

**BACKGROUND ON POLYBROMINATED
DIPHENYL ETHERS (PBDEs)**

Final Report

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1.0 WHAT ARE POLYBROMINATED DIPHENYL ETHERS?

According to Environment Canada (2011) and the U.S. Environmental Protection Agency (2010), polybrominated diphenyl ethers (PBDEs) are a group of 209 chemical compounds, referred to as congeners. All congeners have anywhere from one to ten bromine atoms attached to a diphenyl ether molecule, and so are grouped into ten homologues based on the bromine atom count. For example, a PBDE with five bromine atoms in each molecule is referred to as penta-BDE, while a PBDE with ten bromine atoms in each molecule is called deca-BDE. The three principal PBDE mixtures produced commercially have been identified as penta-, octa-, and deca-BDE (Environment Canada, 2011), based on the dominant homologue group in each mixture.

2.0 USES

Fire-retardant chemicals are added to products to inhibit the ignition of flammable materials and thus impede the spread of fire. PBDEs have been widely used as flame retardants since the 1970s, since their bromine atoms have a high ability to remove free electrons from other atoms, which are partially what drive the combustion process (Crimmins et al., 2012). PBDE use increased significantly in the 1980s as consumer products were increasingly being treated with PBDE-based flame retardants (Diamond et al., 2014). Such products include (Abbasi, et al., 2014; U.S. Environmental Protection Agency, 2010):

- Electronics such as circuit boards, computer cases, and televisions
- Polyurethane foam, used in a wide variety of products such as furniture and car seats
- Textiles, carpets, and plastics
- Motor vehicles

Abbasi et al. (2014) and Swerea IVF (2009) report that brominated flame retardants are the second largest group of flame retardants used worldwide, accounting for about 20 to 25% of overall flame retardant use. Their popularity in the North American market may be associated with the specific requirements of various product flammability standards (Zota et al., 2008). Globally, aluminum hydroxide is the most common flame retardant produced, accounting for almost half of all flame retardant production (Ceresana, 2014; Swerea IVF, 2009).

3.0 PATHWAYS OF PBDE RELEASE

PBDEs are additive flame retardants added to products via physical mixing, rather than via the chemical integration that reactive flame retardants undergo (Swerea IVF, 2009; U.S. Environmental Protection Agency, 2010). Thus, they are capable of migrating from the product into the surrounding environment (Diamond et al., 2014). There are three main product life cycle stages during which migration occurs (Environment Canada, 2011):

- Manufacturing
- Usage
- Post-disposal

Although PBDE-based flame retardants themselves are not manufactured within Canada (Environment Canada, 2015), releases can still occur in industrial settings as they are added to products. An example of such a process is polymer processing (Environment Canada, 2011), which includes various methods such as mixing and compounding, as well as molding, extrusion, and spinning.

Releases during product usage occur when PBDEs volatilize (turn into vapour) into indoor environments from treated plastics (U.S. Environmental Protection Agency, 2010). An example of a related release mechanism is the “flaking off, crumbling, and / or general physical removal” from treated foam (U.S. Environmental Protection Agency, 2010, p. xix).

Post-disposal releases occur from a variety of disposal sites as products containing PBDEs sit or are processed. Landfill leachate is the liquid that seeps out from landfilled waste due to precipitation, biochemical processes within the waste mass, and the inherent moisture content of the waste (Renou et al., 2008; Wiszniowski et al., 2006). In 2010, Environment Canada (2011) tested ten Canadian landfill leachate collection systems for PBDEs, and found low concentrations in both untreated and treated leachate.

In general, higher environmental concentrations are recorded near urban and industrial centres – the greater the urbanization, the higher the environmental concentration (Environment Canada, 2011). However, due to atmospheric transport and deposition, PBDEs are ubiquitous throughout the environment, and can contaminate soil, water bodies, organisms, and food supplies (U.S. Environmental Protection Agency, 2010).

4.0 HUMAN HEALTH IMPACTS

The widespread use of flame retardants containing PBDE compounds in consumer products and their persistent, bio-accumulative nature have led to high exposure amongst humans, from fetal stages through to adulthood. Exposure pathways are present in daily life for much of the population, while health effects may be numerous. However, much of the data is presently inconclusive, and there is no widespread consensus on health effects.

4.1 Exposure

Human exposure to PBDEs occurs mostly through ingestion and inhalation (Huwe et al., 2008; Domingo, 2012). Uptake efficiency, the fraction which reaches the circulatory system after initial exposure, is highest for ingestion (85 – 93%), followed by inhalation (7 – 15%) (Li et al., 2015).

Ingestion occurs primarily due to PBDE presence in meat, fish, and dairy products, likely as a result of bioaccumulation (Darnerud et al., 2008; European Food Safety Authority, 2011; Hites et al., 2004; Law et al., 2014). It is worth noting that the cooking process seems to influence PBDE concentrations. For example, according to Bayen et al. (2005), concentrations of PBDEs in cooked salmon are lower relative to raw versions. Reduction magnitude depends on how efficiently lipids are removed during cooking. In the case of infants, breast milk is an additional ingestion pathway. Daily intake of PBDEs solely via breast milk exceeds the US reference dose in ten percent of all US infants (Lyche et al., 2015).

Inhalation of PBDEs occurs both at home and in the workplace, and in both cases, is due to its presence in everyday objects. Volatilization – essentially transformation into vapour – occurs from materials (i.e. foam, upholstery, plastic components) in various consumer products (U.S. Environmental Protection Agency, 2010; Linares et al., 2015). Additionally, PBDEs are released through the physical flaking off / crumbling of products containing PBDEs (U.S. Environmental Protection Agency, 2010), making house dust a significant source of exposure for both adults and children (Johnson et al., 2013).

Certain occupational groups, such as workers who handle electronic equipment, are reported to have higher than average tissue-level PBDE concentrations (Domingo, 2012; Frederikson et al., 2009; Jakobsson et al., 2002; Qu et al., 2007). Infants and toddlers also experience higher exposure levels via residential surfaces and objects, due to the way in which they move and their high frequency of hand-to-mouth contact (Stapleton et al., 2008; Cohen Hubal et al., 2000).

Another special case of exposure is that of pregnant women and fetuses. PBDEs can be transmitted through the placenta, and thus concentrations within maternal and fetal blood have been found to be similar (Fischer et al., 2006).

4.2 Toxicokinetics

The U.S. Environmental Protection Agency (2010) compiled octanol-water partition coefficients (K_{ow}) for individual PBDE congeners, and found that all PBDEs have a tendency for bioaccumulation, but particularly higher-brominated congeners – those with relatively more bromine atoms. K_{ow} coefficients are generally inversely related to water solubility, and thus bioaccumulation. This is because hydrophobic (insoluble) substances absorb well into organic matter, and thus tend to bioaccumulate in organisms rather than remaining in the aqueous phase. The highest concentrations of lower brominated PBDEs are found in fatty adipose tissues, but significant concentrations are also detected in other tissues, including within the brain (Lyche et al., 2015).

There also appears to be a positive correlation between the extent of bromination and half-life duration. The correlation has been found in several studies (Geyer et al., 2005; Liu et al., 2011; von Meyerinck et al., 1990), although no definitive half-life durations can be provided for humans. Studies have focused on different PBDE mixtures in different types of organisms. However, there is evidence for longer half-life in humans than in rodents (Hakk et al., 2002; Staskal et al., 2006).

4.3 Health effects

Nothing definite is known about the health effects of PBDEs in humans. Practically all of the available information is from studies of laboratory animals. Although there is suggestive evidence that exposure to PBDEs is harmful to health, further epidemiological investigations – particularly with regards to children and long term monitoring – are required to confirm these relationships.

Nevertheless, the health effects possibly associated with PBDE exposure can be divided into the following types:

- Thyroid disorders
- Reproductive health
- Cancers
- Neurobehavioral and developmental disorders

4.3.1 Thyroid disorders

In rodent studies, PBDE exposure during developmental stages of life shows reductions in the thyroxin hormone (T4) (Lyche et al., 2015). This is the main hormone produced by the thyroid, influencing metabolism, growth, and body temperature. It is especially critical for brain development in the young.

Additionally, recent studies (Dallaire et al., 2009; Johnson et al., 2013; Stapleton et al., 2011; Turyk et al., 2008; Zota et al., 2011) have reported varying associations between PBDE exposure and changes in thyroid-stimulating hormone (TSH) levels. TSH causes the thyroid to produce T3 and T4 hormones. Low TSH levels cause low hormone production, while excessively high levels can cause abnormal thyroid enlargement.

Lastly, since maternal thyroid hormone levels play an important role in fetal brain development (Haddow et al., 1999), it has been suggested that PBDEs may affect neurodevelopment and reproduction by disrupting thyroid hormone homeostasis (Harley et al., 2010).

4.3.2 Reproductive health

Presently, there is no overall consensus on the relationship between PBDE exposure and reproductive health. A number of epidemiological studies found either non-significant relations or no relation at all (Kim et al., 2014). However, others have reported both direct and indirect effects. Higher concentrations could be related to early-age menarche and irregular menstrual periods (Chen et al., 2011; Chao et al., 2010a). A few studies report an association between PBDE exposure and lower birth weight, height, head and chest circumferences (Chao et al., 2010b), and reduced fecundability (Harley et al., 2010).

4.3.3 Cancers

There is not enough scientific evidence to confirm whether or not PBDE exposure increases cancer risk in humans. Kim et al. (2014) summarize some studies that looked for correlation between PBDE exposure and the occurrence of various cancers, but none found significant relationships. Neither the U.S. Department of Health and Human Services nor the International Agency for Research on Cancer has classified the carcinogenicity of any PBDEs. The U.S. Environmental Protection Agency (2014a) has found suggestive evidence that deca-BDE has carcinogenic potential.

4.3.4 Neurobehavioral and developmental disorders

A recent study by Eskenazi et al. (2013) found a significant association of maternal, prenatal, and childhood PBDE exposures with poorer attention, fine motor coordination, and cognition in early school age children. Another recent study by Herbstman et al. (2010) showed a significant correlation between prenatal serum PBDE levels and lower scores on tests of mental and physical development at 12-48 and 72 months. An association between PBDE content in breast milk and lower mental development scores was also demonstrated in a study by Gascon et al. (2012).

However, due to differences between and limitations within study designs, the current epidemiological evidence does not support a strong causal association between PBDEs and adverse neurodevelopmental and neurobehavioral outcomes.

5.0 ENVIRONMENTAL IMPACTS

Multiple adverse effects on wildlife health due to PBDE exposure have been demonstrated. Mhadhbi et al. (2011) showed that turbot embryos exposed to PBDEs experienced higher mortality rates and various non-lethal malformations during development (although the concentrations used were higher than actual environmental concentrations). Sjøfteland et al. (2011) found that Atlantic salmon exposed to PBDEs experienced metabolic disturbances and altered cell proliferation processes. Noyes et al. (2011) observed thyroid system impairment in fathead minnows exposed to PBDEs. There are numerous other studies not mentioned here that demonstrate adverse effects of PBDE exposure on other species (Yu et al., 2015; Darnerud et al., 2003).

Adverse effects on vegetation due to PBDE exposure have not been studied as extensively. Wang et al. (2014) did see lower leaf count and reduced chlorophyll content in a mangrove plant when contaminated with PBDEs, which impairs the photosynthetic process.

Within the Canadian context, Environment Canada (2011) states that “tetra-, penta- and hexa-BDEs meet the persistence, bioaccumulation, and inherent toxicity criteria for virtual elimination under the Canadian Environmental

Protection Act of 1999” (p. 2). That is to say, PBDEs persist within various media in the environment over time, accumulate to higher concentrations in organisms through food web processes, and cause adverse effects.

Environment Canada (2011) has developed Federal Environmental Quality Guidelines (FEQGs) for various homologues in different environmental media, including water, fish tissue, sediment, and bird eggs. Concentrations above FEQG levels are expected to result in adverse environmental effects. However, effects may manifest at concentrations below guideline levels, due to uncertainty in the science used to develop any guidelines.

6.0 MOVEMENT PATHWAYS THROUGH THE ENVIRONMENT

The main transportation pathway for PBDEs is through the atmosphere. They can be carried over distances greater than 1,000 kilometres before deposition to land and water (U.S. Environmental Protection Agency, 2010). As a result of this, PBDEs have been found in polar environments where little to no human activity occurs, as well as within the tissue of deep-sea marine life (U.S. Environmental Protection Agency, 2010).

Deposition from the atmosphere transfers PBDEs to land and water. Soil and sediment in particular can end up serving as environmental sinks, because PBDEs tend to bind to them (Environment Canada, 2011). In water, PBDEs are located in both the water column and in sediment, depending on specific congener properties (U.S. Environmental Protection Agency, 2010).

Once deposited into the media, bioaccumulation occurs amongst local organisms. This refers to development of a body burden of a chemical via multiple exposure pathways (i.e. inhalation, ingestion, absorption, etc.). In aquatic settings, the exposure pathways are through contact in the water column, contact with sediment, and through the food web (U.S. Environmental Protection Agency, 2010).

At this point, biomagnification begins to move PBDEs through ecosystem food webs, as numerous contaminated organisms at low trophic levels are consumed by those at higher trophic levels. PBDE concentrations can therefore build up to significant levels over time in top predators, causing adverse effects in relatively higher magnitudes (U.S. Environmental Protection Agency, 2010).

6.1 PBDEs within Great Lakes ecosystems

The number of PBDE-containing products and the magnitude of PBDE concentrations in the Great Lakes region both began to increase significantly in the early 1980s (Abbasi et al., 2014; Environment Canada, 2011). Abbasi et al. (2014) conducted an inventory estimate of PBDEs in products in the use and waste phases on a 50-year timeframe (1970 to 2020), within the Great Lakes basin. They estimated that between 1970 and 2013, total PBDE use within the basin was between 15,000 and 80,000 tonnes, with a peak occurring in 2004. Forty percent of this peak value is expected to remain in the use phase by 2020, mainly in the form of deca-BDE.

Recently observed declines in concentrations of some homologues correspond with national phase-outs of penta- and octa-BDE-based flame retardants. For example, Route et al. (2014) found that PBDE concentrations in northwestern Wisconsin bald eagle nestlings rose steadily from the mid-1990s to the mid-2000s, and then declined from the mid-2000s to 2011. These trends were deemed consistent with global phase-outs. Improved handling and disposal practices may also be a key reduction factor (Crimmins et al., 2012). It is important to note that concentrations in distant, remote areas may not be decreasing, due to the delayed effects of atmospheric transport and deposition (Law et al., 2014).

All five Great Lakes contain some level of PBDEs, but Erie and Ontario have the highest concentrations in water, according to Venier et al. (2014). Due to differences in sampling methods and measured substances, these concentrations cannot be directly compared to FEQG levels. The most common congeners found in all lakes were those used in penta- and deca-BDE commercial mixtures, which were phased out in 2004 and 2013, respectively (Venier et al., 2014).

Huron, Michigan, and Superior are all relatively less contaminated and have statistically insignificant differences in concentrations (Venier et al., 2014). It should be noted, however, that colder average water temperatures, larger volume, and lower productivity in Superior may cause PBDE concentrations to decrease more slowly via volatilization and sedimentation, as Carlson and Swackhamer (2006) found for toxaphene and alpha-HCH. Colder temperatures also cause slower phytoplankton growth rates, providing a longer period of time for contaminants to achieve equilibrium between phytoplankton and the aqueous phase, resulting in a greater proportion of highly-brominated PBDEs in sediment and the local food web (Carlson & Swackhamer, 2006).

Great Lakes predatory fish serve as useful indicators for assessing broader ecosystem conditions. Local species such as lake trout and walleye feed on many other species on lower trophic levels, demonstrating biomagnification and making them representative of broader spatial contamination trends (Gewurtz et al., 2011). PBDE concentrations in Great Lakes fish show similar long-term trends in PBDE concentration as in the surrounding environment, with significant increases from the 1980s to the 2000s, followed by a decline (Environment Canada, 2011). Presently, relatively heavier homologues are more prevalent than lighter ones, due to their higher bioaccumulation potential (Environment Canada, 2011).

7.0 QUANTITY OF PBDEs IN USE

Tonnage data for global PBDE production is scarce, inconsistent, and outdated. Estimates for the early 2000s by the U.S. Environmental Protection Agency (2010) and Siddiqi et al. (2003) place the annual worldwide production tonnage at about 40,000 to 67,000 tonnes per year.

Furthermore, different commercial PBDE mixtures have different production and usage levels. Although tonnage figures vary widely, most literature indicates that deca-BDE is produced and used in the highest quantities by far (U.S. Environmental Protection Agency, 2010; Abbasi et al., 2015; Abbasi et al., 2014; Siddiqi et al., 2003). Deca-BDE was not voluntarily phased out by manufacturers in 2004, unlike penta- and octa-BDEs. The U.S. Environmental Protection Agency (2010) reports that even in 2001, 83% of all PBDE consumed worldwide was deca-BDE, followed by penta- and octa-BDE at 11% and 6%, respectively.

In the Great Lakes basin context, Abbasi et al. (2014) estimated the amount of penta-, octa-, and deca-BDE in the usage phase over a 40-year timespan. Their analysis also finds that deca-BDE is the most common PBDE mixture, with somewhere between 10,000 and 70,000 tonnes in use between 1970 and 2013 and a peak of 12,000 tonnes used in 2008 alone. Penta-BDE follows with 2,000 to 10,000 tonnes used between 1970 and 2004, while octa-BDE trails with 500 to 2,000 tonnes used in the same time period. Both experienced annual peaks at the end of their production runs in 2004.

8.0 TRACKING PBDE REDUCTIONS

No ongoing program to track the inventory of PBDEs presently exists, since such a program would be resource intensive (Diamond et al., 2014). However, within the Great Lakes context, environmental measurements can provide insight into

the effectiveness of actions taken to reduce PBDE use, track short- and long-term loadings to the Great Lakes, and describe spatial trends. To minimize resource use, measurements could be obtained through existing monitoring programs, particularly those where archived samples can be analyzed to establish trends over time. Options include monitoring air, biota tissue, and sediment (Diamond et al., 2014).

8.1 PBDE phase-outs

In the early 2000s, some U.S. states implemented bans on usage of penta- and octa-BDE (Abbasi et al., 2014). In 2003, the European Union banned the sale and usage of products containing more than 0.1% penta- or octa-BDE (Abbasi et al., 2014). Then, in 2004, penta- and octa-BDE were voluntarily phased out of production by major U.S. manufacturers, as evidence of negative health and environmental outcomes mounted (Environment Canada, 2011; Crimmins et al., 2012; Gewurtz et al., 2011; Ma et al., 2013; Abbasi et al., 2014; Route et al., 2014). Further regulatory restrictions include an outright ban by the European Union on their production in 2004 (Route et al., 2014), 2005 U.S. usage and importation restrictions (Route et al., 2014), and a 2008 Canadian ban on their manufacture, importation, and usage (Environment Canada, 2011).

On an international scale, penta- and octa-BDE commercial mixtures were added to the Stockholm Convention on Persistent Organic Pollutants in 2009, a global treaty requiring signatories to reduce and / or eliminate the release of listed chemicals. Additionally, the Stockholm Convention specifically names the dominant constituent homologues for penta- and octa-BDE commercial mixtures, which are tetra-/penta- and hexa-/hepta-, respectively (Stockholm Convention, 2009). Penta- and octa-BDE commercial mixtures and their dominant constituent homologues are also listed for elimination from production and usage under the UN Convention on Long-Range Transboundary Air Pollution (United Nations Economic Commission for Europe, 2010).

Restrictions on deca-BDE are less extensive and have only been implemented more recently. The European Union banned deca-BDE from electronic consumer products in 2008 (Gewurtz et al., 2011; Abbasi et al., 2014). Norway proposed adding deca-BDE to the Stockholm Convention in June 2013, and the proposal is currently under review (Stockholm Convention, 2009). Canada secured commitments from major American manufacturers to stop exports to Canada by 2013 (Environment Canada, 2013). Minnesota recently passed a ban on four flame retardant chemicals, one of which is deca-BDE (The Office of the Revisor of Statutes, 2015). Ultimately, major American manufacturers and importers committed to voluntary phase-out by the end of 2013 (Environment Canada, 2011; Ma et al., 2013; U.S. Environmental Protection Agency, 2010; Abbasi et al., 2014).

These bans and restrictions do not specifically address PBDEs in products already in use or already disposed of. However, fewer products containing PBDEs are being introduced into the market. As usage continues to be phased out and products containing PBDEs are no longer in commerce, the quantity of new PBDEs entering the environment is expected to decrease and, over time, declines in environmental PBDE concentrations should occur. However, because of their persistence and the large quantity still in products, PBDEs will remain in the environment for many years to come.

9.0 PBDE ALTERNATIVES

Recognizing the negative environmental and human health effects of PBDEs, there has been a shift by the flame retardant industry towards alternative chemicals since the initial 2004 phase outs of penta- and octa-BDE (Ma et al., 2013; Abbasi et al., 2014). Ideally, alternatives uphold flammability standards while minimizing negative effects.

Alternative flame retardant chemicals are numerous. An alternatives assessment for deca-BDE conducted by the U.S. Environmental Protection Agency (2014b) evaluated twenty-nine alternatives. Three main types of flame retardants were considered: halogenated (which includes brominated compounds), phosphorus / nitrogen-based, and inorganic. Phosphorus and nitrogen-based retardants cover a wide range of compounds with different oxidation states, and are often chemically combined together. Inorganic retardants are the most common type of flame retardant worldwide; examples include aluminum hydroxide, antimony compounds, and boron compounds.

Some alternative chemicals have already begun appearing in the environment and in biota. Atmospheric concentrations of TBB and TBPH in particular are approaching those of PBDEs in the Great Lakes region (Murray et al., 2014). Two derivatives of tetrabromobisphenol-A have been found to bioaccumulate in the herring gull food chain, and also transfer from gull to egg (Abbasi et al., 2014). Tris(1,3-dichloro-2-propyl)phosphate has been detected in human biomonitoring (Murray et al., 2014). Alternative brominated and organophosphate ester flame retardants have been detected in Great Lakes water, with the highest average concentrations in samples from Erie and Ontario (Venier et al., 2014).

Critically, the consequences of these presences are unknown since there are no toxicity profiles or inventories of PBDE replacement chemicals. Such information is difficult to ascertain due to the sheer number of substances in a wide range of products, where usage is undisclosed (Abbasi et al., 2014).

10.0 ALTERNATIVES ASSESSMENTS

Murray et al. (2014) recommend a general framework for assessing flame retardant alternatives in the Great Lakes region. It should begin with consideration of federal, state / provincial, regional, and local policies and rules regarding the use of flame retardants. Industry / product-specific flammability requirements and existing institutional arrangements and initiatives to do with flame retardants should also be understood. An understanding of the contextual setting surrounding flame retardants informs the evaluation of alternatives, and allows for identification of weak links in regulation that could lead to negative environmental and / or human health impacts (Murray et al., 2014).

Murray et al. (2014) also suggest numerous guiding principles, specifically for a flame retardant alternatives assessment in the Great Lakes region (Table 1).

1. Critically examine need for chemical FRs in the specific application
2. Consider human health and ecological impacts of FR use (ideally in a life cycle context) in development of flammability regulations, codes, and standards
3. Support of virtual elimination goals for chemicals of mutual concern in GLWQA
4. In pursuing alternatives assessment for FRs when deemed necessary: <ul style="list-style-type: none"> • Minimize hazard • Minimize releases, exposures • Maximize disclosure, transparency • Promote development of safer alternatives, including through screening assessments and use of green chemistry • Advance towards chemicals/materials/processes that are technologically feasible, commercially available • Involve stakeholders, public participation

Table 1: Recommended principles for a flame retardant alternatives assessment in the Great Lakes region.

“The first two principles highlight the importance of examining the context of flame retardant use and ensuring that these requirements indeed contribute to improved fire safety while not resulting in broad human and ecological exposures” (Murray et al., 2014, p. 4).

The third principle acknowledges the important role that the Great Lakes Water Quality Agreement can play in controlling flame retardants. More specifically, the assessment can support the General and Specific Objectives, as well as work under Annex 3: Chemicals of Mutual Concern.

10.1 Existing work on and enabling programs for alternatives assessments

The Great Lakes Water Quality Agreement provides a mechanism to conduct an alternatives assessment within Annex 3: Chemicals of Mutual Concern (CMCs). Among other things, it states that the U.S. and Canada shall promote “the use of safer chemical substances and the use of technologies that reduce or eliminate the use or release of chemicals of mutual concern.” The Identification Task Team of the Great Lakes Water Quality Agreement Annex 3 Subcommittee (2015) recently recommended in a summary report that PBDEs be designated as a CMC by the Parties, citing guideline exceedances, stable and slightly increasing concentration trends, and lack of coordination / collaboration for performance measurement and monitoring activities. Although CMC designation does not immediately necessitate an alternatives assessment, it enables the process and concentrates Party efforts on relevant programs and measures.

Apart from the Great Lakes Water Quality Agreement, there are programs on both sides of the border that enable the undertaking of an assessment of PBDE alternatives. The U.S. Environmental Protection Agency’s Design for the Environment program provides guidance for conducting alternatives assessments, and has actually conducted them in partnership with relevant stakeholders. In January 2014, a multi-stakeholder alternatives assessment for deca-BDE was completed, assessing and summarizing hazards for 29 potential deca-BDE alternatives (U.S. Environmental Protection Agency, 2014b). In Canada, alternatives assessments can be conducted under provisions in the Canadian Environmental Protection Act (Murray et al., 2014). Bans and studies on effects have occurred for PBDEs, and a risk management plan for products containing PBDEs was created, but no alternatives assessment work has yet been conducted.

State / provincial involvement can also play a key role in alternatives assessments. Several states in the Great Lakes region are collaborating in the Interstate Chemicals Clearinghouse (ICC) alternatives assessment effort (Murray et al., 2014). ICC is a group of state and local governments that aims to facilitate the development and use of safer chemicals. It provides a collection of chemical alternatives assessments, and publishes an alternatives assessment guide.

Since flammability requirements are often sector specific (i.e. furniture, textiles, electronics, automotive), it would likely be beneficial for an alternatives assessment to engage relevant industry associations in the product supply chain (Murray et al., 2014). Such an approach would increase the likelihood of product and / or process requirements being considered, making findings more applicable. In addition, academia, non-governmental organizations, and the public might also play contributing roles.

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