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# Polybrominated diphenyl ethers: Causes for concern and knowledge gaps regarding environmental distribution, fate and toxicity

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## ABSTRACT

This manuscript critically considers several areas of study of the polybrominated diphenyl ether compounds. Specifically, a brief review of PBDE toxicity is followed by an in depth discussion of PBDE occurrence in abiotic and biotic environmental matrices. Temporal and geographic trends are examined in conjunction with risk assessment factors. Rather than summarize or tabulate the growing body of literature on PBDEs in the environment, the overall goal of this review paper is to highlight broad patterns that may contribute to a more holistic understanding of PBDE behavior in the environment, as well as to identify critical areas of research that warrant further attention.

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## 1. Introduction

On August 2, 2005, an Air France jet overshot the runway at Toronto's Pearson International Airport and burst into flames. Amazingly, all 309 crew members and passengers escaped and in the ensuing reports, safety officials credited fire-resistant materials with slowing the spread of flames and smoke, therefore enabling adequate evacuation time. Such use of flame retardants has been estimated to reduce fire deaths in Europe by as much as 20% and save as many as 280 lives per year in the U.S. This is a great benefit to society when considering that, during the same year in the U.S. alone, fire took the lives of over 3000 people, injured an additional 17,000, and cost approximately 10 billion dollars.

Fire retardant materials are constructed by incorporating chemicals designed to prevent ignition or slow down the initial phase of a fire. The idea of utilizing additives to reduce a material's flammability is by no means a new one. More than 2000 years ago, the Egyptians soaked and dried paper and wood to render them fireproof. Much later, in 1820, Gay-Lussac, on behest from Louis XVIII of France, found that certain ammonium salts were effective for protection of precious textiles, a practice that continues today. However, during the twentieth century, various plastics began to replace the more traditional building materials like wood and metal. Previously used inorganic salts could not be applied to these plastics because they considerably reduced the material's thermal stability. The development of halogen-based organic

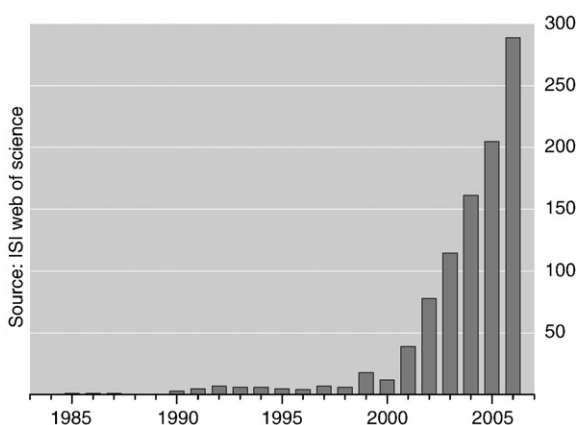
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flame retardants was a major advancement as they could be incorporated into plastic substances and the use of these contemporary flame retardants increased in proportion to the greater use of synthetic polymers.

In 1929, Monsanto led the U.S. commercial production of polychlorinated biphenyls (PCBs), a class of halogenated flame retardants valued for chemical stability and fire resistance. These compounds were processed primarily for use as insulating fluids, plasticizers, and coolants in electrical equipment. PCB manufacture peaked in the 1960s and their production lasted almost two decades until concern over their toxicity and persistence in the environment led the United States Congress to, for the first time, place a ban on a chemical's production. In 1978, brominated flame retardants (BFRs) were established as the new major chemical flame retardant and within this group, the polybrominated diphenyl ethers (PBDEs) took a prominent role (about 40% of the global market of flame retardants). Although similar in structure, there is a critical difference between PCBs and PBDEs. Due to the widespread use of plastic in consumer goods, flame retardants are no longer limited to industrial applications and PBDEs have found their way into our homes, vehicles, and other indoor environments in a way that was never accessible to the PCBs. This, coupled with the fact that by 1990 worldwide production of polybrominated diphenyl ethers had surpassed peak production of PCBs, has raised global attention.

Three technical mixtures of PBDEs have been marketed: penta, octa, and decaBDE, each varying in the degree of bromination and in application. Octa and decaBDE are mainly found in plastic electronics housings and smaller components, while pentaBDE is primarily used in polyurethane foams and textiles. Because they are not chemically bonded to the materials, PBDEs are released during product use and disposal. Unfortunately, the mechanisms controlling the release of PBDEs throughout the lifecycle of these products are not well understood. Currently, the pentaBDE mixture is banned in Europe and the primary U.S. producer has voluntarily phased out production of the penta and octa mixtures. DecaBDE has eluded



**Fig. 1**—Number of publications in scientific journals that report information on PBDEs. This data was compiled from the results of a search of the online journal database ISI Web of Science® (see Supplemental information for search details).

**Table 1**—Timeline of PBDEs

Year	PBDE related events
1871	Hoffmeister synthesizes mono and dibromo congeners—Ann., 1871 (159) 210
1910	Cook synthesizes tetrabromo congener—JACS, 1910 (32) 1285
1960	First patent issued for PBDEs as flame retardants (pentaPBDE)—GB 874006 (1960)
1965	Manufacturing of commercial products containing PBDEs begins
1975	Two year toxicity study on rats finds octa- and deca- essentially non-toxic—Environ. Health Perspect., 1975 (11) 153
1979	PCB manufacture is banned in the USA Presence of decaBDE is first detected in the environment—Ann. N. Y. Acad. Sci., 1979 (320), 678–681
1981	Br <sub>3</sub> –Br <sub>6</sub> PBDE congeners are found for the first time in fish (Viskan River, Sweden) reaching up to 27,000 ng g <sup>-1</sup> —Chemosphere, 1981 (10) 1051–1060
1987	PBDEs are first suggested as global contaminants—Chemosphere, 1987 (16), 2343–2349 U.S. National Institute of Health finds that PBDEs cause an increased number of neoplastic nodes in the liver of rats
1989	Germany voluntarily stops the production and use of decaBDE
1990	PBDEs found in human adipose tissues up to 1 ng g <sup>-1</sup> levels Organohal. Comp., 1990 (2) 347–350
1995	U.S. EPA classifies decaBDE as a possible human carcinogen
1998	Analysis of archived breast milk from Sweden shows significant increase of PBDE levels over the years—Organohal. Comp., 1998 (35) 1–4
1999	Global production of PBDEs estimated to be about 70,000 tons— <a href="http://www.bsef.com">www.bsef.com</a>
2004	Two of the three main PBDE commercial mixtures (pentaBDE and octaBDE) are banned by the European Union Primary North American manufacturer ceases the production of the penta- and octa mixtures
2005	A record-high total PBDE levels are reported in a New York resident adipose tissue: 9,630 ng g <sup>-1</sup> —Environ. Sci. Technol., 2005 (39) 5177–5182
2006	DecaBDE listed as “toxic substance” under the Canadian Environmental Protection Act
2007	Sweden restricts the use of decaBDE in textiles, furniture and cables
2008	Ban of the use of decaBDE in mattresses and furniture begins in the states of Maine and Washington (USA).

legislative bans, in part because industry has successfully argued that it is less toxic than its penta and octa counterparts and its higher stability prevents substantive leaching or degradation into lesser brominated, more toxic compounds. However, Sweden and the states of Washington and Maine have banned this mixture and several other U.S. states have proposed legislation. Our understanding of this congener's toxicity is not complete and there is evidence that decaBDE is not only widespread in the environment but also may be degraded into lesser brominated congeners. In general, our understanding of the fate of PBDEs in the environment has been limited by the dearth of experimental assessments of PBDE sorption and degradation in abiotic and biotic media. Additionally, until recently, the higher brominated congeners from the octa- and deca-mixtures were rarely measured due to issues with analytical methods (Stapleton, 2006). This will be especially

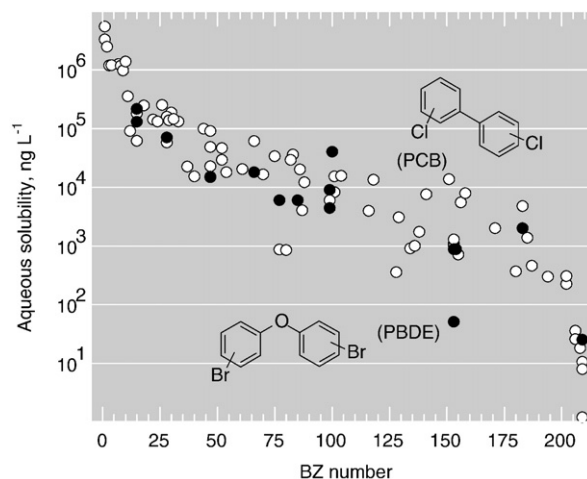
important information for scientists to obtain. Because the products are used in different applications, the distribution, accumulation and biological effects after exposure to these two groups may be different in biotic and abiotic media.

At the present time, PBDEs have been found to be ubiquitous in both the living and non-living parts of the environment. Numerous reports have been published in recent years summarizing their concentrations, time trends, and congener profiles in different environmental and biotic compartments (de Wit et al., 2006; Hale et al., 2003, 2006; Hites, 2004; Law et al., 2006; Wang et al., 2007) (see Fig. 1). In this paper, we discuss the past research of PBDE study, including toxicology and occurrence in the environment and biota, as well as the importance of temporal and geographical trends of PBDEs and risk assessment. Our primary intent is not to summarize individual reports but to take a more holistic look at the historical data in order to reveal broad patterns and to expose areas warranting further investigation (Table 1).

## 2. Toxicological properties

PCBs are well known endocrine disruptors. This, coupled with structural similarities between the PBDEs and their predecessors (see Fig. 2), prompted toxicologists to investigate their potential health effects. Both groups have low water solubilities (see Fig. 3) and tend to be lipophilic. Emerging occurrence data has elevated concern over potential toxic effects of PBDEs and most toxicological studies have focused on the effects of PBDE exposure on the components of the endocrine system, a system that plays a critical role in the regulation of bodily processes, development of the brain and nervous tissues, sexual maturation and function, metabolism and overall homeostasis.

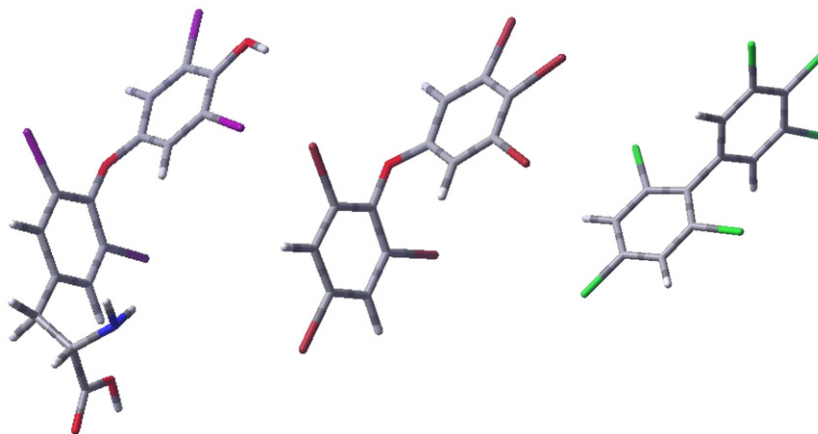
Many synthetic chemicals have the ability to mimic, block or enhance the response to natural hormones in our bodies. Such chemicals are generally labeled endocrine disruptors. One of the major organs of the endocrine system is the thyroid gland and the hormones produced in this gland play a pivotal role in growth and development. Several organo-halogenated chemicals can cause disruptions in the normal function and



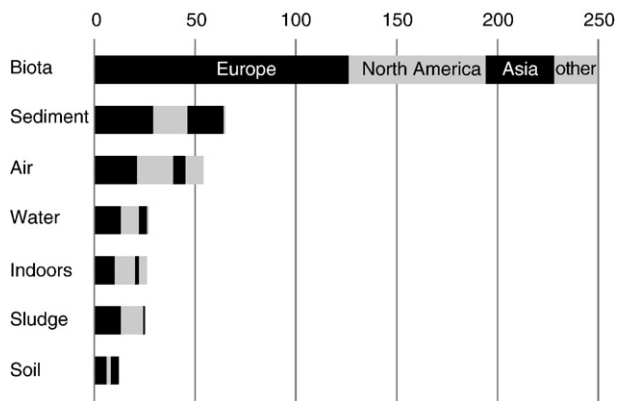
**Fig. 3 – Aqueous solubility of PCB (white circles) and PBDE congeners (black circles). Similar to Koehler numbers for Mozart’s musical compositions, PBDEs and PCBs are catalogued using the Ballschmieder-Zeller number (BZ) shorthand nomenclature also endorsed by IUPAC. Solubility of PBDE-209 is estimated as  $25 \pm 5 \text{ ng L}^{-1}$ . Sources for PCB data included Internet J. Chem. 14 (2001) 11, Fres. J. Anal. Chem 360 (1998) 52, Chemosphere 34 (1997) 275 and Chemosphere 46 (2002) 717. For the PBDE data, the following were used: Chemosphere 67 (2007) 1858, Tittlemier (2002) and Chemosphere 46 (2002) 717.**

alterations in thyroid function leading to changes in basal metabolic rates, protein synthesis, and cell development.

All PBDE technical products, including the decaBDE mixture, have been shown to have thyroid disrupting properties (Birnbaum and Staskal, 2004; Darnerud et al., 2001; NTP, 1986; Zhou et al., 2001). Competitive binding of PBDEs to thyroid hormone receptors is likely due to structural similarities between fire retardants and the endogenous hormones (see Fig. 2). PBDEs can further interfere by altering liver function, leading to changes in thyroid hormones and vitamin A homeostasis, often resulting in over elimination of the thyroid hormone, T4 (Ellis-Hutchings et al., 2006; Zhou et al., 2001).



**Fig. 2 – Chemical structures of T4, PBDE-168, PCB-168.**



**Fig. 4 – Number of reports of PBDE concentrations in various environmental compartments. The number of publications reporting PBDE concentrations for a given environmental compartment were tabulated after an initial literature search for PBDE related publications using Web of Science® (see Supplemental information for details).**

PentaBDEs have also been shown to interfere with sexual development and sexually dimorphic behaviors, both of which are mediated by hormones produced in endocrine glands. Exposure can delay the onset of puberty, decrease follicle formation and increase sweet preference in males (Lilienthal et al., 2006). Although gestational exposure has not been shown to affect the reproductive success of females, adult offspring show a marked decrease in thyroid weights, suggesting *in utero* effects on fetal development (Lilienthal et al., 2006). PentaBDEs also bind to the androgen receptor (AR) which functions in the development of male reproductive tissues as well as other organs (Stoker et al., 2005). Stoker et al. (2005) concluded that delayed puberty onset and decreased size of androgen dependent tissues was a result of competitive binding to AR and the anti-androgenic properties of PentaBDEs (Stoker et al., 2005).

Perinatal exposure to PBDEs is of particular concern, especially with reports of altered cognitive functions later in life. Exposure to PCBs during brain development is known to cause neurologic dysfunction (Eriksson, 1997) and preliminary research suggests that PBDEs will have similar effects (Viberg et al., 2003). Susceptibility to pollutants is highest during development and increases in neurological effects occur when exposure takes place during the period of rapid brain development (Viberg et al., 2003). These effects have been reported for many persistent pollutants including PBDEs (Viberg et al., 2003). The behavioral and developmental effects resulting from neonatal exposure to BDE-209, which is commonly considered to be less toxic, were shown to worsen with age by Viberg et al. (2003). Neonatal exposure to BDE-209 has been reported in animal models (Riu et al., 2008), demonstrating that this congener can cross the placental barrier. Alterations in behavior have also been demonstrated after exposure to the pentaBDE, BDE-99, and were shown to adversely affect learning and memory by Eriksson et al. (2001).

Exposure to PBDEs has also been shown to result in cytotoxic effects. Although such mechanisms occur at the cellular level, the results often affect the associated organ or

organ systems. Cell cultures exposed to PBDEs by Birchmeier et al. (2005) showed decreased viability and increases in both apoptosis and necrosis. While cell death via apoptosis is a natural and controlled process, necrosis is not controlled and leads to tissue damage. Increases in the expression of p53, a major regulator of the cell cycle that can lead to cell cycle arrest and apoptosis, has resulted from exposure to BDE-99, a pentaBDE (Madia et al., 2004).

Based on similarities to other pollutants, PBDEs are expected to be carcinogenic and exposure has been linked to tumor formation and cancer. While concern over decaBDE's carcinogenicity is low, congeners of lower bromination are expected to be relatively carcinogenic (Kociba et al., 1975; NTP, 1986). A study of patients with non-Hodgkin's lymphoma (NHL) showed that compared to non-cancer patients, NHL patients had elevated levels of PBDEs (Hardell et al., 1998). The identification of meaningful toxicity benchmarks for different non-cancer endpoints is one of the largest challenges facing risk assessors, and represents a significant source of uncertainty in the interpretation of PBDE risk assessments. Ironically, decaBDE still remains the only PBDE that has been directly tested for carcinogenicity, in studies conducted more than 15 years ago, although testing of pentaBDEs (BDE-47, 99, 153) is currently in progress under the National Toxicology Program (NTP, 1986). Further, very few studies have examined the toxicity of PBDEs in conjunction with other persistent organic pollutants (POPs), despite evidence that combinations of organohalogen contaminants can have additive, antagonistic, or synergistic effects on toxicity when compared to the effects of the single compounds (Hallgren and Darnerud, 2002). One recent study showed that neonatal exposure to a combination of two PCB and PBDE congeners resulted in greater developmental neurotoxic effects in mice than when exposed to five times higher levels of the PCB congener alone (Eriksson et al., 2006). In addition, PBDEs, in combination with known neurotoxins, have been shown to intensify developmental effects. For instance, it was shown that the combination of BDE-99 and MeHg caused neurotoxic effects at levels which, when administered individually, no effect was observed (Fischer et al., 2008). Finally, toxicological effects resulting from exposure to multiple PBDEs, as compared to exposure to a single PBDE at high levels, should be investigated. Likewise, additive health effects need to be studied in conjunction with potential debromination pathways. Often PBDE congeners found in biota and the environment represent forms not present in the original commercial mixtures (Guardia et al., 2007).

### 3. Occurrence

#### 3.1. Distribution of PBDEs in the abiotic environment

PBDEs have been found in the air, soil, sediment, and water of natural environments, in the buildings where we live and work, in the sewage we produce, and in the automobiles we drive (Hale et al., 2006; Hazrati and Harrad, 2006). In many cases, concentrations of PBDEs are similar or higher than those reported for PCBs (Hale et al., 2006; Hassanin et al., 2004; Hazrati and Harrad, 2006). PentaBDE congeners tend to

dominate in the atmosphere and aqueous media (Hale et al., 2003) while BDE-209 increases in importance in soils, sediments, and sludges (de Wit et al., 2006; Hale et al., 2003, 2006; Law et al., 2006; Wang et al., 2007), an observation that is supported by the various partitioning coefficients of individual PBDE congeners (with increasing bromination, water solubility decreases while  $K_{OW}$  and  $K_{OA}$  increase) (Gouin and Harner, 2003; Palm et al., 2002). In water and air, BDE-209 is generally associated with suspended particulate matter.

Considerably more information is needed to appreciate the global distribution and abundance of PBDEs. Furthermore, the various abiotic environmental compartments have not been equally investigated. While many authors have reported on the presence of PBDEs in air and sediments, comparatively little is known about PBDEs in surface waters, sewage sludges, and soils (Fig. 4) (Hale et al., 2006; Harrad and Hunter, 2006; Streets et al., 2006). Soil is by far the least studied component, but is predicted by fate models to be the dominant environmental sink (Gouin and Harner, 2003; Palm et al., 2002). Studies of indoor environments are less common but particularly needed because they may account for up to 80% of human exposure in some populations (Lorber, 2007).

Most reports of PBDE concentrations in abiotic media are for locations in North America or Europe, with substantial information coming from parts of the Arctic and more recently, from various areas of Asia (Hale et al., 2006; Martin et al., 2004; Wang et al., 2007). Little information exists for Australia and Russia, for the entire continents of South America and Africa, and for the Middle East and many places in Asia. Although North America, Europe, and Asia dominate production of PBDEs and PBDE-containing goods, sizeable exports of these goods and associated wastes to other areas ensure that the environmental burdens are not limited to these core production areas (Hale et al., 2006; Martin et al., 2004; Prevedouros et al., 2004). One of the highest levels of PBDEs detected in air over the North American continent comes from the small capital city of Belize (population ~12,000) (Shen et al., 2006), indicating that PBDEs are not strictly a problem of populous, industrial nations and that the factors controlling their global distribution are not well understood.

### 3.2. Sources and movement of PBDEs within the environment

Concentrations of PBDEs in air, water, and soil have been found to vary across urban gradients (Hale et al., 2006; Harrad and Hunter, 2006; Jaward et al., 2004; Wang et al., 2007), with elevated concentrations near urban centers and decreasing concentrations moving outward from the population center, indicating sources related to human activities. Within highly populated areas, elevated concentrations in air, water, sediment, and soil are associated with proximity to manufacturing, recycling, and waste disposal facilities, signifying that these may serve as point sources of PBDEs in abiotic media (Agrell et al., 2004; Hale et al., 2006, 2002; Leung et al., 2007). PBDEs released into the atmosphere during trash burning, a common practice in the developing world and in rural areas of developed countries, could contribute significantly to environmental PBDE burdens, but little research has been done in support of this (Farrar et al., 2004; Shen et al., 2006).

As the distance from population centers increases, long-range environmental movements of PBDEs become more important. Long-range atmospheric transport (LRAT) of PBDEs is suggested by modeling, by the latitudinal fractionation of individual PBDE congeners, and by the presence of PBDEs in remote sites, where PBDE concentrations are related to regional air mass movements (Breivik et al., 2006; Cheng et al., 2007; de Wit et al., 2006; Gouin and Harner, 2003; Hassanin et al., 2004; Palm et al., 2002). It has been suggested that BDE-209 has a low potential for LRAT, but recent evidence from sediments in remote Canadian lakes shows that BDE-209 can move similar distances in the atmosphere as a broad range of PCB congeners (Breivik et al., 2006), likely via airborne particulate matter. Our limited understanding of LRAT is evidenced by the substantial differences (up to an order of magnitude) between values for LRAT calculated using different models, and by similar differences between modeled and observed values (Breivik et al., 2006). Nonetheless, LRAT likely plays a large role in PBDE movement and distribution at the regional and global scale. For example, atmospheric deposition is believed to be the dominant source of PBDEs observed in 'background' waters and soils (Hassanin et al., 2004; Streets et al., 2006).

Indoors, PBDEs are thought to be released from foam furniture and electronics (Hale et al., 2006), but the controls on indoor air concentrations are not adequately understood (Hazrati and Harrad, 2006). Once released, PBDEs are dispersed into the air, house dust, vacuums, and air filter systems (Hale et al., 2006). Levels of PBDEs in indoor air are often much higher than that of outdoor air (Hale et al., 2006), and indoor air may serve as a major source to (and driver of) outdoor air concentrations (Harrad and Hunter, 2006).

The widespread reports of PBDEs in sludge and effluent from municipal sewage treatment facilities also suggests our homes and offices may be significant sources of PBDEs to the environment (Hale et al., 2006; Song et al., 2006). BDE-209 partitions strongly to particulate matter; thus sewage sludge is enriched in BDE-209 while effluent is dominated by lower brominated, more water-soluble congeners. Despite generally low PBDE concentrations in effluent, the sheer volume of effluent released from a typical sewage treatment plant makes it a substantial source of PBDEs to surface waters (Hale et al., 2006; Song et al., 2006). Soil erosion is another likely source of PBDEs to surface waters (Zou et al., 2007). Once in surface waters, PBDEs can be transported across environmental and political boundaries and accumulate in sediments (Hale et al., 2006), floodplain agricultural soils (Sellström et al., 2005), and aquatic biota (Streets et al., 2006). Enormous volumes of sewage sludge applied to agricultural areas are clearly a major source of PBDEs to soils (Hale et al., 2006; Sellström et al., 2005). Use of municipal wastewater for irrigation may also transfer PBDEs to soil, sediments and air (Goel et al., 2006).

Biota can also redistribute PBDEs throughout the environment following uptake and accumulation from abiotic media. This redistribution occurs at the regional and global scale due to the long-distance migration and dispersal of animals, particularly fish and birds (Blais et al., 2007). Significant redistribution at smaller scales could occur when an organism moves from one component of an ecosystem or landscape to another, such as occurs when an organism spends

one part of its life cycle belowground and another above (e.g., periodical cicadas). Although the quantities of PBDEs transported via biota are likely to be small compared to physical movements (i.e., via air and water), they may be relatively more important due to the potential for direct input into trophic food webs (Blais et al., 2007). For example, Christensen et al. (2005) estimate approximately 70% of the PBDEs in bears originated from migratory salmon, which were picking up their PBDE burdens from the ocean and moving them into the terrestrial environment during their spawning runs up streams/ivers. In their most recent paper, Evenset et al. (2007) show that guano from migratory seabirds may account for as much as 80% of the PCBs in an arctic lake, representing a transport efficiency more than 30 times greater than atmospheric transport. PBDEs may also be transported by seabirds in a similar manner.

In light of the on-going climate change, the influence of environmental conditions, including precipitation and temperature among others on the fate of PBDEs in the environment, merits further attention. Changes in temperature may be particularly important since it affects gas-solid partitioning coefficients and thus, can influence not only PBDE release into the environment but also subsequent movement and retention in different media. For example, temperature variability has been implicated in explaining seasonal patterns of PBDE burdens in air and sludge (Gevao et al., 2008; Goel et al., 2006; Hazrati and Harrad, 2006).

### 3.3. Major knowledge gaps: degradation and bioavailability

Perhaps the greatest source of uncertainty regarding the movement and fate of PBDEs in abiotic media is the extent of degradation in these environments (Gouin and Harner, 2003; Palm et al., 2002). The degradation of BDE-209 is of particular interest as this congener is still in use and its degradation may contribute to the environmental pools of lesser brominated, more toxic congeners. Although it is often suggested that environmental degradation is slow or negligible (Gouin and Harner, 2003; Hale et al., 2006; Sellström et al., 2005; Song et al., 2006), photochemical and biological degradation of PBDEs have been demonstrated in water, soil, sediments, and house dust under laboratory conditions (Ahn et al., 2006a; Eriksson et al., 2004; Gerecke et al., 2005; He et al., 2006; Stapleton and Dodder, 2008; Tokarz et al., 2008; Vonderheide et al., 2006). BDE-209 has also been shown to be susceptible to abiotic degradation by metal oxides that occur naturally in soils and sediments (Ahn et al., 2006b). Yet, the significance of these pathways during storage or transport of PBDEs in the environment is unknown. To date, few studies have addressed PBDE degradation experimentally (with the exception of photodegradation studies) and strong evidence for degradation *in situ* is lacking. This is due in part to limits in instrument sensitivity and selectivity which reduce the ability to detect degradation intermediates and elucidate degradation pathways (Vonderheide et al., 2006). Regarding potential microbial degradation of PBDEs, much information can be gained by tracking the fate of PBDEs artificially amended to soils, sediments, and waters. Use of isotopically labeled PBDEs would be particularly useful and reduce the dependence on standard analytical approaches which have limited sensitivity.

Degradation in soils, sediments, and sewage sludge is of particular importance, since partitioning of PBDEs to these environmental compartments is favored due to their low solubility in water and high partitioning coefficients ( $K_{OW}$  and  $K_{OA}$ ) (Gouin and Harner, 2003; Palm et al., 2002). However, degradation in air and water could have important implications in movement of PBDEs from sources to soil, sediments, and sewage sludge. For example, Raff and Hites (2007) suggest that as much as 90% of lower-brominated PBDE congeners that enter the atmosphere may be removed by photochemical degradation before being deposited in soils and surface waters. Similar modeling work by Kuivikko et al. (2007) shows that the potential for photochemical degradation of BDE-209 in the ocean is much greater than that of lower-brominated congeners. Oxidation of PBDEs by hydroxide radicals in the atmosphere may also be important (Ueno et al., 2008). In general, degradation of PBDEs is expected to be significant in surface water and air, where bioavailability and light penetration are less limiting. Microbial or photolytic degradation in surface waters may be partially responsible for reducing surface water PBDE burdens (often  $\text{pg L}^{-1}$ ) below what might be predicted based on solubility ( $\mu\text{g L}^{-1}$ ). Hydroxylated PBDEs, a potential product of PBDE oxidation by microbes (or other degradation processes), have been recently found to be elevated in surface waters near sewage treatment plants (Ueno et al., 2008).

The extent to which PBDEs are bioavailable in different environmental compartments may determine the amount of microbial degradation that occurs, but no direct studies of PBDE bioavailability have been published and the bioavailability of POPs in general is an area of great uncertainty (Reid et al., 2000). Bioavailability of POPs in soils and sediments is often a function of sorption to mineral particles or organic matter (i.e., ‘aging’) (Reid et al., 2000). Mueller et al. (2006) showed evidence of limited PBDE bioavailability and biodegradation in a soil artificially amended with PBDEs. In their study, less than 10% of PBDEs in sterilized and non-sterile soils were recovered by solvent extraction ten weeks after amendment, suggesting that the majority of PBDEs were bound to soil particles and thus “unavailable.” Given these data, and the widespread use of similar extraction methods for comprehensive removal of PBDEs from environmental matrices, it is likely that total PBDE burdens in these matrices are underestimated. In the same study, when radish and zucchini plants were grown together in PBDE fortified soils, PBDE recovery increased nearly eight times compared to unplanted soil and soil planted with either plant species alone (Mueller et al., 2006). Thus, bioavailability may be influenced by plant competition or other ecological interactions. Rapid degradation of PBDEs in solution by microorganisms extracted from these same experimental soils (Vonderheide et al., 2006) demonstrates that changes in bioavailability could have a strong influence on subsequent degradation by microorganisms.

To fully appreciate the environmental distribution and fate of PBDEs, more efficient assessment/extraction techniques need to be developed. Experimental approaches for assessing bioavailability of persistent organic pollutants, including bioluminescence assays, earthworm accumulation, plant uptake, and mild chemical extractions (Reid

et al., 2000) should be applied to assess the bioavailability of PBDEs to plants, animals, and microbes.

### 3.4. Accumulation of PBDEs in biota

The sequestering of environmental contaminants within an organism may result in greater biota concentrations than those of the surrounding environment. This is especially likely with lipophilic contaminants as compared to water-soluble compounds that (instead) undergo metabolism or elimination from the body. Biomagnification up the food chain then becomes possible after initial introduction. Bioaccumulation and biomagnification are of interest not only because of their importance in determining the fate of PBDEs in the environment, but also because correlations have been noted between PBDE burdens in organisms and various indices of health or fitness including immunosuppression and reproduction.

Occurrence of PBDEs in biota was first reported in 1981 by Anderson and Blomkvist (Andersson and Blomkvist, 1981) in samples collected along the Visken River in Sweden. Since that point, demonstration of biological occurrence has been performed globally and in isolated habitats such as the Arctic. PBDEs have been measured in more than 50 species at different trophic levels in Asia, Europe, North America and the Arctic and in most, the lower-brominated congeners (BDE-47 and/or BDE-99) are dominant, although BDE-209 is also often observed (de Wit et al., 2006). At present, more analysis has been done in the aquatic ecosystem and levels are generally higher in marine organisms, although the terrestrial food chains tend to accumulate more of the higher brominated congeners (Christensen et al., 2005; Jaspers et al., 2006; Voorspoels et al., 2007b).

Analysis of PBDEs in abiotic media has only rarely been reported in conjunction with biotic samples and this represents a critical gap in knowledge. This is because bioaccumulation factors are necessary to adequately compare biotic PBDE burdens across different species or geographical locations and are essential in accurately modeling PBDE fate and transport (Sellström et al., 2005; Streets et al., 2006). A few studies have reported bioaccumulation of PBDEs similar to that of the PCBs (Bethune et al., 2005; Magnusson et al., 2007). There is also evidence that bioaccumulation factors increase with increasing degree of bromination or octanol-water partitioning coefficients ( $K_{ow}$ ) (Streets et al., 2006). More recent studies focus on the ultimate fate of the compounds once initially introduced, i.e., biomagnification and/or the passage of PBDEs to a species' young. Biotransformation pathways have also been studied (Stapleton et al., 2006; Tomy et al., 2004).

Investigation of biomagnification represents an enormous undertaking because it entails the examination of PBDE concentrations in animals of different trophic levels. Further, other factors, such as metabolism, must be taken into account when selecting a species for environmental monitoring purposes (Voorspoels et al., 2007b). In one such comprehensive study, Boon et al. (2002) studied different trophic levels ranging from invertebrate species to marine mammals. They discovered the most marked increase in (lipid-normalized) levels of all six PBDE congeners (an order of magnitude) occurred from fish to marine mammals. Fuglei et al. (2007) studied a predator of a terrestrial food pyramid and demon-

strated higher tissue contaminant levels as a result of biomagnification in the food chain. Bioaccumulation of up to 38,000 ng g<sup>-1</sup> (lipid-normalized) total PBDEs by earthworms in agricultural soils sprayed with sewage sludge indicates the potential for significant biomagnification in other terrestrial food webs (Sellström et al., 2005). This provides a clear link between PBDE distribution in abiotic media, bioaccumulation and the potential for biomagnification. Interestingly, the PBDE tissue burdens of the worm were greatest for BDE-209, the congener whose legislative fate has yet to be decided.

Introduction of contaminants to the metabolism of a species' young is of particular concern. Because of physiological characteristics associated with development, the young are especially susceptible to contaminants. These critical windows of development are major targets for endocrine disruptors and exposure during different developmental stages of the immune, reproductive and nervous systems could provide windows of differential sensitivity to toxic chemicals. For example, the barrier properties of human skin, respiratory tract lining and gastrointestinal tract lining influence absorption in an age-dependent fashion (Hubal et al., 2000). Further, there are toxicokinetic considerations in quantifying the health risks of the young resulting from exposure (Daston et al., 2004). In analysis of the eggs of several predatory bird species, Herzke et al. (2005) showed ng g<sup>-1</sup> levels of PBDEs, confirming this route for exposure of the young. Hence, as the embryo utilizes the yolk/albumin throughout development, it both metabolizes and accumulates the lipophilic PBDEs. In another study, Riu et al. (2008) demonstrated the trans-placental passage of BDE-209 and some of its metabolites (Riu et al., 2008).

In this vein, it is important to note that, outside of the laboratory, few studies have addressed the consequences of PBDE uptake and accumulation by organisms for their fitness, although correlations between PBDE content and reproductive behavior and immunosuppression have been observed in the field (Beineke et al., 2005; Fernie et al., 2008; Kannan et al., 2007; Sonne et al., 2006a, b). DDT's effect on the reproductive system of the bald eagle in the middle of the last century is a historical example of how an environmental contaminant can become widely distributed and subsequently have severe population level consequences for a species. Such data for PBDEs are not yet available and it is not clear whether or not PBDE accumulation in the environment and organisms will have similar consequences or not. More studies are therefore needed to address the population level impacts of PBDE levels in organisms.

Once introduced in an organism, biotransformation of PBDEs can cause errors in interpretation of biotic PBDE burdens by masking differences between true accumulation/magnification of PBDEs and apparent patterns observed by researchers in single time point samples (Riu et al., 2008). Tomy et al. (2004) studied the trout metabolic pathway of a series of BDE compounds and found the metabolites of higher brominated congeners formed via debromination and had the same chemical structure as lower-brominated PBDE counterparts. Debromination was also noted as the major pathway by Vonderheide et al. (2006) in their study of aerobic soil microorganisms. This is an important finding as possible debromination and degradation of the increasingly used BDE-209 may

add to contamination loads of lower-brominated congeners. Additionally, the possibility of metabolic formation of MeO-BDEs and HO-BDEs from PBDEs is under investigation and toxicity studies of these metabolites are underway (Malmberg et al., 2005; Stapleton et al., 2006).

The various migratory patterns of organisms in the environment also makes interpreting biotic PBDE burdens problematic, but few authors have addressed this source of uncertainty. When life cycles dictate that organisms migrate from one geographic area to another, or move from one abiotic media to another, levels of PBDEs within that organism at any given time and place are the result of the integrated exposure taking place during the sum of its life cycle events, not just the exposure imposed by the location from which it was sampled. This makes relating biotic PBDE burdens to that in abiotic media more difficult.

### 3.5. Human exposure

Levels of PBDEs in biotic and abiotic environmental compartments have implications for human exposure. Primary sources of human exposure include dietary intake, dust inhalation, and occupational exposure; the most important factor varies both geographically and among different individuals within a population. Dietary exposure is a significant route of human uptake and is largely attributed to consumption of meat, fish, and dairy products, with fish generally having the highest PBDE levels (Schecter et al., 2006b; Voorspoels et al., 2007a). The contribution of plant foodstuffs to dietary exposure of PBDEs is often ignored and there are few studies of PBDE occurrence in plants or plant products. This may be due to a presumption that plant uptake of PBDEs is similar to that of PCBs: relatively low or non-existent. Indeed, some studies support the theory that plant food products contribute little to dietary exposure. Schecter et al. (2006a) showed that PBDE levels in blood of vegans ( $12.4\text{--}127\text{ ng g}^{-1}$ ) are significantly lower than that of the general U.S. population ( $4\text{--}366\text{ ng g}^{-1}$ ) (Schecter et al., 2006a) and Bocio et al. (2003) showed that PBDE content in fruits and vegetables are lower than in other dietary items (Bocio et al., 2003). Yet, the position of plants at the base of most food chains suggests even small quantities of PBDEs in plant tissues could be magnified up the food chain and passed on to humans. In other words, humans may be potentially and additionally exposed to PBDEs by consuming vegetable/grain-fed animals. It has been demonstrated that some crops do possess the ability to transport PBDEs from their roots to their shoots ( $<5\text{ }\mu\text{g kg}^{-1}$ ) (Mueller et al., 2006) and one study revealed higher PBDE contents in spinach than in pork, chicken or beef (Ohta et al., 2002). Moreover, we are not aware of any reports of PBDE content in plant foodstuffs that were grown on soils of known PBDE content. Thus, it is unclear whether low reported values for PBDEs in crops are the result of plants' lack of ability to translocate PBDEs or simply due to low PBDE burdens in soils or to bioavailability issues. Further investigation into the plant uptake of PBDEs and the contribution of plant food products to dietary exposure are merited, particularly in places where surface application of sewage sludge is performed for fertilization.

Dietary intake, however, cannot alone account for the body burdens measured in the U.S. (Lorber, 2007; Schecter et al.,

2006b). Inhalation of PBDEs from dust has been recently identified as a potentially significant source of PBDEs (Lorber, 2007) and Karlsson et al. (2007) found a positive relationship for the sum of BDE concentrations in dust and blood plasma. Estimates of intake from these two routes are currently being coupled with toxicity data to more accurately predict the effects of an aggregate exposure by the U.S. EPA, through its Office of Pollution Prevention and Toxics and under the Voluntary Children's Chemical Evaluation Program. These results will be of high interest for comparison to the PCB exposure of the U.S. general population, of which 95% is estimated to come from food intake.

Occupational exposure is extremely important for individuals working in the production, recycling, or disposal of PBDE-containing goods. Occupational exposure may be most significant in places like China, where as much as 70% of the world's electronic waste is exported for recycling. Median serum concentrations of BDE-209 in individuals from an electronic recycling region in China ( $310\text{ ng g}^{-1}$  lipid weight) are 50 to 200 times higher than has been reported for any other occupationally exposed population (Bi et al., 2007).

For the average American, European and Asian populations, levels of PBDEs in blood and adipose tissues are on the order of low  $\text{ng g}^{-1}$  lipid weight and again (Fischer et al., 2006; She et al., 2002), concentrations of less brominated congeners are usually higher than those of their higher brominated counterparts (Hites, 2004; Schecter et al., 2003). Congener specific profiles are somewhat different, possibly indicating different technical PBDE usage among regions; in most samples, BDE-47 is the major congener. However, in an increasing number of cases, BDE-153 is becoming dominant (Ingelido et al., 2007) (Eljarrat et al., 2005). Most alarming is that these lipophilic compounds are suspected to be transferred via the placenta and breast milk from the mother to the infant. Schecter et al. (2007) showed a mean concentration of  $23.1\text{ ng g}^{-1}$  lipid in the livers of U.S. fetuses and infants; these levels reflected in-uterine PBDE uptake only, as samples were collected prior to initial feeding. PBDE concentrations in human milk are at  $\text{ng g}^{-1}$  levels and reported concentrations from U.S. women are  $\sim 20$  times higher than those seen in Europe and Asia (Mazdai et al., 2003). These same researchers (Mazdai et al., 2003) showed similar concentrations in maternal and fetal blood samples and Fischer et al. (2006) found higher concentrations in the children, as compared to the parents, all living in the same household. These higher levels have been tentatively correlated with greater rates of dust inhalation (Fischer et al., 2006; Lorber, 2007).

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## 4. Conclusions and needs for future research

Toxicity studies have demonstrated the harmful nature of the PBDEs in biota and it is expected that future work will expand to include interactive effects of these compounds with other environmental contaminants. More work is needed to characterize the abundance and distribution of PBDEs in the environment at local and global scales because uncertainties concerning the fate of PBDEs in the environment create uncertainty regarding which media represents the highest threat to human exposure. Since human exposure is typically



dominated by indoor PBDE burdens and dietary intake, future studies should focus on indoor environments as well as common foodstuffs and their associated bioaccumulation, biomagnification, and biodegradation pathways. Although human exposure via direct contact with soils, sediments, and water is likely low, these environmental media serve as the source of PBDE movement up the food chain and thus, merit further attention. Bioavailability and bioaccumulation may be limited by strong sorption to mineral or organic constituents in soils and sediments and aqueous microbial transformations, yet very few studies have addressed these issues. In general, environmental fate studies could be drastically improved by greater analytical instrument sensitivity, more frequent analysis of higher brominated congeners such as BDE-209, and improved extraction methods that allow for greater efficiency of PBDE recovery from environmental media. Furthermore, additional experimental investigations of PBDE bioavailability and degradation are needed. Studies that track the fate of artificial doses of PBDEs into biotic and abiotic media, including the use of isotopically labeled PBDEs, will be particularly useful.

Overall, future trends for the PBDEs are simply not clear, despite recent changes in their regulatory measures. However, the current indication is that the PBDE burdens in the environment are continuing to increase. Even though the penta and octa mixtures are now prohibited or voluntarily being phased out in many countries, use and production of the deca mixture appears likely to continue largely unabated. Furthermore, PBDE congeners from the penta, octa, and deca mixtures will continue to be released into the environment during the use, disposal, and recycling of existing fire retardant-containing goods for years to come. This is especially troublesome when one considers that large amounts of electronics and furniture, manufactured when PBDEs were most heavily used, are likely being recycled and disposed of at present or will be in the near future. Additionally, many electronic items are still fabricated in countries that are not subject to the same environmental pressures and regulations as U.S. manufacturers. In order to most effectively reduce the potential health impacts of PBDEs, mechanisms of release must be elucidated so that production processes can be modified and strategies developed for end-of-life management of PBDE-containing goods that limit their discharge into the environment and subsequent exposure of both the environment and biota.

## 5. Supplemental information

### 5.1. Methods for journal database surveys

The data presented in both Fig. 1 and Fig. 4 are derived from the same initial literature search conducted using the ISI Web of Science® electronic journal database. Keywords used for the search were “polybrominated diphenyl ether,” “polybrominated diphenyl ethers”, “PBDEs”, and “PBDE”. The search was conducted on May 8, 2007. For Fig. 1, publications from the year 2007 were excluded, yielding a total of 907 relevant publications in scientific journals. Fig. 4 includes data from 2007, which brought the total number of publications included up to 1060.

Following this initial search, the number of publications reporting PBDE abundance in a given environmental compartment (Fig. 4) was compiled using the “search within results” function in ISI Web of Science®. Individual publications may be represented in multiple categories; for example, a study that reports both water and air concentrations would be counted in both the Air and Water categories. The list of keywords used for each environmental compartment follows: Biota (“bioaccumulation, accumulation, biomagnification, organism, organisms, plant, plants, animal, animals, bird, birds, egg, eggs, fish, whale, whales”), Sediment (“sediment, sediments), Air (“air, atmosphere, atmospheric”), Water (“water, wastewater, influent, effluent”), Indoors (“indoors, indoor, inside, office, workplace, home, household, building, plant, facility, dust”), Sludge (“sludge, sludges, sewage, biosolids”) and Soil (“soil, soils”). For each environmental compartment, the geographical location of each report was scored by hand and used to indicate how reports of PBDE burdens in the environment are distributed among the regions of North America, Europe, and Asia.

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