

Stockholm Convention on Persistent Organic Pollutants

**Persistent Organic Pollutants Review Committee
(POPRC)**

DRAFT RISK PROFILE

For

Commercial Octabromodiphenyl Ether

Draft prepared by:

The ad hoc working group on commercial octabromodiphenyl ether

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Draft Risk Profile for Commercial Octabromodiphenyl Ether

Note:

In accordance with the procedure laid down in Article 8 of the Stockholm Convention, this draft was prepared by the Persistent Organic Pollutants Review Committee (POPRC) during its inter-sessional work. Parties and observers to the Stockholm Convention are invited to provide technical and substantive comments on this draft. Comments received will be considered by the ad hoc working group and the revised draft will be made available for the third meeting of the POPRC (19-23 November in Geneva). Please submit your comments to the Secretariat of the Stockholm Convention preferably by e-mail before **July 1, 2007** to:

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EXECUTIVE SUMMARY

The European Union and its Member States, which are Parties to the Stockholm Convention, submitted a proposal in July 2006 for listing octabromodiphenyl ether in Annex A of the Stockholm Convention pursuant to paragraph 1 of Article 8 of the Convention, and the POPRC agreed that the commercial product Commercial octabromodiphenyl ether – actually a mixture as described below - met the screening criteria of Annex D to the Convention. This risk profile reviews the available information on the commercial mixture and its main components: Hexa, Hepta, Octa and NonaBDE.

The polybrominated diphenyl ethers in general are used as flame retardants of the additive type. They are physically combined with the material being treated rather than chemically combined (as in reactive flame retardants). The commercial products cover several congeners and bromination levels. The information provided by the bromine industry indicates that the octa commercial product has been produced in The Netherlands, France, USA, Japan, UK and Israel, but since 2004, it is no longer produced in the EU, USA and the Pacific Rim and there is no information that indicates it is being produced in developing countries. According to the Bromine Science and Environmental Forum (BSEF), OctaBDE was commercialized sometime in the mid 70's. By the early 2000's global production was <4000 tonnes/year and by the time production ceased, demand was <500 tonnes; assuming 30 years of production at 6000 tonnes per year total production volume would be around 180,000 tonnes.

Assuming that the commercial octaBDE is not longer produced, the releases to the environment must be associated to historical processes, as well as to releases during the service life of articles containing the commercial mixtures and at the end of article service life during disposal operations. Switzerland reported for this country diffuse emission from the use of products containing OctaBDE of about 0.37 t/a (based on worst-case estimations) for a total stock of 680 tons.

The persistence of these PBDE in the environment is well documented. The only relevant degradation pathways identified until now are photolysis, anaerobic degradation and metabolism in biota, acting through debromination and producing other BDE which may have higher toxicity and bioaccumulation potential.

The bioaccumulation potential for sediment exposure and particularly for exposure via food is well documented for some c-OctaBDE components. There is also enough toxicokinetic information demonstrating that elimination rates in some vertebrate and invertebrate species are equivalent to those observed for other POPs, with values in the range of 0.01 to 0.5 days⁻¹, equivalent to a half life of about 14 to 70 days assuming first order kinetic. Thus a bioaccumulation potential is confirmed at least for some isomers, as well as biomagnification in some food chains. As debromination into other POP-like chemicals is expected to be a relevant contribution to the dissipation of hexa to nonaBDE, the absence of food-chain biomagnification for a specific congener on a specific taxonomic group does not necessarily decrease the overall concern.

In fact, biota monitoring data in remote areas offer the best demonstration on the potential for long range transport of c-OctaBDE components, in particular for Hexa and HeptaBDE. Theoretically this presence could also be explained by the transport of DecaBDE and its subsequent debromination. However, the comparative analysis of the available information on the physical-chemical properties of the different PBDE homologues indicates that debromination from DecaBDE might contribute to the process but it is not realistic to assume that this explains the process without additional transport from other congeners. Thus, based on the available information a long-range

transport is expected for the c-OctaBDE components, and the role of atmospheric transport is confirmed at least for Hexa and HeptaBDE based on its detection in alpine lakes.

Unfortunately, the available information on the toxicity and ecotoxicity of hexa to nonaBDE is very limited and does not offer enough information for presenting sound toxicological and ecotoxicological profiles for each isomer, mixtures of isomers and commercial mixtures.

No relevant effects have been observed in aquatic, sediment and soil laboratory studies; however, this information cannot be used to conclude that Hexa to NonaBDE are not toxic for these organisms as the measured endpoints and the exposure conditions, employed in these assays are clearly insufficient for a proper assessment of chemicals such as hexa to nonaBDE.

The available information on mammals and birds offer relevant information. The lowest reported NOAEL for traditional endpoints is 2-5 mg/kg bw/d. The effects are relevant for the health and the ecological assessments and therefore useful for assessing risks for humans and wildlife. In addition, immuno-toxicological effects and particularly delayed neurotoxic effects observed after a single dose require specific attention. A critical body burden for hexa BDE 153 of 2000 µg/kg lipid has been estimated based on a NOEL of 0.45 mg/kg; it should be noted that hexa BDE 153 concentrations close to these value have been found in several species and geographic sites and total PBDE concentrations frequently exceed largely this threshold.

The evaluation of the human and environmental risk of commercial OctaBDE associated to its potential for long range transport is not an easy task as the commercial product is a mixture of components with different properties and profiles, which may also be released to the environment due to its presence as components of other PBDE commercial products and also produced in the environment by debromination of commercial decaBDE.

The greatest difficulty appears for the estimation of the potential hazard of the commercial mixture and its components. There are traditional ecotoxicological and toxicological studies where no effects have been observed even at unrealistically high concentrations. However, an in-depth assessment of these studies considering in particular the properties and toxicokinetic of PBDE indicates that the test design, exposure conditions and measured endpoints are not appropriate for a sound assessment of these types of chemicals. Thus, the lack of effects reported in those tests should be considered with care.

In addition, specific studies have reported particular hazards such as delayed neurotoxicity and immunotoxicity which may be particularly relevant in the assessment of both human health and ecosystem risks.

Based on the existing evidence, additional concerns related to the debromination into toxic BDEs, the increasing evidence relating these chemicals with other POPs (similarities between PBDEs and PCBs; relationships with dioxins and furans), and that under Article 8, paragraph 7(a) of the Convention the lack of full scientific certainty shall not prevent a proposal from proceeding, it is concluded that the components of c-OctaBDE, Hexa to NonaBDE, are likely, as a result of LRET, to lead to significant adverse human health and/or environmental effects, such that global action is warranted.

1. INTRODUCTION

The Stockholm Convention is a global treaty to protect human health and the environment from persistent organic pollutants (POPs), of which twelve are currently listed under the Convention. POPs are chemicals that remain intact in the environment for long periods, become widely distributed geographically, accumulate in living organisms and can cause harm to humans and the environment. The European Union and its Member States, which are Parties to the Stockholm Convention, submitted a proposal in July 2006 for listing octabromodiphenyl ether in Annex A of the Stockholm Convention pursuant to paragraph 1 of Article 8 of the Convention, and the POPRC agreed that the commercial product Commercial octabromodiphenyl ether – actually a mixture as described below - met the screening criteria of Annex D to the Convention.

1.1 Chemical identity of the proposed substance

This proposal concerns the c-OctaBDE. There are several components in the commercial product, with different properties and potential risks. Thus this risk profile focuses on the assessment of individual components of the commercial product, and the final compilation for an overall assessment of the commercial product itself.

It is believed that little if any c-OctaBDE is produced since the major supplier located in North America stopped production in 2004. The commercially supplied OctaBDE was complex mixture consisting (as of 2001 within the EU Member States) typically of $\leq 0.5\%$ Pentabromodiphenyl ether isomers, $\leq 12\%$ Hexabromodiphenyl ether isomers, $\leq 45\%$ Heptabromodiphenyl ether isomers, $\leq 33\%$ OctaBDE isomers, $\leq 10\%$ Nonabromodiphenyl ether isomers and $\leq 0.7\%$ Decabromodiphenyl ether. The composition of older products or products from non-EU countries may be different from this.

The c-OctaBDE is sold as a technical grade under the Chemical Abstracts Service (CAS) Registry number for the OctaBDE isomer.

IUPAC Name: Diphenyl ether, octabromo derivative (octabromodiphenyl ether, OctaBDE)

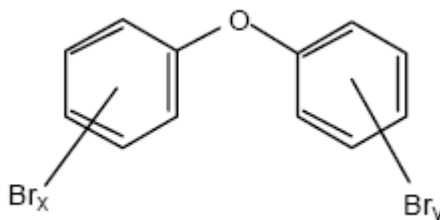
Synonyms: octabromobiphenyl oxide, octabromodiphenyl oxide, octabromo phenoxybenzene and benzene, 1,1' oxybis-, octabromo derivative

CAS Number: 32536-52-0

Molecular formula: $C_{12}H_2Br_8O$

Molecular weight: 801.38

Chemical structure:



Three polybrominated diphenyl ether flame retardants were historically available commercially. They are referred to as penta, octa and decabromodiphenyl ether, but each product is a mixture of diphenyl ethers with varying degrees of bromination.

Several synonyms and abbreviations for polybrominated diphenyl ethers exist and these are shown below:

polybrominated biphenyl ethers \equiv polybromobiphenyl ethers – PBBEs
 polybrominated biphenyl oxides \equiv polybromobiphenyl oxides - PBBOs
 polybrominated diphenyl ethers \equiv polybromodiphenyl ethers - PBDPEs
 polybrominated diphenyl oxides \equiv polybromodiphenyl oxides – PBDPOs

The compositions of the commercial polybrominated diphenyl ethers based on composite samples from the EU suppliers are shown in **Table 1**. These are the substances that have been used in the recent tests and used as a basis for the EU risk assessment reports for the three commercial substances. These data indicate that if tetra- and pentabromodiphenyl ethers are present in the commercial octabromodiphenyl ether or Decabromodiphenyl ether products, they must be present only at very low levels.

Table 1. Composition of commercial polybrominated diphenyl ethers as described in the EU RAR.

Component	% Composition of commercial product			
	Penta-		Octa-	Deca-
	1997	2000	1997	1997
Tribromodiphenyl ether		0.23		
Tetrabromodiphenyl ether	33.7	36.02		
Pentabromodiphenyl ether	54.6	55.10		
Hexabromodiphenyl ether	11.7	8.58	5.5	
Heptabromodiphenyl ether			42.3	
Octabromodiphenyl ether			36.1	0.04
Nonabromodiphenyl ether			13.9	2.5
Decabromodiphenyl ether			2.1	97.4

There is some discrepancy between the composition of octabromodiphenyl ether given in the OECD Voluntary Industry Commitment (VIC) and the composition more recently supplied (**Table 1**), particularly with regard to the levels of the pentabromodiphenyl ether congener. The composition given in the VIC is as follows:

Hexa/pentabromodiphenyl ether 1.4-12.0%¹
 Heptabromodiphenyl ether 43.0-58.0%
 Octabromodiphenyl ether 26.0-35.0%
 Nonabromodiphenyl ether 8.0-14.0%
 Decabromodiphenyl ether 0.0-3.0%

In the VIC it is not clear how much if any pentabromodiphenyl ether actually present. No details of the analyses used were provided. Also, at the time the VIC was set up, production of octabromodiphenyl ether was carried out in the EU. Since then, production moved to sites outside the EU, and some producers have stopped producing octabromodiphenyl ether altogether. This may have had some effect on the composition. From the information presented in Table 1 above, it is clear that if

¹ The Bromine Science and Environmental Forum (BSEF) suggest that the discrepancies are related to analytical limitations at the time, this represents a total of PentaBDE and HexaBDE congeners, however the majority of this % is believed by BSEF to be the hexaBDE fraction.

pentabromodiphenyl ether is present in the commercial product, it will be at much lower levels than the 12% indicated by the VIC. La Guardia et al (2006) have recently reported additional information on the composition of commercial mixtures.

The commercial mixture covered by this entry is therefore a complex combination of isomers and congeners, as defined at POPRC.

There is a tendency in scientific literature to present the identities of polybrominated diphenyl ether congeners using the numbering system based on the polychlorinated biphenyl system. This risk profile will focus on the series of hexa, hepta, octa and nona homologues:

- Hexabromodiphenyl ethers (benzene, 1,1'-oxybis-, hexabromo derivative; hexaBDE) (CAS No. 36483-60-0; IUPAC N° between BDE-128 and BDE-169)
- Heptabromodiphenyl ethers (benzene, 1,1'-oxybis-, heptabromo derivative; heptaBDE) (CAS No. 68928-80-3; IUPAC N° between BDE-170 and BDE-193)
- Octabromodiphenyl ethers (benzene, 1,1'-oxybis-, octabromo derivative; octaBDE) (CAS No. 32536-52-0; IUPAC N° between BDE-194 and BDE-205)
- Nonabromodiphenyl ethers (benzene, 1,1'-oxybis-, nonabromo derivative; nonaBDE) (CAS No. 63936-56-1; IUPAC N° between BDE-206 and BDE-208)

The complexity for setting a risk profile for a complex mixture has been already discussed by the POPRC with reference to the commercial mixture of pentabromodiphenyl ether, and the situation is similar for the commercial mixture of octabromodiphenyl ether. Briefly, there are three main conceptual issues:

- Each isomer and congener may have different physicochemical properties, persistence, bioaccumulation potential, toxicological and ecotoxicological profiles and potential for long range transport
- All, most or several isomers and congeners in the mixture may act through the same mechanism of action and the assessment of the individual risk profiles may not be enough for a proper estimation of the overall risk of the commercial mixture due to additive and synergistic effects.
- The debromination in the environment and biota represents an additional source of bromodiphenyl ethers and related metabolites. The metabolites may be more bioavailable and/or toxic than the parent compounds.

The current complexity when analysing this situation is increased by the reduced availability of information as the physical-chemical, fate, and (eco)toxicological information covers in some cases assays with the commercial mixtures, while in other cases the studies focused on mixtures of isomers and/or homologues, or individual compounds. A full data set for conducting a risk profile is not available for the commercial mixture or for the individual components. Thus the available pieces of information have been combined in this risk profile. The report will present whenever possible the risk expected for the commercial mixture.

1.2 Conclusion of the POP Review Committee of Annex D information

POP RC has evaluated Annex D information and has concluded that proposal fulfils the requirements of Article 8 and Annex D of the Convention. The POPRC decision is included below:

Annex to decision POPRC-2/6

Evaluation of commercial octabromodiphenyl ether against the criteria of Annex D

A. Background

1. The primary source of information for the preparation of this evaluation was the proposal submitted by the European Community and its member States that are Parties to the Convention, contained in document UNEP/POPS/POPRC.2/INF/4.
2. Additional sources of scientific information included critical reviews prepared by recognized authorities, including the European Union risk assessment report on diphenyl ether, octabromo derivative.

B. Evaluation

3. The proposal was evaluated in the light of the requirements of Annex D, regarding the identification of the chemical (paragraph 1 (a)) and the screening criteria (paragraphs 1 (b)–(e)):

(a) Chemical identity:

- (i) Adequate information was provided in the proposal and supporting information. The proposal relates to commercial octabromodiphenyl ether;
- (ii) The chemical structure for the pure compound octabromodiphenyl ether was provided. Commercial octabromodiphenyl ether is a mixture of several polybrominated diphenyl ethers and congeners (pentabromodiphenyl ether isomers, hexabromodiphenyl ether isomers, heptabromodiphenyl ether isomers, octabromodiphenyl ether isomers, nonabromodiphenyl ether isomers and decabromodiphenyl ether isomers);

The chemical identity of commercial octabromodiphenyl ether and the pure compound octabromodiphenyl ether is adequately established;

(b) Persistence:

- (i) There was no degradation in an OECD test (301D) over 28 days (Ref. 3);
- (ii) Elevated concentrations of polybromodiphenyl ethers, including octa and hepta bromodiphenyl ether congeners, were found in agricultural soil more than 20 years after treatment of the soil with contaminated sewage sludge, which is consistent with very long half-lives for components of commercial octabromodiphenyl ether (Ref. 2);

There is sufficient evidence that commercial octabromodiphenyl ether meets the persistence criterion;

(c) Bioaccumulation:

- (i) The log Kow value for the commercial product has been determined to be around 6.29 (Ref. 3). Experimental results presented in the European Union risk assessment report indicates that octa and heptabromodiphenyl ethers have low bioconcentration factors (less than 10–36); these results have been confirmed by data presented and peer reviewed by the Japanese Government. Nevertheless, other brominated diphenyl ethers present in commercial octabromodiphenyl ether have been found to have higher bioconcentration factors, for example 11,700–17,700 for pentabromodiphenyl ethers (Ref. 3) and 1,000–5,600 for hexabromodiphenyl ethers (Ref. 3);

- (ii) and (iii) Field data provide evidence for the potential for bioaccumulation of heptabromodiphenyl ether. Concentrations of 220–270 ng/g lipid weight in eggs of the peregrine falcon in northern Sweden and Greenland have been reported (Refs. 4 and 5). This evidence demonstrates that, despite its large molecular weight, the molecule is found in top predators at levels similar to those of bioaccumulable tetra and penta bromodiphenyl ether. In addition, the estimated half-life in humans is 100 days (Ref. 6), suggesting a potential for bioaccumulation. In soil biota, the soil organism accumulation factor for octabromodiphenyl ether 197 has been calculated as 2 (Ref. 2).

There is sufficient evidence that commercial octabromodiphenyl ether meets the bioaccumulation criterion;

(d) Potential for long-range environmental transport:

- (i) and (iii) The vapour pressure of commercial octabromodiphenyl ether is reported to be 6.59×10^{-6} Pa at 21°C (Refs. 1 and 3). The atmospheric half-life of the pure compound octabromodiphenyl ether is estimated to be 76 days, which means that long-range transport is possible for the substance;
- (ii) Monitoring data show that the hexa and hepta bromodiphenyl ether congeners are present in biota in remote regions (Refs. 7 and 8) and in Arctic air (Ref. 9);

There is sufficient evidence that commercial octabromodiphenyl ether meets the criterion on potential for long-range environmental transport;

(e) Adverse effects:

- (i) There are no data provided on the direct toxicological effects of commercial octabromodiphenyl ether or polybromodiphenyl ether congeners in humans;
- (ii) There is evidence of reproductive toxicity in mammals. The lowest no observed adverse effect level (NOAEL) from the available mammalian toxicity data for the commercial octabromodiphenyl ether product was determined as 2 mg/kg bw/day in a developmental study in rabbits (Ref. 3). Additional information on the developmental toxicity of octabromodiphenyl ether has been published recently (Ref. 10);

There is sufficient evidence that commercial octabromodiphenyl ether meets the criterion on adverse effects;

C. Conclusion

4. The Committee concluded that commercial octabromodiphenyl ether meets the screening criteria specified in Annex D.

The bases for this conclusion were presented in the document UNEP/POPS/POPRC.2/12 – Summary of octabromodiphenyl ether proposal. The key parts of this document, relevant for this risk profile, are reproduced below under paragraphs 1.2.1 to 1.2.5, please note that text included here is the text from the mentioned document and that the discussion at POPRC2 focused on the fulfilment of Annex D criteria and therefore did not cover all these issues.

1.2.1. Persistence

OctaBDE has been found to photodegrade rapidly in a mixture of organic solvents, with a half-life of around 5 hours, but the environmental significance of such a finding is uncertain (European Commission, 2003). Besides, octaBDE is predicted to adsorb strongly onto sediment and soil, which means that only a fraction of this PBDE will be exposed to sunlight, thus having the potential to photodegrade. No information is available on the hydrolysis of octaBDE, but it is not expected to be an important process for octaBDE in the environment.

Regarding biotic degradation, octaBDE is not readily biodegradable in standard tests (no degradation seen over 28 days) and is not expected (by analogy with other brominated diphenyl ethers) to degrade rapidly under anaerobic conditions. Nevertheless, other more highly brominated congeners (deca and nonabromodiphenyl ether) have been found to degrade anaerobically in sewage sludge, although at a very slow rate (Gerecke et al. 2005). The evidence seems to indicate that there is little significant biotic or abiotic degradation of octaBDE.

It's worth noting that degradation of polybrominated diphenyl ethers (PBDEs) can yield byproducts that are lower brominated congeners. For instance, Ahn et al. (2006) showed that decaBDE immobilised on specific soil/sediment and mineral aerosols yielded a number of penta to triBDEs, via octaBDE as an intermediate step. This may pose an additional environmental concern, as these lower brominated diphenyl ethers are usually more toxic and much more bioaccumulative.

1.2.2. Bioaccumulation

The log octanol-water partition coefficient (log Kow) value for the commercial product has been determined to be around 6.29 (European Commission, 2003). Based on its log Kow, octaBDE congener would be expected to be bioaccumulative. However, the experimental result indicates that octaBDE does not bioconcentrate ($BCF < 9.5$), probably due to its large size, which may preclude the crossing of cell walls in organisms.

Nevertheless, other brominated diphenyls present in c-octaBDE have been found to have higher BCFs, for example:

11 700 – 17 700 for pentaBDE (European Commission, 2003); up to 5 600 for hexaBDE (European Commission, 2003).

Thus, lower brominated diphenyls have BCF that meet perfectly the accumulation criteria. As they are not only present in c-octaBDE (penta and hexaBDE make up to 12% of the commercial product) but may also appear as a result of the degradation of the higher brominated diphenyls, c-octaBDE can be considered to be bioaccumulative.

The EU Risk Assessment Report (European Commission, 2003) reports that brominated diphenyl ethers with bromine contents both lower and higher than octaBDE have been detected in some biota samples, notably predatory birds' eggs. Theoretically, higher brominated congeners shouldn't accumulate, as they are large molecules which are not likely to go through cell walls. However, the work of Sellström et al. (2005) shows a noticeable accumulation of these substances (hepta and decaBDE, amongst other BDEs) in wild falcons. Verreault et al. (2005) found accumulation of several octaBDE congeners (both higher and lower brominated) in several environmental samples of two Arctic top predators, and De Wit et al. (2006) reported a variety of PBDEs in the Arctic. Therefore, a similar behaviour could be expected from octaBDE. In addition, other studies (Tomy et al. 2004, Stapleton et al. 2004) mention that biotransformation of PBDEs via debromination can lead to bio-accumulation parameters higher than expected, and a consequent biomagnification risk.

By using the benchmark approach proposed by Scheringer (1997) and Beyer et al. (2000) (which suggests that the intrinsic properties of a substance may be evaluated by studying those of similar substances for which more data exist), it is likely that octaBDE is bioaccumulative.

1.2.3. *Potential for long-range environmental transport*

In the EU Risk Assessment Report (European Commission 2003), the vapour pressure of c-octaBDE is reported to be 6.59×10^{-6} Pa at 21 °C. Brominated diphenyl ethers as a group all have low vapour pressures, the vapour pressure tending to decrease with increasing bromination. In the same report, the atmospheric half-life for octaBDE is estimated to be 76 days which means that long-range transport is possible for this substance.

Table 1: Water solubility (WS), vapour pressure (VP) and Henry's Law Constant (HLC) (at 25 °C) for c-octaBDE and currently listed POPs

Substance	WS mg/L	VP Pa	HLC Pa m ³ /mol
c-octaBDE *	0.0005	6.59×10^{-6}	10.6
POP-min	0.0012 (DDT)	2.5×10^{-5} (DDT)	0.04 (endrin)
POP-max	3.0 (toxaphene)	27 (toxaphene)	3726 (toxaphene)
POP-2nd max	0.5 (dieldrin)	0.04 (heptachlor)	267 (heptachlor)

* EU Risk Assessment Report

Table 1 shows the water solubility, vapour pressure and Henry's law constant for c-octaBDE, in comparison with the maximum and the minimum for currently listed POPs. Henry's law constant, a key property to determine if there is risk of long-range environmental transport for a substance, is well inside the range set by the other POPs. Considering this fact together with its half-life, it can be concluded that c-octaBDE is quite likely to undergo long-range environmental transport.

There are no monitoring data from remote locations available for octaBDE itself. In general, PBDE concentrations have increased exponentially in arctic biota over the past two decades. The lower brominated congeners (e.g. pentabromodiphenyl ethers and hexabromodiphenyl ethers) present in c-octaBDE appear to be subject to long-range environmental transport, possibly via the atmosphere, as they are widely found in sediment and biota in remote areas (Environment Canada, 2004).

For other brominated congeners, hepta and decaBDE have been demonstrated to occur in airborne particles in the high arctic (Wang et al., 2005). The modelling study by Wania and Dugani (2003, as reviewed in European Commission 2004) concluded that decabromodiphenyl ether was likely to be almost exclusively adsorbed to atmospheric particulates that would effectively control the long-range transport behaviour of the substance. Besides, the presence of decaBDE in moss in relatively remote regions of Norway, and in birds and mammals in Polar Regions, has been attributed to long-range particulate transport (European Commission, 2004).

In summary, the data available for lower and higher brominated congeners (some of them also present in c-octaBDE) show that they have potential for long-range environmental transport. Analysis of c-octaBDE's chemical properties seems to support this conclusion, as Henry's law constant is very similar to those of acknowledged POPs. Therefore, it can be expected that c-octaBDE is subject to long-range environmental transport.

1.2.4. *Adverse effects*

The available ecotoxicity data for the c-octaBDE product show little or no effect on aquatic organisms (short-term fish study and a longer-term *Daphnia magna* study), sediment organisms (*Lumbriculus variegatus*) and soil organisms (three species of plant and earthworms *Eisenia fetida*) (European Commission 2003). However, the EU Risk Assessment Report identifies a risk of secondary poisoning in other species (via ingestion of earthworms) for the hexabromodiphenyl ether component in the c-octaBDE product (from use in polymer applications).

The EU Risk Assessment Report (European Commission 2003) reviews the available toxicological studies on octaBDE. In that report, the lowest no observed adverse effect level (NOAEL) from the available mammalian toxicity data for the c-octaBDE product is determined as 2 mg/kg bw/day in a developmental study with rabbits. Using this data, a predicted no-effect concentration (PNEC) of 6.7 mg/kg food was derived in the EU Risk Assessment Report. Within the EU, c-octaBDE has been classified as "Toxic", due to its effects on human health, with the risk phrases "may cause harm to unborn child", and "possible risk of impaired fertility".

The presence of lower brominated diphenyl ethers in the c-octaBDE products is of concern also from the human health point of view as they are likely to have a higher potential to cause adverse effects. WHO (1994) and more recently Birnbaum and Staskal (2004) have reviewed the toxicological data on PBDEs in general.

All the abovementioned studies and assessments provide evidence that c-octaBDE causes adverse effects. The possible formation of brominated dibenzo-p-dioxins and dibenzofurans during combustion or other high temperature processes involving articles containing c-octaBDE is another cause of concern (European Commission, 2003).

1.2.5. Statement of the reasons for concern

The proposal of the European Union and its member States contains the following statement of concern:

"The fact that c-octaBDE consists of several polybrominated diphenyl ethers and congeners makes the assessment of POP characteristics more difficult than in the case of a single compound. However, it can be concluded that c-octaBDE meets the criteria for persistence, potential for long range environmental transport and potential to cause adverse effects. The situation with regards to the screening criteria for bioaccumulation is not so clear cut but the commercial product does contain at least a component group that has been confirmed by the POPRC to meet all the screening criteria (pentabromodiphenyl ether). It also contains hexaBDE, another congener with POP characteristics.

A second aspect of concern is that although the higher brominated BPDEs are persistent, there is evidence that they can degrade under some conditions. Lower brominated diphenyl ether congeners have been identified among the degradation products. Since some of the products may be more bioaccumulative and toxic than the parent compound, any significant formation would be a cause for concern.

An additional risk is the possible formation of brominated dibenzo-p-dioxins and dibenzofurans during combustion and other high temperature processes involving articles treated with c-octaBDE flame retardants.

Marketing and use of octaBDE has been prohibited recently in the EU but it is assumed still to be produced and used as a flame retardant in many countries. As octaBDE and its congeners can move far from their sources, single countries or groups of countries alone cannot abate the pollution caused by it. Due to the harmful POP properties and risks related to its possible continuing production and use, international action is warranted to eliminate this pollution."

1.3 Data sources

The EU risk assessment report (European Commission 2003), the Canadian assessment (Environment Canada, 2004), and references from the WHO (1994) report were the main source of information used by the POP RC in Annex D screening.

Additional information has been submitted by Canada, the Czech Republic, Germany, Lithuania, Norway, Switzerland, Turkey, UK, USA, the NGO Environmental Health Fund on behalf of the International POPs Elimination Network (IPEN), and the industry organization Bromine Science and Environmental Forum. Considering the large

amount of new scientific information produced nowadays, a review of recent scientific literature has also been conducted and used as an essential data source in this report.

- Note: additional information to be incorporated as a separate POP RC INF document)

1.4 Status of the chemical under international conventions

- OSPAR Convention: Octa-BDE takes part of the list of selected substances for the OSPAR lists (no 236). Under the reviewed list, Octa-BDE is put under section C – about the substances put on hold because they are not produced and/or used in the OSPAR catchment or are used in sufficiently contained systems making a threat to the marine environment unlikely.
- UNECE, Convention on Long-range Transboundary Air Pollution (LRTAP) and its Protocol on Persistent Organic Pollutants (POPs): c-OctaBDE is being considered under Protocol procedures for inclusion.

2 SUMMARY INFORMATION RELEVANT FOR THE RISK PROFILE

2.1 Sources

The information provided by the bromide industry indicates that the commercial product has been produced in The Netherlands, France, USA, Japan, UK and Israel, but since 2004, it is no longer produced in the EU, USA and the Pacific Rim and there is no information that indicates it is being produced in developing countries.

The polybrominated diphenyl ethers in general are used as flame retardants of the additive type. They are physically combined with the material being treated rather than chemically combined (as in reactive flame retardants). This means that there is the possibility that the flame retardant may diffuse out of the treated material to some extent.

Industry indicates that octabromodiphenyl ether is always used in conjunction with antimony trioxide. In Europe, it is primarily used in acrylonitrile-butadiene-styrene (ABS) polymers at 12-18% weight loadings in the final product. Around 95% of the total octabromodiphenyl ether supplied in the EU is used in ABS. Other minor uses, accounting for the remaining 5% use, include high impact polystyrene (HIPS), polybutylene terephthalate (PBT) and polyamide polymers, at typical loadings of 12-15% weight in the final product. In some applications, the flame retardant is compounded with the polymer to produce pellets (masterbatch) with slightly higher loadings of flame retardant. These are then used in the polymer processing step to produce products with similar loadings as given above.

The flame retarded polymer products are typically used for the housings of office equipment and business machines. Other uses that have been reported for octabromodiphenyl ether include nylon and low density polyethylene (WHO, 1994), polycarbonate, phenol-formaldehyde resins and unsaturated polyesters (OECD, 1994) and in adhesives and coatings (WHO, 1994).

Assuming that the commercial octaBDE is not longer produced, the releases to the environment must be associated to historical processes, as well as to releases during

the service life of articles containing the commercial mixtures and at the end of article service life during disposal operations.

The information review by La Guardia et al (2006) allows estimations of the relative contribution of each congener in different markets and time periods. As an example, Figure 1 presents the calculations for European commercial products in 2001.

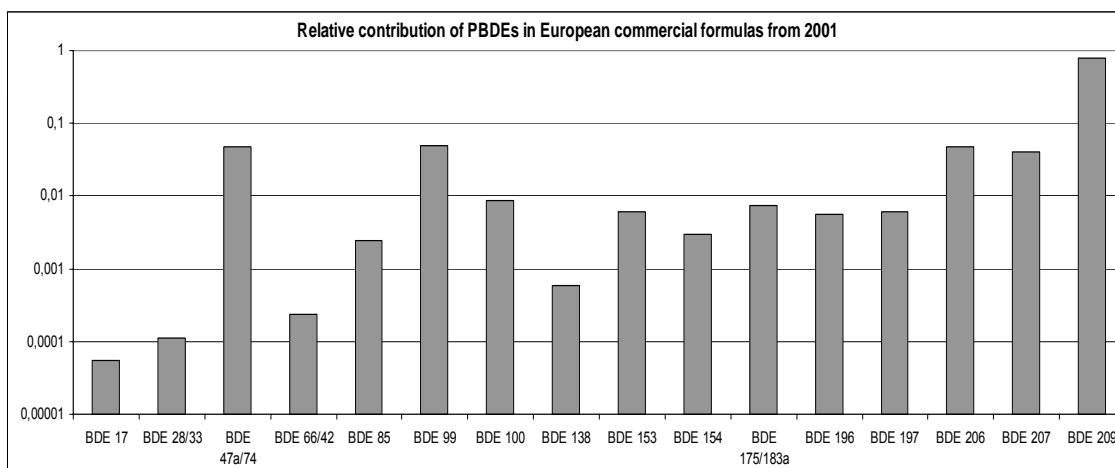


Figure 1. Estimated relative contribution for the different BDE congeners in products in the European market in 2001. Calculated from data published by La Guardia et al., 2006. Note the logarithmic scale.

Although there are some figures on annual production of this mixture, there are no accurate values on the amount of the commercial octa and/or the individual homologues in articles in service and disposed at the world-wide level, but considering the estimated figure of 6 000 tonnes/year (WHO, 1994) the total amount should be expected in the $10^5 - 10^6$ tonnes range. According to the BSEF, OctaBDE was commercialized sometime in the mid 70's. By the early 2000's global production was <4000 tonnes/year and by the time production ceased, demand was <500 tonnes. While Thus, assuming 30 years of production at 6000 tonnes per year gives 180,000 tonnes, a figure within the proposed range.

In a 2002 document (Switzerland info) Switzerland reported for this country diffuse emission from the use of products containing OctaBDE of about 0.37 t/a (based on worst-case estimations) for a total stock of 680 tons.

2.2 Environmental fate

2.2.1 Persistence

No aerobic biodegradation of the hexa- to NonaBDEs is expected based on BIOWIN estimates as recalcitrant with respect to biodegradation, and no degradation, based on oxygen uptake, occurred in a 28-day closed bottle test (OECD 301D).

Gerecke et al. (2005) in a study on DecaBDE reported the degradation of nonaBDE 206 and 207 under anaerobic conditions using sewage sludge inoculum to OctaBDEs; and this degradation has been confirmed in other studies (Gaul et al, 2006; He et al, 2006).

AOPWIN predicts half-lives for reaction with atmospheric hydroxyl radicals ranging from 30.4 to 161.0 d for hexa- to NonaBDEs, respectively. In the atmosphere, octaBDE

is expected to strongly adsorb to suspended particles in the air and be removed via wet and/or dry deposition. Note that predicted half-lives have not been empirically substantiated, but are provided for reference purposes.

The photodecomposition of several BDEs has been studied in different matrices such as methanol/water 80:20 (Eriksson et al. 2001) a sealed polyethylene tube exposed to natural sunlight for up to 120 min (Peterman et al. 2003); or water (Sanchez-Prado et al., 2006); in general degradation was faster for the higher brominated DEs than for the lower brominated congeners. Rayne et al. (2006) suggest a short photochemical half-life for the hexa BDE153 in aquatic systems, with rapid photohydrodebromination to some of the most prevalent penta- and tetra-brominated diphenyl ether congeners.

2.2.2 Bioaccumulation

Bioconcentration factors were reported by European Communities (2003) based on the results of a study by CBC (1982), in which carp, *Cyprinus carpio*, were exposed for 8 weeks to commercial c-octaBDE at 10 or 100 µg/L using polyoxyethylene hydrogenated castor oil as a dispersing agent. If it is assumed that the actual concentrations of the c-octaBDE components were at or around the reported water solubility for the substance of 0.5 µg/L, then the BCF for octaBDE would be <9.5 while the BCF for heptaBDE would be about <1.1-3.8 and the BCF for c-octaBDE would be about <10-36 (European Communities 2003). These BCF values are lower than would be expected from the substance's octanol-water partition coefficients. This was potentially justified by a reduced bioavailability due to the inability of the large molecule to cross cell membranes (European Communities 2003).

The UK has re-analyzed the CITI (1982b) bioconcentration data and suggests BCFs of up to ~5,640 l/kg and ~ 2,580 l/kg for components D and E (both hexaBDPE).

However, toxicokinetic studies and monitoring data on humans and wildlife reviewed by European Communities 2003, Canada Info 2 and others clearly demonstrate the capability of the large BDE molecules to cross cell membranes. In fact UK info indicates that studies on the levels of hexa- to octaBDE in the environment confirm the presence of hepta and octaBDE isomers in some biota samples. Further data (see below) confirms the bioavailability of large BDE molecules. Thus, the discussion on the molecular size is useless and the evaluation should be related to the capability of BCFs to quantify the bioaccumulation potential of these types of molecules (EU_SCHER, 2005).

In fact, oral exposure is expected to be the most relevant exposure pathway for these chemicals. Van Beusekom et al. (2006) reported biota-sediment accumulation factors between 1 and 3 for hexa and heptaBDE on two freshwater fish species in Spain and concluded that 100% of the exposure was associated to food or food plus sediment for bleak (*Alburnus alburnus*) and barbel (*Barbus graellsii*), respectively. The potential for biomagnification has been demonstrated for hexa and heptaBDE (Burreau et al., 2004; Sormo et al., 2006), and more recently suggested for the DecaBDE (Law et al., 2006).

The potential for bioaccumulation and biomagnification of these types of molecules can be calculated using toxicokinetic models, based on metabolism and elimination. Differences among isomers and the reported debromination processes introduce additional uncertainty when reviewing field data.

Ciparis and Hale (2005) have reported a rapid bioaccumulation of hexaBDE in the aquatic oligochaete, *Lumbriculus variegates*, exposed via sediment, with differences between isomers and in the contamination pathway. A biota-sediment accumulation factor of 9.1 ± 1.1 was observed for BDE 154, the highest concentration was found on day 15 and the depuration rate constant was $0.032 \pm 0.016 \text{ days}^{-1}$.

Stapleton et al. (2004) in a dietary study on carps found depuration rates of $0.051 \pm 0.036 \text{ days}^{-1}$ and assimilation efficiencies of $4\% \pm 3$ for the hexaBDE 153.

Stapleton and Baker (2003) and Stapleton et al. (2004b) in dietary studies on common carp (*Cyprinus carpio*) found significant and rapid debromination of heptaBDE183 to hexaBDE154 and to another unidentified hexaBDE congener within the intestinal tissues of the carp after consuming its food. *In vitro* studies have demonstrated the microsomal debromination in fish (Stapleton et al. (2006) and mammals (McKinney et al., 2006).

The role of exposure levels in the elimination rate of several chemicals including hexaBDE 153 has been studied by the LPTC). Université Bordeaux I and the INIA's Laboratory for Ecotoxicology within the context of LRI-Cefic Research Project ECO-1AINIA-1100. Depuration rates of 0.03-0.05 for *Sparus aurata* and *Mytilus edulis*, were obtained (Alonso et al., 2006).

A recent study (Drouillard et al., 2007) has reported a depuration rate constant for the hexaBDE 0.016 days^{-1} in juvenile American kestrels (*Falco sparverius*), with a retention of about 50% of the administered dose.

A controlled feeding trial assessed transfer and accumulation of PBDEs from feed to farmed Atlantic salmon (*Salmo salar*). On average, 95% of the total PBDE content in the feed accumulated in whole salmon including heptaBDE 183 (Isosaari, et al. 2005).

2.2.3 Long range environmental transport

The presence of components of commercial octa BDE in remote areas (e.g. Norway info, Norway Info 2; Canada info 2; Switzerland info2) is considered the best demonstration for the potential for long range transport of these chemicals. As debromination in the environment and biota has been demonstrated, hypothetically, the presence of hexa to nonaBDEs could be explained by a long range transport of decaBDE and its subsequent debromination, however, it is very unlikely to assume a long range transport for decaBDE and not for the nona to hexa congeners.

Previous model predictions suggested a low potential for long-range atmospheric transport for highly brominated BDEs (e.g. Wania and Dugani, 2004). However, in a recent paper on DecaBDE, Breivik et al., (2006) have reported that chemicals that are both sorbed to particles and potentially persistent in the atmosphere, such as BDE-209, may have a larger potential for LRT than anticipated on the basis of earlier model evaluations. This explanation could be also applied to c-OctaBDE components.

Recently Wegmann, et al (2007) applied the OECD Pov and LRTP Screening Tool to the current POPs candidates, including c-octaBDE. The authors noted that they believed that the substance property values for c-octaBDE in Wania and Dugani (2003) were more accurate than the values in the POPRC document and therefore included the Wania and Dugani values in their Monte Carlo uncertainty analysis. Although there were considerable uncertainties, the results indicated that c-octaBDE has Pov and LRTP properties similar to those of several known POPs.

2.3 Exposure

- Summary of relevant information concerning exposure in local areas (both near the source and in remote areas)

2.3.1 Atmosphere

Little information is available about concentrations of c-OctaBDE in the atmosphere. A concentration of 52 000 pg/m³ was predicted for c-OctaBDE in the local atmosphere resulting from emissions from polymer processing (EUSES predictions in European Communities 2003).

Some measurements are available for PBDE congeners present in c-OctaBDE. Strandberg et al. (2001) analyzed air samples from urban, rural and remote sites in the United States near the Great Lakes. The average total c-OctaBDE-related congeners (i.e., sum of BDEs 153, 154 and 190) present in the samples ranged from approximately 0.2 to 0.9 pg/m³.

PBDEs (ranging from tri- to OctaBDEs) were detected in deposition samples collected from sites in The Netherlands, Germany and Belgium, confirming their presence in precipitation (Peters 2003). The PBDE composition of the samples could be linked to the commercial penta- and OctaBDE mixes, with BDEs 47, 99 and 154 the predominant congeners.

Bergander et al. (1995) analyzed air samples from two areas of Sweden remote from industry for the presence of c-OctaBDE. No OctaBDE was detected in either the particulate or gas phase samples (the detection limit not stated), but indications of the presence of hexaBDE and heptaBDE were found in the particulate phase samples.

2.3.2 Water

Luckey et al. (2002) measured total PBDE (mono- to heptaBDE congeners) concentrations of approximately 6 pg/L in Lake Ontario surface waters in 1999, with hexaBDE congeners BDE153 and BDE154 each contributing approximately 5 to 8% of the total.

C-OctaBDE was not detected in 1987 in 75 surface water samples taken in Japan at a detection limit of 0.1µg/L or in 1988 in 147 water samples at a detection limit of 0.07 µg/L (Environment Agency Japan 1991). According to European Communities (2003), the concentrations are considered to be representative of industrial, urban and rural areas of Japan, but it is not known whether any of the sampling sites were in the vicinity of a polybrominated diphenyl ether production site or a polymer processing site.

2.3.3 Sediments

Concentrations of c-OctaBDE in UK sediments ranged from <0.44 to 3030 µg/kg dw (Allchin et al. 1999; Law et al. 1996). The highest levels were in sediments downstream from a warehouse where c-DecaBDE was stored (Environment Agency 1997). C-OctaBDE was detected in 3 of 51 sediment samples from Japan in 1987 at concentrations from 8 to 21 µg/kg (detection limit 7 µg/kg; ww or dw not specified), and in 3 of 135 samples collected in 1988 at concentrations of 15 to 22 µg/kg (detection limit 5 µg/kg; ww or dw not specified) (Environment Agency Japan 1991).

Kolic et al. (2004) presented levels of PBDEs in sediments from tributaries flowing to Lake Ontario, and area biosolids in southern Ontario. Total hexa- and heptaBDEs (i.e., BDE 138, 153, 154 and 183) measured in sediment samples taken from fourteen tributary sites (only 6 sites were reported) ranged from approximately 0.5 to 4.0 µg/kg dw.

2.3.4 Soil

Hassanin et al. (2004) determined PBDEs in undisturbed surface soils (0-5 cm) and subsurface soils from remote/rural woodland and grassland sites on a latitudinal transect through the United Kingdom and Norway. In total, 66 surface soils were analyzed for 22 tri- to heptaBDEs. Concentrations of total PBDEs in the surface soils ranged from 0.065 to 12.0 µg/kg dw. Median PBDE concentrations in the surface soils ranged from 0.61 to 2.5 µg/kg dw, with BDEs 47, 99, 100, 153 and 154 dominating the total concentrations. The median concentration of the sum of these five congeners ranged from 0.44 to 1.8 µg/kg dw. The researchers noted that the congener patterns in the European background soils closely matched that reported for the c-pentaBDE mixture. Northward along the latitudinal transect, there was an increasing relative contribution of BDE 47 and other lighter PBDEs in comparison to the heavier PBDEs measured in the samples.

2.3.5 Waste Effluent and Biosolids

Kolic et al. (2004) presented levels of PBDEs in sediments from tributaries flowing to Lake Ontario, and of biosolids from nearby wastewater treatment facilities in southern Ontario. Total hexa- and heptaBDEs (i.e., BDEs 138, 153, 154 and 183) measured in biosolids ranged from approximately 111 to 178 µg/kg dw.

La Guardia (2001) analyzed 11 sewage sludge samples before land application from Canada and the United States and found that total hexa- to OctaBDE congener concentrations ranged from 40 to 2080 µg/kg dw. Kolic et al. (2003) investigated PBDE levels in sewage sludge from 12 sites in southern Ontario and found hexa- to OctaBDE congener concentrations totaled 124 to 705 µg/kg dw. Hexa- to OctaBDE congeners were not detected in manure samples, and were at very low levels in pulp mill biosolids (up to approximately 3 µg/kg dw).

Martinez et al. (2006) have recently reported concentrations of sum of hexa to nonaBDE in the range of 15.5 to 160 µg/kg dw in sludge from municipal wastewater treatment facilities in Spain, and up to 268 µg/kg dw in industrial facilities.

Gevao et al. (2006) measured PBDEs in coastal sediments receiving industrial and municipal effluents in Kuwait. Total concentrations varied from 80 to 3800 pg/g dw with heptaBDE183 dominating the congener distribution which resembled the commercial formulation, Bromkal 79-8DE. Wastewater discharge from industrial activities appeared to be the primary source of the compounds.

2.3.6 Biota

Concentrations of components found in c-OctaBDEs in biota were reviewed in Law et al. (2003). The concentration of c-OctaBDE (reported as the commercial mixture DE-79) in various biota found in aquatic environments in the UK ranged up to 325 µg/kg ww in the liver of dab (Allchin et al. 1999). Concentrations of OctaBDE in muscle tissue from UK fish ranged from <1 to 12 µg/kg ww (Allchin et al. 1999). In Japan, OctaBDE

was not detected in 75 fish samples taken in 1987 (detection limit 5 µg/kg ww), nor was it detected in 144 fish samples taken from 48 locations in 1988-89 (detection limit 4 µg/kg; ww or dw not specified) (Environment Agency Japan 1991). HeptaBDE, along with other PBDE congeners, was detected in eggs of peregrine falcons, *Falco peregrinus*, from Sweden, at concentrations from 56 to 1300 µg/kg lipid (Lindberg et al. 2004).

Alaee et al. (1999) sampled lake trout from Lakes Superior, Huron and Ontario and found that the total of hexaBDE and heptaBDE congeners ranged from an estimated 11 to 53 µg/kg lipid.

Rice et al. (2002) compared PBDE levels and congener patterns in carp and bass sampled from two industrialized regions in the eastern U.S. The fish were collected from the Detroit River, MI. and the Des Plaines River, IL. in May and June of 1999, and analyzed for the presence of BDEs 47, 99, 100, 153, 154, 181, 183 and 190. Both river systems are considered to receive high contributions from municipal and industrial effluents. BDE47 dominated in fish taken from the Detroit River, comprising an average of 53 to 56% of the total PBDEs by wet weight. BDEs 99, 100, 153 and 154 each contributed between 8 and 9%, and BDEs 181 and 183 each comprised about 5% of the total PBDEs. BDE190 was not detected in either fish species. Only carp were sampled from the Des Plaines River, and these exhibited a markedly different PBDE profile from that seen in the Detroit River fish. HeptaBDEs 181 and 183 were predominant, contributing about 21% and 19%, respectively. BDE47 was third in prevalence, comprising about 17% of the total PBDEs. Levels of the two hexaBDE congeners, BDEs 153 and 154 were 8 to 13%, compared with about 5% for each of the penta- congeners, BDEs 99 and 100. BDE190, not detected in the Detroit River fish, was present at about 12% of total PBDE.

The congener profile identified in the Detroit River fish, with predominance of the tetra- and pentaBDE congeners, was consistent with patterns reported in biota from other parts of North America and the world. The greater prevalence of higher brominated congeners evident in the Des Plaines River fish, however, was atypical. The authors postulated that the unusual congener pattern displayed in fish from the Des Plaines River may relate to the nature of wastewater discharges from manufacturing or waste facilities in the region. The higher quantities of hexa- and heptaBDEs evident in the fish samples may reflect higher discharge volumes of commercial OctaBDE products, so that levels measured in fish represent a combination of commercial pentaBDE and OctaBDE sources. The researchers note that the possible sources for the other heptaBDEs, BDE 181 and BDE 190 that were found in the carp in Des Plaines River are not obvious especially since no commercial products have been documented as containing major quantities of these congeners. The authors speculate that the significant presence of heptaBDE in the Des Plaines River fish may result from active metabolism of BDE209 present in the river sediment; however river sediment concentrations were not obtained in this study. They also suggest that differing contributions to the two river systems from municipal wastewater treatment facilities potentially played a role, but that congener determinations in effluent sources were also not determined in this study.

Norstrom et al. (2002) evaluated the geographical distribution and temporal trends (during the 1981 to 2000 period) of PBDEs in herring gull (*Larus argentatus*) eggs from a network of colonies scattered throughout the Great Lakes and their connecting channels in 2000 (see Section 2.1.6.6 and Appendix D). Although samples were analyzed for octa- to DecaBDE, these were not found at their respective limits of detection (0.01-0.05 µg/kg ww). However, total concentrations of hexa- and heptaBDE congeners (i.e., BDEs 153,154 and 183) increased 6 to 30 fold over the 1981 to 2000

period at the Lake Michigan (from 6.7 to 195.6 µg/kg ww), Lake Huron (from 13.8 to 87.6 µg/kg ww) and Lake Ontario (3.8 to 112.1 µg/kg ww) sites. This increase was not as dramatic as that found for the tetra- and pentaBDE congeners.

Wakeford et al. (2002) conducted sampling of wild bird eggs in western and northern Canada between 1983 and 2000. They determined that the total of hexa- and heptaBDE congeners ranged from 0.148 to 52.9 µg/kg ww in Great Blue Heron (*Ardea herodias*) eggs (on Canada's west coast), 0.03 to 0.68 µg/kg ww in Northern Fulmer (*Fulmarus glacialis*) eggs (in the Canadian arctic) and 0.009 to 0.499 µg/kg ww in Thick Billed Murre (*Uria lomvia*) eggs (in the Canadian arctic). OctaBDE, nonaBDE and DecaBDE congeners were subject to analysis by the researchers, but were not detected (detection limit was not specified) in the any of the samples.

2.3.7 Humans

European Communities (2003) presents some information on the levels of components of c-octaBDE measured in human samples. Large variations among individuals were generally observed, but significant differences between the control population and occupationally exposed groups were also reported.

In a recent study (Toms et al., 2007) the concentrations of PBDEs found in Australian human milk were lower than those reported from North America but higher than those reported from Europe and Asia

- Summary of relevant information concerning exposure as a result of LRET

Measured levels of components of c-OctaBDE in biota from remote areas seem to be the best available information for estimating exposure as result of LRET for these chemicals. Knudsen et al (2005; Norway info) have recently review temporal trends of PBDE in eggs from three bird species, three locations and three sampling times (from 1983 to 2003) from Northern Norway. Spatial differences were only observed for hexaBDE 153, and increases in the measured concentration from 1983 to 2003 were observed for the hexaBDE 153 and 154 and the heptaBDE 183. Mean values were around 1 µg/kg ww for each isomer and maximum values above 10 µg/kg ww were observed for BDE 154 and 183. Inter species differences could be associated to feeding behavior and migration. In general the concentrations were lower than those reported for similar species in industrialized areas and those observed in terrestrial predatory birds.

Exposure to components of c-OctaBDE in remote areas is confirmed and based on the available information should be attributed to a combination of releases and transport of c-OctaBDE, c-pentaBDE (for hexaBDE) and c-DecaBDE (for nonaBDE), and to the debromination of DecaBDE in the environment and biota. There is no sufficient information for assessing these processes in quantified terms. The exposure route is mainly via food, and even for water column animals water exposure is of low, if any, relevance; therefore the BCFs are not suitable parameters for estimating the bioaccumulation potential.

In addition to the feeding strategy, several additional confounding factors are associated to the species to specific differences observed in the isomer distribution pattern of PBDE in wildlife. These factors include, among others, species-specific differences in assimilation, metabolism and depuration of different isomers, even with the same level of bromination.

The presence of hexa and heptaBDE in fish from remote alpine lakes in Switzerland (Switzerland info2) reported to be related to atmospheric deposition confirms the potential for atmospheric long-range transport.

- Information on bioavailability

Despite its large molecular size, the evidence demonstrates the capability of c-OctaBDE components to cross the cellular membranes and to accumulate in biota. Exposure from water is not relevant and significant assimilation rates have been observed for oral and sediment exposures. Although the information is limited, the assimilation and metabolisms of each isomer may vary significantly among species, but also in relation to the administered dose. As a consequence, it is essential to understand the toxicokinetics of these chemicals at environmentally relevant concentrations.

These differences would justify the disparities observed in the assessment of biomagnification potential for different trophic chains.

Like for other chemicals with similar properties, aging processes are expected to reduce the bioavailability, and the experiments conducted on sediment dwelling organisms comparing the bioaccumulation in spiked sediments and from contaminated biosolids offer and indirect support for this hypothesis.

2.4 Hazard assessment for endpoints of concern

Experimental studies

Aquatic Organisms

The EU Risk Assessment report (European Communities 2003), presents a set of studies on the commercial mixture and concludes that for water it seems sensible to assume that no adverse effects on aquatic organisms are likely to occur at concentrations up to the substance's water solubility. However it must be noted, first, that aquatic organisms are also exposed from food and/or sediment; and second, that setting this strong conclusion on chemicals such as PBDEs requires multigenerational or at least full life-cycle assays on the three taxonomic groups covering a large list of sublethal effects, information which is unavailable at this time.

Benthic Organisms

There are two available 28 day spiked sediment studies on *Lumbriculus variegatus* using the c-OctaBDE product (Great Lakes Chemical Corporation 2001a, b). These studies found no statistically significant effects relevant to survival, reproduction or growth at the highest tested concentration (1272 mg/kg dw and 1340 mg/kg dw measured for sediments with 2.4% and 5.9% OC, respectively). Kinetic data from Ciparis and Hale (2005) confirms the expected exposure and bioaccumulation under these conditions.

Soil Organisms

Survival and growth of earthworms, *Eisenia fetida*, were not affected by a 56 day exposure to a commercial OctaBDE formulation in an artificial soil at concentrations up to 1470 mg/kg dw (measured concentration in sediments with 4.7% OC) (Great Lakes Chemical Corporation 2001c).

The toxicity of c-OctaBDE to corn (*Zea mays*), onion (*Allium cepa*), ryegrass (*Lolium perenne*), cucumber (*Cucumis sativa*), soybean (*Glycine max*), and tomato (*Lycopersicon esculentum*) was evaluated in a 21-day emergence and growth study using an artificial sandy loam soil (Great Lakes Chemical Corporation 2001d). No statistically significant effects were observed for any plant species between the controls and the treatments for emergence, survival or growth at any of the tested concentrations (up to 1190 mg/kg dw, measured concentration).

Mammals and Birds

The lowest reported NOAEL for traditional endpoints is a NOAEL of 2 mg/kg/d based on slight fetotoxicity at 5 mg/kg/d (considered relevant in the EU report) or 5 mg/kg bw/d based on increased liver weights and decreased body weight gain among the maternal treatment group and delayed fetal skeletal ossification at 15 mg/kg bw/d (for those reviewers that do not consider relevant the slight fetotoxicity effects) described by Breslin et al. (1989) in a developmental toxicity study with Saytex 111 on New Zealand White rabbits exposed orally via gavage over days 7 to 19 of gestation.

Effects on other endpoints have been described at lower concentrations, including:

- A significant increase in EPN detoxification and *p*-nitroanEROD and isole demethylation in male Sprague-Dawley rats at an oral dose of 0.60 mg/kg bw/day OBDE formulation for 14-days.
- dose-dependent depletion of serum total thyroxine T4 and induced pentoxiresorufin O-deethylase (PROD) activities in rats receiving 10 or more mg/kg bw/day of commercial octabromo (Zhou et al. 2001)
- Delayed neurotoxic effects. Neonatal mice exposed to a single dose of 0.45 mg BDE153/kg bw on postnatal day 10 showed when tested at 2, 4 and 6 months of age altered motor behavior. Spatial learning ability and memory function in the adult mice were also affected Viberg et al. (2001a)
- Eriksson et al. 2002a confirmed neurotoxic effects (aberrant behavioral responses) on developing male mice exposed to 0.45 to 9.0 mg/kg bw of BDE153 on day 10 of development. The effects were comparable to those observed for PCB153 leading the authors to speculate that interactive neurotoxic action may be possible between the two compounds.
- These neurotoxic effects have also been observed after a single oral dose of nonaBDE 206 or OctaBDE 203 administered on postnatal day 3 or 10 to, or PBDE 183; with disturbances in spontaneous behavior, leading to disrupted habituation and a hyperactive condition in adults at the age of 2 months. (Viberg et al., 2006).
- Immunomodulation effects in captive nestling American kestrels (*Falco sparverius*) have been reported by Fernie et al. (2005). Eggs within each clutch, divided by laying sequence, were injected with safflower oil or penta-BDE congeners-47, -99, -100, and -153 dissolved in safflower oil (18.7 microg PBDEs/egg). For 29 days, nestlings consumed the same PBDE mixture (15.6+/-0.3 ng/g body weight per day), reaching PBDE body burden concentrations that were 120x higher in the treatment birds (86.1+/-29.1 ng/g ww) than controls (0.73+/-0.5 ng/g ww). PBDE-exposed birds had a greater PHA response (T-cell-mediated immunity), which was negatively associated with increasing BDE-47 concentrations, but a reduced antibody-mediated response that was positively associated with increasing BDE-183 concentrations. There were also structural changes in the spleen (fewer germinal centers), bursa (reduced apoptosis) and thymus (increased macrophages), and negative associations between the spleen somatic index and PBDEs, and the bursa somatic index and BDE-47. Immunomodulation from

PBDE exposure may be exacerbated in wild birds experiencing greater environmental stresses.

- Fernie et al., 2006 also reported for the same species and test conditions that exposure did not affect hatching or fledging success. PBDE-exposed nestlings were larger (weight, bones, feathers) as they gained weight more quickly and ate more food, the latter in association with their PBDE body burdens. BDE-100 was most influential on nestling growth, being positively associated with size, weight gain, and food consumption. Increasing concentrations of BDE-183 and -153 were related to longer bones and BDE-99 to longer feathers. The larger size of the PBDE-exposed birds may be detrimental to their bone structure and have excessive energetic costs.
- In vitro studies indicates that BDE (including the hexaBDE 153) affected protein kinase C (PKC) and calcium homeostasis in cerebellar granule neuronal cultures in a similar way to those of a structurally-related polychlorinated biphenyl (PCB) (Kodavanti et al., 2005)

Monitoring data on effects

There are several scientific papers comparing population effects observed in the field with measured concentrations of POP like chemicals, including hexa to nonaBDE in individuals from different species.

Unfortunately, wild populations are co-exposed to a mixture of PBDEs as well as to other related brominated and chlorinated persistent pollutants, and with the current level of knowledge epidemiological investigations can just present associations but no cause-effect relationships between the exposure/accumulation of the components of the commercial OctaBDE mixtures and potential adverse effects observed in wildlife.

A similar situation is observed regarding human health data, and no studies offering conclusive evidence on the hazards of hexa to nonaBDE for humans at environmentally relevant exposure levels have been found.

3 SYNTHESIS OF THE INFORMATION

The evaluation of the specific risks of commercial OctaBDE is complex and uncertain as the consequence of:

- the presence of its components in commercial penta- and DecaBDE,
- the additional transformation in the environment through debromination, in particular the evidence suggesting a significant role of debromination of Octa to DecaBDE to other PBDE congeners in biota is highly relevant as food is expected to be the main exposure route for these chemicals, although increases the difficulties for quantitative assessments, and
- mostly, because the lack of a solid body of toxicological and ecotoxicological information for the mixture and its components covering the long-term low level exposure conditions and the sublethal endpoints considered relevant for assessing the risk of a POP candidate.

In this risk profile, hexa to nonaBDE have been considered the relevant components in c-OctaBDE. It should be noted that other BDE are also found in commercial mixtures, but those are expected to be covered by the c-penta and c-deca entries. It should be

noted that theoretically this represent 80 different congeners, while the available information focuses on less than ten congeners and some mixtures.

The persistence of these PBDE in the environment is well documented. The only relevant degradation pathways identified until now are photolysis, anaerobic degradation and metabolism in biota, acting through debromination and producing other BDE which may have higher toxicity and bioaccumulation potential.

The bioaccumulation potential of these highly brominated BDEs cannot be described by the BCF as waterborne exposure is of low if any relevance. As bioaccumulation is the result of assimilation efficiency, metabolism and elimination rate, a reduced assimilation does not necessarily result in low bioaccumulation potential. In fact, if metabolisms and elimination rates are very low, the reduced assimilation will result in longer times for achieving the steady state conditions, but for species with long life spans a significant bioaccumulation may be expected.

The bioaccumulation potential for sediment exposure and particularly for exposure via food is well documented for some hexa to nonaBDE. There is also enough toxicokinetic information demonstrating that elimination rates in some vertebrate and invertebrate species are equivalent to those observed for other POPs, with values in the range of 0.01 to 0.5 days⁻¹, equivalent to a half life of about 14 to 70 days assuming first order kinetic; and therefore, low enough for supporting a justified concern on the bioaccumulation potential of Hexa to NonaBDE. Monitoring programs measuring PBDE concentrations in biota conducted in industrialized and remote areas confirm this potential for bioaccumulation.

It should be noted that differences in both assimilation and metabolism, and therefore in the overall bioaccumulation potential, have been observed not only among homologues, but also between isomers with the same level of bromination. Inter-species differences have also been reported. These differences may explain the discrepancies in terms of food chain biomagnification potential described in the literature.

In summary, a bioaccumulation potential is confirmed at least for some isomers, as well as biomagnification in some food chains. As debromination into other POP-like chemicals is expected to be a relevant contribution to the dissipation of hexa to nonaBDE, the absence of food-chain biomagnification for a specific congener on a specific taxonomic group does not necessarily decrease the overall concern.

In fact, biota monitoring data in remote areas offer the best demonstration on the potential for long range transport of c-OctaBDE components, in particular for Hexa and HeptaBDE. Theoretically this presence could also be explained by the transport of DecaBDE and its subsequent debromination. However, the comparative analysis of the available information on the physical-chemical properties of the different PBDE homologues indicates that debromination from DecaBDE might contribute to the process but it is not realistic to assume that this explains the process without additional transport from other congeners. Thus, based on the available information a long-range transport is expected for the c-OctaBDE components, and the role of atmospheric transport is confirmed at least for Hexa and HeptaBDE based on its detection in alpine lakes.

Unfortunately, the available information on the toxicity and ecotoxicity of hexa to nonaBDE is very limited and does not offer enough information for presenting sound toxicological and ecotoxicological profiles for each isomer, mixtures of isomers and commercial mixtures.

No relevant effects have been observed in aquatic, sediment and soil laboratory studies. The uptake of BDEs at least for the sediment exposure is expected within the time-frame of the experiment, based on toxicokinetic studies. However, this information cannot be used to conclude that Hexa to NonaBDE are not toxic for these organisms. In fact, the measured endpoints and the exposure conditions, employed in these assays are clearly insufficient for a proper assessment of chemicals such as hexa to nonaBDE. Ecotoxicity tests on these types of chemicals should cover if possible several generations or at least a full life cycle, and the measured endpoints must include sublethal effects associated to the accumulation and re-mobilization of the PBDEs during critical periods of development and reproduction, as well as the ecologically relevant consequences of metabolic changes. In addition, all environmentally relevant exposure routes must be addressed. The available tests do not fulfill these conditions, thus, although no toxicity was reported, no concluding statements on the toxicity of hexa to OctaBDE congeners for aquatic, sediment and soil dwelling organisms can be presented.

The available information on mammals and birds offer relevant information. The lowest reported NOAEL for traditional endpoints is 2-5 mg/kg bw/d based on slight fetotoxicity or increased liver weights and decreased body weight gain among the maternal treatment group and delayed fetal skeletal ossification. These effects are relevant for the health and the ecological assessment and therefore useful for assessing risks for humans and wildlife. Nevertheless, the additional available information also creates concerns on the capability of these traditional endpoints for assessing the toxicological profile of hexa to nonaBDE in mammals and other vertebrates.

The metabolic effects, their relevance and they level of coverage by the endpoints measured in the experimental toxicity test should be considered.

However, the immuno-toxicological effects and particularly the delayed neurotoxic effects observed after a single dose require specific attention. Although a quantitative evaluation of these effects in terms of its potential risk for human health and ecosystem is not possible based on the current level of information, the reported observations must be analyzed with care. Certainly, the doses at which the effect have been observed are well above exposure levels in remote areas estimated from current monitoring data for a single congener. However, the effects have been observed for different congeners, and realistic environmental exposure occurs for a mixture of PBDEs. There is not enough information for considering if these effects may be additive or even more than additive in synergistic exposures. The margins between effects observed in the lab and estimated oral exposure levels in the field (based on monitoring data) are not so high when the different isomers/homologues are sum. McDonald (2005) estimated a critical body burden for hexa BDE 153 of 2000 µg/kg lipid based on the NOEL of 0.45 mg/kg reported by Viberg et al 2003 and gives a margin of safety of 7 between this level and the 95 percentile of total PBDE levels in US human populations. It should be noted that hexa BDE 153 concentrations close to these value have been found in several species and geographic sites (see Canada info 2 for a review) and total PBDE concentrations frequently exceed largely this threshold.

Three additional concerns must be mentioned in the risk profile of PBDEs:

- First, the reported debromination of highly brominated BDEs in the environment and biota suggests a potential for the formation of congeners with demonstrated POP properties such as those included in the risk profile of c-pentaBDE.

- Second, there is an increasing evidence suggesting similar toxicological profiles and therefore, equivalent hazards and concerns, between PBDEs and PCBs, although the mode of action seems to be better categorized by AhR-independent mechanisms, as PBDEs do bind but not activate the AhR-AhR nuclear translocator protein-XRE complex (Peters et al., 2006) and appear capable of up-regulating CYP2B and CYP3A in rats at doses similar to that for non-dioxin-like PCB153 (Sanders et al., 2005). As the persistence, bioaccumulation potential and long range transport of the c-octaBDE components are well documented, the confirmation of an equivalent level of hazard for these two groups should be sufficient for confirming a long-range transport associated risk
- Third, the potential for formation of polybrominated dibenzo-p-dioxins and furans (See European Communities, 2003 for a general discussion), although not considered in this risk profile, should also be taken into account.

4 CONCLUDING STATEMENT

The evaluation of the human and environmental risk of commercial OctaBDE associated to its potential for long range transport is not an easy task as the commercial product is a mixture of components with different properties and profiles, which may also be released to the environment due to its presence as components of other PBDE commercial products and also produced in the environment by debromination of commercial decaBDE.

Although the production of c-OctaBDE has ceased in developed countries and there is no information suggesting that the chemical is produced elsewhere; it must be noticed that the product is still present and released from articles in use and during their disposal. Model estimations and measured levels in sewage sludge suggest that current emissions are still significant.

The persistence of the hexa to nonaBDE is well documented and the main route of degradation is debromination forming other BDEs, also of concern. The potential for certain components in c-OctaBDE to bioaccumulate and also for biomagnification in some trophic chains is also sufficiently documented and confirmed by the good agreement between field observations in monitoring programmes and toxicokinetic studies. Monitoring data in remote areas confirm the potential for long-range transport and at least for some congeners the relevance of atmospheric distribution in this process.

The highest difficulty appears for the estimation of the potential hazard of the commercial mixture and its components. There are traditional ecotoxicological and toxicological studies where no effects have been observed even at unrealistically high concentrations. However, an in-depth assessment of these studies considering in particular the properties and toxicokinetic of PBDE indicates that the test design, exposure conditions and measured endpoints are not appropriate for a sound assessment of these types of chemicals. Thus, the lack of effects reported in those tests should be considered with care.

In addition, specific studies have reported particular hazards such as delayed neurotoxicity and immunotoxicity which may be particularly relevant in the assessment of both human health and ecosystem risks.

Based on the existing evidence, additional concerns related to the debromination into toxic BDEs, the increasing evidence relating these chemicals with other POPs (similarities between PBDEs and PCBs; relationships with dioxins and furans), and that under Article 8, paragraph 7(a) of the Convention the lack of full scientific certainty shall not prevent a proposal from proceeding, it is concluded that the components of c-OctaBDE, Hexa to NonaBDE, are likely, as a result of LRET, to lead to significant adverse human health and/or environmental effects, such that global action is warranted.

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