

6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

Polybrominated Biphenyls. PBBs have been identified in at least 9 of the 1,613 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2002). However, the number of sites evaluated for PBBs is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, all are located within the United States.

The production of PBBs in the United States ceased in 1979 (IARC 1986). In the past, PBBs were released to the environment during the manufacture of these compounds and disposal of commercial and consumer products containing these compounds (Hesse and Powers 1978; Neufeld et al. 1977). One of the significant sources of environmental contamination occurred as a result of the accidental mixup of FireMaster BP-6 with cattle food in a number of farms in the lower peninsula in Michigan (see Section 3.2 for additional details concerning this incident). By June 1975, 412 farms had been quarantined. Disposal of contaminated feed, animal carcasses (poultry, dairy cattle, swine), and animal products (dairy, meat, eggs) contributed to environmental contamination (Dunckel 1975; Kay 1977). No information was located on the current levels of contamination at these locations.

PBBs can exist in 209 different congeners, but only about 42 have been synthesized (Sundstrom et al. 1976b). Environmental contamination of PBBs is likely to have occurred mainly from the two commercial products, FireMaster BP-6 and FireMaster FF-1. The principal component in both of these commercial products was 2,2',4,4',5,5'-hexabromobiphenyl or BB-153 (Robertson et al. 1983b).

PBBs are strongly adsorbed to soil and sediment (Filonow et al. 1976; Hesse and Powers 1978) and usually persist in the environment (Jacobs et al. 1978). Adsorption of PBBs generally increases as bromination and organic carbon content in soil and sediment increase (Filonow et al. 1976; Griffin and Chou 1981a, 1981b). As a result, the leaching of commercial mixtures of PBBs from soil is slow. Leaching studies with four Michigan soils, mixed with 100 mg/kg 2,2',4,4',5,5'-hexabromobiphenyl, showed that <0.6% of the compound leached through soils, with leachate quantities equivalent to 20 times the average annual rainfall in Michigan (Filonow et al. 1976). The PBBs in commercial mixtures resist both chemical and biological degradation (Jacobs et al. 1978; Kawasaki 1980; Shelton

Figure 6-1. Frequency of NPL Sites with PBB Contamination



Derived from HazDat 2002

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and Tiedje 1981), although biotic debromination to lower brominated products may occur in anaerobic zones of contaminated sediment and soil (Morris et al. 1992).

PBBs with six or fewer bromine substitutions bioconcentrate in aquatic organisms such as fish, but the octabromo- and decabromobiphenyls do not bioconcentrate significantly in fish (Gobas et al. 1989; Norris et al. 1973; Opperhuizen et al. 1985; Veith et al. 1979; Zitko 1979; Zitko and Hutzinger 1976). Orchard grass, alfalfa, corn, and carrot tops grown in soil contaminated with PBBs showed no uptake of PBBs, and only minor uptake occurred in carrot roots (Jacobs et al. 1976, 1978). Although PBBs were detected in fish-eating birds and predatory animals that had consumed PBB-contaminated food (Heinz et al. 1983, 1985), the biomagnification potential of PBBs in predators resulting from such consumption remains unknown.

PBBs were detected in air, water, sediment, and soil in the vicinity of the manufacturing plants and in groundwater from a landfill site (DeCarlo 1979; Hesse and Powers 1978; Shah 1978). PBBs were also detected in soil near the contaminated farms in lower Michigan (Fries and Jacobs 1980). The distribution of PBBs was limited to the environment in the vicinity of production sites and the contaminated farm sites. Recent studies have identified PBBs in marine mammals from coastal seas and the Atlantic ocean (de Boer et al 1998). Data regarding the current levels of PBBs in ambient air, drinking water, or food were not located.

No estimate on PBB intake by the general population from air, water, and food was located in the literature. However, current intake of PBBs for the general population is expected to be zero or very small. However, populations near the contaminated farms in lower Michigan may still have low exposures from air, water, and food. The level of PBBs in human tissue and body fluids in the exposed population of Michigan has been extensively studied (Brilliant et al. 1978; Cordle et al. 1978; Eyster et al. 1983; Humphrey and Hayner 1975; Lambert et al. 1990; Landrigan et al. 1979; Wolff et al. 1979a, 1982). The finding that PBBs are stored in fatty tissues of the human body and are very slowly excreted (Eyster et al. 1983) indicates a slow decline in the body burden for exposed individuals.

Polybrominated Biphenyl Ethers. The widespread use of PBDEs over the past 30 years has resulted in their ubiquitous presence in the environment. PBDEs are released into the environment from their manufacture and use as additive flame retardants in thermoplastics in a wide range of products (WHO 1994a). PBDEs containing waste may be either incinerated as municipal waste, deposited in landfills, discharged to municipal sewage treatment plants, or emitted to the atmosphere (Darnerud et al. 2001). In

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the future, the disposal of plastic consumables containing PBDEs is likely to increase in the United States (NSC 1999).

PBDEs are strongly adsorbed to soil and sediment and persist in the environment. Adsorption of PBDEs generally increases as bromination and organic carbon content in soil and sediment increase. As a result, PBDEs have little or no mobility in soil and are not expected to leach. Lower BDE homologs, which exist partially in the vapor phase, have the potential for long-range transport in the atmosphere. Biodegradation will not be significant for PBDEs, but these compounds may be degraded by direct photolysis (WHO 1994a).

Monitoring studies indicate PBDEs are transported globally. Atmospheric, water, and biota levels of PBDEs tend to be dominated by lower brominated congeners (e.g., BDE-47). Biota data indicate that PBDE concentrations increase with lower brominated congeners being preferentially bioconcentrated. PBDE concentrations increase with respect to trophic level; thus, organisms that reside higher on the food chain tend to have higher concentrations of PBDE. Body burden data indicate low-level exposures to PBDEs for the general population. Humans are primarily exposed to PBDEs by inhalation of ambient or contaminated air and ingestion of contaminated food. Occupational exposure to PBDEs occurs primarily by inhalation of air containing PBDEs (WHO 1994a).

PBDEs have not been identified in any of the 1,613 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2002). However, the number of sites evaluated for PBDEs is not known.

6.2 RELEASES TO THE ENVIRONMENT

Polybrominated Biphenyls. The production of PBBs in the United States ceased in 1979 (IARC 1986). In the past, PBBs were released to the environment during the manufacture of these compounds and disposal of commercial and consumer products containing these compounds (Hesse and Powers 1978; Neufeld et al. 1977). One of the significant sources of environmental contamination occurred as a result of the accidental mixup of FireMaster BP-6 with cattle food in a number of farms in the lower peninsula in Michigan (see Section 3.2 for additional details concerning this incident). By June 1975, 412 farms had been quarantined. Disposal of contaminated feed, animal carcasses (poultry, cattle, swine), and animal products (meat, milk, eggs) contributed to environmental contamination (Dunckel 1975; Kay 1977).

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Polybrominated Diphenyl Ethers. The widespread use of PBDEs over the past 30 years has resulted in their ubiquitous presence in the environment. The commercial production of PBDEs began in late 1970s (WHO 1994a). In 1999, the total market demand for PBDEs in the United States was 74.9 million pounds. Technical decabromodiphenyl ether constituted about 53.6 million pounds (72%), while technical mixtures of octa- and pentabromodiphenyl ethers were 3.0 and 18.3 million pounds (4 and 24%) of this total, respectively (BSEF 2002). PBDEs may be released into the environment from their manufacture and use in a wide range of consumer products (WHO 1994a). PBDEs are used as additive flame retardants in thermoplastics. Additive flame retardants are physically, rather than chemically, combined with polymers. Thus, there is a possibility that PBDEs may diffuse out of the treated materials to some extent (EU 2001). In addition to other uses, PBDE treated materials are used in foam for furniture and upholstery, computer and business equipment, adhesives and coatings (WHO 1994a). The widespread use of PBDEs over the past 30 years has resulted in their rapid increase in concentrations and ubiquitous presence in the environment. Waste from these products may be either incinerated as municipal waste, deposited in landfills, discharged to municipal sewage treatment plants, or emitted to the atmosphere (Darnerud et al. 2001). The disposal of plastic consumables containing PBDEs to landfills is likely to increase in the United States.

Releases of decabromobiphenyl ethers (DeBDEs) are required to be reported under the Superfund Amendments and Reauthorization Act (SARA) Section 313; consequently, data are available for this compound in the Toxic Release Inventory (TRI) (EPA 1995c). According to the TRI, a total of 865,109 pounds (392,407 kg) of DeBDE was released to the environment in 1999. In addition, an estimated 915,419 pounds (415,227 kg) were transferred off-site, including to publicly-owned treatment works (POTW) (TRI99 2002). The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list.

6.2.1 Air

Polybrominated Biphenyls. In the past, PBBs were released into the air during the manufacture of these compounds in three areas: through the vents of the hydrogen bromide recovery system, from the centrifugation area for recovering PBBs from slurries produced by bromination, and from the drying, pulverizing, and bagging area of the finished product (Di Carlo et al. 1978). An estimated 0.07 pounds/million pounds of the PBBs produced were lost from the hydrogen bromide recovery vent (Di Carlo et al. 1978). No data are available for the air pollution factor (amount released/million pounds produced) at the centrifugation site. The concentrations of FireMaster BP-6 in the Michigan Chemical

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Corporation bagging area were 0.016–0.032 mg/L of air during the bagging operation and 0.003 mg/L of air after the completion of bagging (Di Carlo et al. 1978). In 1977, the maximum air losses of PBBs at production sites were estimated to total 1,125 pounds of PBBs for every 1 million pounds of PBBs produced (Di Carlo et al. 1978).

Another process that could release lower levels of brominated biphenyls in the air is the incineration of PBBs. Pyrolysis of hexabromobiphenyl in the absence and presence of air has produced small amounts of lower brominated biphenyls (Thoma and Hutzinger 1987). No data are available on the importance of this source for the release of PBBs in the air during the incineration of PBBs. However, since the vast majority of products containing PBBs are expected to be out of circulation after more than 25 years since the voluntary ban, incineration will not be a significant source of PBBs to air.

PBBs have been identified in 1 air sample, collected from 1,613 NPL hazardous waste sites, where they were detected in some environmental media (HazDat 2002).

Polybrominated Diphenyl Ethers. PBDEs may be released to the atmosphere during their manufacture. The estimated release of 116,239 pounds (52,725 kg) of DeBDE to the atmosphere from 55 manufacturing, processing, and waste disposal facilities in 1999 accounted for about 13.4% of the estimated total environmental releases (TRI99 2002). These releases are summarized in Table 6-1. The data from the TRI listed in Table 6-1 should be used with caution, however, since only certain types of facilities are required to report (EPA 1995c). This is not a comprehensive list. In 1979, DeBDE was found in the atmosphere as particulate matter in the vicinity of plants manufacturing brominated flame retardants (Zweidinger et al. 1979); the concentration of DeBDE ranged from not detected to 72 ng/m³. Watanabe et al. (1995) measured DeBDE in atmospheric dust from Osaka, Japan. In the regions surrounding metal recycling plants in Taiwan and Japan, a variety of tri-, tetra-, and hexaBDEs were measured in air (Watanabe et al. 1992). The concentrations in Taiwan and Japan were 23–53 and 7.1–21 pg/m³, respectively.

Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Decabromodiphenyl Ether^a

State ^b	Number of facilities	Reported amounts released in pounds per year ^a						Total on and off-site release
		Air ^c	Water	Underground injection	Land	Total on-site release ^d	Total off-site release ^e	
AL	1	10	5	No data	No data	15	No data	15
AR	4	86,610	0	0	366,543	453,153	71,301	524,454
CA	5	291	2	No data	No data	293	136,499	136,792
CT	4	1,027	No data	No data	No data	1,027	19,406	20,433
FL	1	No data	No data	No data	No data	No data	15,954	15,954
GA	4	755	0	No data	No data	755	8,515	9,270
IL	2	267	No data	No data	No data	267	9,146	9,413
IN	5	10	0	No data	No data	10	19,381	19,391
KS	1	293	No data	No data	No data	293	No data	293
KY	2	973	No data	No data	No data	973	11,560	12,533
LA	1	0	No data	No data	390,000	390,000	No data	390,000
MA	12	366	14	No data	0	380	67,745	68,125
MD	1	No data	No data	No data	No data	No data	No data	0
MI	6	2,652	0	No data	2,700	5,352	39,811	45,163
MN	5	0	No data	No data	0	0	13,665	13,665
MO	1	255	No data	No data	750	1,005	750	1,755
MS	1	121	No data	No data	No data	121	2,207	2,328
NC	12	3,647	6,272	No data	90,246	100,165	93,146	193,311
NE	1	No data	No data	No data	No data	No data	No data	0
NH	1	4	5	No data	No data	9	498	507
NJ	3	78	No data	No data	No data	78	96,868	96,946
NY	3	No data	5	No data	1,100	1,105	23,193	24,298
OH	9	389	0	No data	1,054	1,443	126,444	127,887
OR	1	0	No data	No data	10,837	10,837	No data	10,837

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Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Decabromodiphenyl Ether^a (continued)

State ^b	Number of facilities	Reported amounts released in pounds per year ^a						Total on and off-site release
		Air ^c	Water	Underground injection	Land	Total on-site release ^d	Total off-site release ^e	
PA	8	2,330	68	No data	No data	2,398	157,284	159,682
RI	3	265	No data	No data	No data	265	18,337	18,602
SC	9	0	2,615	No data	250	2,865	19,780	22,645
TN	6	2,755	15	No data	22,581	25,351	20,411	45,762
TX	8	2,850	5	0	2,185	5,040	5,959	10,999
VA	5	270	No data	No data	No data	270	19,142	19,412
WA	1	0	No data	No data	No data	0	No data	0
WI	2	1	No data	No data	No data	1	9,688	9,689
Total	128	106,219	9,006	0	888,246	1,003,471	1,006,690	2,010,161

Source: TRI00 2002

^aData in TRI are maximum amounts released by each facility.

^bPost office state abbreviations are used.

^cThe sum of fugitive and stack releases are included in releases to air by a given facility.

^dThe sum of all releases of the chemical to air, land, water, and underground injection wells.

^eTotal amount of chemical transferred off-site, including to publicly owned treatment works (POTW).

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Migration of PBDEs from consumer products may be a significant diffuse source of PBDEs to the atmosphere. Although no studies were found that determined the migration rate of PBDEs from polymers into the air, estimates have been calculated. Estimated migration rates for deca-, octa- and pentaBDE technical mixtures are 0.038, 0.054, and 0.39% per year, respectively (Danish EPA 1999).

PBDEs were not identified in air samples collected from 1,613 NPL hazardous waste sites (HazDat 2002).

6.2.2 Water

Polybrominated Biphenyls. In the past, PBBs were released to water during the manufacturing process. An estimated 0.0046 pounds were lost to sewers for every 1,000,000 pounds of PBBs produced at manufacturing sites (Neufeld et al. 1977). To manufacture PBBs, water was added to the reaction mixture when the desired extent of bromination was achieved. Ultimately, this water was discharged as effluent into surface water. Samples of effluents from the Michigan Chemical Corporation contained PBB concentrations #503 ppm (Di Carlo et al. 1978). Runoff water from the manufacturing plants containing PBBs also contaminated surface water (Di Carlo et al. 1978). Landfill sites used to dispose of wastes from PBB production can also be a source of PBBs in water. Concentrations of PBBs in groundwater from one such landfill in St. Louis, Michigan, were low, but those in water from a drainage ditch and catch basin were much higher (Di Carlo et al. 1978).

PBBs have been identified in 4 and 10 surface water and groundwater samples, respectively, collected from 1,613 NPL hazardous waste sites (HazDat 2002).

Polybrominated Diphenyl Ethers. PBDEs may be released to surface waters during their manufacture. The estimated release of 2,701 pounds (1,225 kg) of DeBDE to water from 11 domestic manufacturing and processing facilities in 1999 accounted for about 0.312% of the estimated total environmental releases (TRI99 2002). An additional 915,419 pounds (415,227 kg) were transferred off-site, including to POTWs (TRI99 2002). These releases are summarized in Table 6-1. The data from the TRI listed in Table 6-1 should be used with caution, however, since only certain types of facilities are required to report (EPA 1995c). This is not a comprehensive list.

Industrial and urban effluents are significant sources of PBDEs to surface waters and sediments. In 1995, sediment samples were collected up- and downstream near an area where the Swedish plastics industry uses brominated flame retardants (Sellström and Jansson 1995; Sellström et al. 1998). Samples were

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analyzed for TeBDEs (50 ng/g dry weight) and PeBDEs (sum of three congeners, 2,300 ng/g dry weight). These PBDEs were found in higher concentrations downstream of the plant than upstream, which indicates that the plastics industry was the most likely source of these compounds. Surficial sediment samples were collected at eight locations along River Viskan near several textile manufacturing facilities that used various brominated flame retardants in the production of textiles. The concentrations of BDE-47, -99, -100, and -209 in sediments increased as samples were collected further downstream and as more industries were passed (Sellström et al. 1998). The lowest levels of PBDEs were found upstream of the textile industries. The combined concentration of BDE-47, -99, and -100 ranged from not detected to 120 ng/g (dry weight); the concentration of BDE-209 ranged from not detected to 16,000 ng/g (dry weight). Allchin et al. (1999) surveyed the concentrations of PBDEs in sediments from several rivers and estuaries in Great Britain. Sediments were collected upstream and downstream of suspected sources of PeBDE and OBDE, including a manufacturer, several industries, landfills, and a reference site. The highest concentrations of BDE-47, BDE-99, PeBDE (as BDE-71), and OBDE (as BDE-79) were in sediments near or downstream from a manufacturing site at Newton Aycliffe on River Skerne. The highest concentrations of DeBDE (as BDE-83) were found downstream of a sewage treatment plant on River Calder. High concentrations were also detected on River Skerne downstream of a manufacturing site. BDE-99 concentrations were identical or slightly higher than BDE-47 in most sediments (Allchin et al. 1999). The sum of five PBDE congeners (BDE-47, -99, -100, -153, and -209) ranged from 0.07 to 10.6 ng/g dry weight in freshwater sediments from Denmark (Christensen and Platz 2001). The highest concentrations were found in sediment close to populated areas.

PBDEs were not identified in water samples collected from 1,613 NPL hazardous waste sites (HazDat 2002).

6.2.3 Soil

Polybrominated Biphenyls. The important former sources of PBBs in soil are manufacturing operations, disposal of PBB-containing finished products, and agricultural operations contaminated in the original episode in 1973–1974. The concentrations of PBBs in soils from bagging and loading areas of the Michigan Chemical Corporation were 3,500 and 2,500 mg/kg, respectively (Di Carlo et al. 1978). Similarly, soil from sites adjacent to the Hexcel Corp and the White Chemical Company, the manufacturers of octabromo- and decabromobiphenyl, contained decabromobiphenyl and other lower brominated biphenyls down to hexabromobiphenyl (Di Carlo et al. 1978). The disposal into landfills of solid wastes generated during the production of PBBs was another important source of PBBs in soil

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(Neufeld et al. 1977). Photodecomposition of FireMaster BP-6 in soil could also be a source of lower brominated biphenyls (Ruzo and Zabik 1975; Trotter 1977) in soil.

Approximately 11.8 million pounds of hexabromobiphenyl were used in commercial and consumer products in the United States, mostly in the production of plastic products. Since the cessation of production of hexabromobiphenyl, all of these products, such as TV cabinet and business machine housings, with a usable life of 5–10 years must have been disposed of by landfilling or incineration (Neufeld et al. 1977). Disposal of these plastic materials in waste disposal sites is an important source of PBBs in soil. The migration of plastic-incorporated PBBs to soil would be very low since PBBs would be tightly bound into the plastic (Neufeld et al. 1977).

The indirect source of PBBs in soil was the contaminated farms in Michigan. Approximately 650 pounds (290 kg) of PBBs were mixed in cattle feeds that were delivered to Michigan farms during 1973–1974 (Fries 1985b). About 50% of this amount was excreted in the feces of the exposed animals and remained on the farms in places of fecal deposition and manure disposal (Fries 1985b). Soil in fields that received contaminated manure contained as high as 300 µg/kg PBBs, whereas soil in resurfaced cattle exercise lots contained as high as 1,000–2,000 µg/kg of PBBs (Fries 1985b).

PBBs have been identified in 12 soil and 5 sediment samples collected from 1,613 NPL hazardous waste sites (HazDat 2002).

Polybrominated Diphenyl Ethers. PBDEs are released to land during their manufacture and disposal. In 1999, 746,169 pounds (338,456 kg) of DeBDE were released to land from 10 domestic manufacturing, processing, and waste disposal facilities reporting releases of the compound to the environment (TRI99 2002). No releases (0 pounds) of DeBDE occurred via underground injection (TRI99 2002). Releases to the environment from facilities that produce, process, or use DeBDE are summarized in Table 6-1. The data from the TRI should be used with caution since only certain types of facilities are required to report (EPA 1995c). This is not a comprehensive list. Solid waste from commercial production of OBDE is typically disposed in landfills (EPA 1994).

PBDEs are release to land as municipal wastes with the disposal of consumer products containing PBDEs. The disposal of consumer products containing PBDEs is likely to increase worldwide due to rapid obsolescence of plastic products. For example, between 1997 and 2004, the number of obsolete computers containing flame retardants is projected to be 315 million (NSC 1999). Based on a monitor weight of 30 pounds, an estimated 350 million pounds of brominated flame retardants will be released to

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landfills (NSC 1999). Although PBDEs will only be a fraction of this total, the amount of PBDEs released to the environment by disposal will still be significant.

Although no information was located, PBDEs may be released to farm land with their disposal as biosolids (i.e., sewage sludge). PBDEs were detected in biosolids from four different regions of the United States (Pardini et al. 2001). The total concentrations of PeBDE in biosolids ranged from 1,100 to 2,290 $\mu\text{g}/\text{kg}$ dry weight; the levels of PeBDE were high and consistent, regardless of the region of origin. The concentration of DeBDE (BDE-209) varied widely among biosolids from different regions; the concentration of BDE-209 ranged from 84.8 to 4,890 $\mu\text{g}/\text{kg}$ dry weight in the biosolid samples. Sewage sludge samples from 13 waste water treatment plants in Germany were sampled (Hagenmaier et al. 1992). The mean concentration of tri- to heptaBDEs was 8.37 ng/g with tri-, tetra-, penta-, hexa-, and heptaBDE at concentrations of 0.65, 3.06, 3.02, 0.49, and 0.22 ng/g, respectively. Levels of penta- and hexaBDEs were highest in these samples.

PBDEs were not identified in soil and sediment samples collected from 1,613 NPL hazardous waste sites (HazDat 2002).

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

Polybrominated Biphenyls. PBBs existed predominantly in the particulate phase in the atmosphere. Particulate phase PBBs are removed from the atmosphere by wet and dry deposition and should not travel long distances in the environment. In water, PBBs are expected to adsorb strongly to suspended solids and sediment, and may bioconcentrate in aquatic organisms. The volatilization of PBBs from water to air is not expected to be important due to attenuation by adsorption in the water column. In soil, PBBs are adsorbed strongly and will be immobile. Volatilization of PBBs from soil to air is not important due to the low volatility of PBBs and strong adsorption of PBBs to soil.

Organic compounds with vapor pressures $>10^{-4}$ mmHg should exist almost entirely in the vapor phase in the atmosphere, while organic compounds with vapor pressures $<10^{-8}$ mmHg should exist almost entirely in the particulate phase (Eisenreich et al. 1981). The estimated vapor pressure of FireMaster BP-6 is 5.2×10^{-8} mmHg at 25 EC (Jacobs et al. 1976). The vapor pressure of octabromobiphenyl is 7.0×10^{-11} mmHg at 28 EC (Waritz et al. 1977). Although no data are available, the vapor pressures of

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decabromobiphenyl at ambient temperatures should be lower than octabromobiphenyl. Therefore, the PBBs produced in the 1970s should have existed predominantly in the particulate phase in the atmosphere. Since the particulate phase PBBs would precipitate out by dry deposition and wet deposition due to washout (Atlas and Giam 1987), PBBs would not be expected to transport long distances in the atmosphere.

There are limited data regarding the transport and partitioning of PBBs in water. Based on an estimated Henry's law constant of 3.9×10^{-6} atm-m³/mol (where Henry's law constant = vapor pressure/water solubility) and an estimation method (Thomas 1990), the estimated volatilization half-life of hexabromobiphenyl is 23 days. Therefore, the transport of PBBs from water to the atmosphere by volatilization is not expected to be important. This is consistent with a fish bioconcentration study where losses of octabromobiphenyl and decabromobiphenyl from water to air were insignificant (Norris et al. 1973). Soil mobility studies have shown that PBBs are strongly adsorbed by soil materials (Filonow et al. 1976; Griffin and Chou 1981a, 1981b). Therefore, transport of water-bound PBBs to particulate matter and sediment in water due to adsorption is a major transport process for PBBs in water. The detection of at least a 1,000-fold higher concentration of PBBs in Pine River sediment (where effluent from Michigan Chemical Corporation was discharged) compared with the level of PBBs in the river water confirms the importance of this transport process (Hesse and Powers 1978).

PBBs may also be transported from water to aquatic organisms where bioconcentration may take place. Data on the bioconcentration of PBBs in fish, which were generated in different laboratories, show wide variation. The experimentally determined bioconcentration factor ([BCF] = concentration in fish over concentration in water) for hexabromobiphenyl (mixtures of unspecified congeners) in the whole body of fathead minnows (*Pimephales promelas*) was 18,100 in a 32-day exposure (Veith et al. 1979). In fillet of fathead minnow, the estimated BCF was >10,000 (Hesse and Powers 1978). The lipid weight-based BCF values of 4,4'-dibromobiphenyl, 2,4,6-tribromobiphenyl, 2,2',5,5'-tetrabromobiphenyl, and 2,2',4,4',6,6'-hexabromobiphenyl in guppies (*Poecilia reticulata*) were 269,153; 114,815; 1,445,440; and 707,946, respectively (Gobas et al. 1989). It is worth noting that the BCF values decrease, rather than increase, as the degree of bromine substitution increases. A similar trend in BCF values for various PBBs was also observed in juvenile Atlantic salmon (*Salmo salar*). For example, the determined whole body BCF values for 2,6-dibromobiphenyl, 2,4-dibromobiphenyl, 3,4-dibromobiphenyl, 2,5,4'-tribromobiphenyl, 2,2',4,5'-tetrabromobiphenyl, 2,3',4',5-tetrabromobiphenyl, hexabromobiphenyl (unspecified congener), and octabromobiphenyl were 1,267; 1,343; 63; 425; 314; 111; 2-48; and 0.02, respectively (Zitko 1979; Zitko and Hutzinger 1976). The determined BCF values for 2,2',3',3',4,4'-hexabromobiphenyl and decabromobiphenyl in whole body guppies (*P. reticulata*) were 10 and 0, respectively

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(Opperhuizen et al. 1985). The BCF value for octabromobiphenyl in filleted rainbow trout (*Salmo gairdneri*) was 0 (Norris et al. 1973). The lack of accumulation for the higher brominated compounds is most likely because they have very limited water solubility and are therefore not available to penetrate membranes (Zitko 1979).

PBBs are adsorbed strongly to soil, and the adsorption increases with an increase in the organic carbon content of soil (Filonow et al. 1976; Griffin and Chou 1981a, 1981b). Neither clay content nor pH of soil correlated with adsorption of hexabromobiphenyl to soil (Filonow et al. 1976). PBBs present in soil-water solution will partition to the soil by adsorption. The presence of certain specific dissolved organic carbon in natural water (e.g., leachate from a landfill) may decrease the adsorption of PBBs in sediments (Simmons and Kotz 1982). Because of the strong adsorption, PBBs will have low mobility in soil, and the leaching of PBBs from soil to groundwater will generally be insignificant (Filonow et al. 1976; Griffin and Chou 1981a, 1981b). However, the mobility of PBBs may greatly increase if methanol or other organic solvents (capable of solubilizing PBBs) are present at significant concentrations in soil (Griffin and Chou 1981b). The phenomena is commonly called “co-solvency.” The transport of PBBs from soil to the atmosphere by volatilization is not important due to the low volatility of PBBs (Jacobs et al. 1976). The transport of PBBs from soil to surface water or another land area via eroded soil contained in runoff water is possible (Jacobs et al. 1976). Orchard grass and carrot tops grown in soil contaminated with PBBs showed no uptake, and carrot roots showed only minor uptake of PBBs (Jacobs et al. 1976, 1978). Therefore, the transport of PBBs from soil to plants via translocation is insignificant.

Polybrominated Diphenyl Ethers. In air, highly brominated PBDEs, which have low vapor pressures and exist in the particulate phase, will be removed from the atmosphere by wet and dry deposition. These PBDEs do not travel long distances in the environment. Lower BDE homologs, which exist partially in the vapor phase, have the potential for long-range transport in the atmosphere. In water, PBDEs are expected to adsorb strongly to suspended solids and sediment, and bioconcentrate in aquatic organisms. The volatilization of PBDEs from water to air is not expected to be important due to attenuation by adsorption in the water column. In soil, PBDEs are adsorbed strongly and will be immobile. They are not likely to leach into groundwater. Volatilization of PBDEs from soil to air is not important due to the low volatility of PBDEs and strong adsorption of PBDEs to soil.

PBDEs with vapor pressures between 10^{-4} and 10^{-8} mm Hg (di- to hexaBDE) should exist in both the vapor and particulate phase in the atmosphere, while PBDEs with vapor pressures $<10^{-8}$ mmHg (hexaBDE to DeBDE) should exist almost entirely in the particulate phase in the atmosphere (Bidleman 1988; Eisenreich et al. 1981). PBDEs have low vapor pressures, with vapor pressure tending to decrease

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with increasing bromination. Watanabe and Tatsukawa (1990) determined the vapor pressures for a range of brominated PBDEs as follows (mm Hg at 25 EC): di- (9.8×10^{-5} – 1.4×10^{-4}); tri- (1.2×10^{-5} – 2.0×10^{-5}); tetra- (1.8×10^{-6} – 2.5×10^{-6}); penta- (2.2×10^{-7} – 5.5×10^{-7}); hexa- (3.2×10^{-8} – 7.1×10^{-8}); and octa- (9.0×10^{-10} – 1.7×10^{-9}). Vapor pressures have also been determined for commercial BDE mixtures, such as PeBDE (2.2×10^{-7} – 5.5×10^{-7} mm Hg) and DeBDE (3.2×10^{-8} mm Hg at 25 EC) (EU 2001; NRC 2000). Since particulate phase PBDEs will precipitate out by wet and dry deposition, these PBDEs would not be expected to transport long distances in the atmosphere (Atlas and Giam 1987). Thus, highly brominated PBDEs (e.g., octa- through decaBDEs), which have low vapor pressures and exist solely in the particulate phase, will not be transported long distances. Although no information was located in the literature, moderately brominated PBDEs (e.g., PeBDEs and hexaBDEs) may have the potential to be transported short distances and might be found close to PBDE point sources. Lower BDE homologs (e.g., TeBDE), which exist partially in the vapor phase, have the potential for long-range transport in the atmosphere (Dodder et al. 2000a). For example, BDE-47, BDE-99, and BDE-153 were detected in air samples from urban, rural, and remote areas of the Great Lakes in the United States (Dodder et al. 2000a).

Concentrations of PBDEs in water are expected to be low due to the low water solubility of PBDEs. For example, the solubilities of the commercial mixtures of PeBDE and DeBDE are 13.3 and 20–30 $\mu\text{g/L}$, respectively (EU 2001; WHO 1994a). As consequence of their low water solubility, monitoring studies have not detected PBDEs in environmental waters (see Section 6.4.2).

PBDEs adsorb strongly onto suspended solids and sediments in the water column. Volatilization of PBDEs from water surfaces will be attenuated by adsorption, and thus is not an important fate process. Sediment-water partition coefficients (K_p) have been measured for several components of commercial PeBDEs (Watanabe 1988). K_p values for tetra-, penta-, and hexaBDEs are 28,293; 49,167; and 62,727 L/kg, respectively, which suggest strong partitioning to sediment. High $\log K_{ow}$ values have been measured for PBDEs as follows (Watanabe and Tatsukawa 1990): di- (5.03); tri- (5.47–5.58); tetra- (5.87–6.16); penta- (6.46–6.97); hexa- (6.86–7.92); octa- (8.35–8.90); and deca- (9.97). Using these $\log K_{ow}$ values, \log organic carbon-water partition coefficients (K_{oc}) were estimated for PBDEs: di- (4.11); tri- (4.35–4.41); tetra- (4.57–4.73); penta- (4.89–5.17); hexa- (5.11–5.69); octa- (5.92–6.22); and deca- (6.80) (Lyman et al. 1990).

PBDEs are transported from water to aquatic organisms, which results in bioconcentration in aquatic organisms. No data on the bioconcentration of PBDEs in fish were located. However, due to abundance of monitoring data illustrating the accumulation of PBDEs in fish (see Section 6.4.4), bioconcentration of PBDEs is expected to occur. In a laboratory study of Baltic blue mussels, BCFs (from water absorption)

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were found to be 1,300,000 for BDE-47, 1,400,000 for BDE-99, and 1,300,000 for BDE-153 (Gustafsson et al. 1999). At several sites along the coast and in the Schelde estuary (the Netherlands), BCFs for blue mussels were determined (Booij et al. 2000). The maximum BCFs were log BCF of 9 for BDE-99 and BDE-100, and approximate log BCFs of 7.4 for BDE-28, 8.4 for BDE-47, and 8.2 for BDE-153.

Bioaccumulation of PBDEs in the aquatic food web is inversely related to the degree of bromination (Burreau et al. 2000b; Jansson et al. 1993). Thus, higher brominated congeners are rarely detected in biota. This is a result of their low solubility and high log K_{ow} values (Hardy 2000). In contrast, tetra- to hexaBDE homologs are most frequently detected in biota (Burreau et al. 1997), which would be expected due to their greater water solubility and relatively high K_{ow} values.

Concentrations of PBDEs in biota are related to the trophic level of the species. For example, Haglund et al. (1997) examined the concentrations of PBDEs in herring, salmon muscle, and gray and ringed seals collected along the Swedish coast of the Baltic sea between 1981–1988. PBDE concentrations were found to increase with trophic level.

PBDEs will be strongly adsorbed to soils based on log K_{ow} values ranging from 5.03 to 9.97 (Watanabe and Tatsukawa 1990). Thus, PBDEs present in soil pore water will bind to soil organic matter. Because PBDEs adsorb strongly to soil, they will have very low mobility (Swann et al. 1983), and leaching of PBDEs from soil to groundwater will be insignificant. Like PBBs, the presence of dissolved organic carbon in natural water (e.g., leachate from a landfill) may increase the mobility of PBDEs. The transport of PBDEs from soil to surface water via eroded soil contained in runoff water is also possible.

Volatilization of PBDEs from moist soil surfaces will be attenuated by adsorption, and is not expected to be an important fate process. Volatilization of PBDEs from dry soil will not be important due to the low volatility of PBDEs (see Table 4-6).

6.3.2 Transformation and Degradation

Photolysis appears to be the dominant transformation process for PBBs and PBDEs. However, the importance of photochemical transformation reactions in the environment cannot be determined due to lack of information. Based on a very limited number of studies, biodegradation does not appear to be significant for either PBBs or PBDEs.

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6.3.2.1 Air

Polybrominated Biphenyls. In air, the two processes that may result in significant degradation or transformation of PBBs are photooxidation by hydroxyl radicals and direct photolysis. The estimated half-life of pentachlorobiphenyl in air due to reaction with hydroxyl radicals is 41.6–83.2 days (Atkinson 1987a). Based on a structure-activity relationship for the estimation of half-lives for the gas-phase reactions of hydroxyl radicals with organic compounds (Atkinson 1987b), the estimated half-lives of hexabromobiphenyl and decabromobiphenyl due to reaction with OH radicals are 182 and 2,448 days, respectively. These half-lives are consistent with the half-life of pentachlorobiphenyl due to reaction with OH radicals. However, the half-lives of brominated biphenyls expected to be present in the particulate phase in the air may be even longer than the estimated half-lives due to gas phase reaction. Therefore, the transformation of the hexa- and other higher brominated PBBs in the atmosphere due to reaction with OH radicals may not be important.

Hexa- and other higher brominated biphenyls are expected to be present in the particle-adsorbed state in the atmosphere. These PBBs photolyze in solution and in soil (Hill et al. 1982; Ruzo and Zabik 1975; Trotter 1977). Since PBBs present in surface soil are known to photolyze, particle-sorbed PBBs present in the atmosphere may also undergo photolysis. The importance of the photochemical reaction under sunlight illumination conditions for the degradation/transformation of PBBs in air cannot be evaluated due the lack of information.

Polybrominated Diphenyl Ethers. In air, PBDEs may undergo indirect photolysis with hydroxyl radicals or direct photolysis with sunlight. Vapor-phase PBDEs may be degraded in the atmosphere by reaction with photochemically produced hydroxyl radicals. The half-lives for this reaction in air are estimated to be 29, 140, and 476 days, respectively, for penta-, octa-, and decaBDE homologs, calculated using a structure estimation method (Meylan and Howard 1993). This estimation is calculated using an atmospheric concentration of 5×10^5 hydroxyl radicals per cm^3 and is based on a 24-hour day of sunlight. The half-lives of PBDEs that are expected to be present in the particulate phase in the air will be longer than the estimated half-lives calculated for the gas-phase reaction. Thus, for the higher brominated PBDEs (e.g., octa- and decaBDEs), indirect photolysis with hydroxyl radicals may not be important.

In water, PBDEs are known to undergo direct photolysis (Norris et al. 1973, 1975a; Watanabe and Tatsukawa 1987). Likewise, PBDEs present in the atmosphere may also undergo photolysis. However, the importance of photolysis of PBDEs in air cannot be evaluated due the lack of information.

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6.3.2.2 Water

Polybrominated Biphenyls. The photolytic degradation of PBBs in solution has been the subject of several studies. Available data in the literature indicate that brominated biphenyls photodegrade by reduction in solvents capable of proton transfer with the formation of lower brominated biphenyls. For example, the irradiation of FireMaster BP-6 and 2,2', 4,4',5,5'-hexabromobiphenyl in methanol at wavelengths >286 nm produced mainly penta- and tetrabromobiphenyl (Ruzo and Zabik 1975). FireMaster BP-6 photolyzed 7 times faster than its chlorinated counterpart, 2,2',4,4',5,5'-hexachlorobiphenyl (Ruzo and Zabik 1975). Although an earlier study tentatively identified dimethoxy tetrabromobiphenyl as a photolysis product of FireMaster BP-6 (Ruzo and Zabik 1975), later work did not detect this compound (Ruzo et al. 1976). Earlier studies indicated that the debromination usually occurs with the stepwise preferential loss of bromine from the *ortho* and *para* positions of the biphenyl ring (i.e., 2, 2', 6, and 6' positions) (De Kok et al. 1977; Ruzo and Zabik 1975; Ruzo et al. 1976; Trotter 1977). Thus, the photolysis of 2,2',4,4',5,5'-hexachlorobiphenyl, the major component of FireMaster BP-6, would be expected to produce 2,3',4,4',5-pentabromobiphenyl and subsequently 3,3',4,4'-tetrabromobiphenyl. More recent work indicates that although photolysis mainly produces debromination products, unlike in the case of an individual PBB congener, reductive debromination of *ortho* substituents is not the predominant photolytic degradation pathway for FireMaster BP-6 (Robertson et al. 1983b).

The study of photolysis of PBBs in the aqueous phase is more relevant to natural environmental situations than photolysis in proton-donating organic solvents. It was suggested that the photolysis of PBBs in aqueous solution would proceed by oxidative process of photohydroxylation, leading to the formation of phenolic compounds (Norris et al. 1973). However, photolysis of 2,4-dibromo- and 2,3',4',5-tetrabromobiphenyl in acetonitrile-water solution showed that debromination was the major reaction (Ruzo et al. 1976). No evidence of the formation of hydroxylated species (phenolic products) was found (Ruzo et al. 1976).

Several investigators assessed the biodegradation potential of PBBs under aerobic conditions, with activated sludge or pure cultures of microorganisms as microbial inoculum, and concluded that although the lower substituted biphenyls might biodegrade in aerobic water and sediment (Kong and Sayler 1983; Sugiura 1992; Yagi and Sudo 1980), the higher substituted biphenyls are resistant to aerobic biodegradation (Kawasaki 1980; Sasaki 1978; Shelton and Tiedje 1981). This is consistent with biodegradation studies in soil (see Section 6.3.2.3). It has been proposed that complete mineralization of

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4-bromobiphenyl to carbon dioxide occurs via a 4-bromobenzoate intermediate by mixed bacterial cultures obtained from PBB-contaminated river sediment (Kong and Sayler 1983). However, complete mineralization was not observed for 2- and 3-bromobenzoate (Kong and Sayler 1984).

Although higher brominated biphenyls do not biodegrade in water or sediment under aerobic conditions, it has been shown that anaerobic microorganisms obtained from populated river sediments can biodegrade higher substituted PBBs, including FireMaster mixtures (Morris et al. 1992). The biodegradation involved debromination at the *meta* and *para* positions, and no *ortho* bromine removal was observed (Morris et al. 1992). However, the possibility of *ortho* bromine removal from higher brominated biphenyls under certain inoculation conditions (e.g., microorganisms from polluted river sediment repeatedly transferred on a pyruvate medium amended with Aroclor 1242) has been suggested (Morris et al. 1992).

Polybrominated Diphenyl Ethers. PBDEs may undergo debromination by direct photolysis. For example, studies indicate that DeBDE is debrominated by UV light and sunlight to lower brominated PBDEs (to triBDE with UV light and to tetraBDE with sunlight) (Norris et al. 1973, 1975a; Watanabe and Tatsukawa 1987). However, it is not known whether direct photolysis of DeBDE happens in the environment. More recent laboratory studies of the photolytic breakdown of DeBDE in toluene have shown that it is successively debrominated by UV light to lower brominated PBDEs (down to TeBDE) and that photolysis occurs very rapidly (Sellström et al. 1998). The photolysis half-life in toluene was <15 minutes. In water alone (i.e., without organic solvent), photodegradation of PBDEs does not result in the formation of lower brominated PBDEs or polybrominated dibenzofurans (PBDFs) (Norris et al. 1973). Photohydroxylation of DeBDE is expected in water, resulting in the formation of hydroxylated products, which are degraded rapidly due to increased UV absorption (WHO 1994a). No information was available on direct photolysis of other PBDEs. By comparison with DeBDE, it is likely that other PBDEs will photodegrade. However, it is not possible to predict the likely extent or rate.

No information was located in the literature about the biodegradation of PBDEs in natural waters. For PeBDE, no degradation (as CO₂ evolution) was seen after 29 days in an OECD 301B ready biodegradation test (EU 2001). The substance tested was a composite sample from two producers with the following composition: 33.7% TeBDE; 54.6% PeBDE; and 11.7% HexaBDE. The test was extended to 93 days to allow sufficient opportunity for adaptation to occur. At the end of 93 days, 2.4% of the theoretical amount of CO₂ had been evolved. Thus, PeBDE is not readily biodegradable. In a laboratory study, Schmidt et al. (1993) found that 4,4'-dibromodiphenyl ether was not used as a carbon source by a pure bacterial culture. Transformation of 4,4'-dibromodiphenyl ether did occur yielding 4-bromophenol

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and 4-bromocatechol. The specific rate of oxidation of 4,4'-dibromodiphenyl ether was 19 nmol of O₂ per minute per mg of protein.

No other information was found in the literature about the transformation and degradation processes for PBDEs in water.

6.3.2.3 Sediment and Soil

Polybrominated Biphenyls. Information on the fate of PBBs in soil is limited. A pure culture of microorganism isolated from soil biodegraded 2-bromobiphenyl via the 2-bromobenzoic acid pathway (Takase et al. 1986). There is little evidence that the higher brominated biphenyls biodegrade in soil under aerobic conditions for an incubation period of #1 year (Griffin and Chou 1981a, 1981b; Jacobs et al. 1976). Some degradation of an undefined congener of pentabromobiphenyl was observed when incubated in soil, but this degradation could not be definitely attributed to biodegradation (Jacobs et al. 1976). As discussed in Section 6.3.2.2, higher brominated PBBs may biodegrade in an anaerobic region of river sediment and possibly soil polluted with PCBs and PBBs to form lower brominated products. Biodegradation of the photolysis products of hexa- and heptabromobiphenyl in soil (which produces lower brominated products) was only minor (. 3% in 1 year) since the photodegradation products were bound to soil (Jacobs et al. 1978).

Degradation of PBBs present in a contaminated soil from a manufacturing site in Michigan was significant (Hill et al. 1982). For example, 2,2',4,4',5,5'-hexabromobiphenyl, the principal component of FireMaster (54–68% in FireMaster) was reduced to 26% of the total PBBs when exposed to a field soil for several years. In two other soils, where the original concentrations of PBBs were much lower, the rate of degradation was much lower. Principal degradation products were 2,3',4,4',5-pentabromobiphenyl, 2,2',4,4',5-pentabromobiphenyl, and two unidentified tetrabromobiphenyls. The degradation was attributed to photochemical reactions. On the other hand, no significant photodegradation of FireMaster was observed after 1 year in contaminated manure spread in field soil from Michigan (Jacobs et al. 1978). The authors provided no explanation for the difference in photoreactivity of PBBs in soils with and without manure. It is important to point out that, due to attenuation and scattering of light, sunlight will not penetrate most soil beyond the surface layer. Therefore, it can be concluded from these studies that although photolysis may be the only viable degradative process for PBBs in soil, photolysis will be limited to the surface layer of soil, and the rate of photolysis will be very slow. PBBs incorporated into

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thermoplastics, which were eventually buried at waste sites are not likely to absorb much light and undergo photolytic degradation.

Analysis (Morris et al. 1993) of sediments from a PBB-contaminated river in Michigan (Pine River) indicates that little degradation of PBBs has occurred since the 1970s. Although microorganisms capable of debrominating PBBs were not present in regions of highest contamination, they were found in sediments downstream from the area of highest contamination. The investigators (Morris et al. 1993) suggest that high levels of contaminants including PBBs may be inhibiting the microbial degradation of PBBs in this river.

Polybrominated Diphenyl Ethers. Information on the transformation and degradation of PBDEs in soil is limited. PBDEs may undergo debromination by direct photolysis in soils and sediment. The extent to which PBDEs photodegrade in soils and sediment is unknown in the environment. If the photodegradation process is significant, it could result in the formation of lower brominated BDEs, which are known to bioaccumulate. Studies indicate that DeBDE is debrominated by UV radiation and sunlight forming lower brominated PBDEs (Norris et al. 1973, 1975a; Watanabe and Tatsukawa 1987). The photolysis half-lives for DeBDE in sediment were 53 and 81 hours for UV and sunlight exposure, respectively. TeBDE appeared at the longest exposure time (244 hours). In sand, the half-life of DeBDE exposed to UV radiation was between 12 and 37 hours. Lower PBDEs and other compounds (not specified) were found after UV radiation (Sellström et al. 1998). In a laboratory study of soil treated with DeBDE and exposed to UV light, a photolysis half-life of 185 hours for DeBDE was measured. The debromination process was the same as that seen for other studies.

No other information was found in the literature about the transformation and degradation processes for PBDEs in soils and sediment.

6.3.2.4 Other Media

No other information was found in the literature about the transformation and degradation processes for PBBs or PBDEs in other media.

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6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Evaluation of the potential for human exposure to PBBs or PBDEs depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Comparisons among various studies are complicated by the fact that authors may report PBB or PBDE concentrations as technical mixtures, as homologs, or as congeners. For PBDEs, it is common to determine the concentration of individual congeners. However, only a limited number of standards are available. Total PBDEs, the definition of what constitutes total PDBEs (i.e., how many and which congeners are summed), is often not the same in the various studies. Chemical analysis procedures are discussed in greater detail in Chapter 7. Recent monitoring data for PBBs are very limited. Historical monitoring data indicate that environmental PBB concentrations are confined to areas near former manufacturing facilities and regions of Michigan effected by the farm catastrophe of the early 1970's (see Section 6.1). Monitoring studies indicate that PBDEs are transported globally. Atmospheric, water, and biota levels of PBDEs tend to be dominated by lower brominated congeners (e.g., BDE-47). Sediments tend to be dominated by higher brominated congeners (e.g., BDE-209). Biota monitoring studies indicate that PBDE concentrations have increased since the late 1970s, with lower brominated congeners (e.g., BDE-47) being preferentially bioconcentrated. Studies indicate that PBDE concentrations increase with respect to trophic level; organisms that reside higher on the food chain tend to have higher concentrations of PBDEs.

6.4.1 Air

Polybrominated Biphenyls. Historically, PBBs were released to the atmosphere during three stages of the manufacturing process, and an estimate of the maximum amount of PBBs expected to be lost to the air during the manufacture of PBBs in the United States is available (see Section 6.2.1) (Neufeld et al. 1977). Monitoring data on the ambient air levels of PBBs are very limited. The concentration of hexabromobiphenyl in air samples collected downwind and crosswind from the White Chemical Company plant in Bayonne, New Jersey was 0.06 ng/m³ (DeCarlo 1979).

Polybrominated Diphenyl Ethers. Atmospheric concentrations of PBDEs tend to be dominated by lower brominated congeners (e.g., BDE-47 and -99). The concentrations of PBDEs in air samples are summarized in Table 6-2. Air samples sampled from urban (Chicago, Illinois), rural (Sleeping Bear Dunes, Michigan and Sturgeon Point, New York), and remote (Eagle Harbor, Michigan) shorelines of the

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U.S. Great Lakes all contained quantifiable levels of BDE-47, -99, -100, -153, and -154 (Dodder et al. 2000a; Strandberg et al. 2001). The most significant congeners were BDE-47 and BDE-99. The concentration of total PBDEs ranged from 6.9 to 77 pg/m³ for all samples; the highest levels were observed near Chicago, Illinois (BDE-47, 48 pg/m³). Throughout the year of 1997, air samples were taken from a rural site in southwestern England called Stokes Ferry and a semirural site in northwestern England called Hazelrigg and analyzed for PBDEs (Peters et al. 1999). Tri- and heptaBDEs were detected; the combined concentrations of BDE-47, -99, and -100 ranged from 7 to 69 pg/m³ at Hazelrigg and from 6 to 58 pg/m³ at Stoke Ferry (de Wit 2002). PBDEs have also been measured in air samples taken from remote stations in the Arctic (e.g., Alert, North West Territory, Canada; Dunai Island, eastern Siberia, Russia) between January 1994 and January 1995 (de Wit 2002). The total concentration of several di- to hexaBDEs ranged from 1 to 4 pg/m³ at Alert the majority of the year; however, in July 1994, the concentration was 28 pg/m³. At Dunai, the major congeners found were BDE-47 and BDE-99. In Sweden during 1990–1991, air samples collected from Ammarnäs in the northern mountains and Hoburgen on the southern tip of Gotland in the Baltic Sea, had measurable amounts of BDE-47, -99, and -100 (de Wit 2002). Total PBDE levels were approximately 1 and 8 pg/m³, respectively. The concentration of BDE-47 was found to be highest in the gas phase, while BDE-99 and BDE-100 were highest in the particulate phase. No DeBDE was found, although the limit of detection limit for DeBDE is much higher than for the lower BDEs.

Table 6-2. Concentrations (in pg/m³) of Several PBDEs in Air Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	Reference
Air	Ammarnäs	6.3	1.6	0.4	8.3	de Wit 2000, 2002
Air	Hobburgen	0.7	0.35	0.07	1.1	de Wit 2000, 2002
Air	Stoke Ferry, United Kingdom	4.7–50	5.5–13	1.1–3.9	6.7–58	Peters et al. 1999
Air	Hazelrigg, United Kingdom	3.2–61	3.1–22	0.62–5.4	4.1–69	Peters et al. 1999
Air	Alert, Canada	No data	No data	No data	1-28	Alaee et al. 1999
Air	Dunai Island, Russia	No data	No data	No data	1-8*	Alaee et al. 1999
Air	Urban, United States	48	25	3.0	77*	Dodder et al. 2000
Air	Rural, United States	6.2–9.2	4.3–5.0	0.6–0.9	2–4.8*	Dodder et al. 2000
Air	Remote, United States	3.7	2.6	0.33	6.9*	Dodder et al. 2000

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

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6.4.2 Water

Polybrominated Biphenyls. The concentrations of PBBs in effluents discharged from the Michigan Chemical Corporation plant in St. Louis, Michigan, to the Pine River during 1974–1977 ranged from <0.01 to 150 µg/L (Hesse and Powers 1978). The concentrations of PBBs in effluents from White Chemical Company, Bayonne, New Jersey, and Hexcel Chemical Corporation, Sayerville, New Jersey, ranged from <0.2 to 210 µg/L (DeCarlo 1979). The concentrations of PBBs in the Pine River #12 miles downstream from the Michigan Chemical Corporation plant in 1974 were 0.01–3.2 µg/L (Hesse and Powers 1978; Neufeld et al. 1977). 2,2',5,5'-Tetra- and 3,3',5,5'-tetrabromobiphenyl were qualitatively detected in water from Lake Ontario, and hexabromobiphenyl (unspecified congeners) was qualitatively detected in water from Lakes Ontario and Huron (Great Lakes Water Quality Board 1983). The concentrations of PBBs in test wells within a PBB landfill site in Michigan ranged from 0.4 to 26.0 µg/L, although the concentrations of PBBs in test wells outside the landfill ranged from <0.1 to 4.4 µg/L (Shah 1978). No other information was located about the concentrations of PBBs in water.

Polybrominated Diphenyl Ethers. In 1999, the concentration of PBDEs in Lake Ontario surface waters ranged between 4 and 13 pg/L with ~90% in the dissolved phase (Luckey et al. 2001). BDE-47 and BDE-99 were the most abundant congeners, together making up >70% of the total PBDEs. No other information was located about the concentrations of PBDEs in water.

6.4.3 Sediment and Soil

Polybrominated Biphenyls. Soil samples from the bagging and loading areas of the Michigan Chemical Corporation plant in St. Louis, Michigan, contained PBBs at concentrations of 3,500 and 2,500 mg/kg, respectively (Di Carlo et al. 1978). PBBs (mostly decabromobiphenyl, but some lower brominated biphenyls down to hexabromobiphenyl) in soil near the Hexcel Chemical Corporation plant in New Jersey and the White Chemical Company plant in New Jersey ranged from 40 to 4.6 mg/kg and from 1.14–4.25 mg/kg, respectively (DeCarlo 1979). PBB levels in surface soil samples from seven dairy farms in Michigan that spread contaminated manure on the fields ranged from 35 to 1,260 µg/kg, while the concentrations in surface soil of control farms (that did not use contaminated manure) were <25 µg/kg (Fries and Jacobs 1980).

Concentrations of PBBs in sediments upstream from the Michigan Chemical Corporation plant were below the detection limit (100 µg/kg) with the exception of one sample (Hesse and Powers 1978). The

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concentration of PBBs in sediment from one upstream sample was 350 µg/kg. Hesse and Powers (1978) explained that this higher value was due to contamination by upstream currents during periods of water level regulation at the St. Louis dam. The concentrations of PBBs in near shore sediment near the Michigan Chemical Corporation plant sewer outfall were #77.0 mg/kg. PBB concentrations in Pine River sediments downstream from the plant showed a gradual decrease from a maximum value of 9.2 mg/kg to a value of 0.1 mg/kg at a location 29 miles downstream from the plant outfall (Hesse and Powers 1978). Similarly, PBB concentrations in sediment samples from swamps and marshes adjacent to the White Chemical Company and Hexcel Chemical Corporation plants in New Jersey ranged from <10 µg/kg to 4.6 mg/kg (DeCarlo 1979). A sludge sample from the discharge treatment plant of the White Chemical Company contained 431 mg/kg of PBBs (DeCarlo 1979).

Polybrominated Diphenyl Ethers. Sediment concentrations of PBDEs tend to be dominated by higher brominated congeners (e.g., BDE-209) (deWit 2002). Temporal trends suggest that concentrations of PBDEs in sediments are increasing. No information was located on the concentration of PBDEs in soils.

Levels of PBDEs in sediments from the United States were not located. The concentrations of PBDEs in sediment samples from other countries are summarized in Table 6-3. Tetra-, penta-, hexa-, and decaBDEs have been observed in river sediments from Japan (Watanabe et al. 1986, 1987, 1995). The combined concentrations of tetra- and pentaBDEs ranged from 21–59 ng/g dry weight. The concentration of DeBDE ranged from <25 to 11,600 ng/g (dry weight) (deWit 2002). In 1999, sediment samples from several locations in the Netherlands contained BDE-47, -99, and -209 (de Boer et al. 2000). Concentrations ranged from 0.3 to 7.1 ng/g (dry weight) for BDE-47, not detected to 5.5 ng/g (dry weight) for BDE-99, and not detected to 510 ng/g (dry weight) for BDE-209. The concentration of PBDEs in suspended particulate matter ranged from not detected to 9 ng/g (dry weight) for BDE-47, not detected to 23 ng/g (dry weight) for BDE-99, and not detected to 4,600 ng/g (dry weight) for BDE-209 (de Boer et al. 2000). The concentration of several brominated flame retardants was measured in sediments collected from the mouths of major European rivers (de Wit 2002). Elevated levels of BDE-47 and BDE-99 were found in Humber and Mersey rivers (Great Britain). In two rivers of the Netherlands, the sum of BDE-47 and BDE-99 ranged 1.61 to 13.1 ng/g (dry weight). The highest hexaBDE levels (as BDE-153) were found in the river Seine (France), three rivers in the Netherlands, and the rivers Schelde (Belgium), Forth (Great Britain) and Ems (Germany); the concentration of

Table 6-3. Concentrations (in ng/g Ignition Loss) of Several PBDEs in Sediment and Suspended Particulate Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs	BDE-209	Reference
Sediment	Japan	No data	No data	No data	21–59 (dw)	<25–11,600	Watanabe et al. 1986, 1987, 1995
Sediment	Baltic Sea	ND–3.4	ND–2.4	ND–1.3	ND–5.4	ND	Nylund et al. 1992
Sediment	Upstream plastics plant	3.7	8.8	1.6	14.1	No data	Sellström and Jansson 1995
Sediment	Downstream plastics plant	780	1,200	270	2,250	No data	Sellström and Jansson 1995
Sediment	River Viskan, upstream and downstream textile industries	<2–50	<1–53	<0.4–19	ND–120	ND–16,000	Sellström et al. 1998a
Sediment	22 European river mouths	<0.17–6.2 (dw)	<0.19–7.0 (dw)	No data	No data	<0.51–1,800	de Wit 2002
Sediment	Seven rivers, Great Britain	<0.3–368 (dw)	<0.6–898 (dw)	No data	No data	<0.6–3,190	Allchin et al. 1999
Sediment	Netherlands, several sites	0.3–7.1 (dw)	<0.2–9 (dw)	No data	No data	<4–510 (dw)	de Boer et al. 2000b
Suspended particulates	Netherlands, several sites	<2–9 (dw)	<0.1–23 (dw)	No data	No data	<9–4,600 (dw)	de Boer et al. 2000b

Source: de Wit (2002)

dw = dry weight; ND = not detected

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BDE-153 ranged from 0.013 to 0.056 ng/g (dry weight) in these sediments. The concentrations of DeBDE were highest in sediment from the Seine, ranging from 2.4 to 3.9 ng/g (dry weight). The concentrations of DeBDE in River Mersey (Great Britain), Schelde and River Liffey (Ireland) ranged from 34 to 1,800 ng/g (dry weight). In the southern Baltic Sea (Bornholm Deep), the upper layer of sediment was analyzed for BDE-47, -99, and -100; the combined concentration of these three congeners was 0.52 ng/g (dry weight) (Nylund et al. 1992).

A well-studied sediment core collected from the southern part of the Baltic Sea proper was analyzed for PBDEs and a number of organochlorine contaminants (Nylund et al. 1992). The analysis revealed a retrospective temporal trend from 1939 to 1987 and showed that the PBDE levels (i.e., sum of BDE-47, -99, and -100) have increased, with a sharp increase after 1980. The PBDE level in the sample from 1989 was 2.9 ng/g (Nylund et al. 1992). Measurable amounts of BDE-28, -47, -66, -99, and -100 were found in sediment cores from a freshwater lake in Germany, the Wadden Sea (the Netherlands), and Drammenfjord (Oslo Fjord, Norway) (Zegers et al. 2000). Samples from the Drammenfjord and freshwater lake also contained BDE-153 and BDE-154, and the Wadden Sea and freshwater lake samples contained BDE-209. The lower brominated PBDEs appear in the 1960s, and BDE-209 appears about 10 years later. The Drammenfjord sediment core shows increasing levels of BDE-47 starting in the 1940s (range, 0.02–0.18 ng/g dry weight) and increasing levels of BDE-99 (range, 0.5–0.28 ng/g dry weight), BDE-100 (range, not detected–0.07 ng/g dry weight), and BDE-154 (range, not detected–0.06 ng/g dry weight) beginning in the 1950s up to 1999. In the sediment core from Lake Woserin, lower brominated BDE congeners are detected beginning in the late 1950s, increase until the late 1970s, and then level off when BDE-209 first appears. A similar leveling-off trend is also observed in the Wadden Sea core (Zegers et al. 2000).

6.4.4 Other Environmental Media

Polybrominated Biphenyls. Although the agriculture episode in Michigan involving contaminated feed occurred in May 1973, PBBs were not identified as the causative factor until April 1974 (Fries 1985b). PBB-containing meats, milk, butter, eggs, and cheese entered the human food chain for almost a year before the PBBs were identified. Concentrations of PBBs (on a fat basis) in milk samples collected from contaminated farms soon after PBB was identified ranged from 2.8 to 595 mg/kg (Cordle et al. 1978; Kay 1977). Concentrations of PBBs in other products processed from the contaminated milk were as follows: butter, 1–2 mg/kg; cheese, 1.4–15.0 mg/kg; and canned milk, 1.2–1.6 mg/kg (Cordle et al. 1978). In 1974, the levels of PBBs in eggs from contaminated farm premises were as high as 59.7 mg/kg (Kay

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1977). The levels of PBBs in poultry and cattle tissues from the contaminated farm collected in 1974 were 4,600 mg/kg and up to 2,700 mg/kg, respectively (Kay 1977). With the seizure and destruction of the contaminated farm animals and products, the levels of PBBs in consumer products showed a steady decline. For example, in 1975, among 18 milk samples, 13 cheese samples, and 14 butter samples taken in Michigan, only 3 butter samples exceeded the FDA guidelines of 0.3 mg/kg fat (Di Carlo et al. 1978). In 1975, PBBs were detected in 245/2,040 meat samples collected in Michigan, with only 24 samples containing levels >0.3 mg/kg fat (Di Carlo et al. 1978). Although 95% of 1,430 meat samples collected in Michigan in 1976 contained detectable PBBs, only 1 sample contained >0.6 mg/kg, and a market basket survey in Michigan showed detectable PBBs in only 1/102 meat samples (Di Carlo et al. 1978).

No PBBs were detected in several varieties of fish (carp, white sucker, Northern pike, bullhead, and bass) from the Alma Reservoir, which is upstream from the Michigan Chemical Corporation plant and above a dam that prevents fish from moving upstream (Hesse and Powers 1978). On the other hand, tissue samples from fish collected from the Pine River, #29 miles downstream from the plant, contained up to 1.33 mg PBBs/kg (wet weight in skinless fillets). There was no apparent change in PBB concentrations in fish between 1974 and 1976 (Hesse and Powers 1978). PBBs could be detected in fish from Pine River and other embayments and tributaries of Lake Huron in 1983. PBB concentrations in carp and other sedentary fish from embayments and tributaries of Lake Huron (including Pine River) and Lake Superior were determined (Great Lakes Water Quality Board 1989; Jaffe et al. 1985). PBBs were detected in the concentration range of 15–15,000 µg/kg (fat basis) in fish from embayments and tributaries of Lake Huron, but not from Lake Superior.

PBB concentrations in whole (with skin) and skinless ducks collected within 2 miles of the Michigan Chemical Corporation plant in 1974–1977 ranged from not detected to 2.70 mg/kg and not detected to 1.8 mg/kg, respectively (Hesse and Powers 1978). Three bottlenose dolphins (*Tursiops truncatus*) collected during 1987–1988 from the U.S. mid-Atlantic contained PBBs at concentrations of 14–20 µg/kg (lipid basis) (Kuehl et al. 1991). The source of the PBBs in the dolphins was not given. The median concentrations of PBBs in 10 specimens of carcass and brain of bald eagles collected from 29 states in 1977 were 0.07 and 0.05 mg/kg, respectively (Kaiser et al. 1980). Twenty-two other specimens did not contain detectable levels (<0.03 mg/kg) of PBBs. The concentrations of PBBs in eggs of fish-eating birds (common tern, little gull, herring gull, and red-breasted mergansers) collected during 1975–1980 from nesting islands in northwestern Lake Michigan and Green Bay contained PBBs in the concentration range of 0.02–0.25 mg/kg (wet weight) (Heinz et al. 1983, 1985).

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White-tailed sea eagles collected from the Baltic Sea contained 280 ng PBBs/g lipid weight (Jansson et al. 1987). The concentration of PBBs in common guillemots (*Uria aalge*) collected in 1979–1981 from the Baltic Sea was 160 ng/g lipid (Jansson et al. 1987). Brunnich's guillemot (*Uria lomvia*), collected from Svalbard in the Arctic, contained 50 ng PBBs/g lipid (Jansson et al. 1987).

In German rivers, elevated levels of nona and octaBBs were present in fish. HexaBB was predominant in fish from the North Sea and Baltic Sea. BB-169 was found at a maximum concentration of 36 mg/kg fat in samples from the Baltic Sea. However, BB-169 was not found in waters from the North Sea or rivers. In Baltic marine fish, the concentrations of BB-153 ranged from 0.2–4.2 mg/kg lipid (de Boer et al 2000a).

In 1981, female ringed seals from Svalbard in the Swedish Arctic contained 4 ng PBBs/g lipid (Jansson et al. 1987). The level of PBBs in Baltic Sea harbor seal (*Phoca vitulina*) was 20 ng/g lipid; North Sea harbor seal contained 3 ng PBBs/g lipid (Jansson et al. 1987). The concentration of hexaBB ranged from 13–61 µg/kg wet weight from harbor seals collected from the North Sea (decaBB <1 µg/kg wet weight). In whitebeaked dolphins from the North Sea, the concentration of hexa-, penta-, and deca-BBs were 13, 8.3, and <0.9 µg/kg wet weight, respectively. Tetra-, penta-, and deca-BBs concentration ranges were 1.1–1.9, 0.4–0.9, and <0.5 µg/kg wet weight, respectively, in sperm whales from the Atlantic Ocean (de Boer et al 1999).

The quantitative determination of the concentrations of PBBs in blood, serum, adipose tissue, milk, and other body tissues or fluids is important in determining the human body burden of these chemicals. Fat is the largest repository of PBBs in the body, and concentrations in fat can provide an index of body burdens and exposure. It is simpler and less invasive to collect samples of serum or breast milk than body fat. However, the collection of milk and serum for the estimation of possible body burden has limitations. Breast milk can be obtained from limited segments of the population. Also the concentration of PBBs in breast milk can show considerable fluctuations because the breast is emptied only periodically (Brilliant et al. 1978; Willett et al. 1988). Serum, however, has lower PBB concentrations than body fat (see Section 3.5.1).

Hexabromobiphenyl was detected (detection limit 6.6 µg/kg) at a frequency of 8–57% in human adipose tissue samples from six Canadian Great Lakes municipalities in 1984 (Williams et al. 1988). The concentration of 2,2',4,4',5,5'-hexabromobiphenyl in adipose tissue samples pooled from tissues of the general population of the conterminous United States ranged from 1 to 2 µg/kg (Lewis and Sovocool 1982). PBB levels in the adipose tissues of 15 quarantined dairy farm residents in mid-Michigan (where

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the mix-up involving FireMaster BP-6 occurred) ranged from 0.104 to 174 mg/kg (Humphrey and Hayner 1975).

In the fall of 1993, the serum levels of BB-167 (2,2',4,4',5,5'-hexaBB) in 32 subjects, approximately 10 of whom consumed sport fish from the Great Lakes, were measured (Anderson et al. 1998). When the data were stratified by lake, on average, the Lake Huron fish consumers had the highest levels of PBBs (0.6 ppb) and Lake Erie fish consumers had the lowest (0.2 ppb). When the data were then stratified by state of residence, on average, Great Lake sport fish consumers who live in Michigan had the highest PBB level (0.7 ppb) and residents of Wisconsin had the lowest level (0.05 ppb).

In Michigan after the agriculture contamination episode in 1973–1974, the median PBB concentrations in blood of exposed adults and children in farms were 0.014 and 0.035 mg/kg (14 and 35 ppb), respectively, compared to corresponding median concentrations of 0.003 and 0.006 mg/kg (3 and 6 ppb) in a control group (Humphrey and Hayner 1975). PBB levels in the blood of quarantined farm workers in Michigan were also higher than in nonquarantined farm residents and the general population of Michigan (Cordle et al. 1978; Kimbrough 1987; Lambert et al. 1990; Landrigan et al. 1979). The concentration ratio of PBBs in adipose tissue over blood plasma for 13 paired specimens was 175 to 1 (Humphrey and Hayner 1975).

A cross-section of the population of Michigan was studied in 1978, 5 years following the agriculture episode involving FireMaster BP-6, to determine the levels of PBBs in human tissues. Levels of PBBs were highest in the part of state in which the episode occurred (median: adipose tissue, 500 µg/kg; serum, 1.7 µg/L) and were lowest in the upper peninsula (median: adipose tissue, 15 µg/kg; serum, 0.2 µg/L), farthest from the source of contamination. Levels in the rest of the state were in between (median: adipose tissue, 240 µg/kg; serum, 0.9 µg/L) (Wolff et al. 1982). The estimated concentration ratio of PBBs in adipose tissue over serum was near 300 among 31 Michigan dairy farm residents (Wolff et al. 1979a). The ratio of adipose tissue to serum PBB concentration was 363 to 1 for the general population and 100 to 1 in lactating women (Brilliant et al. 1978). The kinetics of fat metabolism in lactating women seems to alter PBB partitioning. The ratios of adipose tissue to serum PBB concentration for nonpregnant females and male chemical workers, farm workers and other males, and pregnant females in 3,683 Michigan residents with varying degrees of exposure were 190–260 to 1, 325–329 to 1, and 107–119 to 1, respectively (Eyster et al. 1983). The PBB ratios for cord to serum and placenta to serum in pregnant females were 0.10–0.14 to 1 and 0.10–0.17 to 1, respectively (Eyster et al. 1983). The PBB ratios for feces to serum and bile to serum in farm and chemical workers were 0.53–0.71 to 1 and 0.45–0.63 to 1, respectively (Eyster et al. 1983). The detection of PBBs in bile and feces indicates

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transfer into the intestinal tract. However, the concentration of PBBs in feces represented a minor proportion of the total body burden, indicating a slow rate of excretion (Eyster et al. 1983).

The concentrations of PBBs in the breast milk of females from the lower peninsula of Michigan (exposed area) were generally higher than in breast milk of females from the upper peninsula (farthest from the sources) (Brilliant et al. 1978). PBB levels in breast milk of five females from the exposed farms were 0.21–92.7 mg/kg (Cordle et al. 1978; Humphrey and Hayner 1976). In a cohort of Michigan residents, the ratio of PBBs in breast milk to maternal serum was 107–122 to 1 and in adipose tissue to breast milk was 1.1–1.5 to 1 (Eyster et al. 1983; Landrigan et al. 1979). The concentrations of PBBs found in human tissues and body fluids are given in Table 6-4. Recent levels of PBBs in human breast milk (i.e., 1990 to present) were not located (WHO 1994b).

In an attempt to determine the metabolites of PBBs, whole milk of lactating cows from contaminated areas of Michigan was analyzed for monohydroxy metabolites, but none were found (Gardner et al. 1976). In a later study, the feces of dogs fed FireMaster BP-6 in corn oil was found to contain a metabolite identified as 6-hydroxy-2,2',4,4',5,5'-hexabromobiphenyl (Gardner et al. 1979). However, the hydroxy compound decomposed into two pentabromodibenzofurans inside a gas chromatographic column heated to 230–260 EC.

The effects of processing cow's milk containing PBBs also has been studied (Murata et al. 1977; Zabik et al. 1978). Spray-drying reduced PBB levels in whole and skim milk, whereas pasteurization, freeze-drying, aging of cheese, and condensation were not effective in reducing the level of PBBs in milk products. Pressure cooking meat containing PBBs reduced the level of PBBs in the cooked meat (Zabik et al. 1978).

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Table 6-4. Tissue Levels of PBBs in Michigan Residents

Tissue	Subject(s)	Mean/median concentration ^a	Year	Reference
Serum	Exposed farm workers	14 µg/L	1976	Stross et al. 1979
	Chemical workers	48 µg/L	No data	Stross et al. 1981
	Chemical workers	1.1–\$1,000 µg/L	1976	Anderson et al. 1978d
	Exposed farm workers	BDL to \$1,000 µg/L	1976	Anderson et al. 1978d
	Residents from quarantined farms	26.9 µg/L	1976–1977	Landrigan et al. 1979
	Residents from nonquarantined farms	3.5 µg/L	1976–1977	Landrigan et al. 1979
	Farm product consumers	17.1 µg/L	1976–1977	Landrigan et al. 1979
	Chemical workers and families	43.0 µg/L	1976–1977	Landrigan et al. 1979
	Control group	3.5 µg/L	1976–1977	Landrigan et al. 1979
	General population (lower peninsula)	1.9 µg/L	1978	Wolff et al. 1982
	General population (upper peninsula)	0.2 µg/L	1978	Wolff et al. 1982
	General population (remainder of state)	0.9 µg/L	1978	Wolff et al. 1982
	Chemical workers	25.4 µg/L	No data	Eyster et al. 1983
	Farm and other workers	5.4 µg/L	No data	Eyster et al. 1983
	Mothers in lower peninsula	26.2 µg/L	1976–1977	Landrigan et al. 1979
	Cord serum	Exposed mothers from farms	3.4 µg/L	No data
Nonpregnant women from exposed farms		3.1 µg/L	No data	Eyster et al. 1983
Exposed mothers in lower peninsula		3.2 µg/L	1976–1977	Landrigan et al. 1979
Blood plasma	Mothers in lower peninsula	<1.0 µg/L	No data	Eyster et al. 1983
	Workers from quarantined farms	14 µg/L	1974	Humphrey and Hayner 1975
	Children from quarantined farms	35 µg/L	1974	Humphrey and Hayner 1975
	Adults from nonquarantined farms	3 µg/L	1974	Humphrey and Hayner 1975
Placenta	Children from nonquarantined farms	6 µg/L	1974	Humphrey and Hayner 1975
	Exposed mothers	<1 µg/L	No data	Eyster et al. 1983

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Table 6-4. Tissue Levels of PBBs in Michigan Residents (*continued*)

Tissue	Subject(s)	Mean/median concentration ^a	Year	Reference
Breast milk	Exposed mothers	370 µg/kg (fat basis)	No data	Eyster et al. 1983
	Exposed mothers in lower peninsula	3,614 µg/kg (fat basis)	1976–1977	Landrigan et al. 1979
	Mothers from lower peninsula	68 µg/kg (fat basis)	1976	Brilliant et al. 1978
	Mothers from upper peninsula	<44 µg/kg (fat basis)	1976	Brilliant et al. 1978
Adipose tissue	Population of lower peninsula	500 µg/kg	1978	Wolff et al. 1982
	Population of upper peninsula	15 µg/kg	1978	Wolff et al. 1982
	Population of rest of the state	240 µg/kg	1978	Wolff et al. 1982
	Chemical workers	9,330 µg/kg	No data	Brown et al. 1981
	Farm residents in lower peninsula	3,940 µg/kg	No data	Brown et al. 1981
	Farm residents in lower peninsula	3,260 µg/kg	1976	Stross et al. 1979
	Chemical workers	12,820 µg/kg	No data	Stross et al. 1981
	Workers from quarantined dairy farms	12,500 µg/kg	1974	Humphrey and Hayner 1975
	Pregnant females from lower peninsula	400 µg/kg	No data	Eyster et al. 1983
	Chemical workers	5,290 µg/kg	No data	Eyster et al. 1983
	Farm and other workers in lower peninsula	1,650 µg/kg	No data	Eyster et al. 1983

^aWhen both mean and median values are available, the former values have been used in the table. In some cases, when neither value is available, the range is given in the table.

BDL = below detection limit

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Polybrominated Diphenyl Ethers. Information about the concentrations of PBDEs in food stuffs is very limited. Recently, Ohta et al. (2002) determined the concentration of total PBDEs in vegetables and meat samples from Japan. The concentrations of PBDEs in spinach, potato, and carrot were 134, 47.6, and 38.4 pg/g fresh weight, respectively. The highest concentrations of total PBDEs and BDE-47 were found in spinach. Interestingly, different congener patterns were found among the vegetables analyzed. Compared to root vegetables, which had high concentrations of BDE-153, spinach (representing a leafy vegetable) might be strongly influenced by PBDE contamination in air. The concentration of PBDEs in pork, beef, and chicken were 63.6, 16.2, and 6.25 pg/g fresh weight, respectively. PBDE concentrations were highest in pork samples; however, the reason for this is unknown (Ohta et al. 2002). Four types of commercial fish oils sold in Sweden were found to contain PBDEs (0.2-28.1 ng/g lipid weight) (Haglund et al. 1997). The highest concentration of PBDEs was found in the cod liver oil. These oils were from products marketed as dietary supplements for humans.

The concentrations of PBDEs in biosolids (sewage sludge) and effluent samples are summarized in Table 6-5. PBDEs were detected in biosolids from four different regions of the United States (Pardini et al. 2001). The total concentrations of PeBDE in biosolids ranged from 1,100 to 2,290 µg/kg dry weight; the levels of PeBDE were high and consistent, regardless of the region of origin. The concentration of DeBDE (BDE-209) varied widely among biosolids from different regions; the concentration of BDE-209 ranged from 84.8 to 4,890 µg/kg dry weight in the biosolid samples. Sewage sludge samples from 13 waste water treatment plants in Germany were sampled (Hagenmaier et al. 1992). The mean concentration of tri- to hepta-BDEs was 8.37 ng/g with tri-, tetra-, penta-, hexa-, and heptaBDE at concentrations of 0.65, 3.06, 3.02, 0.49, and 0.22 ng/g, respectively. Levels of penta- and hexaBDEs were highest in these samples.

Monitoring data indicated that the levels of PBDEs are increasing in freshwater aquatic organisms with higher concentrations near point sources. The congener profiles show the highest levels for BDE-47. The presence of PBDEs in freshwater aquatic organisms taken from remote regions suggests that diffuse sources of PBDEs are also important. The concentrations of PBDEs in freshwater fish samples are summarized in Table 6-6. Fish were sampled from two U.S. lakes, Hadley Lake, Indiana near a possible PBDE point source, and Lake of the Ozarks, Missouri, with no known sources (Dodder et al. 2000a). Mean total PBDE concentrations (sum of BDE-47, -99, -100, -153, and -154) were higher in crappie (*Poxomis annularis*) and bluegill (*Lepomis macrochiras*) from Hadley Lake (1,500 and 1,900 ng/g lipid weight, respectively) than from Lake of the Ozarks (340 and 390 ng/g lipid, respectively). BDE-47, -99,

Table 6-5. Concentrations (in ng/g Dry Weight) of Several PBDEs in Biosolids (Sewage Sludge) and Effluent Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Sewage sludge	Gothenburg	15	19	3.5	38	No data	Nylund et al. 1992
Sewage sludge	Klippan	22	18	5.4	45.4	No data	Sellström 1999; Sellström and Jansson 1995
Sewage Sludge	Rimbo	53	53	13	119	No data	Sellström 1999; Sellström and Jansson 1995
Sewage sludge	Three plants, Stockholm	39–91	48–120	11–28	98–239	140–350	Sellström et al. 1999
Sewage sludge	Germany	No data	No data	No data	04–15*	No data	Hagenmaier et al 1992
Sewage treatment plant effluent residues	Netherlands, several sites	11–35	<1	No data	11–35	310–920	de Boer et al. 2000b

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

Table 6-6. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Freshwater Fish Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Whitefish	Lake Störvindeln	15	7.2	3.9	26	No data	Sellström et al. 1993
Arctic char	Lake Vättern	400	64	51	520	No data	Sellström et al. 1993
Pike	Lake Bolmen, 1993	65	42	19	130	No data	Kierkegaard et al. 1993
Pike	Dalslands canal	94–98	60–79	25–36	180–210	No data	Sellström et al. 1993
Pike	River Viskan, upstream and downstream	<46–2,000	<37–1,600	<14–1,000	<130–4,600	Trace	Sellström et al. 1998a
Trout	Dalslands canal	120–460	130–590	33–150	280–1,200	No data	Sellström et al. 1993
Carp	United States	No data	No data	No data	13–22* (fw)	No data	Loganathan et al. 1995
Steelhead trout	Lake Michigan, United States	1,700	600	360	3,000*	No data	Asplund et al. 1999b
Lake trout	Lake Ontario, United States	No data	No data	No data	540*	No data	Alaee et al. 1999
Lake trout	Lake Huron, United States	No data	No data	No data	240*	No data	Alaee et al. 1999
Lake trout	Lake Superior, United States	No data	No data	No data	140*	No data	Alaee et al. 1999

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Table 6-6. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Freshwater Fish Samples (continued)

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Lake trout	Lake Erie, United States	No data	No data	No data	117*	No data	Alaee et al. 1999
Crappie	Hadley Lake, Indiana	250	430	150	1,500*	No data	Dodder et al. 2000
Bluegill	Hadley Lake, Indiana	420	320	240	1,900	No data	Dodder et al. 2000
Crappie	Lake of the Ozarks, Missouri	190	78	59	340*	No data	Dodder et al. 2000
Bluegill	Lake of the Ozarks, Missouri	200	91	59	390	No data	Dodder et al. 2000
Eels	Netherlands	<20–1,400	No data	No data	<50–1,700	No data	de Boer 1990
Bream	Netherlands (several sites)	0.2–130 (dw)	Not detected	No data	No data	No data	de Boer et al. 2000b
Several fish species	Germany	No data	No data	No data	19–983*	No data	Krüger 1988
Osprey	Sweden	1,800	140	200	2,140	No data	Sellström et al. 1993

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

dw = dry weight; fw = fresh weight

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-153, and -154 were primary congeners. From the Lake of Ozarks, BDE-47 was the dominant congener in fish. The total PBDE concentrations in smelt (*Osmerus mordax*) from Lakes Superior and Ontario were 150 ± 9 and 240 ± 30 ng/g lipid, respectively (Dodder et al. 2002). The dominant congeners in these fish were BDE-47 and -99. An analysis of fish tissue samples from selected locations in Washington State showed that total PBDE concentrations ranged 29 ng/g lipid in rainbow trout from a remote spring-fed stream (Douglas Creek, Washington) to 19,000 ng/g lipid in rainbow trout from the urbanized Spokane River, Washington (Johnson and Olson 2001). The tetra- and pentaBDE isomers were the major compounds present. TeBDE to hexaBDE were found in carp (*Cyprinus carpio*) from the Buffalo River (New York), a polluted area around the Great Lakes (Loganathan et al. 1995). TeBDEs dominated the congener pattern with 94–96% of total PBDEs. TeBDE and PeBDE concentrations ranged from 13 to 22 ng/g fresh weight. Asplund et al. (1999a) found tri- to hexaBDEs in steelhead trout (*Oncorhynchus mykiss*) sampled in 1995 from Lake Michigan. The combined concentration of BDE-47, -99, -100, -153, and -154 was 3,000 ng/g lipid weight (Asplund et al. 1999b). Lake trout (*Salvelinus mayanycush*) from Lakes Ontario, Huron, and Superior were also found to have di- to heptaBDEs with combined concentrations of 545, 237, and 135 ng/g lipid weight, respectively (Alaee et al. 1999). Lake trout from Lake Erie had 117 ng/g lipid weight (Luross et al. 2000). Variations in local sources, combined with atmospheric transport, may explain differences that were seen in congener profiles for the different lakes. A retrospective temporal study for the years 1978, 1983, 1988, 1993, and 1998 using archived trout samples from Lake Ontario show a dramatic increase in total PBDE concentrations over time (Luross et al. 2000). At 50 fresh water sites in Virginia, muscle samples from 253 fish samples were collected and analyzed for PBDEs (Hale et al. 2000, 2001). Approximately 85% of the samples contained BDE-47, the predominant congener, at measurable concentrations. Concentrations were $>1,000$ ng/g lipid weight at 9 of 50 sites. The highest combined PBDE concentrations (up to 57,000 ng/g lipid weight) were observed in carp downstream of textile and furniture facilities. BDE-47 levels were greater than PCB-153 concentrations in 58% of the samples analyzed.

Between 1986 and 1988, levels of BDE-47, -99, and -100 were measured in whitefish (*Coregonus spp.*) from a remote mountain lake in Northern Sweden (Lake Storvindeln), Arctic char (*Salvelinus alpinus*) from a heavily populated lake (Lake Vättern) in south-central Sweden with numerous municipal and industrial point sources, and in trout (*Salmo trutta*) and pike (*Esox lucius*) from several sites along Dalslands Canal in west central Sweden (Jansson et al. 1993). No point sources of PBDEs were identified from these sites. Whitefish from the remote lake contained the lowest levels (26 ng/g lipid weight) of PBDEs, whereas the Arctic char, from a heavily populated lake, contained 520 ng/g lipid weight PBDEs. In both samples, BDE-47 was the predominant congener. PBDE concentration levels in pike and trout from the Dalslands Canal ranged from 180 to 210 ng/g lipid weight and from 280 to

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1,200 ng/g lipid weight, respectively. The congener pattern in these samples was similar to the technical mixture, Bromkal 70-5DE, with equal quantities of both BDE-47 and BDE-99. The levels in pike and trout are of the same order of magnitude as in the Arctic char, indicating the spread of PBDEs from diffuse sources (de Wit 2002). In 1979 and 1980, high levels of tri- to hexaBDEs (range, 950–27,000 ng/g lipid weight in muscle tissues) were measured in fish sampled along a river in Sweden (Viskan) where numerous textile industries are located (Andersson and Blomkvist 1981). These textile industries have used PBDEs in the production of textiles. BDE-47 was the predominant congener at 70–80% of the total PBDEs. In 1977, the PBDEs were not detected in fish sampled at the same sites. The elevated levels of BDE-47, -99, and -100 were later confirmed in a follow-up study where fish were caught from approximately the same locations (Sellström et al. 1993). In the current study, BDE-47 was the predominant congener at 65–96% of total PBDEs. Several fish species were sampled (pike, perch, bream, eel, tench, and sea trout) in these studies. In 1995, fresh samples of pike and sediments were collected at four of eight sites along River Viskan in order to search for point sources of contaminants. The combined concentrations of BDE-47, -99, and -100 ranged from not detected to 4,600 ng/g lipid weight, with BDE-47 being the predominant congener (50–90% of total). BDE-209 was found in a few fish at trace amounts. The lowest levels of the PBDEs were found upstream of the industries. The concentrations of PBDEs increased further downstream as more industries were passed (Sellström et al. 1998). Levels of BDE-47 ranged from <20 to 1,700 ng/g lipid in eels (*Anguilla anguilla*) from Dutch rivers and lakes (at 10 locations); BDE-47 comprised 70% of the total PBDEs (de Boer 1990). Bream (*Abramais brama*) sampled from several sites in the Netherlands had concentrations of BDE-47 ranging from 0.2 to 130 ng/g dry weight (de Boer et al. 2000). BDE-99 was below the detection limits. BDE-153 ranged from <0.04 to 4.1 ng/g dry weight. Allchin et al. (1999) conducted a study of PBDEs in plaice (*Pleuronectes platessa*), flounder (*Platichys flesus*), and dab (*Limanada limanada*) collected in the estuaries of rivers in the United Kingdom. Suspected sources of PBDEs in the estuaries include a manufacturer of PeBDE and OBDE, several industries using PeBDE, and several landfills receiving wastes suspected to contain PBDEs. Levels of BDE-47, BDE-99, PeBDE (as technical mixture DE-71), and OBDE (as technical mixture DE-79) in fish ranged from not detected to 9,500 ng/g lipid weight, not detected to 370 ng/g lipid weight, 47 to 1,200 ng/g lipid weight, and not detected to 1,200 ng/g lipid weight. The highest levels were at Tees Bay downstream from a manufacturing plant on the River Tees. These results are similar to the situation found in Sweden along the River Viskan (Andersson and Blomkvist 1981; Sellström et al. 1993).

Freshwater mussels (*Dreissena polymorpha*) were collected at several locations in the Netherlands and analyzed for BDE-47, -99, -153, and -209 (de Boer et al. 2000). Concentration ranges for the congeners

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were 0.7–17, 0.4–11, and <0.1–1.5 ng/g dry weight for BDE-47, -99, and -153, respectively. BDE-209 was below the detection limit.

No identifiable temporal or spacial trends were found for PBDE levels in marine aquatic species. The congener profiles show the highest levels for BDE-47. The levels of several PBDEs in marine aquatic species are summarized in Table 6-7. Fall-caught herring (*Clupea harengus*) muscle from five sites along the Swedish coast was analyzed from BDE-47, -99, and -100; the combined concentration of these three congeners ranged from 17 to 61 ng/g lipid, with BDE-47 being the dominate congener (Sellström et al. 1993). Likewise, the concentration of BDE-47 in Baltic herring ranged from 3.2 to 27 ng/g lipid in different age groups; the combined concentration of BDE-47, -99, and -100 ranged from 3.2 to 32 ng/g lipid (Haglund et al. 1997); 2-year-old herring had the lowest levels and 5-year-old herring had the highest levels. Similarly, Strandman et al. (1999) observed increasing concentrations with age of BDE-47, -99, and -153 in Baltic sprat (*Sprattus sprattus*, age 3–13 years). However, this trend was not seen for herring. BDE-47 was the primary congener with levels ranging from 7.6 to 24 ng/g lipid weight for 1- to 3-year-old sprat, 17–140 ng/g lipid weight for 3- to 13-year-old sprat, and 7.6–24 ng/g lipid weight in herring. The concentrations of BDE-47, -99, and -100 in whole-body composites of herring were 6.21, 0.62, and 0.81 ng/g lipid, respectively; in sprat, the concentrations were 4.32, 0.71, and 0.80 ng/g lipid, respectively (Burreau et al. 1999). Baltic sea herring had similar levels of BDE-47 (46.3 ng/g lipid) compared to 8.4–100 ng/g lipid of BDE-47 found by de Boer (1990) for herring collected from three regions of the North Sea. BDE-47, -99, and -153 concentrations in Baltic salmon (*Salmo salar*) muscle were 167, 52, and 4.2 ng/g lipid, respectively (Haglund et al. 1997). BDE-47, -99, and -100 levels were 47, 7.2, and 6.3 ng/g lipid, respectively, in whole-body composites (Burreau et al. 1999). In another study, the levels of BDE-47, -99, and -100 were determined in muscle, ripe eggs, and blood plasma from Baltic salmon (Asplund et al. 1999a). The mean concentrations of PBDEs in tissues from Baltic salmon (ng/g lipid weight) were as follows: BDE-47 (muscle, 190; ripe eggs, 64; blood, 190), BDE-99 (muscle, 52; ripe eggs, 16; blood, 55), and BDE-100 (muscle, 46; ripe eggs, 18; blood, 59). Cod (*Gadus morhua*) liver samples at three locations of the North Sea had combined levels of BDE-47 and BDE-99 of 1.9–360 ng/g lipid (de Boer 1989). From several sites in the Netherlands, BDE concentrations in flounder were 0.6–20 ng/g dry weight for BDE-47 and <0.01–4.6 ng/g dry weight for BDE-99 (de Boer et al. 2000); concentrations of BDE-153 and BDE-209 were below the detection limit. In 1996, de Boer et al. (2001) measured the levels of two BDE congeners in flounder liver samples from

Table 6-7. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Marine Aquatic Species

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Herring	Baltic Sea	19–38	7.8–17	3.4–6	30–61	No data	de Wit 2002 Sellström et al. 1993
Herring	Baltic Sea	3.2–27	ND–2.9	1.3–1.9	3.2–32	No data	Haglund et al. 1997
Herring	Baltic Sea	7.6–24	4.3–3.9	No data	12.9–28.3*	No data	Strandman et al. 1999
Herring	Baltic Sea	6.3	0.6	0.8	12*	No data	Burreau et al. 1999
Herring	Kattegatt	12	3.4	1.6	17	No data	de Wit 2002; Sellström et al. 1993
Herring	North Sea	8.4–100	No data	No data	No data	No data	de Boer 1990
Sprat (different age groups)	Baltic Sea	17.5–140.8	1.9–9.5	No data	21–149*	No data	Strandman et al. 1999
Sprat	Baltic Sea	4.3	0.7	0.8	8.4*	No data	Burreau et al. 1999
Cod liver	North Sea	170	No data	No data	1.9–360	No data	de Boer 1989
Salmon	Baltic Sea	167	52	44	220	No data	Haglund et al. 1997
Salmon	Baltic Sea	190	52	46	290	No data	Asplund et al. 1999b
Salmon	Baltic Sea	46	7.3	6.4	86*	No data	Burreau et al. 1999
Several fish species	Japan	No data	No data	No data	0.1–17*	No data	Watanabe et al. 1987
Yellowfin tuna	Japan	0.5	0.4	0.25	1.9*	No data	Ohta et al. 2000
Yellowtail	Japan	17	4.5	4.0	30.5*	No data	Ohta et al. 2000

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Table 6-7. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Marine Aquatic Species (*continued*)

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Yellowtail (cultured)	Japan	29	3.3	5.3	44*	No data	Ohta et al. 2000
Salmon	Japan	22	8.1	5.3	46*	No data	Ohta et al. 2000
Several flatfish	Seven river estuaries, Great Britain	73–9,500	16–790	No data	No data	ND	Allchin et al. 1999
Flounder	Netherlands, several sites	0.6–20 (dw)	<0.01–4.6	No data	No data	No data	de Boer et al. 2000b

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

dw = dry weight; ND = not detected

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the Amsterdam and Rotterdam harbors, and off the Dutch coast; BDE-47 and -99 ranged from 15 to 280 and <2 to 24 ng/g lipid weight, respectively. Olsson et al. (1999) detected BDE-47 in perch (*Perca fluviatilis*) from Latvia in a study examining environmental contamination in coastal areas of the former Soviet Union; the concentration of BDE-47 ranged from 6.4 to 10 ng/g lipid weight in the perch.

Watanabe et al. (1987) detected PBDEs in numerous marine fish and shell fish in Japan. TeBDE and PeBDE levels ranged from 0.1 and 17 ng/g fresh weight, with TeBDE being the major congener. DeBDE was also detected in a mussel sample from Osaka Bay (at 1.4 µg/kg wet weight). Recently, Japanese market fish were analyzed for PBDEs. The highest combined PBDE levels (BDE-28, -47, -99, -100, -153, and -154) were in salmon, cultured yellowtail, and wild yellowtail muscle (46, 44, and 30.5 ng/g lipid weight, respectively) and lowest levels in yellowfin tuna (1.9 ng/g lipid weight) (Ohta et al. 2000). BDE-47 was major congener in all samples. In another study, several fish species from Japan were analyzed for 15 BDE congeners (Hori et al. 2000). The PBDE levels ranged from 0.00136 to 2.1 ng/g fresh weight, with BDE-47 as the predominant congener. Seven species of marine fish (conger eel, flounder, gray mullet, horse mackerel, red sea bream, sea bass, and yellowtail) were collected from the Inland Seas of Seto, Japan (Akutsu et al. 2001). Seven PBDEs (BDE-28, -47, -66, -99, -100, -153, and -154) were detected in all samples with BDE-47 being the most abundant congener. Levels of total PBDEs in gray mullets and yellowtails were 63 and 15 ng/g lipid weight, respectively.

Marine mussels (*Mytilus edulis*) collected at several locations in the Netherlands and analyzed for BDE-47, -99, -153, and -209 (de Boer et al. 2000). Concentrations of BDE-47 and -99 ranged from 0.9 to 4.3 ng/g dry weight and from 0.3 to 1.6 ng/g dry weight, respectively. BDE-153 and BDE-209 were not detected. Di- to heptaBDE were analyzed for in hepatopancreas samples from Dungenes crab from several sites on the Strait of Georgia, British Columbia, Canada (Ikonomou et al. 1999). The primary congener detected was BDE-47. The combined concentration of BDE-47 and BDE-99 was approximately 100–350 ng/g lipid weight.

In marine animals, temporal trends show increasing levels of PBDEs with higher levels found near human populated areas. In all marine animal studies, the congeners profile show the highest levels for BDE-47. The concentrations of several PBDEs in marine animals are summarized in Table 6-8. PBDEs have been detected in several species of seal from several different sites. In the Baltic Sea, female grey seals (*Halichoerus grypus*) sampled in 1979–1985 contained 730 ng PBDE/g lipid in their blubber (sum of BDE-47, -99, and -100) (Jansson et al. 1993); male grey seals had 280 ng PBDE/g lipid weight

Table 6-8. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Marine Animals

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Galaucous gull	Bear Island, Norway (Arctic)	290–634	160	No data	No data	No data	de Wit 2002
Brunnich's guillemot	Svalbard	No data	No data	No data	130	No data	Jansson et al. 1987
Cormorant liver	Rhine delta	No data	No data	No data	28,000 (fw)	No data	de Boer 1990
Cormorant	England, United Kingdom	170–3,500	50–250	50–1,500	300–6,400*	No data	Allchin et al. 2000
Grey seal	Baltic Sea	650	40	38	730	No data	de Wit 2002; Sellström et al. 1993
Grey seal	Baltic Sea	308	54	57	419	No data	Haglund et al. 1997
Grey seal	Baltic Sea	No data	No data	No data	208	No data	Andersson and Wartanian 1992
Ringed seal	Baltic sea	256	33	61	350	No data	Haglund et al. 1997
Ringed seal	Baltic sea	No data	No data	No data	320	No data	Andersson and Wartanian 1992
Ringed seal	Svalbard	47	1.7	2.3	51	No data	de Wit 2002; Sellström et al. 1993
Ringed seal	Canadian Arctic	No data	No data	No data	25.8–50*	No data	Alaee et al. 1999
Ringed seal	Holman Island, Northwest Territories, Canada	2.8	No data	No data	2.4–4.9*	No data	Ikonomou et al. 2000
Harbor seal	Baltic Sea	No data	No data	No data	90	No data	Jansson et al. 1987

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Table 6-8. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Marine Animals (continued)

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Harbor seal	Skagerrak	No data	No data	No data	230	No data	Andersson and Wartanian 1992
Harbor seal	North Sea	390–4,900	42–660	25–450	600–6,000	No data	de Boer et al. 1998
Harbor seal	San Francisco Bay, California	46–6,682	17–303	No data	No data	No data	She et al. 2000
Bottlenose dolphin	South Atlantic Ocean	No data	No data	No data	180–220	No data	Kuehl et al. 1991
Bottlenose dolphin	Gulf of Mexico	No data	No data	No data	8,000	No data	Kuehl and Haebler 1995
Whitebeaked dolphin	Netherlands	5,500	1,000	1,200	7,700	No data	de Boer et al. 1998
Harbor porpoise	British Columbia, Canada	50–1,200	No data	No data	350–2,300*	No data	Ikononmou et al. 2000
Harbor porpoise	England and Wales, United Kingdom	227–6,790	No data	No data	440–7,670	No data	Law et al. 2000
Beluga whale	Canadian Arctic	No data	No data	No data	81–160*	No data	Alaee et al. 1999
Beluga whale	Southeast Baffin, Canada	10	0.9	1.6	15*	No data	Stern and Ikononmou 2000
Long-finned pilot whale	Faeroe Islands	410–1,780	160–600	87–280	843–3,160*	No data	Lindström et al. 1999
Long-finned pilot whale	Faeroe Islands	66–860	24–170	12–98	126–1,250*	No data	van Bavel et al. 1999

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Table 6-8. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Marine Animals (*continued*)

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Minke whale	Netherlands	630	160	79	870	No data	de Boer et al. 1998
Sperm whale	Netherlands	130–250	32–64	21–35	187–349	No data	de Boer et al. 1998

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

fw = fresh weight

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(Andersson and Wartanian 1992). Male ringed seals (*Pusa hispida*) from the Baltic Sea had 320 ng PBDE/g lipid weight (Andersson and Wartanian 1992). Baltic gray and ringed seal blubber sampled between 1981 and 1988 contained 419 and 350 ng PBDEs/g lipid (total of BDE-47, -99, and -100), respectively (Haglund et al. 1997). In 1981, female ringed seals from Svalbard in the Swedish Arctic contained 40–51 ng PBDEs/g lipid in blubber (Jansson et al. 1987, 1993; Sellström et al. 1993). Higher levels of PBDEs are generally seen in Baltic Sea ringed seals (320–350 ng/g lipid) (Andersson and Wartanian 1992; Haglund et al. 1997) compared to Arctic ringed seals (26–51 ng/g lipid) (Alaee et al. 1999; Jansson et al. 1987). The level of PBDEs in harbor seals from Skagerrak on the Swedish west coast was 230 ng PBDE/g lipid (Andersson and Wartanian 1992). She et al. (2000) analyzed the concentration of BDE-47, -99, and -153 in harbor seal from the San Francisco Bay area (She et al. 2000). Mean concentrations for BDE-47, -99, and -153 were 1,124, 107, and 50 ng/g lipid weight, respectively. Alaee et al. (1999) found that ringed seal from the Canadian Arctic had mean PBDE concentrations (sum of di- to hexaBDEs) of 25.8 ng/g lipid weight (females) and 50.0 ng/g lipid weight (males). The lower levels in female seals suggests that PBDEs are transferred to young through breast milk. On Holman Island, North West Territory, Canada (Arctic) in 1996, ringed seal had total PBDE concentrations of 2.4–4.9 ng/g lipid for males. The levels of PBDEs were found to increase with age (Ikononmou et al. 2000). In a retrospective temporal trend study, archived samples of blubber from ringed seals from Holman Island, North West Territory, Canada were analyzed for PBDE levels. The concentration of PBDE in samples collected between 1981 and 1996 increased from approximately 0.3 ng/g lipid weight in 1981 to 3.6 ng/g lipid weight in 1996 (Ikononmou et al. 2000).

The levels of PBDEs have recently been determined in harbor porpoises (*Phocaena phocaena*) from British Columbia, Canada (Ikononmou et al. 2000) and from the coasts of England and Wales (Law et al. 2000). In British Colombia (Canada) samples, the total PBDE levels (sum of tri- to hepta-congeners) were 350–2,300 ng/g lipid weight; BDE-47 was found at the highest levels in these samples (range, 50–1,200 ng/g lipid weight) (Ikononmou et al. 2000). Concentrations of total PBDEs (sum of 13 congeners) along the coast of England and Wales, ranged from 450 to 7,670 ng/g lipid weight, with BDE-47 levels ranging from 227 to 6,790 ng/g lipid weight (Law et al. 2000).

During a mass mortality event on the south Atlantic coast in 1987–1988, blubber samples were collected from three bottlenose dolphins (*Tursiops truncatus*); these samples contained 180–220 ng PBDEs/g lipid (Kuehl et al. 1991). Blubber samples, taken from stranded bottlenose dolphins from several locations around the Gulf of Mexico in 1990, contained 3,110 ng PBDEs/g lipid (Kuehl and Haebler 1995). On the Dutch coast in early 1998, de Boer et al. (1998) found PBDEs in blubber of one whitebeaked dolphin

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(*Lagenorhynchus albirostris*); the levels of BDE-47, -99, and -100 were 5,500, 1,000, and 1,200 ng/g lipid weight, respectively.

The concentration of 19 PBDEs was determined in long-finned pilot whale (*Globicephala melas*) from the Faeroe Islands in the north Atlantic (Lindström et al. 1999). Young males and females had the highest levels, ranging from 3,000 to 3,160 ng/g lipid; lower levels were observed for both adult females (840–1,050 ng/g lipid) and males (1,610 ng/g lipid). The predominant isomers in all samples were BDE-47 and BDE-99, accounting for 70% of the sum of 19 congeners. van Bavel et al. (1999) also studied the levels of PBDEs in long-finned pilot whales. They observed a similar trend with young animals having higher PBDE concentrations (740 ng/g lipid weight) and adult animals having lower levels (females, 230 ng/g lipid; males, 540 ng/g lipid). In Beluga whales sampled in 1997 from southeast Baffin (Cumberland Sound), the levels of total PBDEs and BDE-47 were 15 and 10 ng/g lipid weight, respectively (Stern and Ikononou 2000). Between 1982 and 1997, total PBDE concentrations in archived blubber samples of beluga whales from southeast Baffin Canada increased significantly. For this time period, BDE-47, -99, -100, -154, and total PBDEs increased by factors of 6.5, 10.3, 7.9, 30.6, and 6.8, respectively (Stern and Ikononou 2000). Three sperm whales (*Pyseter macrocephalus*) and one minke whale (*Balaenaoptera acutorostrata*) found stranded on the Dutch coast in early 1998 were analyzed for PBDEs (de Boer et al. 1998a, 1998b). Exposure to PBDEs for these animals occurred in the deep Atlantic through the food web. The concentrations of PBDEs in these marine mammals were as follows: sperm whale (BDE-47, 130–250 ng/g lipid weight; BDE-99, 32–64 ng/g lipid weight; and BDE-100, 21–35 ng/g lipid weight) and minke whale (BDE-47, 630 ng/g lipid weight; BDE-99, 160 ng/g lipid weight; BDE-100, 79 ng/g lipid weight); BDE-209 (DeBDE) was below detection limits in all samples.

Increasing levels of PBDEs have been found in marine birds and eggs, with BDE-47 found at the highest levels. Di- and triBDE have been detected, but not quantified, in black skimmer (*Rynchops nigra*) tissues and eggs in the United States (Stafford 1983). Ospreys (*Pandion haliaetus*), found dead at various locations around Sweden, were pooled and analyzed for PBDEs in muscle tissues (Jansson et al. 1993; Sellström et al. 1993). The ospreys' diet was freshwater fish. The combined concentration of BDE-47, -99, and -100 was 2,100 ng/g lipid in samples collected between 1982 and 1986; BDE-47 was the primary congener (86%) in these samples (n=35). High concentrations of PBDEs may reflect biomagnification and/or fish consumption along their migratory routes. The concentrations of PBDEs in common guillemots (*Uria aalge*) collected in 1979–1981 from the Baltic and North Seas were 370 and 80 ng/g lipid, respectively (Jansson et al. 1987). As part of the Swedish National Environmental Monitoring Program, guillemot eggs (St. Karlsö, Baltic Sea) are collected yearly and placed in the Swedish Natural History Museum's Environmental Specimens Bank. The concentrations of BDE-47, -99, and -100 in

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pooled egg samples from the specimen bank showed a significant increase from 1969 to the beginning of the 1990s, with highs of 1,100 ng/g for BDE-47 in 1984 and 190 ng/g for BDE-99 in 1990 (Sellström 1999; Sellström et al. 1993). Between 1992 and 1997, PBDE levels started to decrease statistically. In 1997, the PBDE level (sum of BDE-47, -99, and -100) was 190 ng/g lipid, with BDE-47 as the predominant congener.

The quantitative determination of the concentrations of PBDEs in body tissues and fluids is important in determining the human body burden of these chemicals. Increasing levels of PBDEs have been measured in blood and breast milk in temporal trend studies. Individuals who consumed fish had a somewhat higher concentration of total PBDEs in body fluids compared to individuals who ate less fish. Tables 6-9, 6-10, and 6-11 summarize the concentrations of PBDEs found in blood (serum), adipose tissue, breast milk, and other body tissues or fluids, respectively. No data were located on the levels of PBDEs in body tissues and fluids from individuals living in the United States. However, levels of PBDEs in body tissues and fluids from individuals living in the United States are expected to be similar to individuals from other regions of the world.

Six PBDE congeners (BDE-28, -47, -66, -99, -100, and -153) were quantified in 40 human blood plasma samples from Sweden. The highest concentrations in plasma were for BDE-47 and BDE-99; these congeners made up 70% of the total PBDE concentration. The mean concentration of total PBDEs were 2.1 ± 1.4 ng/g lipid weight (Klasson Wehler et al. 1997). Whole-blood samples from a German environmental specimen bank, collected in 1985, 1990, 1995, and 1999, contained measurable quantities of BDE-28, -47, -66, -85, -99, -100, -153, and -154. An increasing temporal trend was also observed; the mean total PBDE concentration (sum of eight congeners) increased from 3.9 ng/g lipid weight in 1985 to 5.6 ng/g lipid weight in 1999. For the 1999 sample, BDE-47 was the major congener found, with a mean concentration of 3.9 ng/g lipid weight. The total PBDE concentrations were significantly lower in female blood samples (Schröter-Kermani et al. 2000). Serum samples collected from 12 U.S. blood donors in 1988 were analyzed for PBDEs, and BDE-47, -153, -183, and -209 were detected (Patterson et al. 2000; Sjödin et al. 2001). Concentrations of these congeners were similar to those found in the Sjödin et al. (1999b) study for the control group. The median concentrations and ranges of BDE-47, BDE-153,

Table 6-9. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Human Blood Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Human blood	Sweden	No data	No data	No data	2.1*	No data	Klasson Wehler et al. 1997
Human blood	Sweden, computer disassembly workers	2.9 (median)	No data	No data	26*	4.8	Sjödín et al. 1999a
Human blood	Sweden, cleaning personnel/office workers	1.5–1.6 (median)	No data	No data	3.3–4.1*	<0.7 (median)	Sjödín et al. 1999a
Human blood	Sweden, high fish intake	2.1	No data	No data	No data	No data	Bergman et al. 1999; Sjödín et al. 2000
Human blood	Sweden, no fish intake	0.40	No data	No data	No data	No data	Bergman et al. 1999; Sjödín et al. 2000
Human blood	Germany	3.9	No data	No data	5.6*	No data	Schröter-Kermani et al. 2000

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

Table 6-10. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Human Adipose Tissue Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Human adipose tissue	Sweden	8.8	1.1	1.8	11.7	No data	Haglung et al. 1997
Human adipose tissue	Sweden	3.8–16	No data	No data	No data	No data	Lindström et al. 1998
Human adipose tissue	Sweden	2.2	1.6	0.1	5*	No data	Meironyté Guvenius and Norén 1999
Human adipose tissue	United States	No data	No data	No data	No data	ND–0.7	Cramer et al. 1990; Stanley et al. 1991
Human adipose tissue	Finland	7.3	2.3	No data	6.2–22*	No data	Strandman et al. 1999
Human adipose tissue	Spain	1.36	0.42	No data	No data	No data	Meneses et al. 1999
Human adipose tissue	Northern California, United States	7.0–28	3.1–7.3	No data	No data	No data	She et al. 2000

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

Source: de Wit (2002)

ND = not detected

Table 6-11. Concentrations (in ng/g Lipid Weight) of Several PBDEs in Human Breast Milk Samples

Sample type	Location	BDE-47	BDE-99	BDE-100	' PBDEs ^a	BDE-209	Reference
Human breast Milk	Germany	No data	No data	No data	0.6–11*	No data	Krüger 1988
Human breast milk	Sweden	2.3	0.5	0.4	4*	No data	Norén and Meironyté 1998, 2000
Human breast milk	Sweden	2.5	0.7	0.5	4.4*	No data	Darnerud et al. 1998
Human breast milk	Finland	1.31	0.39	No data	No data	No data	Strandman et al. 2000
Human breast milk	Quebec and Ontario, Canada	3.4	1.2	0.44	5.8*	No data	Ryan and Patry 2000
Human breast milk	Martimes, Canada	No data	No data	No data	19*	No data	Ryan and Patry 2000
Human breast milk	Quebec, Canada	No data	No data	No data	18.8*	No data	Ryan and Patry 2000
Human breast milk	Ontario, Canada	No data	No data	No data	2.8*	No data	Ryan and Patry 2000
Human breast milk	Prairies, Canada	No data	No data	No data	5.7*	No data	Ryan and Patry 2000
Human breast milk	Canada, wide	No data	No data	No data	16.2	No data	Ryan and Patry 2000
Human breast milk	Japan	0.18–0.57	0.09–0.13	0.07–0.18	0.65–1.48*	No data	Ohta et al. 2000

^a PBDEs is the sum of BDE-47, BDE-99, and BDE-100, but if more congeners are included, this is marked with an asterisk (*).

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BDE-183, BDE-209, and total PBDEs (sum of four congeners) were 0.63 (<0.4–24); 0.35 (0.08–2.0); 0.17 (0.09–1.3); <1 (<1–34); and 2.2 ng/g lipid weight, respectively (Sjödin et al. 2001). In a study of the influence of diet on concentrations of PBDEs, BDE-47 was measured in blood serum from persons with high fish intake and no fish intake (Bergman et al. 1999; Sjödin et al. 2000). High fish intake groups of Swedish and Latvian men had median BDE-47 concentrations of 2.2 and 2.4 ng/g lipid weight, respectively, whereas the no fish intake groups had median concentrations of 0.4 and 0.26 ng/g lipid weight, respectively (Sjödin et al. 2000).

BDE-47, -153, -154, -183, and -209 were measurable in blood plasma from three groups of workers (i.e., workers at a computer disassembly plant, workers in a computerized office, and a control group) (Sjödin et al. 1999b). The median concentrations (sum of five congeners) were highest for the computer disassembly plant workers (26 ng/g lipid weight); the office workers had a median concentration of 4.1 ng/g lipid weight and the control group had a median concentration of 3.3 ng/g lipid weight. The congener patterns for the control group and office workers were similar, with BDE-47 having the highest levels. For the computer disassembly plant workers, the median concentrations of BDE-183, -153, -154, -47, and -209 were 7.8, 4.5, 1.2, 2.9, and 4.8 ng/g lipid weight, respectively. Blood serum samples from 19 full-time computer technicians were analyzed (Hagmar et al. 2000). The serum concentrations of BDE-153, -183, and -209 in these samples were found to be approximately 5 times higher than the control and office workers in the Sjödin et al. (1999b) study. The median concentration for total PBDEs (for the sum of five congeners) was 10.6 pmol/g (7.0 ng/g) lipid weight. The highest concentrations were seen for BDE-153. Two OBDE congeners and one NoBDE congener were also detected. Connections were observed between fish consumption and serum concentrations for congeners BDE-47, -153, and -183, and between worktime at the computer and congeners BDE-153 and -183.

DeBDE, as well as hexa- through NoBDE, has been found in composite samples from the 1987 National Human Adipose Tissue Survey repository (Cramer et al. 1990; Stanley et al. 1991). The concentrations ranged from not detected to 1 ng/g fat for HexaBDE, 0.001–2 ng/g fat for HeptaBDE and not detected to 8 ng/g fat for OBDE. NoBDE concentrations were estimated to be >1 ng/g fat; DeBDE was estimated to range between not detected and 0.7 ng/g fat. In the late 1990s, breast adipose samples collected in northern California contained quantifiable amounts of BDE-47, -99, and -153 (She et al. 2000). Mean concentrations were 18 ng/g lipid weight for BDE-47, 4.9 ng/g lipid weight for BDE-99, and 2.2 ng/g lipid weight for BDE-153. In the adipose tissue of a 74-year-old Swedish male, the BDE-47 concentration was 8.8 ng/g lipid weight (Haglund et al. 1997). Adipose and liver tissue from two Swedish males were examined for several PBDEs (BDE-28, -47, -85, -99, -100, -153, and -154) (Meironyté Guvenius and Norén 1999). The distribution of congener concentrations in the adipose and liver tissues for each

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individual were similar. BDE-47, -99, and -153 were the predominant congeners with adipose BDE-47 concentrations ranging from 2 to 2.4 ng/g lipid weight, BDE-99 concentrations of 1.6 ng/g lipid weight, BDE-100 concentrations of 0.1 ng/g lipid weight, and BDE-153 concentrations ranging from 1 to 1.3 ng/g lipid weight. The total PBDE concentration (i.e., the sum of the seven congeners) in adipose tissue was 5 ng/g lipid weight. Human liver and adipose tissues from one woman and four men autopsied in Sweden in 1994, were analyzed for PBDEs containing 3–6 bromine atoms (Meironyté Guvenius and Norén 2001). PBDEs were found in all of the tissue samples. The sums of nine congeners (BDE-17, -28, -47, -66, -100, -99, -85, -154, and -153) were 5–18 and 4–8 ng/g lipids in liver and adipose tissue, respectively. The PBDE congeners BDE-47, -99, and -153 occurred at the highest levels and constituted 87–96 and 84–94% of the total sum in liver and adipose tissue, respectively. Strandman et al. (1999) measured the concentration of BDE-47, -99, and -153 in adipose tissue samples from 10 randomly selected individuals in Finland. Mean concentrations were 7.3 ng/g fat for BDE-47, 2.2 ng/g fat for BDE-99, and 2.3 ng/g fat for BDE-153. Levels of PBDEs were measured in adipose tissue samples from 13 individuals (3 women, 10 men) from Tarragona, Spain; the mean concentrations of BDE-47, -99, and -153 were 1.36, 0.42, and 1.83 ng/g lipid weight, respectively. The mean concentrations of PeBDE and HexaBDE were 0.93 and 1.83 ng/g lipid weight, respectively (Meneses et al. 1999).

Norén and Meironyté (1998, 2000) examined the temporal trends of PBDE concentrations in pooled breast milk samples from mothers in Stockholm, Sweden. Between 1972 and 1997, the concentration of PBDEs in human breast milk increased, with a doubling rate of 5 years. In the 1997 sample, the concentration of PBDEs (sum of eight congeners) was 4 ng/g lipid, whereas the 1972 sample contained 0.07 ng/g lipids (Meironyté et al. 1999). The authors suggest that the current exposure of humans to PBDEs may not be only diet; other exposure routes may result from the presence of PBDE in both work or home environments. PBDE levels were studied in breast milk obtained from mothers pregnant for the first time (n=39, ages 22–36 years old) from Uppsala County, Sweden (Darnerud et al. 1998). The mean value of total PBDEs (sum of eight congeners) was 4.4 ng/g fat; the major congener was BDE-47, comprising ca. 55% of the total PBDEs. In human breast milk from 25 German mothers, the levels of PBDEs ranged from 0.6 to 11 ng/g lipid (de Wit 2002). In 1992, the mean concentration of total PBDEs (sum of BDE-28, -47, -99, -100, -153, and -183) was 5.8 ng/g lipid weight for samples (n=6) from mothers from Ontario and Quebec, Canada (Ryan and Patry 2000). Combined samples from 1992 representing four regions of Canada and one representing all Canadian provinces had total PBDE concentrations ranging from 2.6 to 19 ng/g lipid weight; the highest concentrations were observed in the New Brunswick, Nova Scotia, and Prince Edward Island. Breast milk samples from Finland, collected between 1994 and 1998, had concentrations of total PBDEs (sum of BDE-28, -47, -99, and -153) ranging from 0.88 to 5.9 ng/g lipid weight (Strandman et al. 2000). In Japan, breast milk samples had total PBDE

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concentrations (sum of BDE-28, -47, -99, -100, -153, and -154) ranging from 0.66 to 2.8 ng/g lipid weight (Ohta et al. 2002). Women who consumed fish had a somewhat higher concentration of total PBDEs (range, 1.4–2.8 ng/g lipid weight) compared to women who ate less fish (range, 0.67–0.87 ng/g lipid weight). BDE-47 was the major congener in most of the samples; BDE-153 levels were analogous to BDE-47 levels in some samples (Ohta et al. 2002).

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Polybrominated Biphenyls. PBB are no longer produced or used in the United States. Thus, the general population exposure to PBBs will only be from historical releases. For people residing in the lower peninsula of Michigan, especially in the immediate vicinity of the PBB contaminated areas of this region, exposure to PBBs may still be occurring today. However, environmental levels have decreased since the 1970s and current exposure, if any, will be at low levels. For other regions of the United States, the levels of exposure will either be very low or none.

In the past, the general population may have been exposed to PBBs by inhaling contaminated air, ingesting contaminated water and food, and using consumer products containing PBBs. Other than in air in the vicinity of PBB production plants (see Section 6.1), no current or historical data exist that would indicate that PBBs might be present in ambient air. There are no current or historical data on the direct exposure of humans to PBBs from water. The general population may have been exposed to low levels of PBBs from the consumption of contaminated foods, but no estimate is available that quantitated this exposure. Historical monitoring and body burden data indicate that low level exposures to PBBs were limited to the population within the state of Michigan (see Section 6.4 and Table 6-4). The level of exposure to PBBs was slightly higher for the people residing in the lower peninsula of Michigan and highest among people residing in the immediate vicinity of the contaminated dairy farms, where people consumed contaminated meat, eggs, and dairy products (see Section 6.4 and Table 6-4). Consumer exposure in the past (plastics containing PBBs may not be in circulation anymore since PBB production ceased in the 1970s) from using PBB-containing plastics (e.g., typewriters, calculators, projector housings, and movie equipment cases) is expected to be very low since the PBBs were incorporated into the plastic and their mobilization could only have occurred under conditions such as combustion (Di Carlo et al. 1978).

Workers involved in the historical production of PBBs, PBB-containing plastics, and PBB-containing plastic products could have been exposed to PBBs via inhalation of dust and vapor and/or dermal contact.

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Both workplace environmental monitoring and body burden monitoring data of workers (see Table 6-4) (Hesse and Powers 1978; Humphrey and Hayner 1975; Wolff et al. 1979b) indicated that workers in PBB industries were exposed to higher concentrations of PBBs than the general population. Although no evidence has been reported, workers in facilities that combusted or incinerated PBB-containing plastics might have been exposed to higher levels of PBBs.

Polybrominated Diphenyl Ethers. PBDEs are ubiquitous in the environment. Body burden data indicate that there are low-level exposures to PBDEs for the general population. Humans are primarily exposed to PBDEs by inhalation of ambient or contaminated air and ingestion of contaminated food. Dermal exposure to PBDEs could occur by contact with products containing PBDEs such as textiles or polymers. However, little information is known about the potential exposure from this route.

In the United States, widespread exposure is evident by the levels of PBDEs found in tissues from individuals from all areas of the country and in all age groups (see Section 6.4.4). OBDE ranged from not detected to 8,000 ng/kg in adipose tissue samples collected for the National Human Adipose Tissue Survey (NHATS) in 1987 (Cramer et al. 1990; Stanley et al. 1991). HeptaBDE (1–2,000 ng/kg) was also found in the adipose tissue (Stanley et al. 1991). Analysis of adipose tissue samples from subjects of various ages and geographic locations within the United States showed that levels of HexaBDE and HeptaBDE were generally higher for ages 15–44 and 45+ years, whereas levels of OBDE were generally higher for ages 0–14 and 15–44 years. In studies of the general populations of other countries, it has also been shown that exposure to the general population is widespread (Haglund et al. 1997; Meneses et al. 1999).

The concentration of PBDEs in outdoor air ranges from 2 (rural) to 77 (urban) pg/m³ in the United States (Dodder et al. 2000a), which indicates low levels of exposure to the general population. In Sweden, indoor air concentrations of PBDEs in lecture halls, computerized indoor environments, and rooms with electronic devices (e.g., televisions) have low levels of PBDEs (Lindström 1999). Point sources may result in increased concentrations of PBDEs in indoor air. For example, OBDE has been found in indoor areas that contain electronic products containing PBDEs (e.g., televisions and computers) (Bergman et al. 1997). The release of PBDEs from polymers is dependant on the migration ability of the PBDE molecule through the polymer matrix to the polymer surface where emission is possible (Danish EPA 1999). Because PBDEs are large molecules, migration is expected to occur slowly. No experimental studies were located on the emission rate of PBDEs from plastics. Based on worst case emission factors, the estimated emission of deca-, octa-, and pentaBDEs from plastics are 0.038, 0.054, and 0.39% per year, respectively (Danish EPA 1999).

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Consumption of food is expected to be the major route of exposure in humans (Lindström 1999). Consumption of fish has been associated with elevated levels of PBDEs in tissues from the Swedish population (Bergman et al. 1999). In Sweden, fish consumption is about 30 g/day; this translates to an estimated 0.1 µg of PeBDE and 0.3 µg of total PBDEs from fish that is ingested by humans daily (WHO 1994a). The fish of greatest concern to humans are bottom feeders like carp and catfish. Like PCBs, there may be a higher risk of exposure to PBDEs in Native Americans that reside in the Arctic region who consume whale and seal blubber (Jaret 2000).

Workers involved in the production and manufacture of PBDE-containing plastics and plastic products are exposed to PBDEs. Body burden data indicate higher levels for workers exposed to PBDEs than for the general population. Occupational exposure to PBDEs also occurs in workers at plants that dismantle electronic equipment, computer monitor repair technicians, and automobile drivers, as well as other professions (Lindström 1999). Occupational exposure occurs primarily by inhalation. Inhalation of vapor phase PBDEs is expected to be low due to the low vapor pressures of PBDEs (see Table 4-6); however, the inhalation of particulate phase PBDEs is possible during plastic reprocessing where grinding or shredding of polymers with PBDEs occurs. Air samples were taken from an electronics dismantling plant, an office with computers, and outdoors and then analyzed for PBDEs (Sjödin et al. 1999a, 2001a). The electronics dismantling plant had the highest concentrations of PBDEs, with mean concentrations of 2.5 pmol/m³ (1,250 pg/m³) for BDE-47, 4.6 pmol/m³ (2,600 pg/m³) for BDE-99, 6.1 pmol/m³ (3,930 pg/m³) for BDE-153, 26 pmol/m³ (18,800 pg/m³) for BDE-183, and 38 pmol/m³ (36,500 pg/m³) for BDE-209 (Sjödin et al. 1999a, 2001a). Air samples were found to be 4–10 times higher in PBDE concentrations near a plastic shredder when compared to other locations in the plant (range, 0.42–200 ng/m³). Concentrations of PBDEs in the office (range, <0.002– 0.09 ng/m³) were 400–4,000 times lower than in the plant, and PBDEs were not detected in outside air (Sjödin et al. 1999a, 2001a).

BDE-47, -153, -154, -183, and -209 were measurable in blood plasma from three groups of workers (i.e., workers at a computer disassembly plant, workers in a computerized office, and a control group) (Sjödin et al. 1999b). The median concentrations (sum of five congeners) were highest for the computer disassembly plant workers (26 ng/g lipid weight); the office workers had a median concentration of 4.1 ng/g lipid weight and the control group had a median concentration of 3.3 ng/g lipid weight. The congener patterns for the control group and office workers were similar, with BDE-47 having the highest levels. For the computer disassembly plant workers, the median concentrations of BDE-183, -153, -154, -47, and -209 were 7.8, 4.5, 1.2, 2.9, and 4.8 ng/g lipid weight, respectively. Blood serum samples from 19 full-time computer technicians were analyzed (Hagmar et al. 2000). The serum concentrations of

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BDE-153, -183, and -209 in these samples were found to be approximately 5 times higher than the control and office workers in the Sjödin et al. (1999b) study. The median concentration for total PBDEs (for the sum of five congeners) was 10.6 pmol/g (7.0 ng/g) lipid weight. The highest concentrations were seen for BDE-153. Two OBDE congeners and one NoBDE congener were also detected.

6.6 EXPOSURES OF CHILDREN

This section focuses on exposures from conception to maturity at 18 years in humans. Differences from adults in susceptibility to hazardous substances are discussed in 3.7 Children's Susceptibility. Children are not small adults. A child's exposure may differ from an adult's exposure in many ways. Children drink more fluids, eat more food, breathe more air per kilogram of body weight, and have a larger skin surface in proportion to their body volume. A child's diet often differs from that of adults. The developing human's source of nutrition changes with age: from placental nourishment to breast milk or formula to the diet of older children who eat more of certain types of foods than adults. A child's behavior and lifestyle also influence exposure. Children crawl on the floor, put things in their mouths, sometimes eat inappropriate things (such as dirt or paint chips), and spend more time outdoors. Children also are closer to the ground, and they do not use the judgment of adults to avoid hazards (NRC 1993).

Polybrominated Biphenyls. Infants who consume breast milk may have had a higher exposure to PBBs than children who drink formula milk, especially children exposed during the Michigan episode (see Section 6.4.4). No additional information was found in the literature about the exposure of children to PBBs (WHO 1994b).

Polybrominated Diphenyl Ethers. Infants who consume breast milk may have a higher exposure to PBDEs than children who drink formula milk (see Section 6.4.4). Exposure of neonates is evident due to the presence of PBDEs in breast milk of women from around the world as well as one study indicating the presence of PBDEs in placental tissue. The PBDEs detected in breast milk are the tri- to hexaBDEs, but not the hepta- to decaBDEs (LaKind and Berlin 2000), which are the same congeners found in bioaccumulation studies with fish and other mammals. Levels of PBDEs in the breast milk of Swedish women shows an exponentially increasing trend in exposure since the 1970s, with concentrations of PBDEs in breast milk doubling every fifth year (Lindström 1999). The concentrations of PBDEs in breast milk are increasing exponentially from about 300 pg/g lipid in 1976 to about 4,000 pg/g lipid in 1997 in Swedish women (Norön and Meironyté 2000). The sum of four PBDE congeners (BDE-28, BDE-47, BDE-99, and BDE-209) was between 0.88 and 5.89 ng/g lipid in breast milk and between

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1.00 and 4.40 ng/g lipid in placental tissue of 11 Finnish women (Strandman et al. 2000). The four highest concentrations of PBDEs were found in nulliparous women. No additional information was found in the literature about the exposure of children to PBDEs (WHO 1994a).

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Polybrominated Biphenyls. The production of PBBs ceased in 1979, and the usable life of the plastics containing PBBs has expired. Therefore, these plastics are probably no longer in circulation. At the present time and in the near future, populations potentially exposed to low levels of PBBs are those living near hazardous waste sites in which the PBB-containing plastics have been disposed and the residents in and around the contaminated farms in Michigan. The lifetime of PBBs in soil is on the order of years (Jacobs et al. 1978), and the levels of PBBs in fish caught in contaminated waters have declined slowly (Hesse and Powers 1978). Therefore, concentrations of residual PBBs in soil and streams in the vicinity of PBB-containing hazardous waste sites, PBB production facilities, and contaminated farm areas are expected to remain above background levels for many years. The sources of potential exposure to PBBs for residents in these areas are consumption of contaminated meat and dairy products obtained from herds grazing over contaminated soil and consumption of fish from nearby contaminated streams. PBB contamination has triggered the issuance of one human health advisory in the state of Michigan. As of September 30, 1993, recreational and subsistence fishermen who consume appreciably higher amounts of fish caught in the Pine River downstream from St. Louis in Gratiot and Midland Counties (RTI 1993) may be exposed to higher levels of PBBs associated with dietary intake (EPA 1993). The body burden for PBBs in residents of contaminated areas has been higher than in the general population (Brilliant et al. 1978; Cordle et al. 1978; Eyster et al. 1983; Humphrey and Hayner 1975; Kimbrough 1987; Lambert et al. 1990; Landrigan et al. 1979; Wolff et al. 1982). Therefore, babies breast fed by exposed mothers in the contaminated areas may also be at higher risk (Jacobson et al. 1989).

Polybrominated Diphenyl Ethers. Subsistence fisherman who consume PBDE contaminated fish and Native Americans that reside in the Arctic region who consume whale and seal blubber may have a higher risk of exposure to PBDEs (WHO 1994a). No other information was located that identified specific populations with higher exposure levels to PBDEs.

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6.8 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of PBBs and PBDEs are available. Where adequate information is not available, ATSDR, in conjunction with the National Toxicology Program (NTP), is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of PBBs and PBDEs.

The following categories of possible data needs have been identified by a joint team of scientists from ATSDR, NTP, and EPA. They are defined as substance-specific informational needs that if met would reduce the uncertainties of human health assessment. This definition should not be interpreted to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

6.8.1 Identification of Data Needs

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Physical and Chemical Properties.

Polybrominated Biphenyls. Many of the relevant physical and chemical properties of the PBBs are not available (see Table 4-3). More data on the physical and chemical properties of hexabromobiphenyl are available relative to octabromo- and decabromobiphenyl. Even in the case of hexabromobiphenyl, not all relevant data are available, and the quality of data are questionable because the properties of FireMaster BP-6 have been reported as the properties of hexabromobiphenyls. More importantly, very limited data are available on the physical and chemical properties for the individual congeners of hexabromo-, octabromo-, and decabromobiphenyl. The absence of such important data as K_{oc} , vapor pressure, and Henry's law constant, is a major impediment in the prediction of the environmental fate and transport of PBBs.

Polybrominated Diphenyl Ethers. Many of the relevant physical and chemical properties of the PBDEs are not available (see Table 4-4). Very limited data are available on the physical and chemical properties for the individual congeners. Important data, such as K_{oc} , vapor pressure, and Henry's law constant, are necessary for the prediction of the environmental fate and transport of PBDEs.

Production, Import/Export, Use, Release, and Disposal.

Polybrominated Biphenyls. The production of all PBBs in the United States stopped in 1979 (IARC 1986). Data on the past production, import/export, and use of PBBs are available (Neufeld et al. 1977). In the past, PBB-containing plastic was used in consumer products, but the useful life of these products may have ended (Di Carlo et al. 1978; Neufeld et al. 1977), and these products are probably no longer in circulation. In the workplace, the environmental media contaminated by PBBs were air, water, and soil (DeCarlo 1979). Outside of the workplace, soil is expected to be the medium with significant contamination due to disposal of solid waste from production plants and disposal of PBB-containing plastics in landfills (Neufeld et al. 1977). Although it is known that PBB-containing plastics may have been disposed in landfills (Di Carlo et al. 1978), the amount that may have been incinerated is not known. No data were located from studies that determined the efficiency of incineration as a method of disposal of PBBs present in the neat form in industrial wastes or in plastics. Environmental regulations regarding the manufacture and disposal of PBBs have been established (EPA 1988a). According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit chemical release and off-site transfer information to the EPA. The Toxics Release Inventory

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(TRI), which contains this information for 1999, became available in 2002. This database will be updated yearly and should provide a list of industrial production facilities and emissions.

Polybrominated Diphenyl Ethers. Production and use data are available. More information for PBDEs is needed on import/export, release, and disposal. Soil is expected to be the medium with significant contamination due to disposal of PBDE-containing plastics in landfills. Although it is known that PBDE-containing plastics are disposed in landfills, the amount that is incinerated is not known.

Environmental Fate.

Polybrominated Biphenyls. Information regarding the environmental fate of PBBs in air was not located in the literature. The data about the fate of PBBs in air are important for the prediction of transport characteristics of these compounds in air. Photolysis of the PBBs will produce debrominated products in proton donating organic solvents (Ruzo and Zabik 1975; Ruzo et al. 1976), but there is less certainty about the importance of photolysis of PBBs in water (Norris et al. 1973; Ruzo et al. 1976). PBBs will partition from aquatic phase to sediment and suspended solids in water (Hesse and Powers 1978). PBBs will bioconcentrate in aquatic organisms, but the BCF may decrease as the bromine substitution exceeds six (Gobas et al. 1989; Opperhuizen et al. 1985; Zitko 1979; Zitko and Hutzinger 1976). However, the difference in the reported BCF values for hexabromobiphenyl among different investigators is vast (Gobas et al. 1989; Hesse and Powers 1978; Opperhuizen et al. 1985; Veith et al. 1979). PBBs will remain strongly sorbed to soil (Filonow et al. 1976; Griffin and Chou 1981a, 1981b) and will persist in soil because of the lack of suitable degradation pathways (Jacobs et al. 1978). The translocation of PBBs from soil to upper parts in plants was not observed, and the transfer of PBBs from soil to carrot roots was found to be minor (Jacobs et al. 1976, 1978). A recent article by de Boer et al. (1998) found PBBs in deep ocean marine mammals, which suggests that PBBs may be transported globally. More monitoring data for PBBs in the environment are needed to verify the possible global transport of PBBs. Since the toxicity and the environmental fate of PBBs depends on specific PBBs congeners, development of more data regarding congener-specific fate and transport of PBBs in the environment are needed.

Polybrominated Diphenyl Ethers. There is a lack of quantitative data on the photodegradation potential of PBDEs in air, water, and soil in the presence of natural sunlight. No extensive, systematic studies have been done on biodegradation of PBDEs in the environment, especially studies investigating dehalogenation mechanisms. Since the toxicity and the environmental fate of PBDEs depend on specific

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PBDEs congeners, development of more data regarding congener-specific fate and transport of PBDEs in the environment are needed.

Bioavailability from Environmental Media.

Polybrominated Biphenyls. Available information regarding the rate of absorption of PBBs following inhalation, oral, or dermal contact is discussed in the Toxicokinetics Section (Section 3.4). Although no data on the bioavailability of PBBs from inhalation of contaminated air, or ingestion of or dermal contact with water, or inhalation of or dermal contact with soil are available, the bioavailabilities from these routes of exposure are expected to be far less than 100% because these compounds strongly sorb to particulate matter and soil. The estimated bioavailability of higher brominated biphenyls are expected to be even lower than the less brominated biphenyls due to stronger sorption characteristics of the former compounds. The estimated bioavailability of PBBs by farm animals from ingestion of contaminated soil was 56–65% (Fries 1985a).

Polybrominated Diphenyl Ethers. The absorption and distribution of PBDEs as a result of inhalation, ingestion, and dermal exposure are discussed in Sections 3.3.1, 3.3.2, and 3.3.3. Studies that describe the bioavailability of PBDEs from ambient air, surface water, and groundwater, or soil do not exist. Studies determining the effect of particle size and organic matter content on the bioavailability of PBDEs from soil and the role of microparticle-sorbed PBDEs on the bioavailability of PBDEs from drinking water are needed. Such studies would be useful in assessing the health effects of PBDEs on people living near hazardous waste sites.

Food Chain Bioaccumulation.

Polybrominated Biphenyls. PBBs do not readily translocate from soil to plants via root uptake (Jacobs et al. 1976, 1978). Therefore, PBBs may not bioconcentrate in plants. However, plant uptake data are limited, and it will be helpful to develop additional plant uptake data. Brominated biphenyls with bromine substitution 6 or less will bioconcentrate in aquatic organisms (Gobas et al. 1989; Norris et al. 1973; Opperhuizen et al. 1985; Zitko 1979; Zitko and Hutzinger 1976). PBBs are preferentially stored in the adipose tissue of animals (Kimbrough 1987). Although PBBs have been detected in fish-eating birds and predatory animals from the consumption of contaminated food (Heinz et al. 1983, 1985; Hesse and Powers 1978), no systematic study was located that analyzed the biomagnification potential in predators resulting from consumption of contaminated food.

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Polybrominated Diphenyl Ethers. The limited existing data indicate that PBDEs bioaccumulate in aquatic and terrestrial food chains and biomagnify in predators due to consumption of contaminated prey. More information on bioaccumulation and biomagnification of PBDE and its congeners is needed in assessing human health risks.

Exposure Levels in Environmental Media.

Polybrominated Biphenyls. Only limited data on the levels of PBBs in ambient air are available (DeCarlo 1979). Data are available on the levels of PBBs in effluent water from manufacturing plants, in river water, stream sediment, and soil in the vicinity of the plants, in sludge of a waste treatment plant, and in groundwater of a landfill site (Hesse and Powers 1978; Shah 1978). No data on the level of PBBs in drinking water from the contaminated sites were located. No estimate on the human intake of PBBs from any of the various environmental media was located in the literature.

Reliable monitoring data for the levels of PBBs in contaminated media at hazardous waste sites are needed so that the information obtained on levels of PBBs in the environment can be used in combination with the known body burden of PBBs to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites.

Polybrominated Diphenyl Ethers. Information on the relative importance of different routes of exposure to PBDEs is limited. Data on the concentrations of PBDEs in foods, collected using a market-basket approach, are needed to determine concentrations of PBDEs in foods consumed by the general population. Data on the PBDE concentrations in foods grown in contaminated areas, particularly in the vicinity of hazardous waste sites, are also needed. Data on congener-specific PBDE analysis of food, especially plant products, would be useful. More monitoring data on the concentrations of total PBDEs as well as congeners in air in remote, rural, urban, and areas near hazardous waste sites and incinerators are needed. Data on PBDE concentrations in finished drinking water nationwide would be helpful.

Exposure Levels in Humans.

Polybrominated Biphenyls. Body burden data indicate that low-level exposures to PBBs have occurred for people in the state of Michigan. No recent information about average daily intake of PBBs was located. The levels of PBBs in human tissue and body fluids, such as blood, serum, adipose tissue, breast milk, feces, cord blood, biliary fluid, and placenta, of people in the state of Michigan have been

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extensively studied (Brilliant et al. 1978; Cordle et al. 1978; Eyster et al. 1983; Humphrey and Hayner 1975; Lambert et al. 1990; Landrigan et al. 1979; Wolff et al. 1982). However, no recent data are available. Data on the levels of PBBs in tissues and body fluids of residents who live in the vicinity of sites of industrial discharge of PBB wastes were not located. Updated information would be useful to understand current exposure levels of people in the state of Michigan to PBBs. This information is necessary for assessing the need to conduct health studies on these populations.

Polybrominated Diphenyl Ethers. Body burden data indicate that there are low-level exposures to PBDEs for the general population. Information about the average daily intake of PBDEs is limited to populations living in Sweden (Bergman et al. 1999; Lindström 1999; WHO 1994a). PBDE levels are reported in the current literature for blood, breast milk, and adipose tissue of the general population and occupationally exposed individuals (WHO 1994a). However, no surveys have ever been conducted in the United States to evaluate the trend of PBDE concentrations in human tissues over the years. It would be helpful to develop a database of information on congener-specific PBDE levels in tissues of exposed and control cases for studying clinical and epidemiological outcomes. In particular, a comprehensive study that monitors congener-specific concentrations in fish species and relates them directly to congener levels in human tissue would be extremely useful. Additional data regarding the concentrations of PBDEs in body fluids or tissues of people who reside near hazardous waste sites are needed. This information is necessary for assessing the need to conduct health studies on these populations.

Exposures of Children.

Polybrominated Biphenyls. Children may be exposed to PBBs by a variety of exposure pathways. Levels will be highest for children living in the vicinity of the area affected by the Michigan contamination episode. The most important pathway appears to be consumption of contaminated mother's milk (see Section 6.4.4). More data are needed on the levels of PBB exposure in nursing women from consumption of fish and from those of the general population. Exposure and body burden studies related to consumption of fish in the U.S. population are needed to determine exposure levels, particularly in children of recreational and subsistence fishers. Information related to the exposure of children living near hazardous waste sites is also needed. In particular, information is needed that is related to the potential for children to be exposed to PBBs bound to soil and dust particles through pica or unintentional hand-to-mouth activity within homes located in these areas. Quantitative information regarding the bioavailability and amount of PBBs that children are exposed to through contact with contaminated soils are unavailable. Therefore, any information concerning this subject would be useful

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in evaluating children's exposure. Additional information on weight-adjusted intakes would be helpful for determining the health risks for young children. Infants and young children consume a greater amount of food per kilogram of body weight and, therefore, may have a proportionately greater exposure to PBBs than adults.

Polybrominated Diphenyl Ethers. Children may be exposed to PBDEs by a variety of exposure pathways. The most important pathway appears to be consumption of contaminated foods, particularly fish (Bergman et al. 1999; Lindström 1999; WHO 1994a). Children can also be exposed to PBDEs from mother's milk (LaKind and Berlin 2000; Lindström 1999; Norön and Meironyté 2000; Strandman et al. 2000; WHO 1994a). More data are needed on the levels of PBDEs exposure in nursing women from occupational situations or consumption of fish and from those of the general population. Exposure and body burden studies related to consumption of fish in the U.S. population are needed to determine exposure levels, particularly in children of recreational and subsistence fishers. Exposure and body burden studies are also needed in Native American communities that consume high levels of game and marine mammals. Information related to the exposure of children living near hazardous waste sites is also needed. In particular, information is needed that is related to the potential for children to be exposed to PBDEs bound to soil and dust particles through pica or unintentional hand-to-mouth activity within homes located in these areas. Quantitative information regarding the bioavailability and amount of PBDEs that children are exposed to through contact with contaminated soils are unavailable. Therefore, any information concerning this subject would be useful in evaluating children's exposure. Additional information on weight-adjusted intakes would be helpful for determining the health risks for young children, particularly those in Native American populations. Infants and young children consume a greater amount of food per kilogram of body weight and, therefore, may have a proportionately greater exposure to PBDEs than adults.

Child health data needs relating to susceptibility are discussed in 3.12.2 Identification of Data Needs: Children's Susceptibility.

Exposure Registries.

Polybrominated Biphenyls. The Michigan Department of Community Health (MDCH), together with the Centers for Disease Control and Prevention (CDC) and three other federal agencies, began a major study to assess the health effects of PBBs after the Michigan contamination episode. A health questionnaire and blood samples were collected from people affected by the feed contamination incident. MDCH had

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the responsibility to analyze several thousand samples for PBB from 1975 to 1978. MDCH continues contact with this cohort, updates health questionnaires, and collects blood samples to be analyzed (MDCH 2002).

Polybrominated Diphenyl Ethers. No exposure registries for PBDEs were located. This substance is not currently one of the compounds for which a subregistry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for subregistries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

6.8.2 Ongoing Studies

Polybrominated Biphenyls. A search in Federal Research in Progress (FEDRIP 2002) did not identify ongoing research studies for PBBs.

Polybrominated Diphenyl Ethers. A search in Federal Research in Progress (FEDRIP 2002) did not identify ongoing research studies for PBDEs.