption (Table 2, pools A and B vs. pools C and D). We have no detailed dietary information in these subjects, but it is unlikely that their diet, based on rice and beans, differs markedly, except that those in the fishing villages eat more fish and accordingly less meat. Such a marked urban–rural gradient has not been reported previously. In contrast, the levels of PCBs were higher in the women from the rural areas than in the urban women, indicating the importance of dietary exposure through fish consumption (Table 1). A similar pattern was also observed for 4,4'-dichlorodiphenyldichloroethylene (Cuadra et al. 2006). It is notable that the concentrations of both BDE-47 and BDE-99 (~ 600 and 300 pmol/g l.w., respectively) were higher than the level of the dominant PCB congener, CB-153 (195 pmol/g l.w.) in the children working and living at the waste disposal site.

Figure 5. Serum levels of BDE-153 (pmol/g l.w.) in different regions. Data are from: 1: Thomsen et al. (2007); 2: Thuresson et al. (2005); 3: Thomas et al. (2006); 4: Fängström et al. (2005); 5: Covaci and Voorspoels (2005); 6: Inoue et al. (2006); 7: Schecter et al. (2005b); 8: Lopez et al. (2004). For comparison, reported concentrations were transformed into serum concentration in pmol/g l.w. when needed. $\bar{x}$ = mean; $\bar{X}$ = median; GM, geometric mean; GSD, standard deviation of GM; nd, nondetectable; WDS, waste disposal site.

Figure 6. Serum levels of BDE-209 (pmol/g l.w.) in different regions. Data are from: 1: Thomsen et al. (2007); 2: Thuresson et al. (2005); 3: Thomas et al. (2006); 4: Fängström et al. (2005); 5: Covaci and Voorspoels (2005); 6: Inoue et al. (2006); 7: Schecter et al. (2005b); 8: Lopez et al. (2004). For comparison, reported concentrations were transformed into serum concentration in pmol/g l.w. when needed. $\bar{x}$ = mean; $\bar{X}$ = median; GM, geometric mean; GSD, standard deviation of GM; nd, nondetectable; WDS, waste disposal site.
PBDEs and hydroxylated PBDE metabolites in young humans

For the other children, the CB-153 levels were higher by a factor of approximately 10 on a molar level.

Our data clearly indicate that exposure to PBDEs through inhalation is significant in this population, and of greater magnitude than dietary exposure. However, we observed somewhat higher PBDE levels among the nonworking children who ate fish from the lake (pool 3) than in nonconsumers of fish (pools 4 and 5). Thus, fish from the lake may also be a source of exposure to medium BDEs. This is in line with previous findings of a correlation between fish consumption and the level of BDE-47 (Ohta et al. 2002; Cuadra 2005; Cuadra et al. 2006). The need for an integrated risk evaluation is evident when considering the fact that several of the substances have a common critical organ—the developing brain. The levels of exposure to xenobiotics that we have observed are directly relevant to risk assessment with regard to reproductive outcomes. Many of the teenage girls will be mothers in the near future. As many as 21% of the adolescent females in Managua were either pregnant or already mothers in 2001 (INEC 2005). Also, the need to investigate the toxicologic effect of chemical mixtures in connection with malnutrition has been taken up recently. In our population this is clearly relevant, because malnutrition was prevalent.

Conclusions. We studied a vulnerable population of children and adolescents living under extreme socioeconomic conditions in which exposure to multiple hazardous chemicals is evident. The data collected clearly indicate that dust is a significant source of exposure to PBDEs in this population. The unexpected finding of high levels of PBDEs and their OH-PBDE metabolites in urban children in a developing country highlights the need for worldwide exposure assessment of emerging pollutants, not just in the developed countries. This is not only of scientific interest, but indeed a matter of concern for society.

Correction

In the original manuscript published online, BDE-153 and BDE-154 were reversed in position in Figure 2; they have been corrected here.

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