別添6

クロルデコンの危険性の概要

分解性	蓄積性	人健康影響	動植物への影響
【生分解性･加水分解性】 水生環境中であるいは土壌中で、生分解又は加水分解するとは予測されない。 【光分解性】 大気中で直接的光分解を受けることは考えられないと結論している。 ･利用可能な全てのﾃﾞｰﾀに基づき、ｸﾛﾙﾃﾞｺﾝは環境中で高い残留性を示すと考えられる。	【ｵｸﾀﾉｰﾙ/水分配係数】 logKow=4.50-5.41 【BCF（経鰓的生物濃縮係数）】 ･藻類：BCF=6000 ･無脊椎生物：BCF=21600 ･魚類：BCF=60200 【BMF（経口的生物濃縮係数）】 ･ほとんど又は全く代謝浄化せず、水生の食物連鎖において生物濃縮の可能性がある。 ･食物連鎖の研究において、藻からｶｷへの移動は非常に低かったが、ｴﾋﾞからｱﾐ、ｱﾐからｽﾎﾟｯﾄへの明白な栄養段階を通じた移動があることが示された。	【反復投与毒性】 ラット（2 年）：NOAEL 0.05mg/kg/day 0.25mg/kg/day で腎臓影響（蛋白尿、重篤な糸球体硬化） ラット（経口 21 ヶ月）：LOAEL 0.07mg/kg/day 肝細胞の病理組織学的変化、甲状腺ろ胞サイズ、コロイド量胞数低下、甲状腺ろ胞上皮細胞厚高値 ラット（経口 3 ヶ月）：LOAEL 1.17mg/kg/day 肝の病巣壊疽、副腎肥大、振戦、多動性、過剰驚愕反応等 【生殖毒性】 ラット（3 ヶ月）：NOAEL 0.25mg/kg/day 精巣萎縮 ラット（90 日）：LOAEL0.83mg/kg/day で精子の運動性･生存率低下、精子数減少、1.67mg/kg/day で性嚢、前立腺重量低下 マウス（160 日）：LOAEL 2mg/kg/day で排卵停止、膣発情持続、ﾗｯﾄ妊娠 14-20 日に母体経由で 15mg/kg/day 投与した雌児動物においても同様の報告	【慢性毒性】 ミジンコ *Daphnia magna* ：21dNOEC=0.0283 mg/L（繁殖）, 21dNOEC=0.025 mg/L（成長） ミシッドシュリンプ *Americamysis bahia* ：28dMATC=0.000026-0.00034 mg/L（成長） ユスリカ *Chironomus tentans* ：14dNOEC=17.9 mg/kg sediment（発達）

		【催奇形性】 ラット(経口):LOAEL 2mg/kg/day で胎児体重低下、骨化度低下、10mg/kg/day で脳水腫、停留精巣、腎盂肥大、脳室肥大 【発がん性】 ラット(80 週):LOAEL 1.2mg/kg/day 肝細胞腺がん及び上皮がん IARC グループ2B (possibly carcinogenic to human) 【その他】 職業ばく露で振戦、情緒不安定、視力障害、筋力低下、歩行運動失調等、実験動物で、脾臓、胸腺重量、好中球数、NK 活性低下、 EU-Strategy for Endocrine Disruptors 優先化学物質(無処置動物の少なくとも一種類において内分泌かく乱活性を示す科学的根拠がある)に分類	

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Report of the Persistent Organic Pollutants Review Committee on the work of its third meeting

Addendum

Revised risk profile on chlordecone

At its third meeting, the Persistent Organic Pollutants Review Committee revised and adopted the risk profile on chlordecone, on the basis of the draft contained in document UNEP/POPS/POPRC.2/17/Add.2. The text of the risk profile, as amended, is set out below. It has not been formally edited.

K0763736 141207

CHLORDECONE

RISK PROFILE

Adopted by the Persistent Organic Pollutants Review Committee
at its third meeting

November 2007

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Executive summary

The European Community and its member states being parties to the Stockholm Convention have proposed chlordecone to be listed in the Convention. The Persistent Organic Pollutants Review Committee concluded in its meeting in November 2005 that the substance complies with the screening criteria set out in Annex D of the Convention and that a draft risk profile should be prepared to review the proposal further.

Chlordecone is a synthetic chlorinated organic compound, which has mainly been used as an agricultural insecticide, miticide and fungicide. It was first produced in 1951 and introduced commercially in the United States in 1958 (trade names Kepone® and GC-1189). It was available in the United States until 1976. In France, chlordecone was marketed with a trade name Curlone from 1981 to 1993. Historically, chlordecone has been used in various parts of the world for the control of a wide range of pests. It has been used extensively in banana cultivation against banana root borer, as a fly larvicide, as a fungicide against apple scab and powdery mildew and to control the Colorado potato beetle, rust mite on non-bearing citrus, and potato and tobacco wireworm on gladioli and other plants. Given the specific pesticidal uses of chlordecone, it can be expected that all amounts manufactured are ultimately released to the environment.

Chlordecone is not expected to hydrolyse or biodegrade in aquatic environments, nor in soil. Direct photodegradation is not significant. Therefore, chlordecone is considered to be highly persistent in the environment. With BCF-values in algae up to 6,000, in invertebrates up to 21,600 and in fish up to 60,200 and documented examples of biomagnification, chlordecone is considered to have a high potential for bioaccumulation and biomagnification.

The available data are not conclusive when it comes to long-range atmospheric transport of chlordecone in gaseous form. However, atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents as well as biotic transport could also contribute to long-range environmental transport of chlordecone. Due to lack of monitoring data on chlordecone, the assessment of the potential for long-range transport of chlordecone was based on physico-chemical properties and application of long range transport models.

Chlordecone is readily absorbed into the body and accumulates following prolonged exposure. The pesticide is both acutely and chronically toxic, producing neurotoxicity, immunotoxicity, reproductive, musculoskeletal and liver toxicity at doses between 1 - 10 mg/kg bw/day in experimental animal studies. Liver cancer was induced in rats at a dose of 1 mg/kg body weight per day, and reproductive effects are seen at similar dose levels. The International Agency for Research on Cancer has classified chlordecone as a possible human carcinogen (IARC group 2B). Moreover, chlordecone is very toxic to aquatic organisms, with the most sensitive group being the invertebrates.

Based on the available evidence, chlordecone is likely as a result of its long-range environmental transport to lead to significant adverse human health and environmental effects such that global action is warranted.

1 Introduction

The European Community and its member states being parties to the Stockholm Convention have proposed chlordecone to be listed in Annex A to the Convention (UNEP/POPS/POPRC.1/6).

This risk profile has been prepared following the decision of the Persistent Organic Pollutants Review Committee at its first meeting in November 2005 to establish an ad hoc working group to review the proposal further (UNEP/POPS/POPRC.1/10).

In this document all data are presented according to the International System of Units (SI) and, therefore, many have been recalculated from other units in the data sources. Furthermore, all concentrations are presented based on kg or L (*e. g.* µg/kg or mL/L).

1.1 Chemical Identity of the proposed substance

Chlordecone is a synthetic chlorinated organic compound, which has mainly been used as an agricultural insecticide, miticide and fungicide.

1.1.1 Names and registry numbers

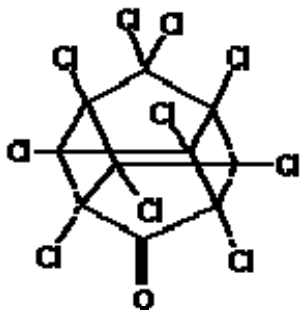
CAS chemical name:
1,1a,3,3a,4,5,5,5a,5b,6-decachloro-octahydro-1,3,4-metheno-2H-cyclobuta-[cd]-pentalen-2-one

Synonyms:
Decachloropentacyclo-[5,2,1,$0^{2,6}$,$0^{3,9}$,$O^{5,8}$]-decan-4-one,
Decachlorooctahydro-1,3,4-metheno-2H,5H-cyclobuta-[cd]-pentalen-2-one
Decachloroketone

Trade names:
GC 1189, Kepone, Merex, ENT 16391, Curlone

CAS registry number:
143-50-0

1.1.2 Structure



Source: **http://webbook.nist.gov**, as quoted in http:// ecb.jrc.it

Chlordecone is chemically closely related to mirex, a pesticide which is already listed under the Stockholm Convention. The chemical structure of chlordecone differs from mirex in that the oxygen of the keto group in chlordecone is replaced by two chlorine atoms in mirex.

1.1.3 Physical and chemical properties

The physical and chemical properties of chlordecone are listed in Table 1.1. It demonstrates that the variation is high between data sources for physical properties like vapour pressure and water solubility. This is confirmed by the fact that the Henry's Law Constant varies by one order of magnitude, depending on the type of data used for the calculation. The source of used data are generally considered to be reliable; the data quality have been assessed in the (inter)national consensus documents (IARC, IPCS HSG, IPCS EHC and US ATSDR) and the quality of the data published by Hansch *et. al.* and Howard has been evaluated (Pedersen *et. al.*, 1995).

Table 1.1 Physical and chemical properties of Chlordecone.

Property	Unit	Value	Reference
Molecular formula		$C_{10}Cl_{10}O$	
Molecular weight	g/mole	490.6	
Appearance at normal temperature and pressure		Tan-white crystalline solid	IARC, 1979[1]
Vapour Pressure	Pa	$3.0x10^{-5}$ (25 °C) $< 4.0x10^{-5}$ (25 °C) $4.0x10^{-5}$ (25 °C)	Kilzer, l *et. al.*, 1979[2] IARC, 1979[1] HSG 41, IPCS, 1990
Water solubility	mg/L	0.35-1.0x 1-2 2.7 (25 °C) 3.0	HSG 41, IPCS, 1990 EHC 43, IPCS, 1990 Kilzer, l *et. al.*, 1979[2] Kenaga, 1980
Melting point	°C	350; (decomposes)	IARC, 1979[1]
Boiling point	°C	No data	
Log K_{OW}		4.50 5.41	Howard, 1991[1] Hansch *et. al.*, 1995[2]
Log K_{aw}		-6.69	Scheringer *et. al* ., 2006
Log K_{oc}		3.38-3.415	Howard, 1991[1]
Henry's Law Constant	Pa m^3/mol	$5.45x10^{-3}$, (25 °C) $2.53x10^{-3}$ (20 °C) $4.9x10^{-3}$ $2.0x10^{-2}$	Calculated[2] Howard, 1991[1] Calculated[3] Calculated[4]
Atmospheric OH Rate Constant	cm^3/molecule-sec	≈ 0 (25 °C)[j]	Meylan & Howard, 1993[2]

* It is likely that the 0.35 number is an outlier. The source (HSG 41 by IPCS) did not provide the reference so it is impossible to track where this number came from. The more robust EHC 43 by IPCS did provide a reference and used 1-2 mg/l. This is in the same range with the other values in peer reviewed articles. ATSDR quotes a value of 3 mg/l from Kenaga.

1: Quoted from US ATSDR, 1995
2: Quoted from http://esc.syrres.com/interkow/webprop.exe
3: Calculated from maximum water solubility and minimum vapour pressure of this table
4: Calculated from minimum reliable water solubility (1 mg/L) and maximum vapour pressure of this table

1.2 Conclusion of the Persistent Organic Pollutants Review Committee on the Annex D information on Chlordecone

The POP Review Committee applied in its first meeting on 7–11 November 2005[1] the screening criteria specified in Annex D to the Stockholm Convention, and decided, in accordance with paragraph 4 (a) of Article 8 of the Convention, that it was satisfied that the screening criteria have been fulfilled for Chlordecone. It decided furthermore, in accordance with paragraph 6 of Article 8 of the Convention and paragraph 29 of decision SC-1/7 of the Conference of the Parties to the Stockholm Convention, to establish an ad hoc working group to review the proposal further and to prepare a draft risk profile in accordance with Annex E to the Convention. It invited, in accordance with paragraph 4 (a) of Article 8 of the Convention, Parties and Observers to submit to the Secretariat the information specified in Annex E of the Convention before 27 January 2006.

[1] See the meeting report at: www.pops.int/documents/meetings/poprc/

1.3 Data sources

This Risk Profile is mainly based on information from the following review reports:

- Environmental Health Criteria (EHC) 43: Chlordecone. IPCS International Programme on Chemical Safety. United Nations Environment Programme. International Labour Organisation. World Health Organization. Geneva 1990 (available at: http://www.inchem.org/documents/ehc/ehc/ehc43.htm)
- Health and Safety Guide No. 41, 1990. IPCS International Programme on Chemical Safety. United Nations Environment Programme. International Labour Organisation. World Health Organization. Geneva 1990 (available at: http://www.inchem.org/documents/hsg/hsg/hsg041.htm)
- Toxicological profile for Mirex and Chlordecone. U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry (ATSDR) August 1995 (available at: http://www.atsdr.cdc.gov/toxprofiles/tp66-p.pdf).

The above extensive review reports were used as the main source of information on this candidate POP chemical. Prior to the drafting of this risk profile, a detailed literature search was undertaken on Chlordecone which did not uncover any further assessment reports on this chemical, either international or at the level of individual countries. Where the reviews above have been cited, the text quoted (or quoted with modifications) includes the references cited in the original review. These references are not shown individually in the reference list.

Following the request of the POP Review Committee for additional information, as specified in Annex E of the Convention, on Chlordecone, information was provided, which was mainly based on the open literature. However, France provided a report prepared for the Assemblée Nationale describing the history of production and use of Chlordecone in Martinique and Guadeloupe (Beaugendre, 2005).

A search for more recent information included a literature search *via* the Danish Technical University Library and the data base FINDit (search terms: Chlordecone, kepone, merex) as well as a data base search in public data bases. The data bases include "Ecotox" (US-EPA, http://www.epa.gov/ecotox/), "NITE" (Japan, National Institute of Technology and Evaluation http://www.safe.nite.go.jp/english/db.html) BUA Reports (http://www.gdch.de/taetigkeiten/bua/berichte.htm) and Environmental Fate Data Base (http://www.syrres.com/esc/efdb.htm). This search was based on the search terms: Chlordecone, Kepone and the CAS number 143-50-0. In addition, the Arctic Monitoring and Assessment Programme[2] and the UNEP Regionally based assessment of Persistent Toxic Substances Global Report[3] were consulted. Most of these gave no further information regarding Chlordecone.

1.4 Status of the chemical under international conventions

Chlordecone is listed in Annex A of the Protocol to the Convention on Long-Range Transboundary Air Pollution (CLRTAP) on Persistent Organic Pollutants. The provisions of the Protocol oblige Parties (currently 25) to phase out all production and uses of Chlordecone. Chlordecone is included in the OSPAR convention as a substance of possible concern[4].

The proposal to include Chlordecone in the UNEP/FAO Rotterdam Convention was reviewed by the Chemical Review Committee (CRC) at its first meeting in February 2005. The CRC agreed that, on the basis of the information currently available, the notifications from Switzerland and Thailand had met all the criteria of Annex II with the exception of criterion (b) (iii)[5]. Accordingly, the CRC concluded that Chlordecone could not be recommended for inclusion in Annex III of the Rotterdam Convention at the current time.

[2] http://www.amap.no/

[3] http://www.chem.unep.ch/pts/gr/Global_Report.pdf

[4] The chemically related compound mirex is already included in the Stockholm convention. Both mirex and Chlordecone are included in the UNECE 1998 Aarhus Protocol on Persistent Organic Pollutants (POPs). Both are included in OSPAR as substances of possible concern.

[5] This requires that the documentation supplied demonstrates that the final regulatory action is based on a risk evaluation involving prevailing conditions within the Party taking the action.

2 Summary information relevant for the risk profile

2.1 Sources

2.1.1 Production

Chlordecone has been produced by reacting hexachlorocyclopentadiene and sulfur trioxide under heat and pressure in the presence of antimony pentachloride as a catalyst. The reaction product is hydrolyzed with aqueous alkali and neutralized with acid; Chlordecone is recovered *via* centrifugation or filtration and hot air drying (Epstein 1978) (Quoted from US ATSDR, 1995).

Chlordecone was first produced in 1951, patented in 1952, and introduced commercially in the United States by Allied Chemical in 1958 under the trade names Kepone® and GC-1189 (Epstein 1978; Huff and Gerstner 1978). The technical grade of chlordecone, which typically contained 94.5% chlordecone, was available in the United States until 1976 (IARC 1979). Chlordecone was also found to be present in technical grade mirex at concentrations up to 2.58 mg/kg and in mirex bait formulations at concentrations up to 0.25 mg/kg (EPA 1978b; IARC 1979a) (Quoted from US ATSDR, 1995).

2.1.2 Trade and stockpiles

Between 1951 and 1975, approximately 3.6 million pounds (1.6 million kg) of chlordecone were produced in the United States (Epstein 1978). (Quoted from US ATSDR, 1995) Chlordecone production was discontinued in the USA in 1976. However, a year later it was reported that a French company was considering the establishment of production facilities in France (Anonymous, 1978b), but no further information on this proposal is available. (Modified from EHC 43, (IPCS, 1984)).

No current data are available regarding import volumes of chlordecone. By 1976, technical chlordecone was not exported from the United States and the compound was no longer produced there. Diluted technical grade chlordecone (80% active ingredient) was exported to Europe, particularly Germany, in great quantities from 1951 to 1975 by the Allied Chemical Company (Epstein 1978) where the diluted technical product was converted to an adduct, Kelevan. Kelevan is a derivative of chlordecone and used for the same purposes. In the environment, it oxidizes to Chlordecone and could therefore also be considered with Chlordecone for listing in the Stockholm Convention. Approximately 90-99% of the total volume of Chlordecone produced during this time was exported to Europe, Asia, Latin America, and Africa. (DHHS 1985; EPA 1978b) (Modified from US ATSDR, 1995) There is no information, indicating that Kelevan is being produced or used at present.

Chlordecone was marketed in France as a formulation, Curlone, by De Laguarique from 1981 to 1993. The formulation was used in Martinique and Guadeloupe following hurricane Allen in 1979 and David in 1980 which led to considerable pest infestations. Chlordecone for this formulation was synthesised in Brazil. The authorisation for Curlone was withdrawn by the French Ministry of Agriculture in 1990. Use was continued until September, 1993. (Beaugendre, 2005) In Canada, no product containing Chlordecone has been registered as a pest control product since 2000.

2.1.3 Uses

Chlordecone has been used extensively in the tropics for the control of banana root borer (Anonymous, 1978a; Langford, 1978). This was its only registered food use. It is regarded as an effective insecticide against leaf-cutting insects, but less effective against sucking insects (Information Canada, 1973). Historically, Chlordecone has been used in various parts of the world for the control of a wide range of pests. It can be used as a fly larvicide, as a fungicide against apple scab and powdery mildew (Information Canada, 1973), and to control the Colorado potato beetle (Motl, 1977), rust mite on non-bearing citrus, and potato and tobacco wireworm on gladioli and other plants (Suta, 1978). Chlordecone has also been used in household products such as ant and roach traps at concentrations of approximately 0.125% (IARC 1979a). The concentration used in ant and roach bait was approximately 25%. (Epstein 1978) (Modified from EHC 43 (IPCS, 1984) and US ATSDR, 1995).

2.1.4 Releases to the environment

Given the specific pesticidal uses of Chlordecone, it can be expected that all amounts manufactured are ultimately released to the environment. The use of Chlordecone as a pesticide in Martinique and Guadeloupe until 1993 resulted in severe contamination of soil and surface water, which are being monitored at present. (Bocquene & Franco, 2005, Beaugendre, 2005).

Major releases of Chlordecone occurred to the air, surface waters, and soil surrounding a major American manufacturing site in Hopewell, Virginia. Releases from this plant ultimately contaminated the water, sediment, and biota of the James River, a tributary to the Chesapeake Bay (Quoted from US ATSDR, 1995).

2.2 Environmental fate

The partitioning of Chlordecone in the environment will be governed by its high log K_{ow} (5.41 or 4.50) and relatively low water solubility (1-3.0 mg/L) resulting in sorption to particulate matter (dust, soil and sediment) and organic material (living organisms).

The combination of these properties and the vapour pressure ($3.0\text{-}4.0 \times 10^{-5}$ Pa) of Chlordecone, results in a relatively low potential for volatilisation as the Henry's Law Constant is between 2.0×10^{-2} and 5.45×10^{-3} Pa m^3/mole (25 °C), depending on the type of data used for the calculation (Table 1.1.).

In the EHC 43 (IPCS, 1984), the volatilisation of Chlordecone is evaluated based on laboratory and field observations that indicate that Chlordecone does not volatilise to any significant extent (Dawson, 1978). However, the release of copious quantities of Chlordecone dust from production facilities has represented a major source of environmental and human contamination. Airborne Chlordecone has been known to spread 60 miles from a point source (Feldmann, 1976), and the potential exists for further dispersion of fine particles (Lewis & Lee, 1976 (Abbreviated from EHC 43 (IPCS, 1984).)

The US ATSDR (1995,), concluded that Chlordecone released to the environment partitions to soil and sediment. Small amounts may remain dissolved in water and Chlordecone released to the atmosphere is eventually deposited on soil or surface waters.

2.2.1 Persistence

In the EHC 43 (IPCS, 1984), early reports that did not include any evidence of Chlordecone degradation in the natural environment (Dawson, 1978; Geer, 1978) were quoted as well as a more recent study, in which microbial action had been shown to transform Chlordecone into monohydro- and possibly dihydrochlordecone (Orndorff & Colwell, 1980a).

EHC 43 (IPCS, 1984), concluded that Chlordecone is an extremely stable compound and is not expected to degrade in the environment to any significant extent. However, there have been reports of trace amounts of monohydrochlordecone being found (Carver *et. al.*, 1978, Orndorff & Colwell, 1980b), but the mechanism of its formation is not clear. Solar irradiation of Chlordecone in the presence of ethylenediamine results in 78% degradation after 10 days (Dawson, 1978) quoted from EHC 43 (IPCS, 1984). However, ethylenediamine is not usually present in the atmosphere, so at the time, there was no information available regarding the photolytic stability of Chlordecone under environmental conditions.

The more recent review (US ATSDR, 1995), concludes that Chlordecone is not expected to be subject to direct photodegradation in the atmosphere. Furthermore, it is concluded that Chlordecone is resistant to aerobic degradation, although some anaerobic biodegradation does occur and that Chlordecone is very persistent in the environment. Chlordecone will strongly bind to organic matter in water, sediment, and soil. When bound to organic-rich soil, Chlordecone is highly immobile; however, when adsorbed to particulate matter in surface water, Chlordecone can be transported great distances before partitioning out to sediment. The primary process for the degradation of Chlordecone in soil or sediments is anaerobic biodegradation (Abbreviated from US ATSDR, 1995) .

Information regarding the persistence of Chlordecone dating after 1995 is scarce, but the use of Chlordecone until 1993 in the Caribbean island of Martinique has resulted in severe contamination and monitoring studies have been initiated. Bocquene & Franco (2005) reported concentrations in samples from 2002 in water (particulate matter) and sediment in rivers of up to 57 µg/kg and 44 µg/kg, respectively. They quoted other investigations for reporting concentrations in river water, sampled in 2000-2001 in the range 1.20 - 2.13 µg/L.

Even though Chlordecone was prohibited from main land France, an exemption was granted that allowed the use of it in the French West Indies until September, 1993. A recent study showed that it is still detected in different ecosystems of Martinique (Coat, S. *et. al.*, 2006). Stocks of Chlordecone may have been used in Martinique after 1993, but it is expected that the use ceased several years ago. However, residues are still measurable in both river water and sediment, where the prevailing anaerobic conditions in the latter allow for the only known biotic degradation of Chlordecone. This is all the more remarkable as the climate in this area is optimal not only for crops and pests but also for biodegradation.

Conclusion

Chlordecone is not expected to hydrolyse or biodegrade in aerobic aquatic environments or in soil; however, there is some evidence of degradation under anaerobic condition. Direct photodegradation is not significant. Based on all available data Chlordecone is considered to be highly persistent in the environment.

2.2.2 Bioaccumulation

Because of the lipophilic nature of this compound (high octanol-water partition coefficient (log K_{ow} 4.50-5.41), Chlordecone has a potential for both bioaccumulation and, with little or no metabolic depuration, also biomagnification in aquatic food chains.

Table 2.1 summarises bioconcentration factors (BCF) selected from the US EPA database Ecotox (US EPA, 2006). The results included are based on measured concentrations and, for organisms different from algae, derived from tests based on flow through exposure. Thereby, the results should reflect the bioconcentration obtained under well defined, constant exposure concentrations. For fish, the results of a series of tests of four days duration were not included, because it is not considered to be likely that equilibrium had been reached[6]. Two additional studies from EHC 43 (IPCS, 1984) are also included.

Table 2.1 BCF values for Chlordecone.

Species	Test Duration	Exposure Concentration µg/L	BCF	Reference[1]
Green algae (*Chlorococcum sp., Dunaliella tertiolecta*)	24 h	100	230-800	Walsh *et. al.*, 1977
Green alga (*Chlorococcum sp.*)	48 h	40	6,000	Bahner *et. al.*, 1977
Diatoms (*Thalassiosira guillardii, Nitzschia sp.*)	24 h	100	410-520	Walsh *et. al.*, 1977
Crustacean (*Callinectes sapidus*)	96 h	110-210	6.2-10.4	Schimmel, 1977
Crustacean (*Palaemonetes pugio*)	96 h	12-121	425-933	Schimmel, 1977
Crustacean (*Palaemonetes pugio, Americamysis bahia*)	21-28 d	0.023-0.4	5,127-13,473	Bahner *et. al.*, 1977
Crustacean (*Palaemonetes pugio*)	16 d	0.041	12,094	Fisher & Clark, 1990
Oyster (*Crassostrea virginica*)	19-21 d	0.03-0.39	9,278-9,354	Bahner *et. al.*, 1977
Midge (*Chironomus tentans*)	14 d	11.8-169.2	21,600	Adams *et. al.*, 1985
Fish (*Brevoortia tyrannus*)	1-18 d	0.14-1.55	2,300-9,750	Roberts & Fisher, 1985
Fish (*Menidia menidia*)	1-28 d	0.08-0.8	21,700-60,200	Roberts & Fisher, 1985
Fish (*Cyprinodon variegatus*)	28 d	< 0.02-1.9	3,100-7,115	Bahner *et. al.*, 1977; Hansen *et. al.*, 1977
Fish (*Leiostomus xanthurus*)	30 d	0.029-0.4	2,340-3,217	Bahner *et. al.*, 1977
Fish (*Pimephales promelas*)	56 d	0.004	16,600	Huckins *et. al.*, 1982[2]
Fish (*Cyprinodon variegatus*)	Life cycle	0.041	1,800-3,900	Goodman *et. al.*, 1982[2]

1: All quoted from the Ecotox database (US EPA, 2006), except for two[2] quoted from EHC 43 (IPCS, 1984)

The information on bioaccumulation from food is limited, but the EHC 43 (IPCS, 1984) report includes two relevant studies; one on food exposure and the other on an estuarine food chain. When chlordecone was fed to juvenile spot for 28 days, the body burden of chlordecone increased additively and equilibrium was not attained (Stehlik & Merriner, 1983). The estuarine food chain study (Bahner *et al.*, 1977) was composed of green algae, oysters, mysids, grass shrimps, sheepshead minnows and spot. The transfer from algae to oysters was very low; but a clear transfer from shrimp to mysids

[6] In OECD Test Guideline 305, the prescribed duration of the exposure phase is 28 days.

and from mysids to spot, indicated that much of the chlordecone was being transferred through the trophic levels. Clearance was slow in shrimp and fish, with tissue levels of chlordecone decreasing by 30-50% in 24-28 days.

US ATSDR (1995), described the bioaccumulation of chlordecone together with that of mirex, stating that they are both highly lipophilic and therefore, have a high bioconcentration potential. They bioaccumulate in aquatic food chains with virtually no degradation of the compounds by exposed organisms (de la Cruz and Naqui, 1973; Epstein, 1978; Huckins *et al.*, 1982; Huggett and Bender, 1980; Kenaga, 1980; Lunsford *et al.*, 1987; Naqvi and de la Cruz, 1973; Nichols, 1990; Oliver and Niimi, 1985 and 1988; Roberts and Fisher, 1985)[7].

Only limited information is available on uptake and bioaccumulation of chlordecone in terrestrial food chains (Naqvi and de la Cruz, 1973), and little uptake of chlordecone by plants was observed (Topp *et. al*., 1986).

Conclusion

With BCF-values of up to 6,000 in algae, of up to 21,600 in invertebrates and of up to 60,200 in fish, and with documented examples of biomagnification, chlordecone is considered to have a high potential for bioaccumulation and biomagnification.

2.2.3 Potential for Long-Range Environmental Transport

The potential for long-range environmental transport can be documented through monitoring data from remote regions (*e.g.* the Arctic) and/or through physical-chemical characteristics of the molecule, which are promoting such transport. The most well known mechanism of long-range transport is atmospheric transport of substances in the vapour phase. However, atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents as well as biotic transport could also contribute (*e. g.* AMAP 2004).

One prerequisite for long-range atmospheric transport is persistence to degradation, and Chlordecone is considered to be highly persistent in the environment (see Section 2.2.1). Chlordecone does not volatilise to any significant extent (see section 2.2). The partitioning of Chlordecone in the environment will be governed by its high log K_{ow} (5.41 or 4.50) and relatively low water solubility (1-3.0 mg/L) resulting in sorption to particulate matter (dust, soil and sediment) and organic materials and living organisms. Therefore, the long range transport is expected to take place through these pathways.

The US ATSDR (1995), states that atmospheric transport of dust containing Chlordecone particles was reported during production years based on results from high volume air sample filters from Hopewell: At approximately 200 yards from the Chlordecon production plant, the contents ranged from 3.0-55 micrograms/m^3, depending on weather conditions and date of collection. At more distant sites in May 1975, levels ranged from 1.4-21 ng/m^3. Specifically, in South Richmond, 15.6 miles north west from Hopewell, the level was 1.41 ng/m^3. At Byrd airport, 14.12 miles north of Hopewell, the level was 1.93 ng/m^3. In Petersburg, 8.19 miles south west from Hopewell, the level was 20.7 ng/m^3. (Epstein, 1978). They conclude further, that airborne Chlordecone has been known to spread 60 miles from a point source (Feldmann, 1976), and that the potential exists for further dispersion of fine particles (Lewis & Lee, 1976) (US ATSDR, 1995).

Transport in aquatic environments is illustrated by results of measurements in clams and oysters from the James River at sampling locations from 8-64 miles from Hopewell, Virginia that contained 0.2-0.8 mg/kg of Chlordecone (Epstein, 1978).

However, no records are available regarding concentrations of Chlordecone in areas at long distances from sites of production or use. Therefore, the assessment of the potential for long-range transport of Chlordecone must be based on physical properties. For this - apart from persistence - the vapour pressure and the Henry's Law Constant are considered to be the most relevant properties. For a comprehensive evaluation of the potential for long-range atmospheric transport, knowledge of the vapour pressure at high as well as at low temperatures (*e. g.* 25 °C and 0 °C) is required. This information is, however, available for only a few substances (AMAP, 2004), so the vapour pressure at 25 °C is used as a measure of the volatility of the substance.

As a rule of thumb, substances with vapour pressures $>1.33x10^{-2}$ Pa will be entirely in the vapour phase and substances with vapour pressures $<1.0x10^{-4}$ Pa will be particulate (US ATSDR, 2004).

A way of evaluating the characteristics and effects of a substance for which not enough information exists is to compare it with better known substances of similar characteristics. This approach (known as "the benchmark approach") was proposed by Scheringer (1997) and Beyer *et. al.*, (2000), has been recently used in some recent studies concerning persistence and environmental transport of pollutants (see, *i. e.* Vulykh *et. al.*, 2006, and Klasmeier *et. al.*, 2006). As a measure of values of properties that would qualify for long-range atmospheric transport, the currently listed POPs are used. However,

[7] These references describe both Mirex and Chlordecone.

information regarding physical-chemical properties for chemicals often varies widely between sources and the quality of data cannot be compared without specific review of the individual studies. This is demonstrated by the available data on the physical-chemical properties of Chlordecone presented in Table 1.1. The two values for the vapour pressure are rather uniform (0.3 and $0.4x10^5$ Pa) but the water solubility found in literature varies by an order of magnitude (0.35–3.0 and the lowest value is considered to be unreliable.[8]

The comparison of Chlordecone with already listed POPs is presented in Table 2.2. As a starting point for this comparison, the highest and lowest values for Chlordecone (Table 1.1) were used. For already listed POPs, information was sought on the UNEP-POPs homepage. Among the currently listed POPs, most of the relevant properties were available for aldrin, chlordane, dieldrin, DDT, hexachlorobenzene, mirex, toxaphene, endrin and heptachlor. Missing information (water solubility of mirex) was sought in US ATSDR (1995) and AMAP (2004). The US ATSDR (1995), quotes values of 0.2 and 0.6 mg/L, while the AMAP (2004), quotes Mackay for very low water solubility: $6.5x10^{-5}$ mg/L. In order to avoid introduction of what seems to be an outlier in the comparison, the value for water solubility of mirex from US ATSDR (1995) was used.

The water solubility and vapour pressure as well as Henry's Law Constants calculated from these values of the currently listed POPs are summarised in Table 2.2 together with information on Chlordecone from Table 1.1.

Table 2.2 Water solubility (WS), vapour pressure (VP) and (calculated) Henry's Law Constant (HLC) (at 25°C) for Chlordecone and currently listed POPs.

Substance	WS mg/L	VP Pa	HLC Pa m^3/mol
Chlordecone-min	1.0	0.00003	0.0049[1]
Chlordecone-max	3.0	0.00004	0.02[2]
POP-min	0.0012 (DDT)	0.000025 (DDT)	0.04 (endrin)
POP-max	3.0 (toxaphene)	27 (toxaphene)	3726 (toxaphene)
POP-2^{nd} max	0.5 (dieldrin)	0.04 (heptachlor)	267 (heptachlor)

1: Calculated from maximum water solubility and minimum vapour pressure
2: Calculated from minimum reliable water solubility and maximum vapour pressure

Table 2.2 shows that the water solubility of Chlordecone is at the level of the most water soluble among the currently listed POPs (toxaphene and dieldrin), while the vapour pressure is comparable to that of DDT. The highest of the two Henry's Law Constants that were calculated for Chlordecone is of the same order of magnitude as that of endrin. It should be noted that in presenting the data in table 2.2 it is not inferred that a chemical (in this case Chlordecone) is considered to meet the long range environmental transport criterion just because it fits within the range of values of currently listed POPs.

Further to this, it should be mentioned that the latest AMAP report on POPs (AMAP, 2004) describes the possibilities of particle borne transport for substances, which have Henry's Law Constants (HLC) close to that of Chlordecone (HLC = 0.0049 or 0.056). Based on HLC-values from AMAP (2004), it is concluded that semi-volatile compounds such as lindane (γ-HCH) (HLC = 0.000149) and chlordane (HLC = 0.342) are distributed between airborne particles and the gaseous phase, depending on the temperature. These can be washed out *via* precipitation and temporarily deposited in seawater or soil and can absorb to water, plant and soil surfaces from the gaseous phase. During favourable warm weather conditions, these compounds evaporate again into the atmosphere and undergo further atmospheric transport. This remobilization is also called the 'grasshopper effect'. The role of stormy weather situations in remobilization of semivolatile compounds into the atmosphere is obvious but still scarcely investigated (AMAP, 2004).

Besides, certain physical-chemical properties of Chlordecone, such as the partition coefficients log K_{ow} (octanol-water partition coefficient) and log Kaw (air-water partition coefficient), are similar to those of some toxaphene components which, added to its persistence in air and water, would mean that coupled long range transport in atmosphere and oceans may take place (*i. e.* the substance is exchanging between atmospheric gas phase and oceanic dissolved phase and can be

[8] Availability of high quality data regarding physical-chemical properties could support more firm conclusions.

transported in either phase). (Wania, F. 2006, personal communication). Chlordecone has a very low Henry's law constant and a high mass fraction is found in water, and therefore it can be inferred that transport with ocean currents contributes to the long-range transport of Chlordecone.

In a recent modeling study, Scheringer *et. al*.,(2006), investigated the persistence and long range transport potential of these potential POPs, including chlordecone and hexabromobiphenyl, using an OECD screening tool which based the evaluation of overall environmental persistence and transport potential on the results of several of the currently available multimedia environmental fate models (see also Klasmeier *et. al*., 2006, and Fenner *et. al*., 2005 for a more detailed explanation). They concluded that the four POP candidates have persistence and long range transport potential properties similar to those of several known POPs in this evaluation. Furthermore, they included the uncertainty regarding the data quality in an uncertainty analysis, which indicated that the result is valid although there are considerable uncertainties in the chemical properties of the four POP candidates. It should be noted that environmental fate modeling results strongly depend on the assumptions made, specifically when essential data such as environmental half-lives are not known. In addition, results for substances like Chlordecone, which are strongly bound to particles and are of very low volatility, are highly dependent on the medium to which they are emitted, i.e., to air, to water, or to soil. The emission to air scenario always yields the highest transfer efficiency, and that value is displayed in the Scheringer *et. al.*, (2006) plots. Transfer efficiency will likely differ by several orders of magnitude when evaluated under soil and water emission scenarios.

Conclusion

In summary, the above discussion shows that the available data on Chlordecone are not conclusive when it comes to long-range atmospheric transport in gaseous form. However, atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents, as well as biotic transport, could also contribute to long-range environmental transport of Chlordecone Coupled atmosphere-ocean transport also seems quite possible.

Due to a lack of monitoring data on Chlordecone the assessment of the potential for long-range transport of Chlordecone must be based on physico-chemical properties and modelling data. The modelling study of Scheringer *et. al.*, 2006, shows clearly that long range environmental transport is possible (and possibly more than actually estimated), even considering the uncertainties surrounding the physico-chemical properties.

In accordance with paragraph 7 (a) of Article 8 of the Convention, and taking into account that a lack of full scientific certainty should not prevent a proposal from proceeding, Chlordecone is likely, as a result of its long-range environmental transport, to lead to significant adverse human health and environmental effects such that global action is warranted.

2.3 Exposure

2.3.1 Environmental concentrations

The available information regarding environmental concentrations of Chlordecone is very limited and includes only areas in the vicinity of production (US) or use (Martinique).

The US ATSDR (1995), illustrates the presence of Chlordecone in the environment following production of the substance. In 1977, 12 years after production of Chlordecone began and 2 years after the production ceased, average concentrations of Chlordecone in estuarine water (dissolved) were <l0 ng/L (ppt) (Nichols 1990). In October 1981, 6 years after production ceased, Chlordecone water concentrations ranged from not detectable to 0.02 µg/L (ppb) (Lunsford *et. al.*, 1987). Groundwater monitoring data are lacking, but because Chlordecone binds tightly to organic matter in soil, leaching into groundwater is not expected to occur extensively (Abbreviated from US ATSDR, 1995).

Recent monitoring data from the United States demonstrate the persistence of Chlordecone, known as Kepone in the United States. The substance is included in the U.S. EPA National Lake Fish Tissue Study to estimate the national distribution of selected residues in fish tissue from lakes and reservoirs in the lower 48 states. There were a total of 881 samples collected and analyzed between 2000 and 2005. For Chlordecone, there were 152 hits (17.25%), ranging from 12.3 and 2008 ppb. (Jensen, 2006).

In Martinique, the widespread use of Chlordecone until 1993 has resulted in contamination of soils and surface water in most of the island (Bocquené & Franco, 2005). These authors reported an investigation from 2002 of the presence of a series of pesticides in the water at the mouth of seven rivers. They measured Chlordecone in particulate matter or sediment of six of the seven rivers at concentrations up to 57 µg/kg in particulate matter, and up to 44 µg/kg in sediment.

Bocquené & Franco (2005), quoted other investigations in which concentrations of Chlordecone in the range 1.20 to 2.13 µg/L were measured in rivers of Martinique in 2002-2001. They also stated that Chlordecone was "ubiquitous" in river water used for drinking water.

Further to this, the report prepared for L'Assemblée Nationale (Beaugendre, June 2005), described the history of the use of Chlordecone in Guadeloupe and Martinique, and mentioned several monitoring programmes which are expected to result in reports at the end of 2005. However, these reports have not been available when drafting this document.

2.3.2 Human exposure

In the US ATSDR (1995), the experience from production of Chlordecone is summarised as follows: Chlordecone has not been detected in human adipose tissue or in blood samples from the general population, although historically it was detected in human milk samples collected in the south-eastern United States (EPA 1978c). Information is available regarding Chlordecone levels in blood of occupationally exposed workers and their families during 1974-1975 employed at the Hopewell, Virginia site. (Cannon *et. al.*, 1978; Epstein 1978; Knishkowy & Baker 1986; Taylor *et. al.*, 1978). (Quoted from US ATSDR, 1995) Further data on human exposure is quoted in section 2.4.1.

Information regarding human exposure resulting from direct use (application) of Chlordecone in the Caribbean Islands is not available. However, monitoring data in agricultural soils, crops, freshwater fish, littoral fish and shellfish indicates that human exposure more than 10 years after the use of chlordecone has ceased in Martinique and Guadeloupe, is still possible In soils having received Chlordecone, residues in crop are proportional to soil contamination and may exceed the recommended national residues limits (50 µg/kg to 200 µg/kg). This concerns mainly root vegetables such as radish (max. measured concentration: 0.055 µg/kg), sweet potatoes (max. measured concentration: 0.300 µg/kg), taro root (max. measured concentration: 0.230 µg/kg), but also aerial part of plants, such as sugar cane (max. measured concentration: 0.690 µg/kg), or pineapple (max. measured concentration: 0.160 µg/kg). In addition, workers are directly exposed to contaminated soils. Concentrations in fisheries products (freshwater and estuarine water) have also been found to exceed in some occasions national residues limits up by a factor of 100 (max. measured concentration: 20 mg/kg). National provisions have been taken in order to prohibit fisheries activities in contaminated area (Cabidoche *et. al*., 2006).

2.4 Hazard assessment for endpoints of concern

2.4.1 Toxicity

Toxicokinetics in experimental animals and in man

The US ATSDR (1995) and EHS 43 (IPCS, 1984) both record that Chlordecone is well absorbed following oral, dermal and inhalation exposure. Toxicokinetic data are mainly available from studies in experimental animals (*e. g.* Blanke *et. al*., 1978; Boylan *et. al.*, 1979; Cohn *et. al.*, 1978; Egle *et. al.*, 1978; Fujimori *et. al.* 1982a; Guzelian *et. al.*, 1981; Hall *et. al.* 1988; Hewitt *et. al.*, 1986b; Kavlock *et. al.*,1980; Plaa *et. al.*, 1987; Richter *et. al.*, 1979; Shah *et. al.*, 1987; Skalsky *et. al.*, 1980; as reported in IPCS, 1984). Following absorption, it is widely distributed in the body, with accumulation in the liver and to a lesser extent in fat, brain and kidneys, both in experimental animal studies and in humans (as reported in US ATSDR (1995) and EHS 43 (IPCS, 1984). Following administration of a single oral dose to rats at 40 mg/kg body weight, the highest concentrations were found in the adrenal glands and liver, followed by the fat and lung (Egle *et. al.*, 1978, quoted from IPCS, 1984). Chlordecone has been reported to be slowly metabolised *via* reductive biotransformation to Chlordecone alcohol in the rat (Blanke *et. al.*, 1978, as reported in EHS 43). Elimination from the body is slow, with a half-life of the order of several months and Chlordecone disappears more slowly from the liver than from other tissues (Egle *et. al.*, 1978, quoted from IPCS, 1984). Elimination is mainly *via* the faeces, a total of 66% of the dose in the Egle study being removed in the faeces and 2% in the urine in the 84 days following administration (Egle *et. al.*, 1978, quoted from IPCS, 1984).

EHC 43 reports that Chlordecone was detected in high concentrations in the liver (range 13.3-173 mg/kg), whole blood (range 0.6-32 mg/litre), and subcutaneous fat (range 2.2-62 mg/kg) of 32 male workers (Cohn *et. al.*, 1976, adapted from IPCS (1984). In occupationally-exposed workers, serum Chlordecone concentrations ranged from 120 to 2109 µg/litre, and dropped to 37 - 486 µg/litre 6-7 months after exposure had ceased (Adir *et. al.*, 1978, reported in IPCS (1984). The half-life of Chlordecone in these workers was estimated to be 63-148 days. Reductive biotransformation to Chlordecone alcohol has also been reported in humans (Blanke *et. al.*, 1978, as reported in EHS 43). Chlordecone was eliminated, primarily in the faeces, at a mean daily rate of 0.075% of the estimated total store in the body (Cohn *et. al.*, 1976, quoted from IPCS, 1984).

Toxicity of Chlordecone in animal studies

Chlordecone is of high acute toxicity in experimental animal studies, with an LD_{50} of approximately 100 mg/kg in the rat and ranging from 65 mg/kg in the rabbit to 250 mg/kg in the dog (taken from IPCS, 1984, Table 2). Acute toxicity effects include tremors indicative of a neurotoxic effect on the nervous and/or musculoskeletal systems, investigated by many authors as reported in US ATSDR (1995). The neurotoxic effects of Chlordecone have been reported in chickens (Naber & Ware, 1965), quail (McFarland & Lacy, 1969), fish (Couch *et. al.*, 1977), hamsters (Martinez *et. al.*, 1976), mice (End *et. al.*, 1979), rats (Epstein, 1978), and man (Martinez *et. al.*, 1978). Acute oral administration of Chlordecone is also associated with reproductive effects (Khera *et. al.*, 1976; Uzodinma *et. al.*, 1984a; Yarbrough *et. al.*, 1981) and hepatotoxicity in some studies (Fujimori *et. al.*, 1983; Mehendale 1977b, 1981b; Teo & Vore 1991) (quoted from US ATSDR (1995).

Repeated exposure to Chlordecone also causes reproductive, neurological, musculoskeletal and liver toxicity at doses as low as 10 mg/kg bw/day, although effects in other organs including kidney, thyroid, adrenals, and testes have also been reported (US ATSDR, 1995, IPCS, 1984). A Lowest-Observed-Adverse-Effect-Level (LOAEL) of 1.17 mg/kg bw/day was recorded in a 3 month feeding study in rats and signs of toxicity included focal necrosis in liver, enlargement of the adrenal gland, tremor, hyperactivity and exaggerated startle response (Cannon and Kimbrough, 1979, as quoted in US ATSDR, 1995). Histopathological changes in the liver, reduction in thyroid follicular size and colloid content and increase in epithelial cell height were reported in a 21-month gavage study in the rat, with a LOAEL of 0.07 mg/kg bw/day in males (Chu *et al*, 1981, as quoted in US ATSDR, 1995). Renal effects (proteinuria and increased severity of glomerulosclerosis) were seen in a 2-year feeding study in rats, with a NOAEL of 0.05 mg/kg/day (Larson *et. al.*, 1979b, as quoted in US ATSDR, 1995). Oral Chlordecone treatment caused decreased spleen and thymus weights, leukocyte counts, natural killer cell activity, and mitogenic responsiveness (EPA 1986c; Smialowicz *et. al.*, 1985; Swanson and Wooley, 1982); decreased natural killer cell activity (Smialowicz *et. al.*, 1985); and significant increase in plaque-forming cells (Chetty *et. al.*, 1993c) (as reported in ATSDR, 1995). The NOAEL was 5 mg/kg bw/day and the LOAEL was 10 mg/kg bw/day.

Hepatocarcinogenicity (hepatocellular carcinoma) of Chlordecone has been demonstrated in rats and mice (males and females) (NCI 1976, Reuber, 1978, 1979, as quoted in IPCS, 1984 and US ATSDR, 1995). Tumours have been observed at doses as low as 1 mg/kg bw/day in the rat and in mice at a dose of 2.6 mg/kg bw/day (NCI, 1976, as quoted in US ATSDR (1995). The International Agency for Research on Cancer (IARC) concluded in 1987 that there was sufficient evidence that Chlordecone is carcinogenic in mice and rats and possibly carcinogenic to humans (Group 2B). Chlordecone is not genotoxic in *in vitro* microbial and mammalian cell gene mutation assays, in a clastogenicity test and in the dominant lethal assay (Mortelmans *et. al.*, 1986; Probst *et. al.*, 1981; Schoeny *et. al.*, 1979, Tong *et. al.* 1981; Williams 1980, Khera *et. al.*, 1976; Simon *et. al.*, 1986, as reported in ATSDR (1995), although it has been reported to interfere with cell-to-cell communication (Tsushimoto *et. al.*, 1982, Caldwell and Loch-Caruso, 1992, as reported in US ATSDR (1995), suggests that it produces liver tumours by an epigenetic, tumour-promoting mechanism involving both hepatic toxicity and hypertrophy, including cytochrome P-450 induction.

Oral administration of Chlordecone to animals causes decreased fertility or fecundity and litter size, reduced sperm count and testicular atrophy (Khera *et. al.*, 1976; Linder *et. al.* 1983; Uzodinma *et. al.*, 1984a; Yarbrough *et. al.* 1981, as reported in US ATSDR (1995). A LOAEL of 0.83 mg/kg/day was recorded for sperm effects in a 90 day feeding study in rats, while effects on seminal vesicles and prostate were apparent at 1.67 mg/kg bw/day (Linder *et. al.*, 1983) (Quoted from US ATSDR (1995).

Chlordecone is also a developmental toxicant. As reported in US ATSDR (1995) and EHC 43 (IPCS, 1984), gestational exposure of rats and mice to low doses of Chlordecone resulted in increased stillbirths and decreased postnatal viability, reduced fetal or neonatal weight and/or skeletal ossification and a low incidence of malformations such as renal pelvis dilatation, undescended testes, enlarged cerebral ventricles, clubfoot, fused vertebrae or ribs, and encephalocele. Chlordecone administered at levels of 2, 6, and 10 mg/kg bw/day to rats and 2, 4, 8, and 12 mg/kg body weight per day to mice on days 7 - 16 of gestation caused 19% maternal mortality in rats at the highest dose and fetuses exhibited reduced weight, reduced degree of ossification, oedema, undescended testes, enlarged renal pelvis, and enlarged cerebral ventricles. (Chernoff & Rogers, 1976, as reported in IPCS, 1984). Lower dose levels induced reductions in fetal weight and degree of ossification. Male rats born to treated dams did not show any reproductive impairment. The reproductive performance of mice fed 0, 10, 30, or 37.5 mg Chlordecone/kg diet was impaired in terms of offspring and litter size (Huber, 1965, as reported in IPCS, 1984). No litters were produced by females fed 40 mg/kg, but litter production did resume within 7 weeks following withdrawal of the Chlordecone, although litters were still smaller than those of untreated controls (quoted from IPCS (1984)). Anovulation and persistent vaginal estrus were observed in female mice given Chlordecone at a dose level of 2 mg/kg bw/day) (Swartz *et. al.*, 1988, as quoted in US ATSDR, 1995), and similar changes were observed in female offspring of maternal rats given 15 mg/kg/day of Chlordecone on gestation days 14-20 (Gellert and Wilson, 1979, as

quoted in US ATSDR, 1995), although no effects on vaginal patency or fertility were observed in female offspring of maternal mice given 20 mg/kg/day during gestation days 8-12 or 14-18 (Gray and Kavlock 1984, as quoted in US ATSDR, 1995).

Toxicity of Chlordecone in humans

Available human data support the conclusion that Chlordecone has a similar toxicity profile in humans to that seen in experimental animal studies. As reported in US ATSDR (1995), a high incidence of nervous system toxicity was seen in a single group of workers exposed to Chlordecone during its manufacture (Cannon *et. al.*, 1978; Martinez *et. al.*, 1978; Sanbom *et. al.*, 1979; Taylor 1982, 1985; Taylor *et. al.*, 1978, taken from US ATSDR (1995)). Exposure of this population occurred by a combination of inhalation, oral, and dermal exposures, although the dermal route was suggested to be the predominant route. The toxicity was manifested as tremors, visual difficulties, muscle weakness, gait ataxia, in coordination, headache, and increased cerebrospinal fluid pressure (US ATSDR (1995). Prolonged exposure to high concentrations of Chlordecone in the workplace has been suggested to cause oligospermia and decreased sperm motility among male workers, although fertility was not impaired (Guzelian 1982a; Taylor 1982, 1985; Taylor *et. al.*, 1978, taken from US ATSDR (1995). A correlation between blood levels, atmospheric levels and sperm effects has however been difficult to prove conclusively (US ATSDR (1995). Epidemiological evidence for carcinogenicity of Chlordecone in exposed humans following inhalation exposure to Chlordecone is extremely limited (US ATSDR, 1995, IPCS, 1984). Liver biopsy samples taken from 12 workers with hepatomegaly resulting from intermediate- or chronic-duration exposures to high concentrations of Chlordecone showed no evidence of cancer (Guzelian *et. al.*, 1980, taken from US ATSDR (1995). However, conclusions from this study are limited by the very small number of workers sampled (US ATSDR, 1995)

Effects on endocrine systems

The effects of Chlordecone on reproduction indicate that this pesticide has effects on endocrine systems. It has been evaluated under the EU-Strategy for Endocrine Disrupters[9] and has been placed in category 1 (evidence of endocrine-disrupting activity in at least one species using intact animals), in the priority list of chemicals established under the EU-Strategy. This categorisation is based on evidence of ED activity in a number of experimental systems including the mouse uterotropic assay, increased uterine weight in rats given multiple injections of Chlordecone postnatally and receptor binding assays, indicative of an oestrogenic effect (as reported in BKH report, 2000, US ATSDR, 1995).

Conclusion on effects assessment and toxicity of Chlordecone

Chlordecone is readily absorbed into the body and accumulates following prolonged exposure. The pesticide is both acutely and chronically toxic, producing neurotoxicity, immunotoxicity, reproductive, musculoskeletal and liver toxicity at doses between 1 - 10 mg/kg bw/day in experimental animal studies. Liver cancer was induced in rats at a dose of 1 mg/kg body weight per day and in mice at a dose of 2.6 mg/kg bw/day, and reproductive effects are seen at similar dose levels. The International Agency for Research on Cancer has classified Chlordecone as a possible human carcinogen (IARC group 2B).

Table 2.3 summarises the outcomes of key toxicological studies on Chlordecone, including the NOAEL/LOAEL derived in each study. The studies included in this Table have been selected from the very large database on toxicological studies on Chlordecone, on the basis of the importance of the endpoint investigated (*e. g.* reproductive toxicity, carcinogenicity, other key target organ toxicity), robustness of the reported studies and the dose level (NOAEL/LOAEL) at which effects were reported. These studies were considered to be particularly relevant for characterisation of the toxicological risks of these compounds, and some of these studies have been used by US ATSDR to define Minimal Risk Levels (MRLs) for Chlordecone (US ATSDR, 1995).

Table 2.3 Summary of key toxicological studies on Chlordecone.

Species	Study type	Effect	LOAEL/NOAEL (mg/kg bw/day)	Reference
Rat Fischer 344	Short-term/acute toxicity 10 day repeat dose gavage study	65% loss in body weight, changes in clinical chemistry parameters	10 mg/kg bw/day (LOAEL) 5 mg/kg bw/day (NOAEL)	EPA, 1986 (as quoted in US ATSDR, 1995).
Rat Fischer 344	Short-term/acute toxicity 10 day repeat dose	Reductions in spleen and thymus weights, numbers of neutrophils, and natural killer cell activity, secondary to generalized toxicity	10 mg/kg bw/day (LOAEL) 5 mg/kg bw/day	EPA, 1986; Smialowicz *et. al.*, 1985, (as quoted in US ATSDR, 1995).

[9] http://europa.eu.int/comm/environment/endocrine/strategy/substances_en.htm

Species	Study type	Effect	LOAEL/NOAEL (mg/kg bw/day)	Reference
	gavage study		(NOAEL)	
Rat Fischer 344	Short-term/acute toxicity 10 day repeat dose gavage study	Increased startle response	2.5 mg/kg bw/day (LOAEL) 1.25 mg/kg bw/day (NOAEL)	EPA, 1986c (as quoted in US ATSDR, 1995).
Rat (Sherman)	3 month feeding study	Focal necrosis in liver, enlargement of the adrenal gland, hyperplasia and hypertrophy of cortical cells, tremor, hyperactivity, exaggerated startle response	1.17 mg/kg bw/day (LOAEL)	Cannon and Kimbrough 1979 (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat, Wistar	2 year feeding study	Renal effects (proteinuria and increased severity of glomerulosclerosis)	0.25 mg/kg bw/day. (LOAEL) 0.05 mg/kg bw/day (NOAEL)	Larson *et. al.*, 1979b (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat Sprague-Dawley	21 month gavage study	Histopathological changes in liver, reduction in follicular size and colloid content and increase in epithelial cell height in thyroid	0.07 mg/kg bw/day (LOAEL), in males	Chu *et. al.*, 1981(as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat, Wistar	3 month feeding study	Testicular atrophy	0.5 mg/kg bw/day. (LOAEL) 0.25 mg/kg bw/day (NOAEL)	Larson *et. al.* , 1979b (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat (Osborne-Mendel) and mouse (B3C6F1)	80 week feeding study	Hepatocellular adenoma and carcinoma	1.2 mg/kg bw/day. (LOAEL, rat) and 2.6 mg/kg bw/day (LOAEL, mouse)	NCI, 1976, Reuber, 1978, 1979(as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat	Multiple injections of Chlordecone to neonatal rats	Uterotrophic response - uterine weights increased in a dose-related manner	10 mg/kg bw/day (LOAEL, Gellert, 1978) $\leq$ 6 mg/kg bw/day (LOAEL, Hammond *et. al.*, 1979)	Gellert 1978[i] Hammond *et. al.*, 1979 (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat, Hotzman strain, ovarectomized immature females	Rats injected x 3 with 0 - 45 mg/kg bw/day Chlordecone $\pm$ 0.01, 0.1, 1 or 10 mg/kg bw/day estradiol benzoate	Uterotrophic response. Effect was additive to that of estradiol benzoate over the dose range studied	Dose of 20 mg/kg bw/day Chlordecone appeared to be threshold for embryo implantation functions	Johnson, 1996
Rat	90-day feeding study	Decrease in sperm motility and viability, decreased sperm, decrease in the weight of seminal vesicles and prostate	0.83 mg/kg bw/day LOAEL for sperm effects 1.67 mg/kg bw/day LOAEL for effects on seminal vesicles and prostate	Linder *et. al.*, 1983 (as quoted in IPCS, 1984 and US ATSDR, 1995).
Mouse, Balbc	130 day feeding study	8% decrease in litter size and 19% increase in pair-days to litter (constant oestrus)	1.3 mg/kg bw/day. (LOAEL)	Huber, 1965 (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rats and mice	2, 6, and 10 mg/kg bw/day by gavage to rats and 2, 4, 8, and 12 mg/kg bw/day to mice on days 7 - 16 of gestation.	Reduced foetal weight, reduced degree of ossification, oedema, undescended testes, enlarged renal pelvis, and enlarged cerebral ventricles. Reductions in fetal weight and degree of ossification at lower dose levels. Maternal mortality at top dose. In the mouse, fetotoxicity was observed only at the highest dose level and consisted of increased fetal mortality and clubfoot.	2 mg/kg bw/day. (LOAEL, rat)	Chernoff & Rogers, 1976). (as quoted in IPCS, 1984 and US ATSDR, 1995).
Balbc mice	160 day feeding study	Increased ovulation, persistent oestrus	2 mg/kg bw/day. (LOAEL)	Swartz *et. al.*, 1988 (as quoted in IPCS, 1984 and US ATSDR, 1995).
Rat	Reproductive toxicity	Increased ovulation, persistent oestrus in female offspring of maternal rats given Chlordecone on gestation days 14-20	15 mg/kg/day (LOAEL)	Gellert and Wilson, 1979, as quoted in US ATSDR, 1995)

Species	Study type	Effect	LOAEL/NOAEL (mg/kg bw/day)	Reference
Humans	Occupational exposure	Histories of tremors, unfounded nervousness or anxiety, and visual difficulties. Also skin rashes	Mean blood levels of Chlordecone in workers reporting adverse effects were 2.53 ppm Skin rashes reported in workers with blood Chlordecone levels in excess of 2 µg/L	Cannon *et. al.*, 1978 (as quoted in IPCS, 1984 and US ATSDR, 1995).

2.4.2 Ecotoxicity

A summary of results of aquatic ecotoxicity tests with Chlordecone from the Ecotox database (US EPA, 2006) is given in Table 2.4.

In addition to this, the EHC 43 (IPCS, 1984), summarised a series of experiments investigating the bioavailability of Chlordecone, noting that it is strongly adsorbed on sediment. Exposure of aquatic organisms is therefore partly *via* the water phase and partly *via* sediment. D'Asaro & Wilkes (1982) examined the effects of sediments previously exposed to Chlordecone at a known concentration, and of James River sediments contaminated with Chlordecone, on an estuarine community established in aquaria supplied with non-filtered sea water. Mysid shrimps showed a dose-related mortality rate, when exposed to sediments previously equilibrated at 0.1, 1.0, or 10 µg Chlordecone/L. Mysids were not affected by James River sediment. Put concentration in sediments, if available Oysters showed dose-dependent reduced shell growth when exposed to Chlordecone-equilibrated sediments, and also responded adversely to river sediment. Lugworms *Arenicola cristata* died after 28 days of treatment with sediment exposed to 10 µg Chlordecone/L, though numbers were not affected by lower doses. Both lugworms and oysters concentrated Chlordecone from the sediment. (Quoted from EHC 43, (IPCS, 1984)).

Table 2.4 Summary of key ecotoxicological studies on Chlordecone.

Taxonomic group and species	End point	Duration	Result mg/L	Reference [1]
Algae *Chlorococcum sp., Dunaliella tertiolecta, Nitzschia sp., Thalassiosira pseudonana*	EC_{50} growth inhibition	7 days	0.35 - 0.60 (formulation)	Walsh *et. al.*, 1977
Algae *Chlorococcum sp., Dunaliella tertiolecta, Nitzschia sp., Thalassiosira pseudonana*	EC_{50} growth inhibition	7 days	350 – 600 (formulation)	Hansen *et. al.*, 1977
Crustaceans *Daphnia magna*	EC_{50} immobility	48 hours	0.120 - 0.690	Barera & Adams, 1983; Adams & Heidolph, 1985; Ziegenfuss *et. al.*, 1986
Crustaceans *Americamysis bahia, Callinectes sapidus, Palaemonetes pugio*	LC_{50}	96 hours	0.01 - 0.210	Nimmo *et. al.*, 1977, 1981; Hansen *et. al.*, 1977; Schimmel, 1977; US EPA, 1976
Crustacean *Daphnia magna*	NOEC reproduction	21 days	0.0283	McKee & Knowles, 1986
Crustacean *Daphnia magna*	NOEC growth	21 days	0.025	Adams & Heidolph, 1985
Crustacean *Americamysis bahia*	MATC growth	28 days	0.000026 - 0.00034	Nimmo *et. al.*, 1981
Insect *Chironomus tentans*	LC_{50}	48 hours	0.17 - 2.3	Adams *et. al.*, 1985; Ziegenfuss *et. al.*, 1986
Fish 9 species	LC_{50}	96 hours, flow through	0.0066 - 0.512	Roberts & Bendl, 1982; Roberts & Fisher, 1985; Schimmel, 1977; Hansen *et. al.*, 1977; Mallat & Barron, 1988; Buckler *et. al.*, 1981
Insect *Chironomus tentans*	NOEC development	14 days	17.9 mg/kg sediment	Adams *et. al.*, 1985

1: All are as quoted in Ecotox, US EPA 2006

In a publication from SETAC a collation of critical tissue residues (CTR) was presented and evaluated (Jarvinen *et. al.*, 1999). The database contains 32 entries for Chlordecone, with data originating from different studies (see Table 2.5). Some of the tissue residues were from studies where no effects were observed, so they may not represent the real CTR. Critical tissue residue values obtained in studies where effects were identified represent 15 CTR values for three fish species. For fathead minnow two studies are available with values of 1.7 and of 3.8-5.4 mg/kg ww. For sheepshead minnow 12 CTRs are available, ranging from 0.13 to 17 mg/kg ww with an average of 5.9 mg/kg ww. Furthermore, one CTR of 2.7 mg/kg ww for spot is available.

Conclusion

In summary, Chlordecone is very toxic to aquatic organisms. The most sensitive group is the invertebrates, which is not surprising for a substance with insecticidal properties. Even if the lowest effect concentration (0.000026 mg/L) was considered to be an outlier, the lowest effect concentrations would be well below 1 mg/L with the results of short term tests (mortality) in the range of 0.01 to 0.69 mg/L and those of long term tests (reproduction and growth) at 0.0025 and 0.0028 mg/L.

Table 2.5 Collation of critical tissue residues (CTR)

Species	Life Stage	Exprte	Expo of Concentration	Results g/g (wet)_)	effect
Cladoceran, Daphnia magna (Fw)	1st instar	Water	175 ng/L	0.133	Survival, Reproduction - No effect
Grass shrimp, Palaemonetes pugio (Sw)	0.09g	Water; Diet	0.04 µg/L; 0.118 µg/g (wet wt)	0.147	Growth - No effect
Blue crab, Callinectes sapidus (Sw)	Juvenile	Diet	2.26 - 2.50 µg/g (wet wt)	2.54 - 4.61	Survival, Growth - No effect
Fathead minnow, Pimephales promelas (Fw)	Larvae-Adult	Water	3.1 µg/L	3.8 - 5.4	Survival, Growth - Reduced
Fathead minnow, Pimephales promelas (Fw)	Larvae-Adult	Water	1.2 µg/L	2.6	Survival, Growth - No effect
Fathead minnow, Pimephales promelas (Fw)	Embryo, 2nd generation	Water; Adult fish	0.31 µg/L; 0.21-0.38 µg/g	1.7	Survival (hatchability) - Reduced
Fathead minnow, Pimephales promelas (Fw)	Embryo, 2nd generation	Water; Adult fish	0.17 µg/L; 0.17-0.46 µg/g	0.26	Survival - No effect
Fathead minnow, Pimephales promelas (Fw)	Larvae, 2nd generation	Water; Adult fish	0.31 µg/L; 0.21- 0.38 µg/g	0.50	Survival, Growth - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Adult	Water	0.8 µg/L	2.5 - 3.6	Survival - Reduced 22%
Sheepshead minnow, Cyprinodon variegatus (Sw)	Adult	Water	1.9 µg/L	11 - 12	Survival - Reduced 80%
Sheepshead minnow, Cyprinodon variegatus (Sw)	Adult	Water	7.8 µg/L	17	Survival - Reduced 100%
Sheepshead minnow, Cyprinodon variegatus (Sw)	Adult	Water	0.16 µg/L	0.65 - 0.90	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo	Adult fish	11-12 µg/g	11	Survival - Reduced 25%
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo	Adult fish	2.5 - 3.6 µg/g	4.7	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Larvae-Juvenile	Water; Adult fish	1.9 µg/L; 11-12 µg/g	8.4	Survival - Reduced 63%
Sheepshead minnow, Cyprinodon variegatus (Sw)	Larvae-Juvenile	Water	2.0 µg/L	7.8	Survival - Reduced 40%

Species	Life Stage	Exprte	Expo of Concentration	Results g/g (wet)_)	effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Larvae-Juvenile	Water	0.8 µg/L	2.0	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Larvae-Juvenile	Adult fish	11-12 µg/g	0.13	Growth - Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Larvae-Juvenile	Water	0.08 µg/L	1.1	Growth – Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo-Adult	Water	0.78 µg/L	5, 6.8*	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo-Adult	Water	0.39 µg/L	2.2, 3*	Growth – Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo-Adult	Water	0.12 µg/L	0.86, 1.2*	Growth - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo-Adult	Water	0.78 µg/L	5, 6.8*	Reproduction – Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo-Adult	Water	0.39 µg/L	2.2, 3*	Reproduction - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo, 2nd generation	Adult Fish + Water	0.78 µg/L	2.3	Survival – Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Embryo, 2nd generation	Adult Fish + Water	0.39 µg/L	1.3	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Fry, 2nd generation	Adult Fish + Water	0.78 µg/L	2.3	Survival - No effect
Sheepshead minnow, Cyprinodon variegatus (Sw)	Fry, 2nd generation	Adult Fish + Water	0.12 µg/L	0.41	Growth – Reduced
Sheepshead minnow, Cyprinodon variegatus (Sw)	Fry, 2nd generation	Adult Fish + Water	0.074 µg/L	0.30	Growth - No effect
Spot, Leiostomus xanthurus (Sw)	Juvenile	Diet	3.3 µg/g (wet wt)	2.7	Survival – Reduced
Spot, Leiostomus xanthurus (Sw)	Juvenile	Diet	3.3 µg/g (wet wt)	0.7	Survival - No effect
Spot, Leiostomus xanthurus (Sw)	Juvenile	Water; Diet	0.04 µg/L; 0.101 µg/g (wet wt)	0.144	Growth, No effect

3 Synthesis of the information

Chlordecone is a synthetic chlorinated organic compound, which has mainly been used as an agricultural pesticide. It is closely related chemically to Mirex, a pesticide which is already listed in Annex A of the Stockholm Convention. Chlordecone is already listed in Annex I of the UNECE Protocol on POPs.

According to available data, Chlordecone can be considered to be highly persistent in the environment. Chlordecone is not expected to hydrolyse or biodegrade in aquatic environments, nor in soil. Direct photodegradation is not significant. Chlordecone does not volatilise to any significant extent.

With BCF-values in algae up to 6,000, in invertebrates up to 21,600 and in fish up to 60,200 and documented examples of biomagnification, Chlordecone is considered to have a high potential for bioaccumulation and biomagnification.

Concerning the potential for causing adverse effects, there is a convincing set of data. Chlordecone is readily absorbed into the body and accumulates following prolonged exposure. It is both acutely and chronically toxic, producing neurotoxicity, immunotoxicity, reproductive, musculoskeletal and liver toxicity at doses between 1 - 10 mg/kg bw/day in experimental animal studies. Liver cancer was induced in rats at a dose of 1 mg/kg body weight per day, and reproductive effects are seen at similar dose levels. The International Agency for Research on Cancer has classified Chlordecone as a possible human carcinogen (IARC group 2B). Moreover, Chlordecone is very toxic to aquatic organisms, most sensitive group being the invertebrates.

The available data on Chlordecone are not fully conclusive when it comes to long-range atmospheric transport in gaseous form. It should be noted that atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents as well as biotic transport could also contribute to long-range environmental transport of Chlordecone.

Due to lack of monitoring data on Chlordecone, the assessment of the potential for long-range transport of Chlordecone is based on physico-chemical properties and especially, on modelling data. While the first of these two approaches may seem somehow insufficient, the modelling data state clearly Chlordecone's LRET potential.

Based on the available data, Chlordecone should be considered as a POP warranting global action.

Production and use of Chlordecone has ceased over the last decades in developed countries, but it is assumed that it can still be produced or used as an agricultural pesticide in some developing countries. If it is still used as pesticide, it will be directly released to the environment. Moreover, the high persistency of the substance has caused high contamination of soil and waters in the areas where it has been used and these contaminated sites can serve as a source of pollution for long times.

4 Concluding statement

It has been demonstrated that Chlordecone meets all the criteria laid down in Annex D of the Stockholm Convention. Moreover, it is chemically very similar to Mirex, an organochlorine pesticide which is already listed in the Stockholm Convention. It is very persistent in the environment and has a great potential for bioaccumulation and in addition there is clear evidence of its biomagnification. While there is no monitoring data from areas remote from sources, the physical and chemical properties, as well as the modelling results, suggest that Chlordecone can be transported long distances bound to particles in air and water, and possibly through coupled transport between these two compartments. Chlordecone is associated with a wide range of harmful effects on both mammals and aquatic organisms.

As Chlordecone can travel in the atmosphere far from its sources, neither a single country nor group of countries alone can abate the pollution caused by this substance. Regional action has already been considered necessary and Chlordecone is totally banned under the UNECE Convention on Long-range Transboundary Air Pollution Protocol on Persistent Organic Pollutants. Although the production and use of Chlordecone seems to be ceased in most countries, its reintroduction remains possible. This could lead to increased releases and levels in the environment.

Based on the available evidence, Chlordecone is likely as a result of its long-range environmental transport to lead to significant adverse human health and environmental effects such that global action is warranted.

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別添7

ヘキサブロモビフェニルの危険性の概要

分解性	蓄積性	人健康影響	動植物への影響
【生分解性】 分解度 4%（OECD TG 301C） 【光分解性】 大気中における分解及び変化は、OH ﾗｼﾞｶﾙによる光酸化と光分解である。OH ﾗｼﾞｶﾙとの反応による推定半減期は 182 日。 【半減期】 ・水中：2 ヵ月を超える ・土壌及び底質中：6 ヶ月を超える	【BCF（経鰓的生物濃縮係数）】 ・ﾌｧｯﾄﾍｯﾄﾞﾐﾉｰ：BCF=18100（32 日間暴露） ・ﾌｧｯﾄﾍｯﾄﾞﾐﾉｰの身：BCF=10000 ・ｺｲ：BCF=4700-16000（重量ﾍﾞｰｽ。60 日間暴露） 【BMF（経口的生物濃縮係数）】 ・餌（ﾆｼﾝ）と捕食者（ﾊﾞﾙﾄｱｻﾞﾗｼ）を較べた食物連鎖：BMF=175（脂質ﾍﾞｰｽ）（PCB と同ﾚﾍﾞﾙの値） ・ﾎｯｷｮｸｸﾞﾏ中の濃度がｸﾞﾘｰﾝﾗﾝﾄﾞ東部のﾜﾓﾝｱｻﾞﾗｼの約 100 倍	【反復投与毒性】 ラット（混餌 7 ヶ月）：0.45mg/kg/day で血清中 T4 濃度低下 ラット（混餌 30 日）：LOAEL 0.05mg/kg/day 甲状腺ろ胞数・ろ胞容積増加、血清中 T3、T4 濃度低下 アカゲザル（混餌 25～50 週）：LOAEL 0.73mg/kg/day 主な毒性は、体重低下、潰瘍性大腸炎、脱毛、肝臓の変化等 【発がん性】 マウス（妊娠 0 日～生後 56 日）：NOAEL 0.15mg/kg/day 児の肝細胞腺がん及び上皮がん IARC グループ 2 B（possibly carcinogenic to human） 【生殖毒性】 ラット（妊娠 0 日～14 日） 28.6mg/kg/day で未着床、新生児生存率低値 アカゲザル：LOAEL 0.012mg/kg/day 主な毒性は、月経周期遅延、流産、死産等	【慢性毒性】 ニジマス *Oncorhynchus mykiss* ：ELS 試験 LD50=3.910 mg/kg

		【その他】 汚染事故で吐き気、腹痛、食欲減退、関節痛、倦怠感、皮膚障害、 EU-Strategy for Endocrine Disruptors 優先化学物質（無処置動物の少なくとも一種類において内分泌かく乱活性を示す科学的根拠がある）に分類	

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Persistent Organic Pollutants Review Committee
Second meeting
Geneva, 6–10 November 2006

Report of the Persistent Organic Pollutants Review Committee on the work of its second meeting

Addendum

Risk profile on hexabromobiphenyl

At its second meeting, the Persistent Organic Pollutants Review Committee adopted the risk profile on hexabromobiphenyl, on the basis of the draft contained in document UNEP/POPS/POPRC.2/9. The text of the risk profile, as amended, is provided below. It has not been formally edited.

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HEXABROMOBIPHENYL

RISK PROFILE

Adopted by the Persistent Organic Pollutants Review Committee
at its second meeting

November 2006

CONTENTS

EXECUTIVE SUMMARY

The European Community and its Member States being Parties to the Stockholm Convention have proposed hexabromobiphenyl to be listed in the Convention. The Persistent Organic Pollutants Review Committee concluded in its meeting in November 2005 that the substance comply with the screening criteria set out in Annex D of the Convention and that a draft risk profile should be prepared to review the proposal further.

Hexabromobiphenyl belongs to a wider group of polybrominated biphenyls (PBBs). The term "polybrominated biphenyls" or "polybromobiphenyls" refers to a group of brominated hydrocarbons formed by substituting hydrogen with bromine in biphenyl. The hexabromo congeners exist as 42 possible isomeric forms. According to the available data, production and use of hexabromobiphenyl has ceased in most, if not all, countries. However, it is possible that hexabromobiphenyl is still being produced in some countries.

Hexabromobiphenyl has been used as a fire retardant in acrylonitrile-butadiene-styrene (ABS) thermoplastics for constructing business, machine housings and in industrial and electrical products and in polyurethane foam for auto upholstery. A considerable part of the substance produced will probably reach the environment sooner or later because of the high stability of these compounds.

According to available data, hexabromobiphenyl can be considered to be highly persistent in the environment. There is evidence of low or no degradation in water, soil and sediment, in the laboratory as well as in the field.

Hexabromobiphenyl is less volatile than many of the currently listed POP substances. However, extensive data on monitoring shows that it is found throughout the Arctic wildlife, demonstrating that it does have a high potential for long range environmental transport.

With measured weight-based BCF values in the range 4,700-18,100 and biomagnification factors in the aquatic food chain exceeding 100, hexabromobiphenyl is considered to be highly bioaccumulative and to have a high potential for biomagnification. These properties are demonstrated by several authors to be comparable to those of hexachlorobiphenyl (a PCB compound), for which the bioaccumulative properties are well documented.

Hexabromobiphenyl is readily absorbed into the body and accumulates following prolonged exposure. Although the acute toxicity of hexabromobiphenyl is low, a number of chronic toxic effects including hepatotoxicity have been observed in experimental animals at doses around 1 mg/kg bw/day following long-term exposure, and effects are seen in the rat thyroid at doses as low as 0.05 mg/kg bw/day. The International Agency for Research on Cancer has classified hexabromobiphenyl as a possible human carcinogen (IARC group 2B). The PBBs are endocrine disrupting chemicals, and effects are seen on reproductive capacity in rats, mink and monkeys. There is epidemiological evidence of hypothyroidism in workers exposed to polybrominated biphenyls and of increased incidence of breast cancer in exposed women. Data on toxicity to other species than laboratory mammals is scarce but suggests the environmental toxicity of hexabromobiphenyl is comparable to that of hexachlorobiphenyl.

Based on the available data, hexabromobiphenyl is likely, as result of its long-range environmental transport, to lead to significant adverse human health and environmental effects, such that global action is warranted.

1 INTRODUCTION

The European Community and its Member States being Parties to the Stockholm Convention have proposed hexabromobiphenyl to be listed in Annex A to the Convention. The original proposal is contained in document UNEP/POPS/POPRC.1/7.

The acceptance of the original proposal for further consideration by the Persistent Organic Pollutants Review Committee implies that the properties of the substance comply with the screening criteria set out in Annex D of the Convention. Therefore, the screening criteria are not discussed in this document. This draft risk profile has been prepared following the decision of the Committee, at its first meeting in November 2005, to establish an ad hoc working group to review the proposal further.

In this document all data are presented according to the International System of Units (SI) and, therefore, many have been recalculated from other units in the data sources. Furthermore, all concentrations are presented based on kg or L (*e. g.* µg/kg or mL/L).

1.1 Chemical Identity of the proposed substance

1.1.1 Names and registry numbers

Hexabromobiphenyl belongs to a wider group of polybrominated biphenyls (PBBs). The term “polybrominated biphenyls” or “polybromobiphenyls” refers to a group of brominated hydrocarbons formed by substituting hydrogen with bromine in biphenyl. The hexabromo congeners exist as 42 possible isomeric forms, which are listed with CAS and IUPAC numbers in US ATSDR (2004) and in document INF 2.

CAS chemical name:	Hexabromo -1,1'-biphenyl
Synonyms:	Hexabromobiphenyl Biphenyl, hexabromo 1,1'- biphenyl, hexabromo - HBB
Trade names:	FireMaster(R) BP-6 FireMaster(R) FF-1

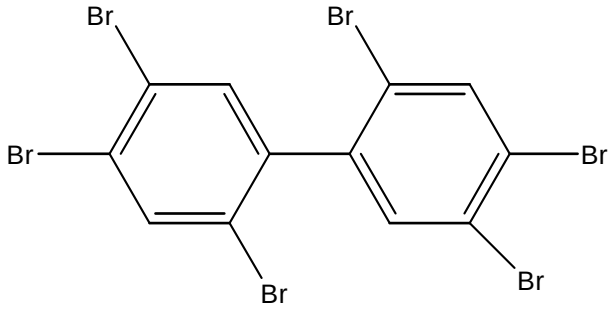
Technical grade PBBs (FireMaster(R)) contain several PBB compounds, isomers and congeners, hexabromobiphenyl being one of the main components. The composition of FireMaster(R) BP-6 changes from batch to batch, but its main constituents are 2,2',4,4',5,5'-hexabromobiphenyl (60-80%), and 2,2',3,4,4',5,5'-heptabromobiphenyl (12-25%) together with lower brominated compounds. Mixed bromochlorobiphenyls and polybrominated naphthalenes have also been observed as minor components of FireMaster(R) (EHC 152 (IPCS, 1994)). FireMaster FF-1 (white powder) is FireMaster BP-6 (brown flakes) to which 2% calcium silicate has been added as an anti-caking agent (EHC 152 (IPCS, 1994)).

Additional data on the composition of identified PBB congeners in FireMaster(R) BP-6 and FireMaster(R) FF-1 is given in US ATSDR (2004).

CAS registry number: 36355-01-8[1] (Common CAS number for hexabromobiphenyl isomers)
59536-65-1 (EHC 192 (IPCS, 1997))[2]
67774-32-7 (EHC 192 (IPCS, 1997))[3]

US ATSDR (2004) provides CAS numbers for a wider number of individual hexabromobiphenyl isomers, as shown in Annex B.

1.1.2 Structure



Structure of 2,2',4,4',5,5'- hexabromobiphenyl (CAS No. 59080-40-9, PBB congener No. 153). (Structural formula source: EHC 192 (IPCS, 1997))

1.1.3 Physical chemical properties

The physical and chemical properties of hexabromobiphenyl are listed in Table 1.1.

Table 1.1 Physical and chemical properties of hexabromobiphenyl.

Property	Unit	Value	Reference
Molecular formula'		$C_{12}H_4Br_6$	
Molecular weight'	g/mol	627.58	
Appearance at normal temperature and pressure		White solid	a)
Vapour Pressure	Pa	$6.9x10^{-6}$ (25° C) $7.5x10^{-4}$ (liquid, sub-cooled)	Jacobs *et. al*., (1976)[a)] Tittlemier *et. al.,* (2002)[a)]
Water solubility	µg/L	11 3	a) Tittlemier *et. al.,* (2002)[a)]
Melting point	°C	72° C	a)
Boiling point		No data	
Log K_{OW}		6.39	Doucette & Andren (1988)[a)]
Log K_{oc}		3.33-3.87	Calculated[a)]
Henry's Law Constant	Pa m^3/mol	$3.95x10^{-1}$ $1.40x10^{-1}$	Waritz *et. al*., 1977[a)] Calculated[a)]

a): Quoted from US ATSDR, 2004

[1] The CAS registry number 36355-01-8 is given as a generic CAS number for PBBs in the 1988 EU Export-Import Regulation and the UNEP Rotterdam Convention.
[2] US ATSDR refers to Firemaster (R) BP-6 as CAS No. 59536-65-1.
[3] US ATSDR refers to FireMaster(R) FF-1as CAS No. 67774-32-7.

Some of the data for the properties listed in Table 1.1 may not be reliable because products of questionable purity were used by earlier investigators to derive them. Therefore, recent physical and chemical property data that have been reported for hexabromobiphenyl in Tittlemier *et. al*., (2002) (Quoted from US ATSDR, 2004) are included in Table 1.1.

1.2 Conclusion of the Persistent Organic Pollutants Review Committee on the Annex D information on Hexabromobiphenyl

The POP Review Committee applied at its first meeting on 7–11 November 2005[4] the screening criteria specified in Annex D to the Stockholm Convention, and decided, in accordance with paragraph 4 (a) of Article 8 of the Convention, that it was satisfied that the screening criteria were fulfilled for hexabromobiphenyl. The Committee decided furthermore, in accordance with paragraph 6 of Article 8 of the Convention and paragraph 29 of decision SC-1/7 of the Conference of the Parties to the Stockholm Convention, to establish an ad hoc working group to review the proposal further and to prepare a draft risk profile in accordance with Annex E to the Convention. It invited, in accordance with paragraph 4 (a) of Article 8 of the Convention, Parties and observers to submit to the Secretariat the information specified in Annex E of the Convention before 27 January 2006.

1.3 Data sources

This Draft Risk Profile is mainly based on information from the following review reports:

- Environmental Health Criteria (EHC) 152: Polybrominated biphenyls. IPCS International Programme on Chemical Safety. United Nations Environment Programme. International Labour Organisation. World Health Organization. Geneva 1994. Available at: http://www.inchem.org/documents/ehc/ehc/ehc152.htm.
- Environmental Health Criteria (EHC) 192: Flame Retardants: A General Introduction. IPCS International Programme on Chemical Safety. United Nations Environment Programme. International Labour Organisation. World Health Organization. Geneva 1994. Available at: http://www.inchem.org/documents/ehc/ehc/ehc192.htm.
- US ATSDR Toxicological Profile for Polybrominated Biphenyls and Polybrominated Diphenyl Ethers (PBBs and PBDEs). 2004. Available at: http://www.atsdr.cdc.gov/toxprofiles/tp68.html

Where the reviews mentioned above have been cited, the text quoted (or quoted with modifications) includes the references cited in the original review. These references are not shown individually in the reference list.

Following the request of the POP Review Committee for additional information, as specified in Annex E of the Convention, information on hexabromobiphenyl was provided by several Parties and observers. This information was mainly based on the open literature or focused on PBDEs.

A search for more recent information included a literature search via the Danish Technical University Library and the data base FINDit (search terms: HBB, hexabromobiphenyl, brominated biphenyls) as well as a data base search in public data bases. The data bases include “Ecotox” (US-EPA, at http://www.epa.gov/ecotox/, “NITE” (Japan, National Institute of Technology and Evaluation at http://www.safe.nite.go.jp/english/db.html, BUA Reports at http://www.gdch.de/taetigkeiten/bua/berichte.htm and Environmental Fate Data Base at http://esc.syrres.com/efdb.htm. This search was based on the search terms: hexabromobiphenyl and CAS numbers 77607091, 36355018, 82865892, 82865905, 59261084, 84303479, 120991482,

[4] See the meeting report at: www.pops.int/documents/meetings/poprc

82865916, 67888997, 84303480, and 60044260. In addition, the Arctic Monitoring and Assessment Programme[5] was consulted.

1.4 Status of the chemical under international conventions

Hexabromobiphenyl is listed in Annex A of the Protocol to the Convention on Long-range Transboundary Air Pollution (CLRTAP) on Persistent Organic Pollutants. The provisions of the Protocol oblige Parties (currently 25) to phase out all production and uses of hexabromobiphenyl. Hexabromobiphenyl, together with other PBBs, is also included in the UNEP/FAO Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade.

2 SUMMARY INFORMATION RELEVANT FOR THE RISK PROFILE

2.1 Sources

2.1.1 Production

The commercial production of polybrominated biphenyls (PBBs) generally involves bromination of biphenyl, a process involving a much more specific reaction and producing a smaller number of product mixtures than chlorination (Sundstrom *et. al*., 1976a) (Quoted from US ATSDR, 2004).

The process of manufacturing PBBs consists of a Friedel-Crafts type reaction in which biphenyl is reacted with bromine in the presence of chloride in an organic solvent, using aluminium chloride, aluminium bromide, or iron as catalyst (Brinkman & de Kok, 1980) (Quoted from EHC 152 (IPCS, 1994)).

2.1.2 Trade and stockpiles

The commercial production of PBBs began in 1970. Approximately 6 million kg of PBBs were produced in the United States from 1970 to 1976. Only three commercial PBB products were manufactured (*i. e.* hexabromobiphenyl, octabromobiphenyl, and hexabromobiphenyl) and these three products were based on a limited number of congeners (Hardy, 2002b). Hexabromobiphenyl constituted about 5.4 million kg (ca 88%) and octa- and decabromobiphenyl constituted ≈ 0.68 million kg together of this total (Neufeld *et. al*., 1977). Michigan Chemical Corporation, St. Louis, Michigan, the sole producer of hexabromobiphenyl in the United States, stopped producing this PBB in 1975. (Quoted from US ATSDR, 2004). Subsequent production of PBBs appears to have been limited to the octa- and decabromobiphenyls.

Production of octa- and decabromobiphenyl continued in the United States until 1979 (IARC 1986; Neufeld *et. al*., 1977). Shortly after the 1973–1974 agriculture contamination accident in Michigan, PBB production in the United States was voluntarily discontinued (Hardy 2000); PBBs are no longer produced in the United States (SRI 2001). Re-initiation of manufacture of PBBs would require approval from the EPA. (Quoted from US ATSDR, 2004)

Two UK companies are reported to have marketed or produced technical-grade decabromobiphenyl in the United Kingdom. In 1977, the production of PBBs in the UK was discontinued. Highly brominated PBBs (Bromkal, 80-9D) were produced in Germany until mid-1985, when the activities concerning bromine-based fire retardants were shifted to the USA. No domestic producer has been identified in the Netherlands. In the early nineties, an Israeli company with two bromine plants in the Netherlands denied the production of PBBs. (Modified from EHC 152 (IPCS, 1994)). There is no information available regarding possible use and production of hexabromobiphenyl in Russia.

[5] See http://www.amap.no/

Until the year 2000, the only PBB in commercial production was decabromobiphenyl, which was manufactured by one company (Atochem) in France (Hardy, 2000). (Modified from US ATSDR, 2004) An author (Darnerud, 2003) has stated that with the closure of the decaBB production in France, the PBB production in the world has ceased.

In the United States, PBBs are not known to be imported or exported anymore except possibly in small quantities for laboratory uses. PBBs have not been imported from other countries into the United States, except in finished products (Neufeld *et. al*., 1977). The two companies that manufactured octa- and decabromobiphenyl in the United States between 1976 (0.805 million pounds) and 1978 exported all of their products to Europe (Neufeld *et. al*., 1977) (Quoted from US ATSDR, 2004) .

EXIDIM, the European Database on the Export Import of Dangerous Chemicals under the Rotterdam Convention has registered a total of 6 export applications for PBBs (which do not however include hexabromobiphenyl) in the years 2003–2006 (1 in 2003 and 2004, 2 each in 2005 and 2006). No imports of PBBs to the European Unions are registered in this period.

Information received by 27 January 2006 as a result of the request for information from Stockholm Convention Parties and observers, included response from Brazil, Australia, Japan, Republic of Lebanon and the USA, all stating that there is no production or use of hexabromobiphenyl in these countries.

In summary, according to the information available, production and use of hexabromobiphenyl has ceased in most, if not all, countries. However, it is possible that hexabromobiphenyl is still being produced in some developing countries or in countries with economies in transition.

2.1.3 Uses

In the United States and Canada, hexabromobiphenyl (FireMaster$^{(R)}$) was the principal PBB product. It was used as a fire retardant in three main commercial products: acrylonitrile-butadiene-styrene (ABS) thermoplastics for constructing business machine housings and in industrial (e.g. motor housing), and electrical (*e. g.* radio and TV parts) products: as a fire retardant in coatings and lacquers, and in polyurethane foam for auto upholstery (Neufeld *et. al*., 1977) (Modified from EHC 152 (IPCS, 1994) and US ATSDR, 2004).

Approximately 5 million tonnes of HBB were produced in the USA from 1970 to 1976; 98 per cent was used as FireMaster BP-6 and the rest as FireMaster FF-1 (Hesse and Powers, 1978). Of the estimated 2,200 tonnes hexabromobiphenyl produced in 1974 (IARC, 1978), about 900 tonnes (Mumma & Wallace, 1975; Neufeld *et. al*., 1977; IARC, 1978) were used in ABS plastic products and an even larger amount in cable coatings (Mumma & Wallace, 1975; Neufeld *et. al*., 1977; IARC, 1978). The exact quantity of FireMaster$^{(R)}$ used in polyurethane foam for automobile upholstery was not published. The two larger consumers ceased using hexabromobiphenyl (one of these in 1972) because PBBs did not decompose in the ultimate incineration of scrapped automobiles (Neufeld *et. al*., 1977) (Quoted from EHC 152 (IPCS, 1994)).

In the EHC 152 (IPCS, 1994), it is stated that at the time, no users of hexabromobiphenyl had been identified (Neufeld *et. al*., 1977; Di Carlo *et. al*., 1978; Brinkman & de Kok, 1980) (Quoted from EHC 152 (IPCS, 1994)) .

2.1.4 Releases to the environment

Data for loss into the environment during normal production are published only for the United States. The following information refers to reviews by Neufeld *et al*., (1977) and Di Carlo *et al*. (1978). Losses of PBBs to the environment at sites of its manufacture can amount to 51 kg/1000 kg of product. These losses occur through:

1) *Emission into the air:*

   In 1977, the maximum air losses as particulate matter at production sites were estimated to total 1.1 kg of PBBs/1000 kg manufactured.

2) *Losses in waste waters* resulting from the quenching and washing of the PBBs as they were recovered from the reaction mass. The losses of PBBs to sewers at manufacturing sites were estimated, in 1977, to be 4.6 µg/kg of product.

3) *Solid losses to landfills* resulting from drying, handling, shipping and transportation. An estimate of PBB losses as solid waste to landfills was 50 g/kg of product.

4) *Losses to the soil*

   Soil samples from the bagging and loading areas of the Michigan Chemical Corp. contained PBBs at concentrations of 3500 and 2500 mg/kg, respectively.

(Abbreviated from EHC 152 (IPCS, 1994))

In 1973, an accidental release of PBBs occurred in Michigan (referred to as the "Michigan disaster" in EHC 152), when two products manufactured by the Michigan Chemical Company were inadvertently confused and 250-500 kg (Di Carlo *et. al*., 1978) of FireMaster(R), instead of NutriMaster(R), a magnesium oxide-based cattle feed supplement, were added to animal feed and distributed to farms within the state. The compound is believed to have been FireMaster(R) FF-1 (*e. g.*, Fries, 1985b), even if in some publications the name FireMaster(R) BP-6 is used (*e. g.*, Neufeld *et. al*., 1977; Di Carlo *et. al*., 1978). This accidental mix up resulted in widespread contamination by PBBs. Chronological reports or reviews of the PBB disaster are given by Carter (1976), Getty *et. al*. (1977), Kay (1977), Di Carlo *et. al*., (1978), Damstra *et. al*., (1982), Zabik (1982), and Fries (1985b) (Quoted from EHC 152 (IPCS, 1994)) .

Approximately 5350 tonnes of hexabromobiphenyl were used in commercial and consumer products in the United States, most in the production of plastic products with an estimated use life of 5–10 years (Neufeld *et. al*., 1977). Since the cessation of production, all of these products, such as TV cabinet and business machine housings, are expected to have been disposed of by land filling or incineration (Neufeld *et al*., 1977) (Quoted from US ATSDR, 2004).

Hexabromobiphenyl can enter the environment from the widespread use of flame-retarded products. A considerable part of the substance produced will probably reach the environment sooner or later because of the high stability of these compounds. Furthermore, some of these chemicals may form toxic polybrominated dibenzofurans during combustion processes.

2.2 Environmental fate

2.2.1 Persistence

The EHC review (1994), concludes that polybrominated biphenyls are stable and persistent in the environment. The degradation of PBBs by purely abiotic chemical reactions (excluding photochemical reactions) is considered unlikely.

In air, the two processes that may result in significant degradation or transformation of PBBs are photo-oxidation by hydroxyl (OH) radicals and direct photolysis. Based on a structure-activity relationship for the estimation of half-lives for the gas phase reactions of hydroxyl radicals with organic compounds (Atkinson 1987b), the estimated half-life of hexabromobiphenyl due to reaction with OH radicals is 182 days. The importance of the photochemical reaction under sunlight illumination conditions for the degradation/transformation of PBBs in air cannot be evaluated due to the lack of information. (Abbreviated from US ATSDR, 2004)

The EHC 152 (IPCS, 1994) refers to laboratory experiments in methanol, showing rapid photodegradation of 2,2',4,4',5,5'-hexabromobiphenyl (90% degradation after 9 minutes) and resulting in mainly lower brominated PBBs. However, in the US ATSDR (2004), it is questioned whether this photolysis could take place in water due to the lack of active groups. Therefore it is questionable whether hexabromobiphenyl can be degraded rapidly in air.

Biodegradation in water under aerobic conditions is low, although the lower substituted biphenyls might biodegrade in aerobic water and sediment (Kong and Sayler, 1983; Sugiura, 1992; Yagi, and Sudo, 1980), the higher substituted biphenyls are resistant to aerobic biodegradation (Kawasaki, 1980; Sasaki, 1978; Shelton and Tiedje, 1981) (quoted from US ATSDR, 2004). This is further supported by the measurement (by GC) of negligible biodegradation of hexabromobiphenyl in a four week ready biodegradability test (OECD TG 301C), resulting in 4% reduction in total concentration as measured by GC (Governmental Japanese database NITE, 2006) resulting in an extrapolated half-life in water>2 months.

Under anaerobic conditions, it has been shown that microorganisms in river sediments obtained from populated areas can biodegrade higher substituted PBBs, including FireMaster mixtures (Morris *et al*. 1992) to form lower brominated products (quoted from US ATSDR, 2004). However, the potential of sediment microflora from remote areas has not been investigated, so it cannot be evaluated whether anaerobic debromination may be a considerable cause for degradation under anaerobic conditions.

PBBs have been reported to be persistent under field conditions. The information on the fate of PBBs in soil is limited. Soil samples from a former PBB manufacturing site, analysed several years after accidental release, still contained PBBs. However, the congener composition differed from the original PBB mixture, indicating partial degradation of the PBB residue in the soil samples. According to the 1994 EHC Review, follow-up surveys over a three-year period following the termination of PBB production showed no significant decline in PBB levels in sediments from a river. In laboratory investigations, mixtures of PBBs appear to be fairly resistant to microbial degradation. (Quoted from EHC 152 (IPCS, 1994)) This implies that the degradation half-life in soil and sediment is>6 months.

The US ATSDR (2004), refers to studies in soils with high levels of FireMaster, in which degradation of hexabromobiphenyl was “significant” during a period of several years but it was not complete. However in other soils, in which the concentrations were lower, or to which manure was added, degradation was even slower. The degradation was attributed to photodegradation even if this process will only take place at the soil surface (US ATSDR, 2004).

Conclusion

In spite of photodegradation in methanol, it is questionable whether hexabromobiphenyl can be degraded rapidly in air. There is evidence of low or no degradation in water (DT_{50}>2 months), soil and sediment (DT_{50}>6 months) in the laboratory as well as in the field. Therefore, hexabromobiphenyl is considered to be highly persistent.

2.2.2 Bioaccumulation

The EHC review states that PBBs are lipophilic and able to bioaccumulate. This is also supported by monitoring results from wildlife studies. For example, fathead minnows (*Pimephales promelas*) caged in a river where water levels of PBB remained consistently at less than 0.1 µg/l concentrated these contaminants in their bodies more than 10,000 fold in two weeks of exposure (EHC 152 (IPCS, 1994)).

As expected from their lipophilicity, PBBs show a marked tendency to accumulate in animals. US ATSDR, (2004), states that PBBs may also be transported from water to aquatic organisms in which bioconcentration may take place. Data from different laboratories on the bioconcentration of

PBBs in fish show wide variation. The experimentally determined bioconcentration factor (BCF) for hexabromobiphenyl (mixtures of unspecified congeners) in the whole body of fathead minnows (*Pimephales promelas*) was 18,100 in a 32-day exposure (Veith *et. al*., 1979). In fillet of fathead minnow, the estimated BCF was>10,000 (Hesse and Powers, 1978). Weight-based BCF values in the range 4,700-16,000 were recorded in a 60 day test with the carp *Cyprinus carpio* placed in concentrations of hexabromobiphenyl of 0.1-1 µg/L respectively (Governmental Japanese database NITE, 2006).

Furthermore, a potential for biomagnification has been demonstrated by Jansson *et. al*., (1993), who reported a biomagnification factor (BMF) for 2,2',4,4',5,5'-hexabromobiphenyl (PBB congener 153) of about 175 comparing lipid-based concentrations in prey (herring) and predator (Baltic seal). This BMF was at the same level as that of the PCB congener 153. These findings were supported by Vorkamp *et. al*., (2004)[6], who found lipid-based concentrations of hexabromobiphenyl (PBB 153) in polar bear to be a factor of about 100 higher than in ringed seal from East Greenland. They conclude further, that the PBBs (and PBDEs) seem to biomagnify along the marine food chain in a manner similar to PCBs and that PBBs show indications of a higher biomagnification potential than PBDEs (Vorkamp *et. al*., 2004).

Conclusion

With measured weight-based BCF values in the range 4,700-18,100 (most of which exceed 5,000) and demonstrated biomagnification in the aquatic food chain, hexabromobiphenyl is considered to be highly bioaccumulative and to have a high potential for biomagnification. These properties are demonstrated by several authors to be comparable to those of hexachlorobiphenyl, for which the bioaccumulative properties are well documented. Evidence appears to be satisfactory to conclude high bioaccumulation and biomagnification.

2.2.3 Potential for Long Range Environmental Transport

The partitioning of hexabromobiphenyl in the environment will be governed by its high log K_{ow} (6.39) and low water solubility (3 µg/L) resulting in sorption to particulate matter (dust, soil and sediment) and organic material (including living organisms). Furthermore, the combination of these properties and the relatively low vapour pressure ($6.9x10^{-6}$ to $7.5x10^{-4}$ Pa) of hexabromobiphenyl, results in a low potential for volatilisation. The latter is specified in US ATSDR (2004) as follows: Based on an estimated Henry's law constant of $3.95x10^{-1}$ Pa m^3/mol (where Henry's law constant = vapor pressure/water solubility) and an estimation method (Thomas, 1990), the estimated volatilization half-life of hexabromobiphenyl is 23 days. Therefore, the transport of PBBs from water to the atmosphere by volatilization is not expected to be important.

The assessment of the potential for long-range transport of hexabromobiphenyl could be done by comparing the properties of hexabromobiphenyl to those of the currently listed POPs. As a starting point for the assessment of hexabromobiphenyl, the highest and lowest of the values in Table 1.1 were used (for vapour pressure, only the value at 25 °C) and, for comparison, the information on the UNEP-POPs homepage. Among the currently listed POPs, most of the relevant properties were available for aldrin, chlordane, dieldrin, DDT, hexachlorobenzene, mirex, toxaphene, endrin and heptachlor. Missing information (water solubility of mirex) was sought in US ATSDR (1995), so as not to introduce what seems to be an outlier in the comparison by using the value of $6.5x10^{-5}$ mg/L from AMAP (2004).

[6] These investigations are part of the Arctic Monitoring and Assessment Programme (AMAP).

The water solubility and vapour pressure as well as Henry's Law Constants calculated from these values of the currently listed POPs are summarised in Table 2.1 together with information on hexabromobiphenyl from Table 2.1.

Table 2.1 Water solubility (WS), vapour pressure (VP) and (calculated) Henry's Law Constant (HLC) (at 25 °C) for hexabromobiphenyl and currently listed POPs.

Substance	WS mg/L	VP Pa	HLC Pa m^3/mol
Hexabromobiphenyl-min	0.011	$6.9x10^{-6}$	0.39
Hexabromobiphenyl-max	0.003	$6.9x10^{-6}$	1.44
POP-min	0.0012 (DDT)	$2.5x10^{-5}$ (DDT)	0.04 (endrin)
POP-max	3.0 (toxaphene)	27 (toxaphene)	3726 (toxaphene)
POP-2^{nd} max	0.5 (dieldrin)	0.04 (heptachlor)	267 (heptachlor)

Table 2.1 shows that the water solubility of hexabromobiphenyl is at the level of the least water soluble among the currently listed POPs (DDT), while the vapour pressure of HBB is one order of magnitude lower than that of DDT. The two Henry's Law Constants calculated for hexabromobiphenyl are well inside the range marked by the currently listed POPs, being at least one order of magnitude higher than the lowest (endrin). It should be noted that in presenting the data in table 2.1 it is not inferred that a chemical (in this case hexabromobiphenyl) is considered to meet the long-range environmental transport criterion just because it fits within the range of values of currently listed POPs.

Based on the vapour pressure alone, the potential for long-range airborne transport of hexabromobiphenyl is low compared to most of the currently listed POPs, while a comparison of the Henry's Law Constants places hexabromopbiphenyl in a position close to endrin.

The EHC 152 (1994), argues that the vapour pressure of hexabromobiphenyl is $6.9x10^{-6}$ Pa and, thereby the potential for volatilisation is low. There is no information available about measured half-life of hexabromobiphenyl in the atmosphere. In the laboratory photodegradation of 2,2',4,4',5,5'-hexabromobiphenyl was rapid (90% degradation after 9 minutes) mainly resulting in lower brominated PBBs (EHC 152 (IPCS, 1994)). On the other hand, the rates and extent of photolytic reactions of PBBs in the environment have not been determined in detail. The few field observations available indicate a high persistence of the original PBBs or a partial degradation to less brominated, and often more toxic, photoproducts.

In support of the assessment of the potential for long-range environmental transport, monitoring data demonstrate that this substance has managed to reach remote areas like the Barents Sea and Greenland. In the Arctic, hexabromobiphenyl has been measured in samples of animals in several investigations. The results are summarised in Annex A, Table A.1.

In whitefish from Lapland (North Scandinavia) and ringed seal from Svalbard, concentrations of 0.29 and 0.42 µg/kg lipid, respectively, were reported by Jansson *et. al*., (1993). In another paper, Jansson *et. al*., (1987) reported concentrations of hexabromobiphenyl (Firemaster BP-6) in ringed seal from Svalbard to be 4 µg/kg lipid and concentrations in guillemot muscle of 50 µg/kg lipid. It is not clear whether these results are from different investigations. For comparison, Krüger (1988), measured 0.8 µg/kg of PBB 153 in unspecified seal samples from the same area (Quoted from US ATSDR, 2004).

In samples of large char collected in 1999-2001 from one of two lakes in Bear Island in the Barents Sea, Evenset *et. al*., (2005) measured concentrations of 4.11-51.5 µg/kg lipid of hexabromobiphenyl (PBB 153). These figures should be used with some caution since levels of other POPs are always very high in char from this lake, maybe due to a local biotransfer process through neighbouring bird species. These levels are the same as or higher than levels of PBB 153 (0.2-9.4 µg/kg lipid) in lake trout sampled in 1997 from Lakes Ontario, Erie, Huron and Superior, which were measured by Luross *et al*., (2002) (Table 2.2).

Vorkamp *et. al.* (2004) , measured concentrations of PBDEs in samples from Greenland and the Faroe Islands of sediment and seven species of animals representing different trophic levels of the food chain. As a pilot investigation, analyses for five PBBs including PBB 153 were made in selected samples of blubber or fat from ringed seal, mink whale and polar bear from Greenland as well as pilot whale and fulmar from the Faroe Islands. PBBs were detected in all samples, except sediment samples, shorthorn sculpin samples and samples of ringed seal from West Greenland. In all other samples, PBB 153 was generally the dominant congener. The concentrations measured in samples from (East) Greenland were in the range 0.34–44.26 µg/kg lipid with the lowest values found in the seal and the highest in polar bear. In the Faroese samples, the range of concentrations of PBB 153 was 8.71-25.54 µg/kg lipid weight with the highest values found in fulmar, a fish predator (Vorkamp *et. al*., 2004).

For comparison, concentrations of PBB 153 in grey seal and osprey from the Baltic Sea were 26 and 22 µg/kg lipid weight; respectively (Jansson *et. al*., 1993). Thus, concentrations of PBB 153 as µg/kg lipid weight in seals from the Arctic (0.34-0.74) are considerably lower than in seals from the Baltic Sea (26 µg/kg lipid weight), while concentrations in predatory birds from the two areas (fulmar and osprey) are of the same order of magnitude, being 25 and 22 µg/kg lipid weight; respectively.

Vorkamp *et. al*., (2004), conclude that PBBs and PBDEs seem to biomagnify along the marine food chain in a similar manner to PCBs. PBBs show indications of a higher biomagnification potential than PBDEs. Even though their absolute concentrations are lower than those of PBDEs, the PBDE/PBB ratio increases in the order ringed seal<pilot whale<mink whale<fulmar<polar bear, leading to almost equal concentrations of PBDEs and PBBs in polar bear. Apparently, the compounds follow the same spatial trend as previously observed for organochlorine compounds, with higher concentrations in East Greenland than in West Greenland (Vorkamp *et. al*., 2004). This indicates that the long-range transport of hexabromobiphenyl may be slow.

Monitoring information on PBBs from areas outside the Arctic, Northern Europe and America is scarce, as only one reference has been found. Hexabromobiphenyl (PBB 153) was not detected (LOD between 0.02 and 0.1 µg/kg wet weights) in samples of muscle and liver from several species of fish from the eastern Mediterranean region of Turkey (Erdogrul *et. al*., 2005).

In summary, the 1994 EHC, review concludes that long-range transport of PBBs in the atmosphere has not been proven, but that the presence of these compounds in Arctic seal samples indicates a wide geographical distribution (EHC 152 (IPCS, 1994)). Several authors report levels of hexabromobiphenyl (and other brominated biphenyls) in arctic animals, especially in fish eating predators and predators at higher trophic levels.

In a recent modelling study, Scheringer *et. al.*, (2006), investigated the persistence and long range transport potential of four potential POPs, including chlordecone and hexabromobiphenyl. They concluded that these POP candidates have persistence and long range transport potential properties similar to those of several known POPs. Furthermore, they included the uncertainty regarding the data quality in a Monte Carlo analysis, which indicated that the result is valid although there are considerable uncertainties in the chemical properties of the four POP candidates.

Conclusion

Although hexabromobiphenyl is less volatile than any of the currently listed POPs, it is found throughout the Arctic wildlife, demonstrating that it does have a high potential for long range environmental transport. The potential for long range environmental transport of hexabromobiphenyl is further supported by the modelling study of Scheringer *et. al.*, 2006.

2.3 Exposure

Because production of hexabromobiphenyl is assumed to have ceased (section 2.1.2) the assessment of the exposure will focus on general exposure instead of current production sites.

2.3.1 Concentrations in abiotic environmental media

Recent monitoring data in soil, water and sediments for PBBs are limited. Historical monitoring data from the United States indicate that environmental PBB concentrations are confined to areas near former manufacturing facilities and regions of Michigan affected by the farm accident of the early 1970's (see Section 2.2.3) (US ATSDR, 2004) .

The only available data for environmental concentrations of PBBs in areas outside the vicinity of former production sites are those from sediment samples from Greenland (Vorkamp *et. al*., 2004), where PBBs (including PBB 153) were not detected in any sample (the limits of detection/quantification are, however, not well defined in the paper).

2.3.2 Concentrations in biota

In the vicinity of Michigan

Concentrations in biota in the vicinity of the Michigan production and contamination accident sites were measured in a multitude of samples during the decade following the cessation of production. The US ATSDR (2004) includes the following: In the late 1980's, PBBs were detected in the concentration range of 15–15,000 µg/kg (lipid basis) in fish from embayments and tributaries of Lake Huron, but not from Lake Superior. Recently, Luross *et. al*. (2002) determined the concentrations of several PBB congeners in lake trout from Lakes Huron, Superior, Erie, and Ontario. 2,2',4,4',5,5'-Hexabromobiphenyl (PBB-153) and 2,2',4,5,5'-pentabromobiphenyl (PBB-101) were found at the highest levels at concentrations ranging from 0.189 to 2.083 µg/kg wet weight and from 0.042 to 0.633 µg/kg wet weight, respectively. Several other congeners were also detected in these lake trout samples (Quoted from US ATSDR, 2004). The concentrations of PBBs in eggs of fish-eating birds (common tern, little gull, herring gull, and red-breasted mergansers) collected during 1975–1980 from nesting islands in northwestern Lake Michigan and Green Bay contained PBBs in the concentration range of 0.02–0.25 mg/kg (µg/g) wet weight (Heinz *et al*. 1983, 1985) (quoted from the US ATSDR, 2004).

Other areas

Monitoring data from areas outside the Arctic (see chapter 2.2.3) and the most exposed region of the US are summarised in Table A.2. in Annex A.

EHC 152 (1994) includes the following investigations on residues of (hexa)bromobiphenyl in biota:

- In Europe, 2,2',4,4',5,5'-hexabromobiphenyl (PBB 153) was found in fish from German and Swedish rivers at concentrations ranging from 0.3 to 0.6 µg/kg lipid (Krüger, 1988; Jansson *et. al*., 1992). A trout sample from a breeding farm contained much lower levels of PBBs than the fish samples from the rivers (Krüger, 1988).

- Swedish reindeers (pooled samples) showed PBB 153 levels as low as 0.04 µg/kg lipid (Jansson *et. al*., 1992).
- PBBs (as a group) were not found in otters *(Lutra canadensis)* from a region relatively remote from industrial sites in north eastern Alberta (Canada) (Somers *et. al*., 1987).
- Fish samples (freshwater and marine species) collected in 1983 from an industrial area of Japan (Osaka) did not contain "PBBs" (not specified) (Watanabe & Tatsukawa, 1990).
- In Europe, PBBs have been detected in seals *(Phoca vitulina; Pusa hispida),* guillemots *(Uria aalge; U. lomvi),* and white-tailed sea eagles *(Haliaeetus albicilla).* The concentrations (estimated by comparison with the technical product Firemaster BP-6) ranged from 3 to 280 µg/kg lipid (Jansson *et. al.*, 1987). The concentrations of PBBs in comparable samples from the Baltic Ocean were all higher than concentrations in samples from the Arctic Ocean. The same was true for polybrominated biphenyl ethers and PCBs (Jansson *et. al*., 1987).
- Concentrations of PBB 153 determined in marine fish ranged from 0.2 to 2.4 µg/kg lipid (Krüger, 1988; Jansson *et. al*., 1992). PBB 153 levels of 0.4-26 µg/kg lipid were found in seals (Krüger, 1988; Jansson *et. al*., 1992).
- Detailed isomer-specific PBB analyses were carried out by Krüger (1988), in fish (several species) from the Baltic and North Seas and from sections of the Lippe and Rur rivers in North Rhine-Westphalia, Germany. Seal samples from Spitsbergen (Norway) were also included in this investigation. All samples contained PBBs. The smallest number of PBB congeners was found in seals (n=5) from an area remote from industrial sites. The main components were different hexabrominated isomers with 2,2',4,4',5,5'-hexabromobiphenyl reaching a mean concentration of 0.8 µg/kg fat. The mean concentrations of several PBB congeners and isomers (penta- to nonabrominated biphenyls) measured in fish (n=35) ranged, mostly, between 0.01 and 2 µg/kg fat. The pattern of PBB congeners found in fish differed in a characteristic manner, depending on the different capture sites. While relatively high amounts of nona- and octabromobiphenyls (besides polybrominated biphenyl ethers) were present in fish from German rivers (n=17; several species), hexabrominated biphenyls were predominant in fish from the North Sea and the Baltic Sea (n = 17; several species). In all samples from the Baltic Sea (n=6), 3,3',4,4',5,5'-hexabromobiphenyl was found in relatively high concentrations (maximum concentration: 36 µg/kg fat), but it was not detected in samples from the North Sea and from rivers. The concentrations of the other hexabrominated biphenyls were mostly higher in fish from the Baltic Sea than in fish from the North Sea.

(Quoted from EHC 152 (IPCS, 1994))

US ATSDR (2004) supplements with:

- Three bottlenose dolphins (*Tursiops truncatus*) collected during 1987–1988 from the U.S. mid-Atlantic contained PBBs at concentrations of 14–20 µg/kg lipid basis (Kuehl *et. al*., 1991). The source of the PBBs in the dolphins was not given.
- The median concentrations of PBBs in carcass and brain of 10 specimens of bald eagles (*Haliaeetus leucocephalus*) collected from 29 states in 1977 were 0.07 and 0.05 mg/kg (µg/g), respectively (Kaiser *et. al*., 1980). Twenty-two other specimens did not contain detectable levels (<0.03 mg/kg [µg/g]) of PBBs.
- In whitebeaked dolphins from the North Sea, the concentration of hexa-, penta-, and deca-BBs were 13, 8.3, and <0.9 µg/kg (µg/kg) wet weight, respectively. Tetra-, penta-, and deca-BBs concentration ranges were 1.1–1.9, 0.4–0.9, and <0.5 µg/kg wet weight, respectively, in sperm whales from the Atlantic Ocean (de Boer *et. al*., 1999).

The German Baltic fish samples (as the only samples in that investigation) also contained PBB 169 at a concentration of 15.16 µg/kg lipid (EHC 152 (IPCS, 1994)).

In the Belgian samples from corpses of birds of prey, the variation in concentrations of hexabromobiphenyl was high. Thus, the maximum concentrations measured in muscle and liver were 150 and 180 µg/kg lipid; respectively (Jaspers *et. al*., 2006).

Jansson *et al*. (1993), measured hexabromobiphenyl (PBB 153) in samples of reindeer (a herbivore) from northern Sweden at a level of 0.037 µg/kg lipid. In two other herbivores (rabbit and moose) from Southern Sweden, PBBs were not detectable (level of detection not well defined).

2.3.3 Concentrations in human tissues and breast milk

Michigan

The human exposure to hexabromobiphenyl subsequent to the Michigan accident is discussed in EHC 152 (1994) as well as in US ATSDR (2004). The general trends of the findings are described as follows in EHC 152 (1994):

- Nearly 100% of the adipose samples randomly selected throughout the state had detectable PBB concentrations. Thus, statewide exposure of Michigan residents to PBBs can be demonstrated.
- Levels of PBBs in serum (Landrigan, 1980; Wolff *et. al*., 1982), breast-milk (Brilliant *et. al*., 1978; Miller *et. al*., 1984), and adipose tissue (Wolff *et. al*., 1982) were highest in the area of the accident (lower peninsula), and lowest in the upper peninsula, farthest from the source.
- Compared with residents of quarantined farms, direct consumers of products from quarantined farms, and PBB production workers, the tissue burdens among the general population of Michigan were 1-3 orders of magnitude lower. Moreover, for example, only 36% of the general population had serum PBB concentrations greater than 1 µg/L, compared with 78% among farmers (Anderson *et. al*., 1979; Wolff *et. al*., 1982).
- PBB levels appear to be higher in males than females (Meester & McCoy, 1976; Landrigan *e.t al*., 1979; Landrigan, 1980; Wolff *et. al*., 1978; 1980; Kreiss *et. al*., 1982; Eyster *et. al*., 1983) and higher in children (below the age of 10 years) than in adults (Humphrey & Hayner, 1975; Landrigan *et. al*., 1979; Landrigan, 1980; Barr, 1980; Wolff *et. al*., 1982) (Quoted from EHS 152 (IPCS, 1994)).

The subsequent development is described in EHC 152 (1994):

- In most cases, PBB concentrations did not appear to be decreasing significantly over time. Wolff *et. al*. (1979b), did not find any significant variation in the serum PBB levels of nine dairy farm residents during 18 month of observation.
- Paired serum samples, one collected in 1974 and the other in 1977, were also available for 148 members of the Michigan PBB cohort. The data indicate that levels were generally stable over the 3-year period with a mean change of 16 µg/litre (Landrigan *et. al*., 1979). In another study of the Michigan PBB-cohort, the decrements in median serum levels of PBBs between matched pairs over one (1977-78) and two (1977-79) year intervals were both only 1 µg/litre (Kreiss *et. al*., 1982). No significant change in blood plasma PBB levels was observed over a 5-month period in 41 residents of quarantined farms (Humphrey & Hayner, 1975). In contrast, Meester & McCoy (1976) reported a marked decline over 3 years (1974-76) in serum levels of PBBs. These authors also found that the average decrease in PBB concentrations in the fat of 16 individuals was about 40% in a period of 6 months. No changes in PBB levels were seen over an 11-year period (1976-87) in fat samples from a patient with long-term exposure to PBBs from the early 1970s as a result of the Michigan PBBs accident. The average fat level of PBBs was 0.8 mg/kg (Sherman, 1991).

- In 1981, PBBs were found in 13-21% of serum samples from 4-year-old Michigan children. Their mothers belonged to a group that was surveyed either with regard to the consumption of Lake Michigan sport fish (mean PBB level detected in children: 2.4 ng/ml) or with regard to former exposure to quarantined farm products (mean PBB level detected in children: 3.0 ng/ml) (Jacobson *et. al*., 1989) (Quoted from EHC 152 (IPCS, 1994)).

Other areas

The EHC 152 (1994), stresses the lack of available monitoring studies from areas outside Michigan, as few human monitoring data are available for the US population outside of Michigan. One study deals with the population in the vicinity of industrial areas involved in PBB production or use (Stratton & Whitlock, 1979), the other with farmers of the state of Wisconsin who were examined as control group in connection with the Michigan PBB studies (Wolff *et. al*., 1978).

PBBs were found in all studies, but, because of the limited data, the significance is unclear. The highest PBB levels were found in the hair of humans living near PBB industry. Of the nine samples analysed, five had detectable PBB levels. Both male and female hair samples contained PBBs (Stratton & Whitlock, 1979).

There is very little human monitoring data on PBBs in the populations of countries other than the United States. Krüger *et. al*., (1988) reported PBB contamination of breast-milk from women in Europe in a survey from North Rhine-Westphalia, Germany. The milk samples (n=25) contained a typical pattern of certain PBB congeners. It included penta- to octabromobiphenyls in concentrations ranging from 0.002 to 28 µg/kg, based on milk fat. The most abundant component was 2,2'4,4',5,5'-hexabromobiphenyl (PBB 153) followed by a peak consisting of two heptabromobiphenyl isomers (2,2',3,4',5,5',6- and 2,2',3,4,4',5,6'-heptabromobiphenyl, PBB 187 and 182 respectively). Differences in the pattern were only found in the milk given by a Chinese woman and in that given by a woman having been exposed to several fires in industry.

Concentrations of PBB 153 in human and cow's milk, both collected from the same region (North Rhine-Westphalia), were 1 µg/kg and 0.03 µg/kg, respectively, measured on a lipid basis (Krüger, 1988). (Quoted from EHC 152 (IPCS, 1994))

2.3.4 Human exposure

The US ATSDR (2004), considers the current human exposure to PBBs to be very low, because PBBs are no longer produced or used. Thus, the general population exposure to PBBs will only be from historical releases. For people residing in the lower peninsula of Michigan, especially in the immediate vicinity of the PBB contaminated areas of this region, exposure to PBBs may still be occurring today. However, environmental levels have decreased since the 1970s and current exposure, if any, will be at low levels. For other regions of the United States, the levels of exposure will either be very low or none (Quoted from US ATSDR, 2004) .

In Arctic and North Atlantic regions, where the traditional diet includes top predators (*e. g.* seal in Greenland and pilot whale in the Faroe Islands), exposure has not ceased. Especially the level of PBBs in pilot whale blubber of up to 17 µg/kg lipid indicate the presence of hexabromobiphenyl in food. Pilot whale blubber is consumed as a delicacy in the Faroe Islands.

2.4 Hazard assessment for endpoints of concern

2.4.1 Toxicity

As described in Section 1.1.1, the descriptor “hexabromobiphenyl” covers 42 different hexabrominated biphenyls or congeners, as individually listed in Annex B. The EHC review (IPCS, 1994) indicates that the hexabrominated biphenyls are the most toxic of the chemical class

of polybrominated biphenyls (PBBs) and that the higher homologues (hepta-, octa-, nona- and decabrominated biphenyls) are of progressively lower toxicity. Toxicological studies on hexabromobiphenyl have been carried out mainly on the congener 2,2',4,4',5,5'-hexabromobiphenyl (PBB 153), which is the major component of the PBB mixture FireMaster®, and on FireMaster® itself. The toxicity of FireMaster® appears to be primarily associated with the minor components 2,3,3',4,4',5-hexabromobiphenyl, 2,3',4,4',5,5'-hexabromobiphenyl, 3,3',4,4',5,5'-hexabromobiphenyl (PBB 169) and 2,3',4,4',5-pentabromobiphenyl (IPCS, 1994). The predominant congeners in FireMaster® (2,2',4,4',5,5'-hexabromobiphenyl and 2,2',3,4,4',5,5'-heptabromobiphenyl), are less toxic (IPCS, 1994). Other toxic contaminants in technical PBB mixtures include the polybrominated naphthalenes (HBNs). Hexabromonapthalene has been identified as a toxic contaminant of Firemaster BP-6 or FF-1 at levels of approximately 150 ppm (Birnbaum *et. al.*, 1983, as reported in US ATSDR, 2004). The toxicological effects of the PBBs in humans and in animal studies, as described in the scientific literature, are considered to be attributable mainly to exposure to hexabromobiphenyl congeners (EHC 152 (IPCS, 1994) and US ATSDR, 2004)), although a possible contribution of the HBNs to toxicity cannot be ignored.

Mechanism of action

Hexabromobiphenyl, in common with all PBBs, is a potent inducer of hepatic cytochrome P-450 metabolizing enzymes in the liver. The mechanism of action underlying a number of the toxicological effects of some of these compounds, including induction of metabolising enzymes, immunotoxicity, hepatotoxicity and reproductive toxicity, is considered to be due to interaction with the cellular Ah receptor (also the target of the polychlorinated dioxins, furans and dioxin-like PCBs), causing altered gene expression (Poland & Glover, 1977, 1980; Poland *et. al*., 1979; Goldstein, 1980; Moore *et al*., 1980; McKinney & Singh, 1981; Parkinson & Safe, 1981; Bandiera *et. al*., 1982, 1983; McKinney & McConnell, 1982; Nebert *et. al*., 1982; Poland & Knutson, 1982; Robertson *et. al*., 1982b, 1984c,d; Safe *et. al*., 1982, 1985; Aust *et. al*., 1983; Dannan *et. al*., 1983; Lai, 1984; Safe, 1984, as quoted in IPCS, 1994).

Toxicokinetics

Hexabromobiphenyl is readily absorbed into the body, the primary route of human exposure being via food, due to accumulation and biomagnification in the food chain (IPCS, 1994; US ATSDR, 2004). The majority of animal toxicology studies have used the oral route of exposure and little information is available on exposure via the inhalation and dermal routes, although worker exposure is likely to occur mainly via these routes (Wolff *et. al*., 1979a, as quoted in IPCS, 1994). Following absorption, hexabromobiphenyl is widely distributed in the body and accumulates, with the highest concentrations found in adipose tissue and to a lesser extent the liver (IPCS, 1994).

Exposure *in utero* occurs via transfer of PBBs to offspring by placental transfer and infants are also exposed via milk. Human milk has been found to contain levels of 2,2',4,4',5,5'-hexabromobiphenyl 100 times higher than those found in maternal blood (Brilliant *et. al*., 1978; Landrigan *et. al*. 1979; Eyster, 1983, as reported in IPCS, 1994).

Metabolism and excretion of the hexabromobiphenyls is low (IPCS, 1994; US ATSDR, 2004), and the compounds therefore show marked bioaccumulation and persistence in all species. Average half-lives for 2,2',4,4',5,5'-hexabromobiphenyl in humans have been estimated to be between 8 and 12 years (IPCS, 1994), while shorter half-lives have been reported in rats, monkeys, and other species (see Table 68 in IPCS, 1994). It has been suggested that humans may retain certain congeners to a greater degree than experimental animals (*e. g.* Fries (1985b, as quoted in IPCS, 1994), a phenomenon that is also found with the polychlorinated dioxins and furans.

Darnerud (2003), argues that the pattern of toxicity of PBBs should be similar to that of PCBs apart from the change in effects brought about by the chlorine-bromine substitution. Consequently, the planar PBBs are expected to be most toxic (as they bind to the Ah receptor) and toxicity to decrease through mono-ortho congeners to di-ortho congeners. This should be supported by experimental evidence, as 3,3',4,4',5,5' hexabromobiphenyl was found to be the most toxic PBB congener in several systems (Darnerud, 2003).

Toxicity of hexabromobiphenyl in animal studies

In experimental animal studies, hexabromobiphenyl shows relatively low acute toxicity (LD_{50}>1 g/kg body weight) (see Table 70, IPCS, 1994). Toxicity is higher following repeated exposure (IPCS, 1994), due to progressive accumulation of the compounds and a characteristic delay in lethality after exposure is seen (Di Carlo *et. al*., 1978; Gupta & Moore, 1979, (as quoted in IPCS, 1994). At lethal doses, death is reported to be due to a "wasting syndrome" with marked loss in body weight rather than to specific organ pathology (Hutzinger *et. al*., 1985a; McConnell, 1985, as quoted in IPCS, 1994). However, prolonged exposure of laboratory animals to doses in the range of<1 mg/kg bw/day to 100 mg/kg bw/day results in liver, kidney and thyroid changes, accompanied by effects in the nervous and immune systems, porphyria and skin disorders (IPCS, 1994).

A summary of outcomes of a number of the key toxicological studies on hexabromobiphenyl, including the NOAEL/LOAEL derived in each study is provided in Annex A, Table A.3 to this document. The studies included in Annex A, Table A.3 have been selected from the very large database on toxicological studies on hexabromobiphenyl, on the basis of the importance of the endpoint investigated (*e. g.* reproductive toxicity, carcinogenicity, other key target organ toxicity), robustness of the reported studies and the dose level (NOAEL/LOAEL) at which effects were reported. Table 2.2 below provides information on pivotal toxicological studies (also included in Annex A Table A.3) that provide information on the toxicity of hexabromobiphenyl at low levels of exposure, considered to be particularly relevant for characterisation of the toxicological risks of these compounds. Some of these studies have been used by US ATSDR to define Minimal Risk Levels (MRLs) for hexabromobiphenyl (US ATSDR, 2004).

Effects in toxicological studies included decreased circulating thyroid hormones in a 10-day gavage study in rats with a NOAEL of 1 mg/kg bw/day (Allen-Rowlands *et. al*., 1981, as quoted in US ATSDR, 2004), decreased lymphoproliferative responses in rats at a dose level of 3 mg/kg/day (LOAEL) (Luster *et. al*., 1980, as quoted in US ATSDR, 2004), and generalised toxicity in male Rhesus monkeys at 0.73 mg/kg bw/day (LOAEL) (Allen *et. al*., 1978; Lambrecht *et. al*. 1978 (as quoted in US ATSDR, 2004)). PBBs produced porphyria in rats and male mice at doses as low as 0.3 mg/kg bw/day. The no-effect level was 0.1 mg/kg bw/day.

These results show that hexabromobiphenyl produced long-term toxicity in experimental animals at very low doses, a critical effect for the purposes of risk characterization being the effects seen in the thyroid in rats at doses as low as 0.05 mg/kg bw/day, comprising increased number and decreased size of follicles, accompanied by changes in levels of circulating T_3 and T_4 hormone (Akoso *et al*., 1982, as quoted in US ATSDR, 2004).

Hepatocarcinogenicity of hexachlorobiphenyl has been demonstrated in a number of studies including repeated dose studies in Fischer-344/N rats and B6C3F1 mice (males and females) administered FireMaster$^{(R)}$ FF-1 at dosages of 0, 0.1, 0.3, 1, 3, or 10 mg/kg bw/day (NTP 1983, NTP, 1992, as quoted in US ATSDR, 2004). Tumors included hepatocellular adenoma and carcinoma and, in female rats, cholangiocarcinoma. The lowest dose of FireMaster$^{(R)}$ that produced tumors (primarily adenomas rather than carcinomas) in rats was 3.0 mg/kg bw/day for 2 years, and in mice the dose was10 mg/kg bw/day (NTP 1983, as quoted in US ATSDR, 2004). Mice receiving 0.15 mg/kg bw/day in a study involving pre- and perinatal exposure in addition to lifetime exposure

did not suffer any adverse effects (NTP, 1992, as quoted in US ATSDR, 2004). The International Agency for Research on Cancer (IARC) in 1987 concluded that there was sufficient evidence that hexabromobiphenyl is carcinogenic in mice and rats and possibly carcinogenic to humans (Group 2B). Hexabromobiphenyl is not genotoxic in *in vitro* microbial and mammalian cell gene mutation assays (see Table 88 in IPCS, 1994), although it has been reported to interfere with cell-to-cell communication (Sleight, 1985 as quoted in IPCS, 1994). These results, coupled with the results of tumor promotion studies (*e. g*. Schwartz *et. al*., 1980; Jensen *et. al*., 1982, 1983, 1984; Jensen & Sleight, 1986; Rezabek *et. al*., 1987; Dixon *et. al*., 1988, as quoted in IPCS, 1994) indicate that these chemicals cause cancer by epigenetic mechanisms, involving both hepatic toxicity and hypertrophy, including cytochrome P-450 induction (IPCS, 1994).

Oral administration of hexabromobiphenyl was associated with adverse effects on reproductive parameters in a range of experimental animals (see Table 86 and 87 in IPCS, 1994). The most common adverse effects on reproduction were failure in implantation and decreases in pup viability of offspring. These effects were seen at a dose level of 28.6 mg/kg bw/day in a 15-day reproductive toxicity study in rats, with dosing between gestational day 0-14 (Beaudoin, 1979, as quoted in US ATSDR, 2004) and in mink at concentrations of 1 mg/kg diet (Aulerich and Ringer, 1979 as quoted in IPCS, 1994). Increased menstrual cycle duration and prolonged implantation bleeding were observed in female monkeys fed approximate daily dose levels of 0.012 mg/kg bw/day for 7 months before breeding and during pregnancy. Fetal deaths were also observed after approximately 1 year of exposure. Effects were attributed to decreases in serum progesterone (Lambrecht *et, al*., 1978; Allen *et, al*., 1978; 1979, (as quoted in US ATSDR, 2004).

Table 2.2 Pivotal toxicological studies on the toxicity of hexabromobiphenyl.

Species	Study type	Effect	LOAEL/ NOAEL
Rat	Short-term/acute toxicity 10 day repeat dose gavage study	decreased thyroid serum T4 hormones	3 mg/kg bw/day (LOAEL) 1 mg/kg bw/day (NOAEL)
Rat, Sprague Dawley	30-day dietary feeding study	increased number and decreased size of thyroid follicles	0.05 mg/kg bw/day (LOAEL)
Mice B6C3F1	In utero and post partum exposure from Gd 0-ppd 56	hepatocellular adenoma and carcinoma in offspring	1.5 mg/kg bw/day (LOAEL) 0.15 mg/kg bw/day (NOAEL)
Rhesus Monkey	25-50 wk dietary feeding study	34% weight loss in adult male, 0% weight gain in juvenile, proliferation of mucosal cells, chronic inflammation, severe ulcerative colitis, alopecia, keratinization of hair follicles and sebaceous glands, clinical chemical and hepatic changes	0.73 mg/kg bw/day (LOAEL, males)
Rat, Sprague Dawley	7 month dietary feeding study	decreased thyroid serum T3 and T4 hormones	0.45 mg/kg bw/day (LOAEL)
Monkey, Rhesus		increased menstrual cycle duration in 4/7;implantation bleeding in 2/7). 1/7 fetuses were aborted, 1/7 fetuses stillborn, 12% decreased birth weight and 22% decreased postnatal weight gain in 4/7 survivors	0.012 mg/kg bw/day (LOAEL)

Toxicity of hexabromobiphenyl in humans

Information on toxicological effects of PBBs (and by inference, hexabromobiphenyl) in humans has mainly been derived from the Michigan accident described in Section 2.1.4 of this draft Risk Profile (Carter (1976), Getty *et. al*., (1977), Kay (1977), Di Carlo *et. al*., (1978), Damstra *et. al*., (1982), Zabik (1982), and Fries (1985b), as quoted in EHC 152 (IPCS, 1994)). This accident resulted in widespread exposure of consumers for periods approaching 1 year, before the contamination of food by PBBs was identified and affected foodstuffs were removed from the food chain.

Adverse health effects reported included changes in liver enzymes, nausea, abdominal pain, loss of appetite, joint pain and fatigue (Anderson *et. al.*, 1978b, 1979, as reported in IPCS, 1994), together with reports of skin disorders, including acne and hair loss, in the period following the contamination. (IPCS, 1994). Similar skin disorders have also been reported in workers with occupational exposure to PBBs (Anderson *et. al.*, 1978a, as reported in IPCS, 1994), and also following exposure to the polychlorinated dioxins and furans.

Detailed epidemiological studies have been carried out on the health status of exposed individuals including immunological status, cancer incidence, reproductive effects and effects on development of young children. These studies have in the main failed to establish a definite link between any of these effects and exposure to PCBs, although some studies have reported decreased immune function in Michigan farm residents (Bekesi *et. al.*, 1979, 1987) and effects have also been reported on pubertal development in young females (see endocrine-disrupting effects below).

There are no reports of acute hexabromobiphenyl intoxication in humans, and there is also no consistent epidemiological evidence for hepatocarcinogenicity in exposed humans. A relationship between increasing serum levels (>2 ppb) of PBBs and increasing risk of breast cancer was indicated in case-control studies of women exposed during the Michigan contamination episode (Henderson *et. al.*, 1995; Hoque *et. al.*, 1998), but according to US ATSDR, 2004 (and quoted from this source) the results are only suggestive due to factors such as the small number of cases, insufficient information on known breast cancer risk factors, and confounding exposures to other organochlorine chemicals.

Effects on endocrine systems

The PBBs (and by inference, hexabromobiphenyl) are considered to have effects on endocrine systems. They have been evaluated under the EU-Strategy for Endocrine Disrupters[7] and have been placed in category 1 (evidence of endocrine-disrupting activity in at least one species using intact animals) in the priority list of chemicals established under the EU-Strategy. This categorisation is based on evidence of delayed vaginal opening in new-born rats, epidemiological evidence of hypothyroidism in workers exposed to polybrominated biphenyls and of increased incidence of breast cancer among women exposed to polybrominated biphenyls (as reported in BKH report, 2000). In an assessment (Blanck *et. al.*, 2000) of pubertal development in girls and young women exposed in utero and via breast milk to high levels of PBBs (>7pbb), it was found that this population had an earlier age to menarche than a similar breastfed population exposed to lower levels of PBBs, or than a highly-exposed population who were not breastfed. Earlier pubic hair development was also seen in the more highly exposed population, suggesting an effect of PBBs on pubertal events (Blanck *et. al.*, 2000).

[7] http://europa.eu.int/comm/environment/endocrine/strategy/substances_en.htm

Conclusion on effects assessment and toxicity of hexabromobiphenyl

Hexabromobiphenyl is readily absorbed into the body and accumulates following prolonged exposure. Although the acute toxicity of hexabromobiphenyl is low, a number of chronic toxic effects including hepatoxicity have been observed in experimental animals at doses around 1 mg/kg bw/day following long-term exposure, and effects are seen in the rat thyroid at doses as low as 0.05 mg/kg bw/day. Cancer was induced in animal studies at a dose of 0.5 mg/kg bw/day and the no-observed-effect level was 0.15 mg/kg bw/day. The International Agency for Research on Cancer has classified hexabromobiphenyl as a possible human carcinogen (IARC group 2B). The PBBs (and by inference, hexabromobiphenyl) are endocrine disrupting (ED) chemicals, and effects are seen on reproductive capacity in rats, mink and monkeys. Effects were seen in monkeys fed 0.012 mg/kg bw/day for 7 months before breeding and during pregnancy, the lowest effect level reported for hexabromobiphenyl in toxicology studies. There is epidemiological evidence of hypothyroidism in workers exposed to polybrominated biphenyls and of increased incidence of breast cancer in exposed women.

It can be concluded that hexabromobiphenyl is a bioaccumulative chemical with a range of potentially adverse effects on health, including carcinogenicity, reproductive toxicity, endocrine and other hormone-disrupting effects, at very low levels of exposure.

2.4.2 Ecotoxicity

Only few data are available on effects of PBBs on other organisms than mammals. Toxicity tests with technical decabromobiphenyl (Adine 0102) and bacteria (*Pseudomonas putida)* and the water flea *Daphnia magna* are quoted in EHS 152 (1994). The results were an EC10 of 53 mg/L for *Pseudomonas putida* (cell multiplication) and an EC50>66 mg/liter for *Daphnia magna* (immobilization, 24 hours). Because these concentrations exceed the solubility of HBB in water, the data may be of limited relevance to evaluating the environmental effects. However, the fact that the NOEC is reported to be <2 mg/L indicates that the water fleas were affected at the lowest concentration tested.

MacPhee & Ruelle (1969) and Applegate *et. al*., (1957), report results from short term tests with hexabromobiphenyl (CAS No. 36355-01-8) and several species of fish in the range 5-10 mg/L (Quoted from the Ecotox data base (US EPA, 2006)). These concentrations are also above the water solubility and may also be of limited environmental relevance.

In a field study on water birds, correlations between behavioural effects and reproductive success were not unambiguously correlated to body burdens of PBBs. (EHS 152 (IPCS, 1994)).

In an untraditional fish early life stage test, Hornung *et al*., (1996), injected halogenated organic contaminants into rainbow trout eggs. For 3,3',4,4',5,5'- hexabromobiphenyl they found an LD_{50} of 3,910 µg/kg. This result is not comparable to those of traditional fish tests, where exposure is via the water but it is comparable to results of other test with similar exposure. Hornung *et. al*. (1996), made such experiments to compare the toxicity of PBBs and PCBs and found that both 3,3',4,4'-tetrabromobiphenyl and 3,3',4,4',5,5'-hexabromobiphenyl were 10-fold more potent than identically substituted polychlorinated biphenyls.

Based on this, it seems to be relevant to expect the environmental toxicity of hexabromobiphenyl to be comparable to that of hexachlorobiphenyl.

3 SYNTHESIS OF THE INFORMATION

Hexabromobiphenyl belongs to a wider group of polybrominated biphenyls (PBBs). It has mainly been used as a fire retardant. Hexabromobiphenyl is already listed in Annex I of the UNECE Protocol on POPs.

According to available data, hexabromobiphenyl can be considered to be highly persistent in the environment. There is evidence of low or no degradation in water, soil and sediment, in the laboratory as well as in the field. Therefore, hexabromobiphenyl is considered to be highly persistent.

Hexabromobiphenyl is less volatile than many POP substances. However, extensive data on monitoring shows that it is found throughout the Arctic wildlife, demonstrating that it does have a high potential for long range environmental transport.

With measured weight-based BCF values in the range 4,700 - 18,100 and biomagnification factors in the aquatic food chain exceeding 100, hexabromobiphenyl is considered to be highly bioaccumulative and to have a high potential for biomagnification. These properties are demonstrated by several authors to be comparable to those of hexachlorobiphenyl (a PCB compound), for which the bioaccumulative properties are well documented.

Hexabromobiphenyl is readily absorbed into the body and accumulates following prolonged exposure. Although the acute toxicity of hexabromobiphenyl is low, a number of chronic toxic effects including hepatotoxicity have been observed in experimental animals at doses around 1 mg/kg bw/day following long-term exposure, and effects are seen in the rat thyroid at doses as low as 0.05 mg/kg bw/day. The International Agency for Research on Cancer has classified hexabromobiphenyl as a possible human carcinogen (IARC group 2B). The PBBs are endocrine disrupting chemicals, and effects are seen on reproductive capacity in rats, mink and monkeys. There is epidemiological evidence of hypothyroidism in workers exposed to polybrominated biphenyls and of increased incidence of breast cancer in exposed women. Data on toxicity to other species than laboratory mammals is scarce but suggests the environmental toxicity of hexabromobiphenyl is comparable to that of hexachlorobiphenyl.

Based on the available data, hexabromobiphenyl should be considered as a POP warranting global action.

Production and use of hexabromobiphenyl has ceased over the last decades but it cannot be excluded that it is still produced or used in some countries. In addition to emissions during manufacture or use, hexabromobiphenyl can enter the environment from the widespread use of flame-retarded products. A considerable part of the substance produced will probably reach the environment sooner or later because of the high stability of these compounds. Furthermore, some of these chemicals may form toxic polybrominated dibenzofurans during combustion processes.

4 CONCLUDING STATEMENT

It has been demonstrated that hexabromobiphenyl clearly meets all the criteria laid down in Annex D of the Stockholm Convention: It is very persistent in the environment. It has a great potential for bioaccumulation and in addition there is clear evidence of its biomagnification. Due to its physical and chemical properties and based on findings in environmental samples, it is verified that hexabromobiphenyl can be transported long distances in air, far from its sources. Hexabromobiphenyl is a possible human carcinogen and can also be regarded as a substance capable of disrupting the endocrine system.

As hexabromobiphenyl can travel in the atmosphere far from its sources, neither a single country nor group of countries alone can abate the pollution caused by this substance. Regional action has already been considered necessary and hexabromobiphenyl is totally banned under the Convention on Long-range Transboundary Air Pollution Protocol on Persistent Organic Pollutants. Although the production and use of hexabromobiphenyl seems to be ceased in most countries, its reintroduction remains possible. This could lead to increased releases and levels in the environment.

Based on the available data, hexabromobiphenyl is likely, as result of its long-range environmental transport, to lead to significant adverse human health and environmental effects, such that global action is warranted.

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ANNEX A

Table A.1 Concentrations of hexabromobiphenyl (PBB 153) in Arctic predators.

Year of sampling	Location	Species	Tissue	Concentration µg/kg lipid
1999-2002	East Greenland	Polar bear (*Ursus maritimus)* [1]	Blubber	33-44
1998	Faroe Islands	Fulmar (*Fulmarus glacialis*) [1]	Fat	16-26
2001	Faroe Islands	Pilot whale (*Globicephala melas*) [1]	Blubber	8.7-17
< 1987	Arctic Ocean	Guillemot (*Uria aalge*) [2]	Muscle	50 [6]
2002	East Greenland	Ringed seal (*Phoca hispida*) [1]	Blubber	0.34-0.42
1998-2002	West Greenland	Ringed seal (*Phoca hispida*) [1]	Blubber	n.d.
< 1987	Svalbard	Ringed seal (*Phoca hispida*) [2]	Blubber	4 [6]
1981	Svalbard	Ringed seal (*Phoca hispida*) [3]	Blubber	0.42
< 1988	Svalbard	Seal sp. [4]	? (mean)	0.8
1998	East Greenland	Minke whale (*Balaenoptera acutorostrata*) [1]	Blubber	0.56-1.2
1999-2001	Barents Sea	Arctic char (*Salvelinus alpinus*) [5]	Muscle	n.d.-52
1986	Lapland	Whitefish (*Coregonus sp.*) [2]	Muscle	0.29
2002	East Greenland	Shorthorn sculpin (*Myoxocephalus scorpius*) [1]	Liver	n.d.
2002	West Greenland	Shorthorn sculpin (*Myoxocephalus scorpius*) [1]	Liver	n.d.

n.d. = Not detected. Limits of detection are not well described in the references.

1: Vorkamp *et al*., 2004,
2: Jansson *et al*., 1987,
3: Jansson *et al*., 1993,
4: Krüger, 1988 (Quoted from EHC 152),
5: Evenset *et al*. 2005.
6: FireMaster(R) BP-6

Table A.2 Concentrations of hexabromobiphenyl (PBB 153) in biota, collected in subarctic and temperate regions outside the vicinity of Michigan.

Year of sampling	Location	Species	Tissue	Concentration µg/kg lipid
Aquatic species				
1979-85	Baltic Sea	Grey seal (*Halichoerus grypus*) [2]	Blubber	26
< 1987	Baltic Sea	Harbour seal (*Phoca vitulina*) [7]	Blubber	20
< 1987	~North Sea	Harbour seal (*Phoca vitulina*) [7]	Blubber	3
< 1987	Baltic Sea	Guillemot (*Uria aalge*) [7]	Muscle	160
1987-88	US mid Atlantic	Bottlenose dolphin (*Tursiops truncatus*) [8]	?	14-20
< 1999	North Sea	Whitebeaked dolphin (*Lage-norhynchus albirostris*) [10]	?	13 (wwt)
1987	S. Sweden	Arctic char (*Salvelinus alpinus*) [2]	Muscle	0.42
1986	Bothnian Bay	Herring (*Clupea harengus*) [2]	Muscle	0.092
1987	Baltic Proper	Herring (*Clupea harengus*) [2]	Muscle	0.16
1987	Skagerak	Herring (*Clupea harengus*) [2]	Muscle	0.27
< 1988	Germany	River fish (average) [1]	?	0.60
< 1988	Baltic Sea	Fish [1]	?	2.39
< 1988	North Sea	Fish [1]	?	1.31
1997	USA, Great Lakes	Lake trout (*Salvelinus nanay-cush*) (range of means) [6]	Whole fish	0.19-2.08
Predatory birds				
< 1987	Baltic Sea	White tailed sea eagle (*Hali-aeetus albicilla*) [7]	Muscle	280
1977	USA, 29 states	Bald eagle (*Haliaeetus leuco-cephalus*) [9]	Carcass	< 0.03 – 0.07 (wwt?)
1977	USA, 29 states	Bald eagle (*Haliaeetus leuco-cephalus*) [9]	Brain	< 0.03 – 0.05 (wwt?)
1982-86	S. Sweden	Osprey (*Pandion haliaeetus*), corpses [2]	Muscle	22
2003-2004	Belgium	7 species of predatory birds, corpses (range of medians) [3]	Muscle	2-35
2003-2004	Belgium	7 species of predatory birds, corpses (range of medians) [3]	Liver	2-43
1998-2000	Belgium	Little owl (*Athene noctua*) [5]	Unhatched eggs	1-6
1991-2002	Norway	6 species of predatory birds (range of medians) [4]	Unhatched eggs	0.2-9.4 µg/kg wwt
Terrestrial herbivores				
1986	S. Sweden	Rabbit (*Oryctlagus cuniculus*) [2]	Muscle	n.d.
1985-86	S. Sweden	Moose (*Alces alces*) [2]	Muscle	n.d.
1986	N. Sweden	Reindeer (*Rangifer tarandus*) [2]	Suet (fat)	0.037

n.d. = Not detected. Limits of detection are not well described in the references.

1: EHC 152 (IPCS, 1994), 2: Jansson *et al*. 1993, 3: Jaspers *et al*., 2006, 4: Herzke *et al*., 2005, 5: Jaspers *et al*., 2006, 6: Luross *et al*., 2002, 7: Jansson *et al*. 1987, 8: Kuehl *et al*. 1991 (quoted from US ATDSR, 2004), 9:Kaiser *et al*., 1980 (quoted from US ATSDR, 2004), 10: de Boer *et al*., 1999 (quoted from US ATSDR, 2004).

Table A.3. Summary of key toxicological studies on hexabromobiphenyl.

Species (test material)	Study type	Effect	LOAEL/ NOAEL	Ref.
Rat Fischer 344/N (FF-1)	Short-term/acute toxicity, 14-day repeat dose, 5 single daily doses per week	Body weight loss, emaciation, hepatotoxicity, renal & adrenal changes, atrophy of thymus; necrosis of splenic lymphoblasts)	1000 mg/kg/day (LOAEL)	Gupta and Moore 1979 (as quoted in US ATSDR, 2004).
Rat	Short-term/acute toxicity10 day repeat dose gavage study	decreased thyroid serum T4 hormones	3 mg/kg bw/day (LOAEL) 1 mg/kg bw/day (NOAEL)	Allen-Rowlands *et al*. 1981(as quoted in US ATSDR, 2004).
Rat, Sprague Dawley (BP-6)	30-day dietary feeding study	increased number and decreased size of thyroid follicles	0.05 mg/kg/day (LOAEL)	Akoso *et al*. 1982 (as quoted in US ATSDR, 2004).
Mouse B6C3F1 (FF-1)	Short-term/acute toxicity, 14-day repeat dose, 5 single daily doses per week	Hepatocyte enlargement and single-cell necrosis	0.3 mg/kg bw/day (NOAEL)	Gupta *et al*. 1981 (as quoted in US ATSDR, 2004).
Guinea Pig (PBB not specified)	30-day dietary feeding study	vacuolation and fatty changes in liver	0.04 mg/kg bw/day	Sleight and Sanger 1976, (as quoted in US ATSDR,2004).
Balb/c) Mouse (BP-6)	Short-term/acute toxicit, 10 day oral dietary study	suppressed antibody-mediated response to SRBC, thymic atrophy)	130 mg/kg bw/day (LOAEL)	Fraker and Aust 1978, (as quoted in US ATSDR, 2004).
Rat Fischer 344/N (FF-1)	6 month gavage study, 5 single daily doses per week	decreased lymphoproliferative responses and decreased delayed hypersensitivity responses)	3 mg/kg bw/day (LOAEL)	Luster *et al*. 1980 (as quoted in US ATSDR, 2004).
Rhesus Monkey (FF-1)	25-50 wk dietary feeding study	34% weight loss in adult male, 0% weight gain in juvenile, proliferation of mucosal cells, chronic inflammation, severe ulcerative colitis, alopecia, keratinization of hair follicles and sebaceous glands, clinical chemical and hepatic changes	0.73 mg/kg bw/day (LOAEL, males)	Allen *et al*. 1978; Lambrecht *et al*. 1978 (as quoted in US ATSDR, 2004).
Rat, Sprague Dawley (BP-6)	7 month dietary feeding study	decreased thyroid serum T3 and T4 hormones	0.45 mg/kg bw/day (LOAEL)	Byrne *et al*. 1987, (as quoted in US ATSDR, 2004).

Note: FF-1 and BP-6 in column 1 refer to FireMaster(R) FF-1 and FireMaster(R) BP-6, the PBBs used in the toxicity study described.

Table A.3 (continued) Summary of key toxicological studies on hexabromobiphenyl.

Species (test material)	Study type	Effect	LOAEL/ NOAEL	Ref.
Rat Fischer 344/N (FF-1)	25 wk gavage study, 5 single daily doses per week	gastric ulcers, decreased serum thyroid T4 hormone) hepatic, haematological disorders, thymic atrophy, progressive nephropathy	0.3 mg/kg bw/day (LOAEL) 0.1 mg/kg bw/day (NOAEL)	NTP 1983, (as quoted in US ATSDR, 2004).
Rat Sprague-Dawley Holtzman (FF-1)	4 week gavage study, 5 single daily doses per week	decreased motor activity	6 mg/kg bw/day (LOAEL) 3 mg/kg bw/day (LOAEL	Geller *et al*. 1979, (as quoted in US ATSDR, 2004).
Rat, Sprague Dawley (BP-6)	6 month gavage study, 5 single daily doses per week	delayed acquisition of locomotion and reduced open field activity in offspring).	2 mg/kg bw/day (LOAEL) 0.2 mg/kg bw/day (NOAEL)	Henck *et al*. 1994, (as quoted in US ATSDR, 2004).
Monkey, Rhesus (FF-1)		increased menstrual cycle duration in 4/7; implantation bleeding in 2/7). 1/7 fetuses were aborted, 1/7 fetuses stillborn, 12% decreased birth weight and 22% decreased postnatal weight gain in 4/7 survivors	0.012 mg/kg bw/day (LOAEL)	Lambrecht *et al*. 1978; Allen *et al*. 1978; 1979, (as quoted in US ATSDR, 2004).
Rat, Wistar (BP-6)	15-day reproductive toxicity study, dosing between gestational day 0-14	no implantations in 2/5 rats	28.6 mg/kg bw/day (LOAEL) 14.3 mg/kg bw/day (NOAEL)	Beaudoin 1979, (as quoted in US ATSDR, 2004).
Rat, Sprague Dawley	Gavage study in pregnant rats, dosing between gestional day 7-15	Reproductive: Delayed vaginal opening in pups	0.04 mg/kg bw/day (NOAEL)	Harris *et al*. (1978) (as quoted in BKH Final Report 2000)
Rat, Sprague Dawley (BP-6)	40 day dietary feeding study	Reproductive deficits in learning behavior in offspring, 6 months after prenatal and lactational exposure)	0.2 mg/kg bw/day (LOAEL)	Henck and Rech 1986, (as quoted in US ATSDR, 2004).

Note: FF-1 and BP-6 in column 1 refer to FireMaster(R) FF-1 and FireMaster(R) BP-6, the PBBs used in the toxicity study described.

Table A.3 (continued) Summary of key toxicological studies on hexabromobiphenyl.

Species (test material)	Study type	Effect	LOAEL/ NOAEL	Ref.
Rat, Fischer 344/N (FF-1)	6 month gavage study, 5 single daily doses per week dosages of 0, 0.1, 0.3, 1, 3, or 10 mg/kg/day	hepatocellular adenoma and carcinoma, cholangiocarcinoma (females only	3 mg/kg bw/day (LOAEL)	NTP 1983, (as quoted in US ATSDR, 2004).
Mice B6C3F1 (FF-1)	6 month gavage study, 5 single daily doses per week dosages of 0, 0.1, 0.3, 1, 3, or 10 mg/kg/day	hepatocellular adenoma and carcinoma	10 mg/kg bw/day (LOAEL)	NTP 1983, (as quoted in US ATSDR, 2004
Mice B6C3F1 (FF-1)	In utero and post partum exposure from Gd 0-ppd 56	hepatocellular adenoma and carcinoma in offspring	1.5 mg/kg bw/day (LOAEL) 0.15 mg/kg bw/day (NOAEL)	NTP 1992, (as quoted in US ATSDR, 2004).
Humans	Females accidentally exposed in the Michigan incident	relationship between serum PBBs and risk of breast cancer	relationship between serum PBBs of > 2 pbb and risk of breast cancer when compared with the reference group (<2 ppb),	Henderson *et al*. 1995, (as quoted in US ATSDR, 2004).
Humans	Michigan farm residents accidentally exposed in the Michigan incident	Significant reduction of in vitro immunological function		Bekesi *et at*. 1979, 1985 (as quoted in US ATSDR, 2004) Bekesi *et al*, 1987
Humans	Females accidentally exposed in the Michigan incident	Possible disturbance in ovarian function as indicated by menstrual cycle length and bleed length		Davis *et al*., 2005
Humans	Offspring of females accidentally exposed in the Michigan incident	breastfed girls exposed to high levels of PBB in utero had an earlier age at menarche	Effects at > or =7 ppb in breast milk	Blanck *et al*., 2000, (as quoted in US ATSDR, 2004)

Note: FF-1 and BP-6 in column 1 refer to FireMaster(R) FF-1 and FireMaster(R) BP-6, the PBBs used in the toxicity study described.

ANNEX B
HEXABROMOBIPHENYL ISOMERS

IUPAC Number[8]	Name	CAS Registry number[9].
	Hexabromobiphenyl	36355-01-8
128	2,2',3,3',4,4' hexabromobiphenyl	82865-89-2
129	2,2'3,3',4,5 hexabromobiphenyl	
130	2,2',3,3',4,5' hexabromobiphenyl	82865-90-5
131	2,2',3,3',4,6 hexabromobiphenyl	
132	2,2',3,3',4,6' hexabromobiphenyl	119264-50-5
133	2,2',3,3',5,5' hexabromobiphenyl	55066-76-7
134	2,2',3,3',5,6 hexabromobiphenyl	
135	2,2',3,3',5,6' hexabromobiphenyl	119264-51-6
136	2,2',3,3',6,6' hexabromobiphenyl	
137	2,2',3,4,4',5 hexabromobiphenyl	81381-52-4
138	2,2',3,4,4',5' hexabromobiphenyl	67888-98-6
139	2,2',3,4,4',6 hexabromobiphenyl	
140	2,2',3,4,4',6' hexabromobiphenyl	
141	2,2',3,4,5,5' hexabromobiphenyl	120991-47-1
142	2,2',3,4,5,6 hexabromobiphenyl	
143	2,2',3,4,5,6' hexabromobiphenyl	
144	2,2',3,4,5',6 hexabromobiphenyl	119264-52-7
145	2,2',3,4,6,6' hexabromobiphenyl	
146	2,2',3,4',5,5' hexabromobiphenyl	
147	2,2',3,4',5,6 hexabromobiphenyl	
148	2,2',3,4',5,6' hexabromobiphenyl	
149	2,2',3,4',5',6 hexabromobiphenyl	69278-59-7
150	2,2',3,4',5,6' hexabromobiphenyl	93261-83-7
151	2,2',3,5,5',6 hexabromobiphenyl	119264-53-8
152	2,2',3,5,6,6' hexabromobiphenyl	
153	2,2',4,4',5,5' hexabromobiphenyl	59080-40-9
154	2,2',4,4',5,6' hexabromobiphenyl	36402-15-0
155	2,2',4,4',6,6' hexabromobiphenyl	59261-08-4
156	2,3,3',4,4',5 hexabromobiphenyl	77607-09-1
157	2,3,3',4,4',5' hexabromobiphenyl	84303-47-9
158	2,3,3',4,4',6 hexabromobiphenyl	
159	2,3,3',4,5,5' hexabromobiphenyl	120991-48-2
160	2,3,3',4,5,6 hexabromobiphenyl	
161	2,3,3',4,5',6 hexabromobiphenyl	
162	2,3,3',4',5,5' hexabromobiphenyl	
163	2,3,3',4',5,6 hexabromobiphenyl	
164	2,3,3',4',5',6 hexabromobiphenyl	82865-91-5
165	2,3,3',5,5',6 hexabromobiphenyl	
166	2,3,4,4',5,6 hexabromobiphenyl	
167	2,3',4,4',5,5' hexabromobiphenyl	67888-99-7
168	2,3',4,4',5',6 hexabromobiphenyl	84303-48-0
169	3,3',4,4',5,5' hexabromobiphenyl	60044-26-0

(US ATSDR (2004)[10])

[8] Ballschmiter and Zell 1980
[9] From EHC 152 (IPCS, 1994).
[10] Note: the US ATSDR List does not include the two CAS numbers included in EHC 192 1997

別添8

商業用ペンタブロモジフェニルエーテルの危険性の概要

分解性	蓄積性	人健康影響	動植物への影響
【生分解性】 ・（Tetra, Penta, HexaBDE）難分解性（BIOWIN） ・（PentaBDE） 分解せず（OECD TG 301B で CO_2 発生なし） 【半減期】 ・大気中：11-19 日（EPIWIN） ・水中： 150 日（EPIWIN） ・土壌中：半減期 150 日（EPIWIN） ・好気性底質中： 600 日（EPIWIN） ・1970 年代初期にﾖｰﾛｯﾊﾟの海洋の底質に沈降した PentaBDE 同属体が現在も相当量存在しており、底質中での残留性が高いことが示されている。	【ｵｸﾀﾉｰﾙ/水分配係数】 log KOW=6.5-7.4 【BAF（経鰓及び経口による生物濃縮係数）】 ｾﾞﾌﾞﾗｶﾞｲ：BAF=1.8 【BMF（経口的生物濃縮係数）】 ・ｳﾐﾊﾞﾄ/ﾆｼﾝ：BMF=17 ・ﾊｲｲﾛｱｻﾞﾗｼ/ﾆｼﾝ：BMF=4.3 ・ｻｹ/ﾆｼﾝ：BMF=3.8 ・動物ﾌﾟﾗﾝｸﾄﾝ/底生生物：BMF=7.1 ・ﾎｯｷｮｸﾀﾞﾗ/動物ﾌﾟﾗﾝｸﾄﾝ：BMF=0.04-3.4 ・ﾜﾓﾝｱｻﾞﾗｼ/ﾎｯｷｮｸﾀﾞﾗ：BMF=13.7 ・ﾎｯｷｮｸｸﾞﾏ/ﾜﾓﾝｱｻﾞﾗｼ：BMF=0.3-11 ・多数の調査から、上位捕食者において懸念される濃度の PentaBDE が存在することが示されている。北極圏では、ﾜｼｶﾓﾒ、ﾎｯｷｮｸｸﾞﾏ、ﾜﾓﾝｱｻﾞﾗｼ、ｼﾛｲﾙｶなどの上位捕食鳥類および哺乳類中から高ﾚﾍﾞﾙの PentaBDE が検出されている ・土壌又は底質中のPentaBDEは、容易に食物連鎖に取り込まれ、人など食物連鎖上位者の脂肪組織中に生物濃縮する。	【反復投与毒性】 ラット（90 日）：NOEL 2mg/kg/day 未満 主な毒性は、肝臓肥大等（DE71） 【生殖毒性】 ラット（妊娠♀単回）：0.06mg/kg で児に自発行動変化（多動性） 0.3mg/kg で児に精巣体積・精子数の低値 （BDE99） 【催奇形性】 ラット（妊娠 6 日単回）：0.3mg/kg でばく露の母動物（F1）2 個体から得られた F2 児で、外観・骨格異常 （BDE99） 【その他】 実験動物で甲状腺ホルモン系への影響	【慢性毒性】 ミジンコ *Daphnia magna* ：繁殖阻害が認められた。

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Report of the Persistent Organic Pollutants Review Committee on the work of its second meeting

Addendum

Risk profile on commercial pentabromodiphenyl ether

At its second meeting, the Persistent Organic Pollutants Review Committee adopted the risk profile on commercial pentabromodiphenyl ether, on the basis of the draft contained in document UNEP/POPS/POPRC.2/7. The text of the risk profile, as amended, is provided below. It has not been formally edited.

K0653879 141206

PENTABROMODIPHENYL ETHER

RISK PROFILE

Adopted by the Persistent Organic Pollutants Review Committee
at its second meeting

November 2006

CONTENTS

Executive Summary

A substantial range of studies on pentabromodiphenyl ether has been identified and the findings summarised in this risk profile. The new findings reported here support the conclusion reached by the Persistent Organic Pollutants Review Committee in 2005 that PentaBDE's properties fulfill the screening criteria in Annex D of the Stockholm Convention.

Commercial pentabromodiphenyl ether (C-PentaBDE) refers to mixtures of bromodiphenyl ether congeners in which the main components are 2,2', 4,4'- tetrabromodiphenyl ether (BDE-47 CAS No. 40088-47-9) and 2,2',4,4',5-pentabromodiphenyl ether (BDE-99 CAS No. 32534-81-9), which have the highest concentration by weight with respect to the other components of the mixture.

Commercial pentabromodiphenyl ether mixtures (C-PentaBDE) are used for flame retardant purposes as additives in consumer products. The commercial mixtures contain brominated diphenyl ether congeners with three to seven bromines in the molecule, but molecules with four and five bromines predominate. The proportion of the different polybromodiphenyl ether (PBDE) congeners in C-PentaBDE varies in different regions of the world.

PentaBDE is released into the environment during the manufacture of the commercial product, in the manufacture of products containing PentaBDE, during their use and after they have been discarded as waste. Even though production of C-PentaBDE is phased out or being phased out worldwide, different products containing it will still be in use in several years to come, resulting in continuing releases to the environment. The products will in the end of their lifetime become wastes with the potential of additional releases.

The main source in North America and Western Europe has been the C-PentaBDE incorporated in polyurethane foam, used in domestic and public furniture. This use is now mainly phased out. The information is too limited to draw conclusions on the importance of other uses, like textiles, electrical and electronic products, building materials, vehicles, trains and aeroplanes, packaging, drilling oil fluid and rubber products. While some representative examples are covered, detailed information on use is lacking for many regions of the world.

Major releases to air are emissions from products during use, through volatilization of PentaBDE and dust-borne PentaBDE. Emissions of PentaBDE can also occur from recycling and dismantling activities such as dismantling of vehicles, buildings and constructions. Emissions can occur from electronic waste recycling plants and shredder plants. Potentially toxic products such as brominated dibenzo-*p*-dioxins and furans might be generated during incineration of articles containing C-PentaBDE.

The releases of PentaBDE are to air, water and soil, but the major part ends up in soil. The distribution between the environmental compartments is: soil>>>water>air. Several studies using sediment cores show that PentaBDE is very persistent in marine sediments, still occurring after 30 years. In the main, PentaBDE in the environment is bound to particles; only a small amount is transported in its gaseous phase or diluted in water but such transport over long periods can be effective in distributing the PentaBDE widely in the environment, especially into Arctic regions. Occurrence in the Arctic environment is demonstrated in several monitoring studies in air and biota.

Due to its high persistency in air, the main route for long-range transport of PentaBDE - as with so many substances that are sufficiently volatile, persistent and bioaccumulative - is through the atmosphere. Modelling and environmental studies indicate that the transport is through a series of deposition/volatilization hops towards the poles but particulate transport is known to be important, too. Long-range transport through water and emigrating animals is also likely.

Several studies show that PentaBDE in soil and sediments is bioavailable, enters the food chain and that it bioaccumulates and biomagnifies in the food webs, ending up in high levels in top predators.

PentaBDE is widespread in the global environment. Levels of components of C-PentaBDE have been found in humans in all UN regions. Most trend analyses show a rapid increase in concentrations of PentaBDE in the environment and in humans from the early 1970s to the middle or end of the 1990s, reaching plateau levels in some regions in the late 1990s, but continuing to increase in others. The levels in North America and the Arctic are still rising. Vulnerable ecosystems and species are affected, among them several endangered species. Some individuals of endangered species show levels high enough to be of concern. Toxicological studies have demonstrated reproductive toxicity, neurodevelopmental toxicity and effects on thyroid hormones in aquatic organisms and in mammals. The potential for the toxic effects in wildlife, including mammals, is evident.

Potential exposure to humans is through food, and through use of products and contact with indoor air and dust. PentaBDE transfers from mothers to embryos and lactating infants. A Canadian assessment of risk quotients suggests that the highest risks accrue to species high in the food chain. Information is lacking on the effects in humans of short-term and long-term exposure, although it is to be expected that vulnerable groups can be pregnant women, embryos and infants. Considerably higher levels are found in humans from North America in general. About 5% of general populations have been found to be subjected to elevated exposure. This, together with the estimates of the long half-life of PentaBDE congeners in humans, raises concern for long-term effects on human health.

Based on the information in this risk profile, PentaBDE, due to the characteristics of its components, is likely, as a result of long-range environmental transport and demonstrated toxicity in a range of non-human species, to cause significant adverse effects on human health or the environment, 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. Norway, which is a Party to the Stockholm Convention, submitted a proposal in January 2005 to list pentabromodiphenyl ether in Annex A to the Stockholm Convention, and the POPRC agreed that the commercial product 'pentabromodiphenyl ether' ('PentaBDE') – actually a mixture as described below - met the screening criteria of Annex D to the Convention.

1.1 Chemical identity of the proposed substance

Commercial pentabromodiphenyl ether (C-PentaBDE) refers to mixtures of bromodiphenyl ether congeners in which the main components are 2,2', 4,4'- tetrabromodiphenyl ether (BDE-47 CAS No. 40088-47-9) and 2,2',4,4',5-pentabromodiphenyl ether (BDE-99 CAS No. 32534-81-9), which have the highest concentration by weight with respect to the other components of the mixture.

The numbering system for the PBDEs is the same as that used for polychlorobiphenyls (PCBs) (Ballschmiter *et al.* 1993).

The acronym PBDE is used for the generic term polybromodiphenyl ether, covering all congeners of the family of brominated diphenyl ethers. It is sometimes abbreviated to BDE.

1.2 Conclusion of the Review Committee regarding Annex D information

The Committee has evaluated Annex D information at its first meeting in Geneva in November 2005 (UNEP/POPS/POPRC.1/10) and has concluded that the screening criteria have been fulfilled for C-PentaBDE (Decision POPRC-1/3).

1.3 Data sources

This risk profile is elaborated using Annex E information submitted by countries and nongovernmental organizations, national reports from web sites for environment protection agencies in different countries, contact and submissions from Norwegian research institutes, the bromine industry, EMEP and AMAP.

Eleven countries have submitted information (Australia, Brazil, Canada, Japan, Norway, Mexico, Poland, Republic of Lebanon, Spain, Switzerland and United States of America). Seven countries submitted information on production and use. Only one country submitted information on releases; another reported that they did not have release data. All except one country provided monitoring data. There was no information on stock-piles from submitting countries and only a few have submitted information on trade. Two observers submitted information - World Wide Fund for Nature (WWF) and the International POPs Elimination Network (IPEN).

1.4 Status of the chemical under other international conventions

1.4.1 The OSPAR Convention

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the OSPAR Convention) is guiding international cooperation on the protection of the marine environment of the North-East Atlantic. The OSPAR Convention was signed in Paris in 1992 and entered into force on 25 March 1998. The OSPAR Commission is made up of representatives of the Governments of 17 Contracting Parties and the European Commission, representing the European Community. In 1998, the OSPAR Commission placed PBDEs on its “List of Chemicals for Priority Action.” An OSPAR

Commission background document on PBDEs was reviewed by Sweden in 2001. The next full review of this document is not planned before 2008. At the 4th North Sea Conference, it was decided to phase out the use of brominated flame retardants by 2020.

1.4.2 The UNECE Convention on Long-range Transboundary Air Pollution

United Nations Economic Commission for Europe (UNECE) works for sustainable economic growth among its 55 member countries. The UNECE Convention on Long-range Transboundary Air Pollution was signed by 34 Governments and the European Community in 1979 in Geneva. Under it, Parties shall endeavour to limit and, as far as possible, gradually reduce and prevent air pollution including long-range transboundary air pollution. It entered into force in 1983 and has been extended by eight specific protocols. There are today 50 countries that are parties to the Convention. The Protocol for persistent organic pollutants (POPs) was adopted on 24 June 1998 in Aarhus (Denmark). It focuses on a list of 16 substances that have been singled out according to agreed risk criteria, for total ban, elimination at a later stage or restrictive use. C-PentaBDE was nominated as a new POP to the Convention in 2004 by Norway. In December 2005 it was considered by the Executive Body of the Convention to meet the screening criteria for POPs, set out in EB decision 1998/2. They requested that the UNECE Task Force on POPs continue with the review and fuether explore management strategies..

1.4.3. The Rotterdam Convention

The Rotterdam Convention is a multilateral environmental agreement designed to promote shared responsibility and cooperative efforts among Parties in the international trade of certain hazardous chemicals. It is an instrument to provide importing Parties with the power to make informed decisions on which chemicals they want to receive and to exclude those they cannot manage safely.

The text of the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade was adopted at the Diplomatic Conference held in Rotterdam on 10 September 1998. The Convention entered into force on 24 February 2004 and became legally binding for its Parties. Today there are 102 states that are parties to the Convention. The EU notified PentaBDE to the Rotterdam Convention in 2003. For it to become a candidate, bans of the substance must be notified by two parties under the Convention.

1.4.4 Other international forums of relevance

The Arctic Council is a high-level intergovernmental forum that provides a mechanism for addressing the common concerns and challenges faced by the Arctic governments and the people of the Arctic. Member states are Canada, Denmark (including Greenland and the Faeroe Islands), Finland, Iceland, Norway, Russia, Sweden and United States of America. Six international organizations representing many Arctic indigenous communities have the status of Permanent Participants of the Arctic Council.

Significant monitoring and assessment of pollution in the Arctic is performed under the auspices of the Arctic Council (The Arctic Monitoring and Assessment Programme, AMAP). This work is important in identifying pollution risks, their impact on Arctic ecosystems and in assessing the effectiveness of international agreements on pollution control, such as the Stockholm Convention on Persistent Organic Pollutants (POPs). AMAP has shown that PentaBDE is one of the important pollutants of the Arctic.

In the autumn of 2004, the Arctic Council adopted a new Arctic project concerning the reduction of brominated flame retardants. The project will be managed by Norway.

2. Summary information relevant to the risk profile

2.1 Sources

2.1.1. Production and use

Based on the last information on total market demand of C-PentaBDE presented at the Bromine Science and Environmental Forum (BSEF), the estimated cumulative use of C-PentaBDE since 1970 was 100 000 metric tons (tones). The total market demand decreased during the later years of this period, for example from 8,500 tons in 1999 to 7,500 tons in 2001 (BSEF, 2001).

Table 2.1. C-PentaBDE volume estimates: Total market demand by region in metric tons (BSEF, 2001).

	America	Europe	Asia	Rest of the world	Total
1999	8,290	210	-	-	8,500
2001	7,100	150	150	100	7,500

These consumption figures need to be seen in the context of the global demand for polybrominated flame retardants of all types, which vastly outweighs the demand for C-PentaBDE. Thus, world totals of PBDE were 204,325 (1999), 203,740 (2001), 237,727 (2002) and 223, 482 (2003) tonnes (BSEF 2006).

C-PentaBDE has been produced in Israel, Japan, U.S. and EU (Peltola *et al.* 2001 and van der Goon *et al.* 2005). Since 2001 actions to regulate or voluntarily phase-out C-PentaBDE have been conducted in several countries.

Production in EU ceased in the former EU (15) in 1997 (EU 2000). Usage in the EU (15) has been declining during the second half of the 1990s and is estimated to be 300 metric tonnes in 2000 (used solely for polyurethane production) (EU 2000). The use of PentaBDE was banned in the EU (25) in 2004. Use in electrical and electronic appliances ceased on 1 July 2006.

In the United States of America, in June 2006, the U.S. Environmental Protection Agency (EPA) issued a significant new use rule on tetra-octaBDE and any combinations of these chemicals resulting from a chemical reaction, which requires persons to notify EPA before commencing manufacture or import for any use. C-PentaBDE will be banned in the state of California from 2008. The sole US manufacturer voluntarily ceased production, but use may be continuing and will cease only when stocks are fully exhausted. Although a patent on production of C-PentaBDE was taken out in China as recently as 1999 for a PBDE mixture that differs from the traditional penta-mix, the substance is being phased out in that country. Remaining production in China is estimated as less than 100 MT/year and will cease in 2007 when the substance is banned in that country.

A major bromine producer in Israel, Israel Chemicals and Industrial Products (formerly the Dead Sea Bromine Group), declares in a public statement on its web site that their products do not contain PentaBDE. This aligns the producer with the ban in the EU, which is an important market for the company's flame retardants.

There is today no production in Japan. The use of C-PentaBDE was voluntarily withdrawn from the Japanese market in 1990 (Kajiwara et al. 2004). Some developing countries around the East China Sea are potential “hot spots” releasing PentaBDE into the marine environment (Ueno *et al.* 2004). Many industrial manufacturers of computers, television sets and other electric household equipment are situated in the coastal areas of Asian developing countries (Ueno *et al.* 2004). There are indications on a phase-out of C-PentaBDE in manufacture of new electrical and electronic products in the Asian

region, although uses there were always subsidiary to the major uses in polyurethane foams. The extent of this is uncertain. Waste electric products used in developed countries have been exported to Asian developing countries, such as China, India and Pakistan. This waste material has been recycled for recovery of valuable metals (Ueno *et al.* 2004) and continuation of this trade can remain a source to PentaBDE releases. No restrictions have so far been implemented in developing countries in the Asia Pacific and the southern hemisphere.

The release of 'banked' PentaBDE during recycling of foam products has its parallel in the release of CFCs and other ozone depleting substances which have similarly remained in the foam during its useful lifetime.

Results from a survey of Canadian industries regarding certain substances on the country's Domestic Substances List conducted for the year 2000 indicated that no PBDEs were manufactured in Canada, but approximately 1300 tonnes of C-PentaBDE (for incorporation into finished articles) was imported into the country (Environment Canada 2003). Based on quantities reported, C-PentaBDE was the PBDE imported in greatest volume, followed by the commercial decabromodiphenyl ether product. A very small amount of octabromodiphenyl ether was imported in 2000. The volumes reported do not include quantities imported in finished articles. In 2004, it was proposed that PentaBDE be added to the Virtual Elimination list in Canada.

In the U.S. the sole producer voluntarily ended their production of C-PentaBDE in 2004. In 2001 alone, almost 70,000 metric tons of PBDEs were produced globally, almost half of which was used in products sold in the US and Canada. Before the phase-out in U.S. the majority of C-PentaBDE formulation produced globally was used in North America (>97 %). At the end of 2004 in the US, approximately 7.5% of the more than 2.1 billion pounds of flexible polyurethane foam produced each year in the US contained the C-PentaBDE formulation (Washington State 2005).

In Australia in 2004, the National Industrial Chemicals Notification and Assessment Scheme (NICNAS) advised that all importers were phasing out imports of PentaBDE by the end of 2005, and this was reconfirmed by the major importers in mid-2005.

C-PentaBDE is used or has been used in the following sectors (Alaee *et al.* 2003, Danish EPA 1999, EU 2000, Prevedouros *et al.* 2004b, Swiss Agency for the Environment 2002, Birnbaum and Staskel, 2004):

- Electrical and electronic appliances (EE appliances) – computers, home electronics, office equipment, household appliances and other items containing printed circuit laminates, plastic outer casings and internal plastic parts such as small run components with rigid polyurethane elastomer instrument casings.
- Traffic and transport – cars, trains, aircraft and ships containing textile and plastic interiors and electrical components.
- Building materials – foam fillers, insulation boards, foam insulation, piples, wall and floor panels, plastic sheeting, resins etc.
- Furniture – upholstered furniture, furniture covers, mattresses, flexible foam components.
- Textiles – curtains, carpets, foam sheeting under carpets, tents, tarpaulins, work clothes and protective clothing.
- Packaging – polyurethane foam based packaging materials.

The most common use, accounting for 95-98% of C-PentaBDE since 1999, has been in polyurethane foam (Hale *et al.* 2002). This foam may contain between 10 and 18% of the C-PentaBDE formulation. Polyurethane foam is mainly used for furniture and upholstery in domestic furnishing, automotive and

aviation industry. Other uses are in rigid polyurethane elastomers in instrument casings, in epoxy resins and phenolic resins in electrical and electronic appliances, and construction materials. For some years now, the more highly brominated Deca-BDE has been preferred in these applications. C-PentaBDE has also been incorporated in minor amounts in textiles, paints, lacquers, in rubber goods (conveyer belt, coating and floor panels) and in oil drilling fluids. Levels range from 5-30% by weight. Up to the early 1990s, C-PentaBDE was used in printed circuit boards, usually FR2 laminates (phenolic resins) in Asia. Such FR2 laminates are used in household electronics (television, radio, video), vehicle electronics, white goods (washing machines, kitchen appliances, for example). In the early 1990s the amount C-PentaBDE used in textile treatment was 60 % of total use in the EU, but this application is now banned.

C-PentaBDE has been identified as an additive flame retardant in textiles in national substance flow analyses in the ECE region (Danish EPA 1999). Manufacturers of furniture textiles have stated that the textile contained 0.45% PentaBDE in a Norwegian flow analysis reported in 2003. Stringent rules on flammability apply to textiles used in the public sector, the transport sector and business sector, but rules for domestic use are less consistent.

According to information obtained from the bromine industry the use of C-PentaBDE as hydraulic fluid (as a component of a mixture) in petroleum borings and mining was discontinued 10-20 years ago.

Australia has reported uses in manufacture of polyurethane foams for refrigerators and packaging, and in epoxy resin formulations supplied into aerospace market and for use as potting agents, laminating systems and adhesive systems. The US has reported use of C-PentaBDE in the aircraft industry. There is no use of C-PentaBDE in newer aircraft, and thus no exposure of the public, but C-PentaBDE is still used in military aircraft.

2.1.2 Global demands for brominated flame retardants in the future

According to a market analyst consulting company, the global demand for flame retardants is expected to grow at 4.4% per year, reaching 2.1 million metric tons in 2009, valued at $4.3 billion. Growth will largely be driven by gains in developing countries in Asia (China, in particular), Latin America and Eastern Europe. Strong increases are forecast for most of the flame retardants. Globally, demand will be greatest for bromine compounds, due mainly to strong growth in China. Electrical and electronic uses will grow fastest. Higher value products will continue to make inroads as substitutes for less environmentally friendly compounds, especially in Western Europe, and chlorine compounds will begin to be replaced in China by bromine- and phosphate-based and other flame retardants (Fredonia Group 2005).

After a severe falloff in demand in 2001, electrical and electronic applications will continue to recover. Demand growth for flame retardants will be strongest in such applications. As electronic circuits become smaller, and more densely packed electronics are subjected to ever higher temperatures, the need for flame retardants will increase. Construction markets will be the second fastest growing globally, but in China second place will be held by motor vehicles, followed by textiles, both of which industries are growing rapidly in that country. Plastics will continue to replace other materials such as metals and glass in a wide range of products, in order to lower both cost and weight and to allow improved design and more flexible production. Plastic usage is already widespread and growing in fields such as transportation, building products and electronics. Plastics must be made flame retardant for many applications, and as a result some 75% of all flame retardants are used in plastics (Fredonia Group 2005).

Environmental restrictions vary by region. In Western Europe, Japan and to a lesser extent in North America, such restrictions will especially limit growth of chlorinated compounds. A ban on some brominated flame retardants in Western Europe is not expected to spread substantially to other regions, but it will drive the development of alternatives in electrical and electronic equipment for sale on the world market. Dozens of Asian, European and US companies announced in 2005 that they have

developed or are developing electrical and electronic equipment that does not contain C-PentaBDE. In Asia, 51% of electronic manufacturers already make products compliant with the ban on PentaBDE in the EU, and 42% expected to have products that are compliant by 1 July 2006. Officials from electronics companies and industry consultants expected that the difficulty of keeping product streams separate would ensure that most electronic equipment sold on the world market would be compliant by 2005 (International Environment Reporter 2006).

2.1.3 Releases to the environment during production

PentaBDE is released into the environment during the manufacturing process, in the manufacture of products, during their use and after they have been discarded as waste. In addition to working towards a manufacturing process that does not cause emissions, it is also important to consider the contributions of emissions from products during use as well as after they have been discarded. Most of the PentaBDE is released as diffuse pollution during and after the service life of articles incorporating C-PentaBDE and as small-scale point source pollution from the waste management chain of the end products.

PentaBDE is synthesised from diphenyl ether by brominating it with elemental bromine in the presence of a powdered iron Friedel-Craft catalyst. The producers of PentaBDE have reported that the major routes of PentaBDE from this process to the environment are filter waste and rejected material, both of which are disposed of in landfills. Waste water releases of PentaBDE may also occur from spent scrubber solutions (Peltola *et al.* 2001).

According to the EU risk assessment of PentaBDE, the emissions in polyurethane production are assumed to occur prior to the foaming process, when handling the additives (discharges to water) and during the curing (emissions to air). Releases to air may occur during the curing phase of foam production, during which the foam stays at elevated temperature for many hours, depending on the production block size. Emission to air at this stage is estimated to be 1 kg/tonne PentaBDE, but it is assumed that some of the volatilized PentaBDE condenses in the production room and ends up in the waste water. The EU risk assessment concludes that 0.6 kg of PentaBDE is released in this way, and 0.5 kg into air, for each tonne of C-PentaBDE used in polyurethane foam production.

Table 2.2 Global production and use of C-PentaBDE in polyurethane foam production, and estimation of associated releases in 2000 (foam containing 10-18% PentaBDE).

Polyurethane foam production	**Quantity of PentaBDE**	**Release of PentaBDE into waste water**	**Emissions of PentaBDE to air during production**
150,000 tonnes/year	15,000-27,000 tonnes/year	9,000-16,200 kg/year	7,500-13,500 kg/year

An important source of release has been associated with the use of liquid flame retardant additives such as C-PentaBDE in production of polymer foams. Approximately 0.01% (that is, 100 g /tonne) of the raw material handled during mixing is estimated to be released to wastewater. There is also potential for release due to volatilization during the curing phase as described above, since foam reaches temperatures of 160^{o}C for several hours. Wong *et al.* (2001) examined the atmospheric partitioning characteristics of BDEs 47, 99 and 153, and predicted that tetra- and pentabromo-congeners will become gaseous at warmer air temperatures. Therefore, although the low measured vapour pressure values for the PBDEs indicate that volatilization is minimal at normal air temperatures, there is potential for release to air at the elevated temperatures reached during curing (European Communities 2001). The European Communities (2001) study estimates the overall release of PentaBDE to be approximately 0.11%, with about one half of this going to air and the other half to wastewater.

2.1.4 Releases to the environment during product use

C-PentaBDE is used solely as an additive in physical admixture with the host polymer, and can thus migrate within the solid matrix and volatilize from the surface of articles during their life cycle (EU 2000). Approximately 3.9 % of the PentaBDE present in articles was estimated to be released each year through volatilization during their assumed service life of 10 years in the EU risk assessment, but each congener will have its own characteristic migration and volatility coefficients. Based on the quantities of shown in Table 2.2, and the 3.9% loss rate, it can estimated that 585-1053 tonnes of PentaBDE enters the environment in this way each year.

Wilford *et al.* (2003) conducted controlled chamber experiments in which they passed air through samples of C-PentaBDE -treated foam products containing 12% PBDE w/w. They found that PBDEs volatilized from polyurethane foam at measurable levels. Average total PBDE levels of 500 ng/m^3/g foam were released from the chamber. For BDE-47, BDE-99 and BDE-100 (4,5 and 5 bromines, respectively), the loss rates were 360, 85 and 30 ng/m^3/g foam, respectively. The average temperature range during sampling was 30-34^{o}C.

Given the use of C-PentaBDE in domestic items such as furniture, carpeting and appliances, exposure to indoor air house dust containing PentaBDE has been examined in a number of studies (Shoeib *et al.* 2004, Wilford *et al.* 2005). US researchers (Stapleton *et al.* 2005) report results for a study conducted in 2004 in the Washington, DC, metropolitan area and one home in Charleston, South Carolina. The concentrations of PBDEs in house dust from sixteen homes ranged from 780 ng/g dry mass to 30,100 ng/g dry mass. The dominant congeners were those associated with C-PentaBDE and DecaBDE. It was estimated that young children (1-4 years) would ingest 120-6000 ng/day of PBDEs. For five of the homes, clothes dryer lint was also analyzed, showing PBDE concentrations of 480-3080 ng/g dry mass. The exposures are higher than those observed in Europe, a fact that the researchers attribute to the fact that most markets for C-PentaDBE has been in the United States.

The information in the preceding paragraph highlights the fact that while PentaBDE can volatilize from the products in which it is incorporated, as well as during their whole life-cycle, and during recycling or after disposal, a major route for dissemination of this chemical into the environment will be in the form of particles on which it is absorbed or adsorbed. When emitted from products, the flame retardants are likely to adsorb to particles, and these may adhere to surfaces within appliances or on other surfaces in the indoor environment, or they may spread to the outdoor environment during airing of rooms. Industrial environments where equipment is dismantled may suffer much higher exposures (Danish EPA 1999). There are also releases from products due to weathering, wearing, leaching and volatilization at the end of their service life during disposal or recycling operations (dismantling, grinding or other handling of waste, transport and storage, for example). The annual releases in the EU region from the product life-cycle of polyurethane products were estimated to be distributed among the different compartments as follows: 75% to soil, 0.1% to air and 24.9% to surface water (EU 2000).

The inclusion of C-PentaBDE in materials used for car undercoating, roofing material, coil coating, fabric coating, cables, wires and profiles, and shoe soles can result in slow release to the environment. Emission factors for such releases in the EU risk assessment were judged to be 2-10% during the lifetime of the product, with the higher factors applying to uses with high wear rates such as car undercoating and shoe soles. A further 2% was assumed to be emitted during disposal operations. Taking these into account, the losses in the EU region were estimated to be 15.86 tonnes/year to soil, 5.26 tonnes/year to surface water, and 0.021 tonnes/year to air. No actual measurements were found in the literature with which one might compare these estimates.

Hale *et al.* (2002) demonstrated that flame-retardant treated polyurethane foam exposed to direct sunlight and typical Virginia summer conditions with temperatures up to 30-35^{o}C and humidity of 80% or greater, became brittle and showed evidence of disintegration within four weeks. The authors postulate that the resulting small, low density foam particles would be readily transportable by

stormwater runoff or air currents. Such degradation processes may provide an exposure route to organisms via inhalation or ingestion of the foam particles and their associated PentaBDE.

2.1.5 Emissions from waste containing C-PentaBDE

Waste can be generated from production of C-PentaBDE, from processes for manufacture of C-PentaBDE -containing materials, and from end-of-service-life management of products containing PentaBDE.

In production, the C-PentaBDE producers have stated that the major source of release was from filter waste and reject material, but quantities are small to negligible. In general, the waste was disposed of to landfill (EU 2000), although it is noted that waste containing more than 0.25% PentaBDE is classified as 'hazardous waste'.

After curing and cooling, blocks of polyurethane foam generally have to be cut to the required size, although for some applications the foam is produced in a mould of the desired shape so cutting is not required. Some flame retardant is lost in the scrap foam that results from the cutting process. Such foam scrap is often recycled into carpet underlay (rebond), particularly in the United States. Interestingly, the EU exports about 40,000 tonnes/year of scrap foam to the US for such use (EU 2000). In other uses, scrap foam is ground and used as filler in a number of applications such as cars eats or used for addition to virgin polyol in slab foam production. It is also possible that some foam scrap will be disposed of to landfill, or even incinerated.

During the production of printed circuit boards a substantial part of the laminate is cut off and becomes solid waste. In most countries, however, C-PentaBDE is no longer used in this application. There is limited information about waste generated in other applications of C-PentaBDE, such as its use in electrical and electronic appliances. While some such appliances are recycled on account of their metal content, many are burned in municipal waste incinerators and this often the fate of non-metallic portions of this waste stream. In the EU, from December 2006, plastics containing brominated flame retardants must be separated from such waste prior to recovery and recycling.

Used vehicles, often containing solid or foam components with C-PentaBDE are stored outdoors and then dismantled in shredder plants. In some countries, restrictions require that components containing substances like PentaBDE be treated as hazardous waste. Wastes generated from production of building materials, textiles and furniture are disposed of in landfills, or incinerated. This is easy enough for small, easily dismounted components, but most material containing flame retardants is harder to segregate and so these materials end up in the waste from shredder plants and are usually landfilled.

Movement of polymer foam particles containing PentaBDE within the landfill could provide a mechanism for transport of the brominated material to leachate or groundwater. It is not currently possible to assess the significance of such processes. However, given the physico-chemical properties of the substance, it is considered unlikely that significant amounts of PentaBDE will leach from landfills, since it has low water solubility, high octanol-water partition coefficient, and adsorbs strongly to soils (EU 2000). Norwegian screening studies have found levels of PentaBDE of concern in landfill leachates (Fjeld *et al.* 2003, Fjeld *et al.* 2004, Fjeld *et al.* 2005). The quantity of PentaBDE disposed of annually in the EU, and going to landfill or incineration, is estimated to be approximately 1,036 tonnes (EU 2000).

In a Dutch project, the emissions of PentaBDE in the EMEP region were estimated and distribution between sources was as follows: 0.33 tonnes/year from industrial combustion and processes, 9.45 tonnes/year from solvent and product use and 0.05 tonnes/year from waste incineration (van der Gon *et al*. 2005).

At the operating temperatures of municipal waste incinerators almost all flame retardants will be destroyed, but based on experience with other organic compounds, trace amounts could be passing the combustion chamber (Danish EPA 1999). Studies of recipients to municipal solid waste incinerators have detected above-background levels of PentaBDE in both gaseous and particulate fractions in the air in the vicinity of the facility (Agrell *et al.* 2004, Law 2005, ter Schure *et al.* 2004b). Potentially toxic products like brominated dibenzo-*p*-dioxins and dibenzofurans may be produced during incineration of articles containing C-PentaBDE (Danish EPA 1999, Ebert and Bahadir 2003, Weber and Kuch 2003, Birnbaum and Staskel 2004) and possibly released to the environment.

Analyses of dismantled FR2 circuit boards in electrical scrap show that about 35% of the PBDE used was PentaBDE, and for estimation purposes it was assumed that 25% of FR2 laminates in older appliances had been treated with the C-PentaBDE (Swiss agency 2002). Prevedouros *et al.* (2004) estimated production, consumption, and atmospheric emissions of PentaBDE in Europe between 1970 and 2000 based on literature data. According to that study, the flow of PentaBDE in discarded electrical and electronic appliances in Europe is in the range 17-60 metric tons per year for the period 2000-2005. However, a Swiss experimental study of such flow in a modern recycling plant showed values higher than expected on the basis of the literature study. This could mean that the literature has under-estimated the PBDE content of such appliances, and the study acknowledges that companies seldom provide all the information necessary to make accurate estimates (Swiss agency 2002). This same study reported a flow analysis for the life cycles of Penta-, Octa- and Deca-BDE as well as tetrabromobisphenol A (TBBPA). Waste electrical and electronic equipment was the biggest contributor, ahead of automotive shredder residues and construction waste. The plastics in vehicles produced in 1980 contained 0.089 g/kg of PentaBDE (excluding that contained in electrical and electronic components), whereas plastic in those built in 1998 had 0.044 g/kg. At the beginning of this period, almost all unsaturated polyurethane resins were treated with brominated flame retardants, primarily DecaBDE and TBBPA, but also PentaBDE. Even larger quantities, up to 50 g PentaBDE/kg of resin, were used in rail vehicles produced in 1980.

The average concentration of PentaBDE in appliances is estimated to be 34 mg/kg, with the highest concentration – 125 mg/kg – in the plastic fraction (Morf *et al.* 2005). In plants with off-gas filtering, a large proportion of the PentaBDE will be found in the collected fraction (Morf *et al.* 2005). On the other hand, in a facility without an efficient air pollution control device such as that in the modern facility studied, a significant flow of dust-borne PentaBDE may be released to the environment. A case in point was presented by Wang *et al.* (2005), who detected levels of PentaBDE in soil and sediment collected in the vicinity of an open electronic waste disposal and recycling facility located in Guiyu, Guandong, China.

The Swiss study showed that 5% of polyurethane foams produced in 1990 were used in the building industry, and contained up to 220 g/kg of C-PentaBDE. About 10-20% of the thermoplastic sheeting used in construction was treated with brominated flame retardants at levels of 1.3-5% by weight (Danish EPA) but no information about C-PentaBDE content is available. Some polyvinyl chloride sheeting would also have been treated with C-PentaBDE, typically at 49 g/kg. PentaBDE can be assumed to be emitted during dismantling activities but no information is available about the extent of such emissions.

2.2 Environmental fate

2.2.1 Persistence

Estimated half-life values of PDBE in different environmental compartments are scarce in the literature. In table 2.3 half-life estimates found in literature are summarized.

*Table 2.3 Half-lives of PentaBDE (**BDE-99**) in different environmental compartments, estimated with the use of Syracus Corporation's EPIWIN program.*

Environmental compartment	Half-life estimate (d)	References
Soil	150	Palm 2001, Palm *et al.* 2002
Aerobic sediment	600	Palm 2001, Palm *et al.* 2002
Water	150	Palm 2001, Palm *et al.* 2002
Air	19 11	Palm *et al.* 2002 Vulykh *et al.* 2004

It is noted that caution should be used in relying on half-life estimates derived from this program, now called EPI Suite (http://www.epa.gov/opptintr/exposure/docs/episuite.htm). The EPI Suite's intended use is chemical screening only and may not be appropriate for consideration of substances for global control. Because of interest in this matter, it is likely that half-life data from new studies will be published but the picture provided by existing data seems unlikely to change substantially. The nature of degradation products of the PBDEs is also likely to be elucidated in future, leading to consideration of their toxicity.

With respect to biodegradation, Tetra-, Penta- and Hexa-BDE are predicted to be "recalcitrant" by the BIOWIN program. Using the EPIWIN program, estimated half-lives for PentaBDE are 600 days in aerobic sediment, 150 days in soil, and 150 days in water (Palm 2001). This degree of persistence is supported by the fact that no degradation (as CO_2 evolution) was seen in 29 days in an OECD 301B ready biodegradation test using PentaBDE (Schaefer and Haberlein 1997).

Schaefer and Flaggs (2001) carried out a 32-week anaerobic degradation study using a mixture of ^{14}C-labelled and unlabelled BDE-47 (a TetraBDE) incorporated into sediments. The study showed that <1% of the total radioactivity was recovered as $^{14}CO_2$ and $^{14}CH_4$, indicating that essentially no mineralization had occurred. Overall, the study found that levels of degradation were not statistically significant; however, the HPLC analytical method with radiometric detection indicated that some products had been formed in the 32-week samples. Between one and three such peaks were identified in 26 of 42 samples analyzed. Work is underway to identify these products. It is likely that BDE-47 has the potential to degrade very slowly under anaerobic conditions.

Several studies using sediment cores show that PentaBDE congeners deposited in European marine sediments at the beginning of 1970s are still present in significant amounts, indicating high persistency in sediments (Covaci *et al.* 2002a, Nylund *et al.* 1992, Zegers *et al.* 2000, Zegers *et al.* 2003). The industrial production and use in Europe started in the beginning of the 1970s, with a reduction in more recent years. This is reflected in the sediment core profiles, with no occurrence before this date, and an increase in levels after, with a levelling off in more recent years. In the most recent studies (Zegers *et al.* 2003) sediment cores from Norway, the Netherlands and Germany
were studied. Concentrations of PBDEs, normalized to total organic carbon content, were in the range 10-20 μg/g total carbon.

2.2.2 Bioaccumulation

2.2.2.1 Studies on bioaccumulation and biomagnification in local food webs

Several studies have focused on PentaBDE's potential for bioaccumulation and biomagnification. The studies show an increase of concentrations in biota with increasing trophic level in pelagic and Arctic food webs. The calculated bioconcentration factors (BCFs), bioaccumulation factors (BAFs) and biomagnification factors (BMFs) indicate PentaBDE's potential for bioaccumulation and biomagnification. In Table 2.4 the calculated values in the literature are summarized. The octanol/water

partition coefficient (log K_{OW}) for PentaBDE in those studies is 6.5 – 7.4. The more recent studies are described in the following text.

Table 2.4 Calculated bioaccumulation factors (BAFs) and biomagnification factors (BMFs) for one PentaBDE (BDE-99) in the literature from environmental studies in pelagic and Arctic food webs. The data are calculated using the mean lipid weight concentrations, except for the study performed by Sørmo et al. 2006, in which the values in brackets are BMFs calculated from mean whole body concentrations.

Variable	Organism	Area	Value	Reference
BAF	*Dreissena polymorpha*	Lake Mälaren, Sweden	1.8	Lithner et al. 2003
BMF	Guillemot egg/herring	Baltic sea	17	Sellström 1996
	Grey seal/herring	Baltic sea	4.3	Sellström 1996
	Salmon/sprat	Baltic sea	10	Burreau *et al.* 1999
	Salmon/sprat	Baltic sea	5.9	Burreau *et al.* 2000
	Atlantic Salmon/Small Herring	The Northern Atlantic Sea	3.8	Burreau *et al.* 2000
	Net plankton/Benthic organisms	Lake Ontario, Canada	7.1	Alaee *et al.* 2002
	Benthic organisms/Forage fish	Lake Ontario, Canada	0.8	Alaee *et al.* 2002
	T. libellula/Copepods	Svalbard, Arctic Norway	0.65 (1.3)	Sørmo *et al.* 2006
	G.wilkitzkii/Copepods	Svalbard, Arctic Norway	47.6 (19.0)	Sørmo *et al.* 2006
	Polar cod/Copepods	Svalbard, Arctic Norway	2.1 (1.6)	Sørmo *et al.* 2006
	Polar cod/*T. inermis*	Svalbard, Arctic Norway	1.9 (1.2)	Sørmo *et al.* 2006
	Polar cod/*T. libellula*	Svalbard, Arctic Norway	3.4 (1.3)	Sørmo *et al.* 2006
	Polar cod/*G.wilkitzkii*	Svalbard, Arctic Norway	0.04 (0.1)	Sørmo *et al.* 2006
	Ringed seal/*T. inermis*	Svalbard, Arctic Norway	26.8 (54.5)	Sørmo *et al.* 2006
	Ringed seal/*T. libellula*	Svalbard, Arctic Norway	43.1 (60.0)	Sørmo *et al.* 2006
	Ringed seal/*G.wilkitzkii*	Svalbard, Arctic Norway	0.6 (3.9)	Sørmo *et al.* 2006
	Ringed seal/Polar cod	Svalbard, Arctic Norway	13.7 (56.6)	Sørmo *et al.* 2006
	Polar bear/Ringed seal	Svalbard, Arctic Norway	0.3 (0.29)	Sørmo *et al.* 2006
	Polar bear/Ringed seal	Arctic Canada	3.4	Muir *et al.* 2006
	Polar bear/Ringed seal	Arctic Canada	11	Muir *et al.* 2006
	Polar bear/Ringed seal	Arctic Canada	8.0	Muir *et al.* 2006
	Polar bear/Ringed seal	Greenland	1.0	Muir *et al.* 2006
	Polar bear/Ringed seal	Svalbard, Arctic Norway	5.9	Muir *et al.* 2006

PBDE analyses of zebra mussels (*Dreissena polymorpha*) were included in a larger study undertaken in and around the city of Stockholm, Sweden (Lithner *et al.*, 2003). Mussels were collected from a background site and transplanted in baskets to other downstream sites in Lake Mälaren, Saltsjön and in several small lakes. Freshwater flows from Lake Mälaren, through the middle of Stockholm, then out into the brackish Baltic Sea via Saltsjön. Five PBDE congeners (BDE-47, BDE-99, BDE-100, BDE-153 and BDE-154) were determined. The congener pattern was dominated by BDE-47 and BDE-99 (four and five bromines, respectively) and was similar to the C-PentaBDE. Bioaccumulation factors (BAFs) for the various compounds studied were estimated using data from suspended particulate matter (SPM) collected in sediment traps in 1998-99 at the same sites in Riddarfjärden and Saltsjön (Broman *et al.*, 2001). The concentrations on SPM were assumed to reflect water concentrations. BAFs were calculated using lipid weight concentrations in mussels and organic carbon based concentrations in the SPM.

When compared to other compounds (PCBs, DDTs, HCB), the BDEs had the highest BAFs, ranging from 1 to 2. The BAF (= level in mussel/level in SPM) for PentaBDE was 1.8.

Concentrations of BDE-47 and BDE-99 in Lake Ontario pelagic food web show increasing concentrations with increasing trophic position (Alaee *et al.* 2002). In this study, concentrations of PBDEs in archived plankton, *Mysis*, *Diporeia*, alewife, smelt, sculpin and lake trout samples collected in 1993 were determined. The trophodynamics of PBDEs in the Lake Ontario pelagic food web were also investigated. Lake Ontario pelagic food web consists of three trophic levels. The lake trout (*Salvelinus namaycush*) is a top predator fish species in Lake Ontario, feeding on forage fish including alewife (*Alosa pseudoharengus*), rainbow smelt (*Osmerus mordax*) and slimy sculpin (*Cottus cognatus*); in turn these fish feed on *Mysis* and *Diporeia*, which feed on phytoplankton, and zooplankton sampled as net plankton. Concentrations were increasing at each step up the food chain. The exception to this trend was the biomagnification of BDE-99 from benthic organisms to forage fish, which had a biomagnification factor of 0.8. This is an indication of the breakdown of BDE-99. In fact, the PBDE profile in the plankton; *Mysis* and *Diporeia* resembled the C-PentaBDE formulation, which indicates that BDE-99 bioaccumulates in the invertebrates and starts to be metabolized by forage fish.

Further studies of metabolism involving reductive debromination are discussed in Section 2.3.5.

Whittle *et al.* (2004) conducted surveys of PBDE levels in fish communities of Lake Ontario and Lake Michigan in 2001 and 2002 and evaluated biomagnification in the local pelagic food web (net plankton/*Mysis*/*Diporeia* → forage fish (smelt/sculpin/alewife) → lake trout). Their analysis, which included a total of forty one PBDE congeners, showed that BDE 47, 99 and 100 were prominent at each trophic level. The biomagnification factors (BMFs) representing total PBDEs for forage fish to lake trout ranged from 3.71 to 21.01 in Lake Michigan and from 3.48 to 15.35 in Lake Ontario. The BMF for plankton to alewife as 22.34 in Lake Ontario.

A recent study of an Arctic food chain shows the same result (Sørmo *et al.* 2006) as Alaee´s study. Concentrations of PBDEs were investigated in an Arctic marine food chain, consisting of four invertebrate species, polar cod (*Boreogadus saida*), ringed seals (*Pusa hispida*) and polar bears (*Ursus maritimus*). The most abundant PBDEs, BDE-47 and BDE-99, were found in detectable concentrations even in zooplankton, the lowest trophic level examined in this study. Most of the investigated PBDEs biomagnified as a function of tropic level in the food chain. A noticeable exception occurred at the highest trophic level, the polar bear, in which only BDE-153 was found to increase from its main prey, the ringed seal, indicating that polar bears appear to be able to metabolize and biodegrade most PBDEs. The authors suggested that this discrepancy in the fate of PBDEs among the different species may be related to greater induction of oxidative detoxification activities in the polar bear. Absorption and debromination rates may be more important for bioaccumulation rates of PBDEs in zooplankton, polar cod and ringed seals. BDE-99 showed no biomagnification from pelagic zooplankton to polar cod, probably as a consequence of intestinal or tissue metabolism of BDE-99 in the fish. Also among pelagic zooplankton, there was no increase in concentrations from calanoid copepods to *T. libellula*. Lipid-weight based concentrations (LWCs) and whole-body based concentrations (WBCs) of PBDEs were used to assess biomagnification factors (BMFs). Whole body concentrations gave the most realistic BMFs, as BMFs derived from LWCs seem to be confounded by the large variability in lipid content of tissues from the investigated species. This study demonstrates that PentaBDEs have reached measurable concentrations even in the lower trophic levels (invertebrates and fish) in the Arctic and biomagnifies in the polar bear food chain.

Polybrominated diphenyl ethers (PBDEs) were determined in adipose tissue of adult and sub-adult female polar bears sampled between 1999 and 2002 from sub-populations in Arctic Canada, eastern Greenland, and Svalbard, and in males and females collected from 1994 to 2002 in northwestern Alaska (Muir et al. 2006). Only four congeners (BDE-47, BDE-99, BDE-100, and BDE-153) were consistently identified in all samples. BDE-47 was the major PBDE congener representing from 65% to 82% of the

ΣPBDEs. Age was not a significant covariate for individual PBDEs or ΣPBDE. Higher proportions of BDE-99, BDE-100, and BDE-153 were generally found in samples from the Canadian Arctic than from Svalbard or the Bering- Chukchi Sea area of Alaska. All four major PBDE congeners were found to biomagnify from ringed seals to polar bears. The polar bear-seal BMFs were relatively consistent despite the large distances among sites. The exceptions were the BMFs for BDE-99, BDE-100, and BDE-153 in East Greenland which had lower BMFs than those at all other sites. This may imply differences in the transformation of PBDEs in the marine food web leading to polar bears or to food web differences. Species differences in bioaccumulation and biotransformation of PBDEs have been noted for fish and this could lead to differences in congener patterns in fish-eating mammals and their predators.

Studies of the biomagnification of Tri- to DecaBDEs were carried out in three different food chains, two in the Baltic Sea and one in the Atlantic Ocean (Law 2005). All of Tri- to HeptaBDE congeners biomagnified, but the maximum biomagnification was for the PentaBDEs.

Matscheko *et al.* (2002) investigated the accumulation of seven PBDEs, eight PCBs and polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCCD/Fs) by earth worms collected from Swedish soils in spring and autumn 2000. The selected sampling sites were agricultural lands receiving applications of sewage sludge, and a field flooded by a river known to contain the target substances in its sediment. Reference sites were rural and urban soils with no known sources of the target substances other than background. Earthworms (primarily *Lumbricus terrestris, Lumbricus spp, Aporrectodea caliginosa, A. rosea* and *Allolobophora chlorrotic*) were collected from all field sites, starved for 24 h to clear gut contents, and then analyzed for the presence of the target substances. Biota-soil accumulation factors (BSAFs) were calculated as the ratio of concentration of target substance in worm lipids to that in soil organic matter. BSAFs for BDE-47, BDE-66, BDE-99 and BDE-100 ranged from 1 to 10. They were comparable to those determined for the PCBs but higher than those for PCCD/Fs. BSAFs of greater than 10 were determined at one agricultural site, where factors of 11, 18 and 34 were calculated for BDE 99, 47 and 100 respectively. Data collected for BDE-153, BDE-154 and BDE-183 were not used, as levels in the earthworm blanks were deemed to be unacceptable high.

2.2.2.2 Monitoring results indicating bioaccumulation

A large range of studies show concentrations of concern in top predators. High levels in top predators are usually an indication on the potential of a compound to bioaccumulate in the top predator food chain.

Several studies (Jaspers *et al.* 2004, Herzke *et al.* 2005, Lindberg *et al.* 2004, D`Silva *et al.* 2004, Law *et al.* 2005, Sinkkonen *et al.* 2004, Sellström *et al.* 2003) indicate that PentaBDE is widespread in top predatory birds in Europe, such as peregrine falcon (*Falco pereg*rine), merlin (*Falco columbarius*), goshawk (*Accipiter gentiles*), golden eagle (*Aquila chrysaetos*), and buzzard (*Buteo buteo*). High levels are detected in top predatory eggs of white-tailed sea eagle, peregrine falcon, osprey, and golden eagle (Herzke et al. 2005, Lindberg *et al.* 2004). High levels have also been detected in European harbour porpoises (*Phocoena phocoena*) (Thron *et al.* 2004 and Covaci *et al.* 2002).

In the Arctic, C-PentaBDE is detected in high levels in top predatory birds and mammals (Verrault *et al.* 2005, Verrault *et al.* 2004, Norström *et al.* 2002, Herzke *et al.* 2003, Vorkamp *et al.* 2004a and b, Wolkers *et al.* 2004, Thron *et al.* 2004, Thomas *et al.* 2005, Ikonomou *et al.* 2002), such as glaucous gulls (*Larus hyperboreus*), polar bears (*Ursus maritimus*), ringed seals (*Phoca hispida*) and beluga whales (*Delphinapterus leucas*).

2.2.3 Long-range environmental transport

2.2.3.1 Environmental studies on transport and distribution

There are several factors indicating long-range transboundary transport of PentaBDE in the environment. It has a high persistency in air, with a half-life of 11-19 days (Palm *et al.* 2002, Vulykh *et al.* 2004)). Monitoring studies have detected a widespread occurrence in the European atmosphere (ter Shure *et al.* 2004, Lee *et al.* 2004, Jaward *et al.* 2004, Harrad and Hunter 2004, Harrad *et al.* 2004) and Arctic (AMAP 2002 and AMAP 2005, Peltola *et al.* 2001).

Sampling of air in the Great Lakes region of North America was undertaken in 1997-1999 and reported by Strandberg *et al.* (2001). PBDEs, mainly BDE-47 and BDE-99, were detected in all samples from four locations, and there was little variation over the time period. PBDE concentrations ranged from 5 pg/m^3 near Lake Superior to about 52 pg/m^3 in Chicago. At the temperatures of collection, 20±3^{o}C, approximately 80% of the tetrabromo congeners were in the gas phase, but 70% of the hexabromo congeners were associated with particles.

Results for the far-northern Pacific covered particulate matter collected in July-September 2003 from the Bohai Sea to the high Arctic, 37^o – 80^o N (Xin-Ming Wang *et al.* 2005). The dominant congeners were BDE-47, BDE-99, BDE-100 (all present in the commercial pentamix) and BDE-209, with concentrations falling from mid- to high-latitudes, probably resulting (according to the authors) from dilution, deposition and decomposition of the PBDEs during long-range transport. Total PBDE concentrations were in the range 2.25 – 198.9 pg/m^3 with a mean of 58.3 pg/m^3. The source of the PBDEs is believed to be the North American continent from which they distill to an Arctic 'cold trap'.

The emphasis on any assessment of the dispersal of PentaBDE into the environment has to be on long-range transport, specially to Arctic regions, but there also is a growing body of data on dispersal of the substance and related congeners within regions. Air sampling in Southern Ontario in the Spring of 2000, before bud burst, showed PBDE concentrations of 88-1250 pg/m^3, with the lighter congeners (DBE-17, -28 and -47) dominating (Gouin *et al.* 2002). The concentrations fell to 10-20 pg/m^3, a change that the researchers attributed to, firstly, enhanced levels caused by expiration from the winter snowpack, followed by possible sorption by emergent foliage. Other studies in Ontario (Harner *et al.* 2002) found air levels of total PBDE in the range 3.4-46 pg/m^3. In later work, organic films on indoor and outdoor windows in Southern Ontario were examined for their content of PBDEs by Butt *et al.* (2004). While the PBDE content was dominated by BDE-209 from the decabromo mixture, there were significant quantities of congeners deriving from the C-PentaBDE. Back calculation gave total PBDE concentrations in outdoor air of 4.8 pg/m^3 and 42.1 pg/m^3 for indoor air.

Jaward *et al.* (2004a) studied a total of 71 passive air samples using semi permeable membrane devices (SPMDs) for eight BDE congeners (BDE-28, BDE-47, BDE-49, BDE-75, BDE-99, BDE-100, BDE-153 and BDE-154) during a six week period in 2002 at remote/rural/urban locations across 22 countries in Europe. BDEs were detected in approximately 50% of the samples, and the equivalent ΣBDE air concentrations estimated from the passive sampler data ranged from 0.5 to 250 pg m^3. The focus of the most elevated concentrations was the UK, which has a history of PBDE production and has also been a major user of PBDE formulations due to stringent fire regulations within the country. The UK is clearly a regional source for BDEs to the European atmosphere and, in contrast, levels reaching Europe from the west (over the Atlantic Ocean) are low. Other high values were detected in urban centres in mainland Europe – samples from Athens, Bilthoven (Netherlands), Geneva, Milan and Seville, for example. Non-detectable/very low values occurred in remote/background sites, especially in Iceland, Ireland, Norway and Sweden, and values in Eastern Europe were generally low. BDE-47 and BDE-99 contributed ca. 75% to ΣBDE, similar to their proportion in the Bromkal 70-5DE C-PentaBDE.

In the US, high-volume samplers were used to examine concentrations of gaseous and particulate PBDEs at five sites (urban, semi-urban, agricultural and remote) from the Midwest to the Gulf of Mexico, every twelve days during 2002-2003 (Hoh and Hites 2005). The mean concentration of total PBDEs at the Chicago site was 100±35 pg/m^3, some 3-6 times higher than those at other sites and significantly higher than measurements made in 1997-1999 (Strandberg *et al.* 2001). The mean concentration of PentaBDE was 31 pg/m^3 at the Chicago site, some 2-4 times the values for other sites.

Fugacity model results indicate that PBDEs will largely partition to organic carbon in soil and sediment and that their persistence will be strongly influenced by degradation rates in these media (although these are not well known). Only a small proportion of PBDEs exist in air and water. If this is the case, it suggests that these compounds have limited LRAT potential (Prevedouros *et al.* 2004a, Gouin and Harner 2003). This corresponds with PentaBDE's affinity for carbon, low solubility in water (1.0 µg/L) and low vapour pressure (7.6 x 10^{-6} Pa). However, Gouin and Harner (2003) suggest that because of their physical–chemical properties, PBDEs may experience active surface–air exchange as a result of seasonally and diurnally fluctuating temperatures. Subsequently, this may result in the potential for LRAT of the PBDEs through a series of deposition/volatilization hops, otherwise known as the ‘‘grasshopper’’ effect. This assumption is supported by environmental data. Lee et al. (2004) detected atmospheric concentrations of BDEs at two rural/semirural sites in England, and one remote site on the west coast of Ireland in 2001 and in 2000, respectively. ΣBDE concentrations at Mace Head, Ireland, were 0.22 to 5.0 pg/m^3 with a mean of 2.6 pg/m^3 and were controlled primarily by advection. ΣBDE concentrations at Hazelrigg (NW England) were 2.8 to 37 pg/m^3 with a mean of 12 pg/m^3, and at Chilton (SW England) were 3.4 to 33 pg/m^3 with a mean of 11 pg/m^3. The congener profile was, on average, similar to that of the C-PentaBDE. At the two English sites in the summer, PBDE concentrations were strongly influenced by temperature, indicating that land/air exchange processes play an important role in determining atmospheric concentrations.

The concentrations of PBDEs were determined in soil samples collected along a latitudinal transect through the UK and Norway, at remote/rural woodland (both coniferous and deciduous) and grassland sites (Hassanin *et al.* 2004). Concentrations for ΣBDE ranged from 65 to 12,000 ng/kg dry weight. BDE congeners BDE-47, BDE-99, BDE-100, BDE-153 and BDE-154, covering the major constituents of the C-PentaBDE, dominated the average congener pattern in the soils. This was interpreted as evidence that transfer of the congeners from materials treated with the commercial product from source to air to soil occurs with broadly similar efficiency, and that there is little degradation of the congeners by processes acting either during atmospheric transport or within the soils themselves. There was evidence of latitudinal fractionation of the BDE congeners, with the relative amounts of BDE-47 and the lighter congeners increasing to the north (with increasing distance from source areas) while the proportion of BDE-99 and the heavier congeners decreased. Plots of BDE congener concentrations against percentage soil organic matter yielded different slopes for different congeners. Steeper slopes were generally observed for lighter congeners such as BDE-47, indicating that they have undergone some air-surface exchange (“hopping”), whilst those of heavier congeners such as BDE-153 were close to zero, indicating that they are retained more effectively by soil following deposition. A Japanese study detected seasonal variations in the partitioning of PBDEs between the gas and particulate phase. The fraction of particulate PBDEs was higher in samples collected in winter than those in the summer (Hayakawa *et al.* 2004). PentaBDE is expected to be transported in the environment mostly by being absorbed onto particles due to its low volatility, low solubility and high affinity for carbon compounds. There are results from environmental studies which indicate that PBDEs are transported on air borne particles, and that they are susceptible to wet deposition (ter Schure *et al.* 2004a, ter Schure and Larsson 2002). Further transport depends on the fate of the particles. Fate after depositions on land depends on the level of wind erosion, that can vary with the season. Fate after deposition into the sea depends on oceanographic processes, such as water layering and transport by currents in the surface layers.

Ter Schure *et al.* (2004a) collected air and atmospheric bulk deposition samples on the island of Gotska Sandön in the Baltic Proper during a 10 week period in autumn 2001. The sampling site was chosen because of its central position in the Baltic Sea, and because of the absence of local point sources of

pollution. Ten PBDE congeners were determined (BDE-17, BDE-28, BDE-47, BDE-85, BDE-99, BDE-100, BDE-153, BDE-154, BDE-183 and BDE-209). The median ΣBDE concentration (ΣBDE is the sum of the concentrations of the congeners determined in each study) was 8.6 pg/m^3, and the BDEs were mainly associated with particles. A comparison to levels of PCB in the atmosphere indicated that, as PCB concentrations in Baltic air have been declining, the input of BDEs by atmospheric deposition to the Baltic Proper now exceeds that of the PCBs by a factor of almost 40 times.

BDEs were determined in precipitation falling in southern Sweden during a two week period in 2000 (ter Schure and Larsson, 2002). The particle-associated and "dissolved" phases were separated during sampling and 65 ± 18% of ΣBDE was found to be particle-associated. The volume weighted mean concentration of ΣBDE (nine congeners) in rain was 209 pg/l, and the total deposition rate was 2 ± 1 ng ΣBDE/m^2/day. The congener profile in both phases of the total deposition was dominated by BDE-209, and thereafter BDE-47, BDE-99 and BDE-183, representing inputs from all three commercial PBDE formulations. The authors found that particle associated BDEs are effectively removed during small precipitation episodes, and that particle scavenging was an important mechanism for the wet deposition of BDEs.

A model assessment of potential for long-range transboundary atmospheric transport and persistence of PentaBDE have been carried out by EMEP (Co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe). The values of LRTP were considered to be strongly influenced by environmental processes, such as degradation, deposition, gas/particle partitioning, and gaseous exchange with underlying surface. The main process of removal from the atmosphere for the two congeners BDE-47 and BDE-99 was found to be deposition to land and seawater, 78% to land and 15% to sea for BDE-47 and 77% to land and 21% to sea for BDE-99. Only 7% of BDE-47 and 2% of BDE-99 was degraded. The calculated half-life in air was 7 days for BDE-47 and 11 for BDE-99. The findings showed a spatial distribution of BDE-47 that covers the Arctic, Europe, the Mediterranean Sea and northern Africa. BDE-99 spreads over longer distances and spreads to the Arctic, Atlantic Ocean, Asia and Africa. Transport distances (TD) were calculated for the two congeners. The TD was 2300 km for BDE-47 and 2800 km for BDE-99 (EMEP 2004).

Wania and Dugani (2003) examined the long-range transport potential of PBDEs using a number of models – TaPL3-2.10, ELPOS-1.1.1, Chemrange-2, and Globo-POP-1.1 – and various physical and chemical properties – for example, solubility in water, vapour pressure, log K_{OW}, log K_{oa}, log K_{aw}, and estimated half-lives in various media. They found that all models yielded comparable results, with tetrabromodiphenyl ether showing the greatest atmospheric transport potential and decabromodiphenylether the lowest. The researchers estimated a characteristic transport distance (CTD) ranging from 1113 to 2483 km for the tetrabromo, 608 to 1349 for the pentabromo, 525 to 854 for the hexabromo, and 480 to 735 for the decabromo congener. The CTD was defined as the distance a parcel of air has travelled until 1/*e* (approximately 63%) of the chemical has been removed by degradation or deposition processes (Gouin and Mackay 2002).

The EU risk assessment (EU 2000) concluded that the major part of releases end up in soil. From soil, PentaBDE can be expected to be moved mainly through leaching with water in the suspended solids fraction or through wind erosion where it occurs. A small part in the soil can be volatilized, especially in the warm season, and so may be considered a plausible alternative mechanism for transport in addition to volatilization and advective transport of vapor identified in the literature. Although PentaBDE has low water solubility, it has been detected in lakes and seas, and can be transported with water in the soluble and particle phases (Peltola *et al.* 2001). Occurrence in migratory birds and fish indicate the possibility of transport by migration of animals, but the main route seems to be through the atmosphere.

2.2.3.2. Levels in remote areas

The detected levels in the Arctic atmosphere, biota and environment are strong indicators of the PentaBDEs potential for long-range transport (Verreault *et al.* 2005, Verreault *et al.* 2004, Norstrøm *et al.* 2002, Herzke *et al.* 2003, Vorkamp *et al.* 2004a and b, Wolkers *et al.* 2004, Thron *et al.* 2004, Thomas *et al.* 2004, Ikomomou *et al.* 2002, Christensen *et al.* 2002, de Wit *et a*l. 2004, AMAP 2002 and AMAP 2005).

There are several studies showing the occurrence of PentaBDE in remote areas in Europe as well (Vives *et al.* 2004, Hassanin *et al.* 2004 and Zenegg *et al.* 2003). Levels in remote regions are considered to be an indication on long-range transport.

PentaBDE (as total BDE) has been detected in Canadian and Russian Arctic air at concentrations up to 28 pg/m^3 (Alaee *et al.* 2002). Strandberg *et al.* (2001) reported concentrations of total PBDE (BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, BDE-190 and BDE-209) in air from the Great Lakes area during the period 1997-1999. Average concentrations based on four samples from each of four locations ranged from 4.4 pg/m^3 near Lake Superior in 1997 to 77 pg/m^3 in Chicago in 1998. The average air concentration of total PBDEs (1997, 1998 and 1999) for the sampling sites ranged from 5.5 to 52 pg/m^3. Tetra- and pentabromo congeners accounted for approximately 90% of the total mass of PBDE in this study. At 20±3^{o}C, about 80% of the tetrabromo congeners and 55-65% of the pentabromo congeners were in the vapour phase while about 70% of the hexabromo congeners were associated with the particulate phase.

A larger study was performed detecting BDEs in trout (three species) from eleven high mountain lakes in Europe (566 to 2,485m altitude) (Vives *et al.*, 2004). These lakes were selected as being far from local pollution emission sources, and it was considered that the only source of BDEs to these lakes was as a result of atmospheric transport and deposition. The major congeners identified (of 39 determined) were BDE-47 and BDE-99, followed by BDE-100, BDE-153, BDE-154 and BDE-28, and these congeners were found in all samples analysed. The highest concentrations of ΣBDE in fish muscle and liver were found in Lochnagar, Scotland, 1.2 and 11 µg/kg wet weight, respectively (177 and 366 µg/kg on a lipid basis). No correlation was observed between the occurrence of these compounds and altitude, latitude or temperature, and the authors inferred that the environmental distribution of the BDEs has not, as yet, reached a steady-state.

2.3 Exposure

2.3.1 Levels

PentaBDE has spread widely in the global environment. A large quantity of monitoring data exist with detected levels in marine and terrestrial birds, sea and terrestrial mammals, sediments, soil, seafood and fish. A global study by Ueno *et al.* (2004) of PentaBDE in skipjack tuna (*Katsuwonus pelamis*) shows a wide spread occurrence in the offshore waters of various regions in the world. Table 2.5 gives an overview over the levels found in different parts of the world.

Contamination of the environment and biota in remote regions can be a threat to vulnerable species and ecosystems. In the Arctic, together with other pollutants of concern, PentaBDE is detected in high levels in top predatory birds and mammals (Verreault *et al.* 2005, Verreault *et al.* 2004, Norstrøm *et al.* 2002, Herzke *et al.* 2003, Vorkamp *et al.* 2004a and b, Wolkers *et al.* 2004, Thron *et al.* 2004, Thomas *et al.* 2004, Ikomomou *et al.* 2002) showing that the Arctic food webs are seriously affected. Wolkers *et al.* (2004) detected levels of PentaBDE in beluga whales (*Delphinapterus leucas*) in the Arctic, a species protected by the Convention on migratory species (the Bonn convention). ΣBDE concentrations (geometric mean; 22 congeners) were 234, 161 and 29 µg/kg in juvenile, adult male and adult female beluga.

In fact, there are detected high levels of PentaBDE in several species, with populations of concern protected by the Bonn convention. Several studies (Jaspers *et al.* 2004, Herzke *et al.* 2005, Lindberg *et al.* 2004, D`Silva *et al.* 2004, Law *et al.* 2005, Sinkkonen *et al.* 2004, Sellström *et al.* 2003, Kannan *et al.* 2005, Ramu *et al.* 2005 and Wolkers *et al.* 2004) indicate that PentaBDE is widespread in peregrine falcon (*Falco pereg*rine), merlin (*Falco columbarius*), goshawk (*Accipiter gentiles*), golden eagle (*Aquila chrysaetos*), buzzard (*Buteo buteo)*, beluga whales (*Delphinapterus leucas*), irrawaddy dophins (*Orcaella brevirostris*), and Indo-Pacific humpback dolphin (*Sousa chinensis*), all protected by the Bonn convention. High levels of PBDEs are also detected in peregrine falcon eggs in Sweden (Lindberg *et al.* 2004), for which individual ΣBDE concentrations were as high as 39,000 µg kg^{-1} lipid weight, some of the highest concentrations seen in wildlife so far.

The populations of harbour porpoises (*Phocoena phocoena*) in the North and Baltic seas are protected through the Bonn Convention. Studies have detected high levels in those populations (Thron *et al.* 2004 and Covaci *et al.* 2002). In a study by Thron *et al.* (2004) animals with poor body condition (lower mean blubber thickness) had much higher concentrations than other individuals. Only females showed decreasing concentrations with age, indicating elimination via transfer from mother to offspring.

The harbour porpoise is, together with peregrine falcon and merlin, also on the list for strictly protected (endangered) species in the convention on the conservation of European wildlife and natural habitats (the Bern Convention). The white-tale sea eagle is on the list for endangered species in the Bern Convention. Levels of concern are detected in both individuals and eggs (Herzke *et al.* 2005). Beluga whales and irrawaddy dolphins are on list for protected (vulnerable) species. High levels are found in white-beaked dolphin (*Lagenorhyncus albirostris*), another endangered species. The parties of this convention undertake to take appropriate measures to ensure the conservation of endangered and vulnerable species and their habitats.

*Table 2.5 Levels of PentaBDE (**BDE-99**) in different parts of the world (LW=Lipid weight, DW=Dry weight).*

Country/Region	Organism/compartment	Levels of PentaBDE	References	Comments
Europe	Atmosphere Gas phase	10-120 pg/m^3	Jaward *et al.* 2004	22 countries
Japan	Atmosphere Particulate Gas phase	0.05-0.9 pg/m^3 0.05-19' pg/m3	Hayakawa *et al.* 2004	Measured in the summer
Sweden	Sediments	<0.7-51.4 ng/g DW	Palm *et al.* 2002	Rivers at point source
United Kingdom	Soil	78 – 3200 pg/g DW	Hassanin *et al.* 2004	
Western Europe	Sediments	<0.2-6.9 ng/g DW	Palm *et al.* 2002	Estuaries
Japan, Osaka	Sediments	9-28 ng/g DW	Palm *et al.* 2002	
North Pacific Ocean	Skipjack tuna	0.18-2.1 ng/g LW	Ueno *et al.* 2005	
Japan	Skipjack tuna	1.1-1.7 ng/g LW	Ueno *et al.* 2005	Offshore waters
East China Sea	Skipjack tuna	2.4-4.7 ng/g LW	Ueno *et al.* 2005	
Philippines	Skipjack tuna	2.1 ng/g LW	Ueno *et al.* 2005	Offshore waters
Brazil	Skipjack tuna	1.9 ng/g LW	Ueno *et al.* 2005	Offshore waters
Canada	Atlantic tomcod	77 ng/g LW	Law *et al.* 2003	
Chilika Lake, India	Irrawaddy dolphin	0.12-0.78 ng/g LW	Kannan *et al.* 2005	Endangered species
Hong Kong, China	Indo-Pacific humpback dolphin	33.6-720 ng/g LW	Ramu *et al.* 2005	Coastal waters 12% of ΣPBDEs
United Kingdom	White beaked dolphin	1480 ng/g LW	Law *et al.* 2003	Endangered species
Hong Kong, China	Finless porpoises	27.6-117.6 ng/g LW	Ramu *et al.* 2005	Coastal waters 12% of ΣPBDEs
Japan	Northern fur seal	2.64-4.56 ng/g LW	Kajiwara *et al.* 2004	Pacific coast 12% of ΣPBDEs

Svalbard, Arctic Norway	Polar bear	0.7-4.7 ng/g LW	Gabrielsen *et al.* 2004	
Canadian Arctic	Polar bear	1.04-11.3 ng/g LW	Muir *et al.* 2006	
Bjørnøya, Arctic Norway	Glacous gulls	0-7.9 ng/g LW	Herzke *et al.* 2003	
Norway	White-tailed sea eagle	6-184 ng/g LW	Herzke *et al.* 2005	In eggs. Endangered Species
Sweden	Peregrine falcons	110-9200 ng/g LW	Lindberg *et al.* 2004	Endangered species
Australia	Melon-headed whale	4.8 ng/g LW	Law *et al.* 2003	
Canada	Beluga whale	108 ng/g LW	Law *et al.* 2003	Vulnerable species
Netherlands	Mussels	0.3-11 ng/g LW	Law *et al.* 2003	Marine+freshwater
Sweden	Frog	5.6 ng/g LW	De Wit *et al.* 2004	
Canada	Zooplankton	0.46 ng/g LW	Law *et al.* 2003	

2.3.2 Trends

Most trend analysis show an increase in concentrations of PBDEs in the environment and in humans from the beginning of the 1970s, with a peak around the mid-1990s and a stabilisation or subsequent levelling off in Europe (Covaci *et al.* 2002, Fängström *et al.* 2005, Thomsen *et al.* 2005 and Knudsen *et al.* 2005), but with a continuous increase in the Arctic (Vorkamp *et al.* 2005, AMAP 2002 and AMAP 2005). PentaBDEs are reported in the studies to follow the same trend as ΣPBDEs. This increase has also been seen in North America, in air, soil and sediment, and wildlife, but insufficient data exist to allow comment on trends in the human population.

In the Asia-Pacific region a study on northern fur seals on the Pacific coast of Japan shows an increase of PBDEs to about 150 times between 1972 and 1994, and then levels decreased to about 50% in 1998 (Kajiwara *et al.* 2004). The reduction in PBDEs values was assumed to be due to the voluntary phase out of C-PentaBDE in Japan in 1990. BDE-99 levels showed the same pattern as ΣPBDEs.

Analysis of archived herring gull eggs (sampled in 1981, 1983, 1987, 1989, 1990, 1992, 1993, 1996, 1998, 1999 and 2000) enabled Norstrom *et al.* (2002) to establish temporal trends in PBDE concentrations over the period 1981-2000. At Lake Michigan, Lake Huron and Lake Ontario sampling sites, concentrations of tetra- and pentabromodiphenyl ethers (that is, BDE-47, BDE-99 and BDE-100) increased by 71-112-fold over these two decades (from 4.7 to 400.5 μg/kg ww at Lake Ontario; from 8.3 to 927.3 μg/kg ww at Lake Michigan; from 7.6 to 541.5 μg/kg ww at Lake Huron). These increases were found to be exponential at all three locations ($r^2 = 0.903 – 0.964$, $p < 0.00001$).

Wakeford *et al.* (2002) undertook sampling of eggs of the great blue heron in 1983, 1987, 1991, 1996, 1998 and 2000 in southern British Columbia and found that total PBDE concentrations (sum of tetra-, penta- and hexabromo-congeners) increased from 1.31 to 287 μg/kg ww between 1983 and 1996, but then dropped slightly to 193 μg/kg ww in 2000. They also undertook sampling of the eggs of thick billed murre in the Canadian North in 1975, 1987, 1993 and 1998, and observed a trend of gradually increasing PBDE concentrations (sum of tetra-, penta- and hexabromo-congeners) in these eggs from 0.43-0.89 μg/kg ww in 1975, to 1.83-3.06 μg/kg ww in 1998.

PBDEs have been detected in a variety of marine mammals. Alaee *et al.* (1999) reported average PBDE (di-to hexaBDE) concentrations in the blubber of marine mammals from the Canadian Arctic as 25.8 μg/kg lipid in female ringed seals (*Phoca hispida*), 50.0 μg/kg lipid in male ringed seals, 81.2 μg/kg lipid in female beluga (*Delphinapterus leucus*) and 160 μg/kg lipid in male beluga. BDE-47, a tetrabromodiphenyl ether, was the predominant congener, followed by the pentabromo BDE-99. Ikonomou *et al.* (2000, 2000b) reported PBDE concentrations in biota samples from the west coast and Northwest Territories of Canada. The highest concentration of of total PBDE residues, 2269 μg/kg lipid, was found in the blubber of a harbour porpoise form the Vancouver area. With a concentration of about

1200 μg/kg, one congener, BDE-47, accounted for slightly more than half of the total PBDE in the sample. Ikonomou *et al.* (2002a) analyzed temporal trends in Arctic marine mammals by measuring PBDE levels in the blubber of Arctic male ringed seals over the period 1981-2000. The mean total concentrations increased exponentially, from 0.572 μg/kg lipid in 1981 to 4.622 μg/kg in 2000, a greater than eightfold increase. They determined that Penta- and HexaBDEs are increasing at approximately the same rate (doubling time 4.7 and 4.3 years, respectively), more rapidly than TetraBDEs, for which the doubling time was 8.6 years. Once again, BDE-47 was predominant, followed by BDE-99 and BDE-100.

A marked increase in tissue PBDE levels was also evident in blubber samples collected from San Francisco Bay harbour seals over the period 1989 to 1998 (She *et al.* 2002). Total PBDEs (the sum of BDEs 47, 99, 100, 153 and 154) rose from 88 μg/kg lipid to a maximum of 8325 μg/kg lipid over this short period. Stern and Ikonomou (2000) examined PBDE levels in the blubber of make SE Baffin Bay beluga whales over the period 1982-1997, and found that the levels of total PBDEs (tri-to hexa-congeners) increased significantly, Mean total PBDE concentrations were about 2 μg/kg lipid in 1982, and reached a maximum value of about 15 μg/kg lipid in 1997. BDE-47 was the dominant congener, with a mean concentration of approximately 10 μg/kg lipid in 1997. Total PBDE residues (concentrations for individual congeners not provided) in the blubber of St Lawrence estuary belugas sampled in 1997-1999 amounted to 466 (±230) μg/kg ww blubber in adult males, and 655 (±457) μg/kg ww blubber in adult females. These values were approximately twenty times higher than concentrations in beluga samples collected in 1988-1990 (Lebeuf *et al.* 2001).

The results from a modelling exercise utilizing the European variant (EVn) BETR multimedia environmental fate model were presented for the C-PentaBDE product by Prevedouros *et al.* (2004). To predict future atmospheric concentration trends, the model was used in its fully dynamic mode over the period 1970-2010. It predicted that atmospheric concentrations would have peaked around 1997, and then declined with an overall “disappearance” half-life of 4.8 years. The model steady state simulations gave generally good agreement with measured data for BDE- 47 and BDE-99. The empirical data for North America presented above, however, show continuing increases in concentrations, at least up the year 2000, and so while the model results match some European data with fair agreement, they are not in accord with data from North America.

Three dated sediment cores from locations in Western Europe were analyzed for 14 BDE congeners (Zegers *et al.*, 2003). Cores from the Drammenfjord (Norway), the western Wadden Sea (The Netherlands) and Lake Woserin (Germany) showed a time dependent pattern in the distribution of BDEs since the start of production of PBDE formulations. Two of the three commercial formulations could be distinguished. The penta-mix formulation is clearly present from the beginning of the 1970s. This is in agreement with data for the industrial production of this formulation. In the cores from the Netherlands and Germany, concentrations of BDE congeners associated with the C-PentaBDE were levelling off in the most recent layers (1995 & 1997), whereas those in the Drammenfjord were still increasing in 1999. The absence of all BDE congeners in the older (deeper) layers of all three cores, as well as in several 100 to 150 million year old layers of clay from Kimmeridge, UK, indicated that these BDE congeners are not produced naturally.

Human exposure to polychlorobiphenyls and PBDEs in Japan in 1980 and 1995 showed that levels of the latter had increased substantially over the twenty-year period, although there was great variation between regions. The main congeners detected in serum were BDE-47 and BDE-99. Most total PBDE levels had more than doubled, and in one area increased twenty-fold, with 1995 values falling in the range 0.6 – 41.4 ng/g lipid Koizumi *et al.* 2006).

2.3.3. Bioavailability

Environmental studies on bioavailability have detected uptake of PentaBDE in soil organisms (Matscheko *et al.* 2002), sediment dwelling organisms (Magnusson *et al.* 2003) and aquatic organisms (Lithner *et al.* 2003, Voorspoels *et al.* 2003, Marsch *et al*. 2004, Kierkegaard *et al*. 2004, and Sinkkonen *et al.* 2004), making PentaBDE's way into the food webs evident. Subsequent bioaccumulation and biomagnification of the compound has been detected and described in Section 2.2.2.

Soil exposed to PBDEs in various ways was analyzed for BDE-47, BDE-66, BDE-99, BDE-100, BDE-153, BDE-154 and BDE-183 (Matscheko *et al.*, 2002). Earthworms collected at all soil sampling sites were analyzed as well. The BDE congener profile in all soil samples was dominated by BDE-47 and BDE-99. Accumulation of the compounds in earthworms from the sites yielded a direct relationship between the concentrations in the soil and concentrations in the worms. The biota-soil accumulation factors (BSAFs) of BDE congeners BDE-47, BDE-99 and BDE-100 were around 5 (organic matter/lipids). Thus, earthworms living in contaminated soils will accumulate tissue BDE concentrations and, as these animals represent the base of the terrestrial food chain for many organisms, this form a pathway for the accumulation of BDEs in organisms at higher trophic levels.

The western Scheldt estuary is subject to a variety of suspected PBDE sources, such as a brominated flame retardant manufacturing plant, Antwerp harbour, and the textile industry located further upstream. PBDE concentrations in samples of biota, including crab, shrimp, starfish, benthic fish (such as dab, goby, plaice and sole) and gadoid fish (such as bib and whiting) from the estuary were compared to those in samples from the Belgian North Sea beyond the mouth of the estuary (Voorspoels *et al.*, 2003). Eight BDE congeners (BDE-28, BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, BDE-183 and BDE-209) were determined. Concentrations observed in the estuarine samples were up to 30 times higher than in those from the Belgian North Sea, with an increasing gradient towards Antwerp. Concentrations in the North Sea ranged from 0.02 to 1.5 µg/kg wet weight in benthic invertebrates and goby, from 0.06 to 0.94 µg/kg wet weight in fish muscle, and from 0.84 to 128 µg/kg wet weight in fish liver. The corresponding ranges in samples from the estuary were from 0.2 to 30, 0.08 to 6.9, and from 15 to 984 µg/kg wet weight, respectively. The ratio BDE-99/BDE-100 was found to be highly location- and species-dependent, possibly relating to differences in metabolism. In shrimp, the value of this ratio (4:1) was very similar to that observed in the Bromkal formulation and in estuarine sediment, and was similar in shrimp from both the North Sea and the estuary, implying both that these congeners are readily bioavailable and that shrimp lack the ability to metabolize either congener. On a lipid weight basis, concentrations of BDE-47 ranged from 3 to 108 µg/kg lipid weight in samples from the North Sea, and from 8 to 1,550 µg/kg lipid weight in estuarine samples. BDE-47 was the most abundant congener in all samples, comprising 43 to 75% of ΣBDE.

Thomas *et al.* (2004) conducted an input-output balance study of BDEs on three captive, juvenile grey seals. The animals were fed a diet of herring for six months, and the study was performed during the last three months of this period. BDE analysis was undertaken using GC-ECNIMS. Consistently high absorption (89 - 99%) was observed for all PBDE congeners studied (BDE-28, BDE-47, BDE-49, BDE-99, BDE-100, BDE-153, BDE-154 and BDE-209).

2.3.4 Human exposure

Studies, assessments and reviews referred to in this section have shown that the main routes for human exposure are food, and exposure to dust in indoor air at home and workplaces due to levels in products like furniture and electronic devices. Fish and agriculture products are the main food sources of PentaBDE for humans, and mother's milk for the nursing child. Fatty fish from contaminated areas are a major source (Sjödin *et al.* 2003). PentaBDE has been detected in various foods (VKM 2005, Burniston *et al.* 2003 and Bocio *et al.* 2003) as well as in indoor dust (Shoeib *et al.* 2004 and Wilford *et al.* 2005). Levels in foods in the US have been reported by Schecter *et al.* (2004), Schecter *et al* 2006, and Huwe *et al.* (2005). There are several hazard assessments in EU and US, looking into the exposure of humans

(VCCEP 2003, COT 2004, VKM 2005). They conclude that the available hazard or exposure information is inadequate to fully characterize the risks.

About 5% of the individuals in general populations have been found to be subjected to elevated exposure (Thomsen *et al.* 2005 b). This, together with estimates of the half life of C-PentaBDE congeners in humans, raises concern for long-term effects on human health. The half-lives for these congeners in humans have been estimated to be 1,040 days (BDE-99) and 573 days (BDE-100) (Geyer *et al.* 2004).

Domestic house dust is likely to be a significant source where furniture, carpet or appliances contain C-PentaBDE. This has been discussed in Section 2.1.1. It is not clear which sources are the greatest, and there could be wide variations depending on lifestyle and diet.

Several studies have detected levels of PentaBDE in sewage sludge (Matscheko *et al.* 2002, Fabrellas *et al.* 2004, Motche and Tanner 2004 and Sjödin *et al.* 2003, Hale 2002). Sewage sludge is considered to be one of the main sinks for PBDEs. The application of sewage sludge to agricultural land is one of the reasons for detected levels of PentaBDE in food products. This can explain the detected levels in vegetables and root crops in experimental studies. Levels in fish and root crops can be the source of exposure to domestic animals like chickens and pigs, and the source of PBDEs in meat products for human nourishment.

A Canadian global study showed that PentaBDE is widespread in human milk in populations all over the world (Ryan 2004). There are data on levels in human blood serum and milk from USA, Canada, Mexico, Japan, the EU region, the Arctic region and Scandinavia. A meta-analysis by Hites (2004), using data published up to mid-2003, showed that serum and milk levels in the US were much higher than those in Europe - ~35 ng/g *vs* ~ 2 ng/g lipid – and were doubling on average every 4-6 years. BDE-47 and BDE-99 were the major congeners detected. Considerably higher levels are found in humans from North America in general. About 5% of general populations have been found to be subjected to elevated exposure. Thus, together with estimates of the half-life of PentaBDE congeners in humans, raiss concern for long-term effects on human health (Thomsen *et al.* 2005b).

Levels increasing from the 1980s to the 2000s have been observed in mother's milk from Sweden as well as in blood from Germany and Norway (Sjödin *et al.* 2003). A more recent study in Sweden (Fängström *et al.* 2005) assessed the temporal trends of polybrominated diphenyl ethers (PBDEs), in mothers' milk in the Stockholm area. The pooled samples were covering the time period 1980 to 2004, with emphasis on samples from the last ten years. Concentrations of BDE-47, BDE-99 and BDE-100 reached a peak in the mid-1990s and are now clearly showing decreasing levels. The concentrations are however still much higher than in 1980.

The objective of a recent Norwegian study was to complete and extend a previous study on time trends of PBDEs in Norwegian pooled serum samples (Thomsen *et al.* 2005a) and put together an overview of the PBDE body burden in the general population from 1977 to 2004. The temporal trend of the sum of seven PBDEs (BDE-28, BDE-47, BDE-99, BDE-100, BDE-153, BDE-154 and BDE-183) in the pooled serum from the present study are in close agreement with the levels found in a previous study by the same authors. In general, for similar time periods the levels in breast milk seem to be somewhat lower than in the serum, but the same overall trend is observed. This confirms that the PBDE body burdens in these regions have risen rapidly from 1977 to about 1997, but now seem to have stabilized or even to have decreased. This is in accordance with the trends observed in Swedish breast milk, as an indicator of the European situation, but may not be true of levels in North America. The PBDE level was previously found to be about twice as high in a serum pool from infants up to four years of age compared to serum pools from elderly persons. This finding was confirmed in the Norwegian study. However, in 2002, children between the ages of 5 and 14 years showed higher levels of PBDEs than the average adult.

Contemporary PBDE concentrations in Europe and Asia are remarkably similar, with low median values on a lipid basis for all countries and relatively small variations. The situation in North America is completely different with median values for individual studies in the range of 20-50 ng/g LW (Ryan 2004). However, in parallel with the regional differences that were reported above for biota, the levels in breast adipose tissue taken from women living in San Francisco Bay area in 2000 were almost two orders of magnitude higher than what has been reported in human milk from Sweden (Sjödin *et al.* 2003). A more recent study of levels in human adipose tissue in New York was published by Johnson-Restrepo *et al.* (2005). The study of 40 males and 12 females of a range of ages and ethnicities showed wide variations in lipid PBDE concentrations, with mean values substantially higher than the medians. Median concentrations were: BDE-47, 29.3 ng/g lipid; BDE-99, 10.3 ng/g lipid; BDE-100, 12.0 ng/g lipid.

In a preliminary screening of PBDEs in plasma and milk samples from Mexican women, the levels were well above European levels of PBDEs reported so far (López *et al.* 2004). The mean level of PBDEs (with BDE-209 excluded) in Mexican women living in urban areas was approx. 20 ng/g LW in plasma. The levels in women living in rural areas in Mexico were however comparable with women living in rural areas in Sweden. (BDE-209 levels were only detected in women living in the Mexican city).

Ryan (2004) detected a big individual variation in levels in the general population in a study from Canada. The values span more than three orders of magnitude, with a few values showing a much greater level. Levels detected in the Canadian Arctic in Ryan's study (2004) were increasing. Values in human milk from the Faroe Islands showed the same trend (Fängström *et al.* 2004).

Two studies in Australia indicated that levels of PBDEs in Australian breast milk and blood serum are higher than those in Europe but lower than those found in North America (Harden *et al.* 2004 and 2005).

*Table 2.6 Data on mean levels of PentaBDE (**BDE-99**) (ng/g LW) in humans from different parts of the world.*

Data	Country/region	Levels	References	Year	Comments
blood	The Netherlands	0.8	Weiss *et al.* 2004	unknown	
blood	Norway	1.0	Thomsen *et al.* 2004	1999	
blood	Mexico	2.0	López *et al.* 2004	2003	Urban population
blood	Australia	2.3	Harden *et al.* 2004	2003	
milk	Germany	0.2	Harden *et al.* 2004	2000	
milk	Sweden	0.3	Fängström *et al.* 2005	2003	Urban population
milk	Mexico	0.6	López *et al.* 2004	2003	Rural population
milk	Sweden	0.5	López *et al.* 2004	2003	Rural population
milk	United Kingdom	0.9	Harden *et al.* 2004	?	median
milk	Faroe Islands	1.0	Fängström *et al.* 2004	1999	Rural population
milk	Australia	1.9	Harden et al. 2005	2002/2003	
milk	Canada	4	Ryan *et al.* 2002	2002	Rural population
milk	USA	28	Päpke *et al.* 2001	2000	Urban population

Although they are less relevant that environmental data, results from occupational studies bear out the facility with which the PBDEs are taken up by human bodies. In Sweden, occupational exposure to PBDE has been identified among electronics recycling personnel (Sjødin *et al.*, 1999) and in technicians responsible for repair and maintenance of computers (Jacobsson *et al.*, 2002) as well as in nearby soil and sediment (Wang *et al.* 2005). Also workers in industry manufacturing C-PentaBDE, or polyurethane foam and electronic equipment containing it can be exposed to PentaBDE. There is an extensive literature on such exposures.

2.3.5 Debromination

There is growing interest in the fate of PBDEs in the environment. In experiments reported by Stapleton *et al.* (2004), carp were fed food spiked with individual BDE congeners for 62 days, and tissue and excreta were examined. At least 9.5±0.8% of BDE-99 in the gut was reductively debrominated to BDE-47 (one less bromine) and assimilated in carp tissues. Similarly, 17% of the heptabromo congener BDE-183 was reductively debrominated to hexabromo congeners. The authors noted that body burdens of PBDEs may thus reflect direct uptake from exposure as well as debromination of more highly brominated congeners. Highly selective reductive microbial debrominations were observed in experiments reported by He *et al.* (2006). Hepta- and Octa-BDEs were produced in cultures of *Sulfurospirillum multivorans* to which DecaBDE had been added, but OctaBDE was not attacked in a similar system. Cultures of an alternative organism, *Dehalococcoides sp.*, failed to attack the DecaBDE but an OctaDBE mixture was extensively changed, yielding a mixture of Hepta- through Di-BDEs which included the PentaBDE, BDE-99. The authors draw attention to the potential for conversion of higher congeners in the environment to more toxic congeners with fewer bromine substituents. Further studies particularly environmental monitoring studies focussing on congeners for which the primary source is likely to be debromination reactions, are required to clarify the role of debromination in determining the final mix of PBDE congeners in the environment.

Hydroxylated BDEs (OH-BDEs) have been detected and identified as metabolites in several species after exposure to specific BDE congeners but have also been found to occur as natural products in marine sponges and ascidians (Marsch *et al.* 2004). Methoxylated BDEs (MeO-BDEs) have also been reported as natural products present in marine sponges and green algae. It would seem that the origin of these substances can be natural, anthropogenic or both. Nine OH-BDEs and six MeO-BDEs were identified in blood of Baltic Sea salmon (*Salmo salar*) using newly synthesized standards (Marsch *et al.*, 2004). All of the identified OH- and MeO-BDEs were substituted with four or five bromine atoms and five of them also had one chlorine substituent. Fourteen have the methoxy or hydroxy group substituted in the position *ortho*-to the diphenyl ether bond. The structures of several of the compounds support natural rather than anthropogenic origins. However, at least one of the OH-BDEs (4'-OH-BDE-49) may be a hydroxylated metabolite of BDE-47. Estrogenic activity of some hydroxylated PBDEs has been reported by Meerts *et al.* (2001).

Other studies of metabolism of PBDEs are summarized in Section 2.2.2.1.

2.4 Hazard assessment for endpoints of concern

Evidence to date suggests that the major congeners of the C-PentaBDE formulation, BDE-47 and BDE-99, are likely to be more toxic and bioaccumulative than other PBDE congeners. Although the toxicology of PBDEs is not completely understood, some studies on PentaBDE have demonstrated reproductive toxicity, neurodevelopmental toxicity and effects on thyroid hormones. The neurotoxic effects of PBDEs are similar to those observed for PCBs and so children exposed to PBDEs are likely to be prone to subtle but measurable developmental problems. It is presumed that PBDEs are endocrine disrupters, but research results in this area are scant (Siddiqi *et al.* 2003).

While further studies follow internationally-accepted guidelines might be needed to make a full risk assessment of the situations of children, there are sufficient data for development of the present risk profile.

It is acknowledged that these conclusions rest to some extent on examination of reviews, rather than reanalysis of primary data, but in general the studies under review have followed internationally accepted experimental protocols. Nonetheless, there is no significant disagreement between some reported results and later analyses, such as that of the US Voluntary Children's Chemical Evaluation Program (VCCEP) (2005).

2.4.1 Ecotoxicity

Recent studies show that exposure to BDE-47 can cause growth inhibition in colonies of the plankton algae (*Skeletonema costatum*) and a depression on reproductive output of the zooplankton *Daphnia magna* (Källqvist *et al.* 2006).

A recent paper by Timme-Laragy *et al.* (2006) showed adverse effects on fish development at low concentrations. However, the endpoints that were affected in this report (behavioural learning) are not usually accepted risk assessment endpoints. Other endpoints that would be acceptable, such as growth or survival, were not affected.

Canada was able to perform a risk quotient analysis for each congener, integrating known or potential exposures with known or potential adverse effects. In its simplest form, the risk quotient may be described by the equation:

$$\text{Risk quotient} = \frac{\text{exposure reference value}}{\text{toxicity reference value}}$$

and it is customary to use conservative values in order to highlight the worst case.

Exposures were estimated local to emission sources including areas receiving urban drainage (wildlife consumers) and downstream of a polymer processing facility (benthic organisms). Adjustment factots of 100-1000-fold were applied to critical toxicity values to reflect extrapolation from laboratory to fielod conditions, intraspecies and interspecies variations in sensitivity, and because compounds are bioaccumulative and persistent.

A risk quotient value >1 signifies the likelihood or potential for adverse effects to occur, while those <1 imply no danger to organisms. The Canadian results shown in Table 3.1 are based partly on Canadian empirical data and partly on surrogate data from Swedish and US sources.

Table 3.1 Risk quotient values for PentaBDE (Environment Canada 2006, Canadian Wildlife Table 8).

Commercial Product	**Pelagic organisms**	**Benthic organisms**	**Soil organisms**	**Wildlife consumers**
C-PentaBDE	$4x10^{-3}$	45.2	0.13-0.26	149

These values reflect the bioaccumulation of PentaBDE which causes organisms higher in the food chain to be exposed to greater risk.

2.4.2 Effects in mammals

In a review article on toxic effects of brominated flame retardants, Darnerud (2003) drew on a range of primary literature to conclude that exposure to PBDEs gives rise to adverse effects in experimental *in vivo* models, and depending on type of product different effects are seen, occurring at varying dose levels. Generally, the C-PentaBDE products cause effects at the lower dosages. The critical effects of PentaBDE are those on neurobehavioral development and, although somewhat less sensitive, thyroid hormones in offspring (from 0.6 to 0.8 and 6 to 10 mg/kg body wt., respectively) (Darnerud 2003). Note that some data reported in Table 2.7 show levels below these. More recent information, especially for North America, is available in Birnbaum and Staskal (2004).

Blubber biopsy and blood samples were collected from weaned grey seal (*Halichoerus grypus*) pups and juveniles during 1998 and 1999 (Hall *et al.*, 2003). Fifty four post-weaned pups and fifty five first year juveniles (of which thirteen were recaptured post- weaned pups) were studied. The median concentrations of ΣBDE (14 congeners) were 0.17 and 0.46 µg/kg lipid weight in the blubber of the pups and the juveniles, respectively. The study indicated that thyroid hormone levels in the blood of grey seals during their first year of life were significantly, and positively, related to ΣBDE concentrations in blubber, after accounting for the effects of possible confounding variables. Such an association is not, in itself, sufficient evidence for a causal relationship, but is in accordance with the hypothesis that these compounds can act as endocrine disrupters in grey seal pups.

Darnerud (2003) concluded in his review that for PentaBDEs, the critical effects among the available studies seem to be developmental neurotoxicity and, although generally at somewhat higher doses, altered thyroid hormone homeostasis. Regarding the neurotoxicity in mice, no clear mechanism could be defined but effects of the PentaBDEs both via thyroid hormone disruption and directly on signal transmission in brain have been discussed. For example, a number of PBDEs were capable of inducing cell death of cerebellar granule cells in culture (Reistad *et al.*, 2002, Reistad and Mariussen 2005). The LOAEL value for PentaBDE could be set to 0.6–0.8 mg/kg body wt., based on the most sensitive effect observed, neurobehavioral effects during early development (Darnerud 2003, although it is not the task of the POPRC to set a regulatory level, for construction of which resort would need to be made a wider range of data.

In a hazard assessment by the Committee on Food Safety in Norway (VKM 2005) the following toxic effects of exposure to BDE-99 or the C-PentaBDE formulation was reported: neurotoxicity, effects on neurobehavioral development, effects on the thyroid hormone system and hispatological alterations in the tyroid and liver.

*Table 2.7 Overview of No Observed Effect level (NOEL) and Lowest Observed Effect Level (LOEL) after oral administration of **BDE-99** congener or C-PentaBDE formulations. Bold values are the lowest LOEL or NOEL detected.**

PentaBDE	**Duration**	**Dose**	**NOEL mg/kg/ day**	**LOEL mg/kg/day**	**Endpoint**	**Species**	**Reference**
BDE-99	s.d	0.8 or 12.0 mg/kg	n.d.	0.8	Neurotoxicity Behaviour, motor activity level and learning	mouse	Eriksson *et al.* 2001
BDE-99	s.d	0.6, 6, or 30 mg/kg	n.d.	0.6	Developmental- and neurotoxicity Behaviour - hypoactive	mouse	Branchi *et al.* 2002
BDE-99	s.d	0.4, 0.8, 4.0, 8.0, or 16 mg/kg	**0.4**	**0.8**	Developmental- and neurotoxicity Behaviour	mouse	Viberg *et al.* 2004 Sand *et al.* 2004
BDE-99	s.d.	0,06 and 0,3 mg/kg to pregnant female	**n.d.**	**0,06**	Developmental- and neurotoxicity Behaviour (increased activity)	rat, F1 gen.	Kuriyama *et al.* 2005
BDE-99	s.d.	0,06 and 0,3 mg/kg to pregnant female	**0,06**	**0,3**	Reduced testis size and number of sperms	rat, F1 gen.	Kuriyama *et al.* 2005

Penta mix DE-71	30 d	0.01, 0.05, 0.1, 0.5, or 1.0 mg/kg/day	**1**	n.d.	Growth, food intake, hematology, histopatology Clinical chemistry	rat	Great lakes Chemical Corporation 1985
Penta mix DE-71	30 d	0, 3, 30, or 60 mg/kg/day	3	30	Liver weight, puberty, reproduction, liver enzymes, T_4-reduction	Male rat	Stoker *et al.* 2004
Penta mix DE-71	30 d	0, 3, 30, or 60 mg/kg/day	n.d.	3	T_4-reduction	Female rat	Stoker *et al.* 2004
Penta mix DE-71	35 d	0, 1, 10 or 30 mg/kg/day	1	10	T_4-reduction Liver enzymes	pregnant rat	Zhou *et al.* 2002, Zhou *et al.* 2001
Penta mix DE-71	90 d	0-0.44 mg/kg/day	n.d.	0.44	Liver enzymes	rat	Carlson 1980
Penta mix DE-71	90 d	0, 2,10, or 100 mg/kg/day	0-2	2-10	Hepatocyto-megali Tyreoidea hyperplasi	rat	Great lakes Chemical Corporation 1984

n.d. = not defined, s.d. = single dose

* Most of the studies are in line with the OECD test guidelines and for those are not, the quality of the study is assessed to be adequate.

The PBDE mixture known as DE-71 (71% bromine by mass, and containing BDE-47, BDE-99, BDE-100, BDE-153, BDE-154) delays the puberty and suppresses the growth of androgen-dependent tissues in male Wistar rat following a peri-pubertal exposure. These effects suggest that DE-71 may be either inducing steroid hormone metabolism or acting as an androgen receptor (AR) antagonist (Stoker *et al.* 2005).

Talsness *et al.* (2005) evaluated the effects of environmentally relevant concentrations (low doses) of BDE-99 on the female reproductive system in rats. Ultra structural changes compatible with altered mitochondrial morphology were observed in the ovaries of the F1 offspring. No statistically significant changes in ovarian follicle counts were observed. External and skeletal anomalies were detected in offspring (F2) from two different dams (F1) with early developmental exposure to 300 µg BDE-99/Ikg BW. Exposure to BDE-99 resulted in female reproductive tract changes in the F1 generation which were apparent at adulthood.

In utero exposure to a single low dose of BDE-99 disrupts neurobehavioral development and causes permanent effects on the rat male reproductive system apparent in adulthood (Kuriyama *et al.* 2005). Also in this study, the effects of developmental exposure to BDE-99 on juvenile basal motor activity levels and adult male reproductive health were assessed. The exposure to low-dose BDE-99 during development caused hyperactivity in the offspring at both time points (postnatal days 36 and 71) and permanently impaired spermatogenesis by the means of reduced sperm and spermatid counts. The doses used in this study of 60 and 300 µg/kg BW are relevant to human exposure levels, being approximately 6 and 29 times, respectively, higher than the highest level reported in human breast adipose tissue. This is the lowest dose of PBDE reported to date to have an *in vivo* toxic effect in rodents and supports the premise that low-dose studies should be encouraged for hazard identification of persistent environmental pollutants. The study by Viberg *et al.* (2004) shows that neonatal exposure to BDE-99 can induce developmental neurotoxic effects, such as changes in spontaneous behaviour (hyperactivity), effects that are dose-response related and worsen with age. The changes are seen in C57/B1 mice of both sexes. Spontaneous behaviour (locomotion, rearing, and total activity) was observed in two-, five- and eight-month-old mice.

2.4.3 Toxicity to humans

Several hazard assessments have been produced in EU and in US. The conclusions in the hazard assessments elaborated are qualified by the lack of sufficient knowledge of the toxicology of PentaBDE to enable assessment of the risk to humans (COT 2004, VKM 2005 and VCCEP 2003). The toxicological importance for humans of detected effects in laboratory animals is not clear. There is still not enough knowledge of the mechanisms, half-life and metabolism of PentaBDE in experimental animals and humans (VKM 2005).

The conclusion in the hazard assessment by the Committee on Food Safety in Norway was that the exposure through food and mother's milk is considerably lower than the observed NOEL in laboratory mammals (VKM 2005). It is believed that long-time exposure to lower doses of PentaBDE can cause health effects, since PentaBDE accumulates in the human body. Since the half-life of PentaBDE in humans is not known it is not possible today to conclude on long-time exposure effects. This is true even for the US situation, where levels may be 10-20 times those observed in Europe, but pharmacokinetics, toxicology, exposure and other critical data are lacking.

Vulnerable groups could however be pregnant women, embryos and infants, because of effects on the thyroid hormone balance, and the embryo's development of the central nervous system. During pregnancy, maintenance of the thyroid hormone balance is a physiological challenge. Embryos and infants are particularly vulnerable for reductions in thyroid hormone levels (VKM 2005). Infants are exposed to PentaBDE through the diets of their mothers' milk, since PentaBDE is lipophilic and accumulates in the milk (VKM 2005).

3. Synthesis of information

3.1 Summary

PentaBDE meets all of the Annex D screening criteria, and details are included (for the sake of completeness) in Table 3.2, below.

In the absence of production controls, the levels detected in humans, other species and the environment have been observed to rise steeply and this increase is observed in remote locations as well as closer to sites of production and use. In the US, where C-PentaBDE was in high use until recently and where it remains in such materials as polyurethane foam incorporated into consumer products, there has been a build-up in human tissue.

PentaBDE in soil or sediment is readily incorporated into the food chain and bioaccumulates in the fatty tissues of top predators, including humans.

There are toxicological studies of concern that demonstrate neurodevelopmental impacts in animals at low tissue levels that are of relevance to levels observed in populations. Such body burdens remain under close review.

An assessment of the impact of PBDEs on the environment was recently concluded by Environment Canada (2006), taking into account critical studies and lines of evidence that support the conclusion that these commercial substances entering the environment have or may have an immediate or long-term harmful effect on the environment or its biodiversity.

4. Concluding statement

Pentabromodiphenyl ether (C-PentaBDE) is a synthetic mixture of anthropogenic origin with no known natural occurrence. It can be concluded therefore that the presence of components of PFOS in the environment is the result of anthropogenic activities. Long range transport must be responsible for its presence in areas such as the Arctic region, remote from sites of production and release. PentaBDE degrades slowly in the environmernt and can bioaccumulate and biomagnify in mammals and piscivorous birds.

The phase out of C-PentaBDE production and use has led to a reduction in current use but many materials in use, such as polyurethane foams and plastics in electronic equipment, contain PentaBDE which is slowly released to the environment. This release will be accelerated at end-of-life of such materials, especially during recovery and recycling operations.

Although levels of PentaBDE in human blood and milk, and in other environmental species, are falling in Europe, they continue to increase in North America and the Arctic region.

Based on the information in this risk profile, C-PentaBDE, due to the characteristics of its components, is likely, as a result of long-range environmental transport and demonstrated toxicity in a range of non-human species, to cause significant adverse effects on human health and the environment, such that global action is warranted.

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別添9

商業用オクタブロモジフェニルエーテルの危険性の概要

分解性	蓄積性	人健康影響	動植物への影響
【生分解性】 分解せず（OECD TG 301D） 【半減期】 ・大気中：（Hexa-Nona BDE）30.4-161.0 日（OH ﾗｼﾞｶﾙとの反応）（AOPWIN）	【BCF（経鰓的生物濃縮係数）】 ・ｺｲ：（HexaBDPE）BCF=2580-5640 ・ｺｲ：（HeptaBDE）BCF＜1.1-3.8 ・ｺｲ：（OctaBDE）BCF＜9.5 ・ｺｲ：（c-OctaBDE）BCF）＜10-36 【BMF（経口的生物濃縮係数）】 ・飼育中のﾀｲｾｲﾖｳｻｹの餌に含まれる HeptaBDE 183 をﾓﾆﾀ-した結果、95%がｻｹに蓄積。 【BSAF（生物相-底質濃縮係数）】 ・2 種の淡水魚：（HexaBDE）BSAF =1,（HeptaBDE）BSAF =2 ・（BDE 154） BSAF =9.1±1.1	【反復投与毒性】 ラット（28 日）：10mg/kg/day で T4 濃度減少（octa-BDE:30.7%, hepta-BDE:45.1%,） 【催奇形性・発生毒性】 ウサギ（経口 妊娠 7～19 日）：5mg/kg/day で胎児毒性、15mg/kg/day で児の肝重量増加、体重増加量減少、骨形成遅延 マウス（生後 10 日目単回）：0.45mg/kg で 2、4 及び 6 月齢での異常行動並びに成長後の空間認識能・記憶の影響（BDE153）	アメリカチョウゲンボウ*Falco sparverius*：18.7μg PBDEs/egg 及び15.6±0.3 ng PBDEs/g bw/dayで29日間曝露した雛鳥において、PHA応答（T細胞媒介性免疫）が増大し、抗体媒介性反応が減少した。脾臓（胚中心の減少）、滑液嚢（アポトーシスの減少）、胸腺（マクロファージの増大）に構造的変化あり。脾臓の体細胞指標とPBDEs間及び滑液嚢の体細胞指標とBDE-47間に負の相関性あり。

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Stockholm Convention on Persistent Organic Pollutants
Persistent Organic Pollutants Review Committee
Third meeting
Geneva, 19–23 November 2007

Report of the Persistent Organic Pollutants Review Committee on the work of its third meeting

Addendum

Risk profile on commercial octabromodiphenyl ether

At its third meeting, the Persistent Organic Pollutants Review Committee adopted the risk profile on commercial octabromodiphenyl ether, on the basis of the draft contained in document UNEP/POPS/POPRC.3/14. The text of the risk profile, as amended, is set out below. It has not been formally edited.

K0763734 171207

COMMERCIAL OCTABROMODIPHENYL ETHER

RISK PROFILE

Adopted by the Persistent Organic Pollutants Review Committee
at its third meeting

November 2007

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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 (c-OctaBDE) – 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 (c-OctaBDE) 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.

Although the commercial OctaBDE seems to be not longer produced, releases during the service life of articles containing the commercial mixtures and at the end of article service life during disposal operations are still relevant. 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 c-OctaBDE components 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.

Assessing the bioaccumulation potential of c-OctaBDE components constitutes a main challenge in this risk profile. A high potential for bioaccumulation (including a moderate potential for bioconcentration) and food-web biomagnification has been demonstrated for HexaBDE; and it is fully in line with the reported elimination rates. The food-web biomagnification has been reported for HeptaBDE, although at a lower extend than expected from the Kow; this fact can be explained by metabolism resulting in a relatively short half-life (experimentally demonstrated and explained by the authors by debromination). The presence of Octa and NonaBDE in biota is well document but its potential for bioaccumulation from water and food is much lower than expected from their Kow. Reduced availability, metabolisms or both can justify this fact. The number of scientific papers demonstrating debromination of Octa to DecaBDE to other PBDEs is continuously increasing; this is critical for the assessment as would indicate that the supposed low bioaccumulation potential could be in reality the consequence of metabolism to bioaccumulative PBDEs. A quantitative estimation cannot be presented yet, but the debromination process has been already reported for aquatic organisms, mammals and birds. This is an active research field, and new results will need to be assessed by the POPRC as they appear in refereed literature.

Biota monitoring data in remote areas offer the best demonstration on the potential for long range transport of the c-OctaBDE components, Hexa and HeptaBDE. The role of atmospheric transport is confirmed based on its detection in alpine lakes. The potential for long range transport has been observed for DecaBDE. The lack of confirmation for Octa and NonaBDE may be related to the lower relative contribution and/or metabolism via debromination.

No relevant effects have been observed in aquatic, sediment and soil laboratory studies; however, 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 HexaBDE 153 of 2000 µg/kg lipid has been estimated based on a NOEL of 0.45 mg/kg; it should be noted that HexaBDE 153 concentrations close to these value have been found in several species and geographic sites and total PBDE concentrations frequently exceed this threshold by a large margin.

The evaluation of the human and environmental risk of commercial OctaBDE associated to its potential for long range transport must consider that 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. 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; although a quantitative evaluation of these effects in terms of hazard for human health and ecosystem is not possible based on the current level of information, it may become feasible soon if additional scientifically sound information is produced at a similar rate than in recent years.

Based on the existing evidence, it is concluded that the Hexa and HeptaBDE components of the octabromodiphenyl ether are likely, as a result of LRET, to lead to significant adverse human health and/or environmental effects, such that global action is warranted.

The increasing evidence related to debromination of Octa and Nona BDE into BDEs with POPs properties and considering that under Article 8, paragraph 7(a) of the Convention states that the lack of full scientific certainty shall not prevent a proposal from proceeding, it is concluded that the Octa and NonaBDE components of the octabromodiphenyl ether 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.

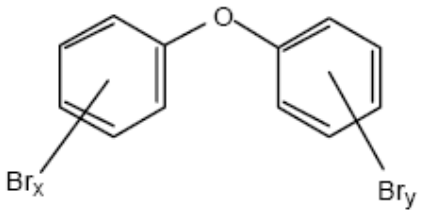
It is believed that little if any c-OctaBDE is produced since the major supplier located in North America stopped production in 2004. The commercially supplied OctaBDE was complex mixture consisting (as of 2001 within the EU Member States) typically of ≤0.5% Pentabromodiphenyl ether isomers, ≤12% Hexabromodiphenyl ether isomers, ≤45% Heptabromodiphenyl ether isomers, ≤33% OctaBDE isomers, ≤10% Nonabromodiphenyl ether isomers and ≤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: (x+y=8)



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

The abbreviations PBDE and BDE preceded by the number of bromine atoms (e.g. HeptaBDE) will be used in this document. The commercial mixtures will be identified by a c- (e.g. c-OctaBDE).

The compositions of the commercial polybrominated diphenyl ethers based on composite samples from the EU suppliers are shown in Table 1-1below. These are the substances that have been used in the recent tests and used as a basis for the EU risk assessment reports (RAR) for the three commercial substances. 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. This risk profile will focus on the series of Hexa, Hepta, Octa and Nona homologues, as the Penta and Deca homologues are covered by their respective commercial mixtures. 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:

- Hexabromodiphenyl ethers (benzene, 1,1'-oxybis-, hexabromo derivative; HexaBDE) (CAS No. 36483-60-0; IUPAC Nº between BDE-128 and BDE-169)
- Heptabromodiphenyl ethers (benzene, 1,1'-oxybis-, heptabromo derivative; HeptaBDE) (CAS No. 68928-80-3; IUPAC Nº between BDE-170 and BDE-193)
- octabromodiphenyl ethers (benzene, 1,1'-oxybis-, octabromo derivative; OctaBDE) (CAS No. 32536-52-0; IUPAC Nº between BDE-194 and BDE-205)
- Nonabromodiphenyl ethers (benzene, 1,1'-oxybis-, nonabromo derivative; NonaBDE) (CAS No. 63936-56-1; IUPAC Nº between BDE-206 and BDE-208)

Table 1-1. 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

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. 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 information was particularly scarce for Hepta- to NonaBDEs but there is an increasing interest in the scientific community for covering these congeners. A quantitative assessment is still not possible nowadays, but may become feasible soon if additional scientifically sound information is produced at a similar rate than in recent years,

1.2 Conclusion of the POP Review Committee of Annex D informat*ion*

The POPRC has evaluated Annex D information and has concluded that proposal fulfils the requirements of Article 8 and Annex D of the Convention (POPRC-2/6)

1.3 Data sources

The EU risk assessment report (EC, 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, Japan, 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 (BSEF), as well as during the consultation period. 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.

1.4 Status of the chemical under international conventions

- OSPAR Convention: OctaBDE is included in the list of selected substances for the OSPAR lists (no 236). Under the reviewed list, OctaBDE 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.

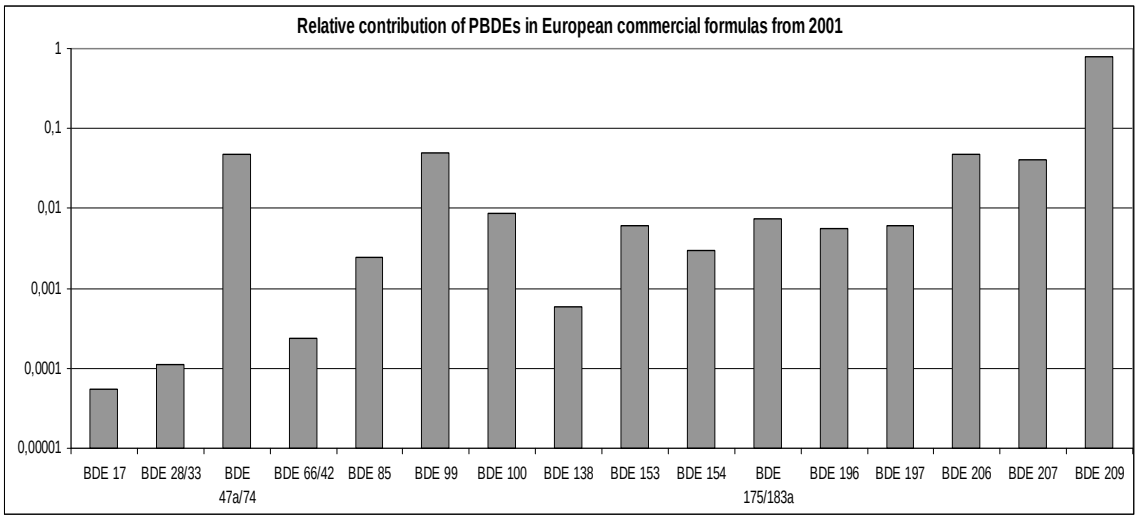
The polybrominated diphenyl ethers in general are used as flame retardants of the additive type. They are physically combined with the material being treated rather than chemically combined (as in reactive flame retardants). This means that there is the possibility that the flame retardant may diffuse 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 1-1 presents the calculations for European commercial products in 2001.

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.

Figure 1-1. 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.



Morf et al., (2002) reported for Switzerland 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 (EC, 2003).

Gerecke et al. (2005) reported the degradation of NonaBDE 206 and 207 and DecaBDE to OctaBDEs under anaerobic conditions using sewage sludge innoculum; 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. However, in the atmosphere, Hexa to NonaBDEs are 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 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

The bioaccumulation potential differs strongly among the components of the commercial mixture. For facilitating, the assessment, the different bioaccumulation processes will be presented independently.

2.2.2.1. Bioconcentration from water

Bioconcentration from water is considered relevant only for HexaBDE. The UK has re-analyzed the CITI (1982) bioconcentration data and suggests BCFs of up to ~5,640 l/kg and ~ 2,580 l/kg for components D and E (both HexaBDE).

Bioconcentration factors were reported (EC, 2003) for carp. Assuming 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; for HeptaBDE about <1.1-3.8 and for c-OctaBDE about <10-36. These BCF values are lower

than would be expected from the substance's octanol-water partition coefficients. This can be explained by a reduced bioavailability, metabolisms or both.

2.2.2.2 Bioaccumulation and biomagnification from food exposures.

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.

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).

The potential for biomagnification has been demonstrated for Hexa and HeptaBDE (Burreau et al., 2004; 2006; Sormo et al., 2006; Tomy et al., 2004), and more recently suggested for the DecaBDE (Law et al., 2006).

Food-web biomagnification was not been observed for Octa and NonaBDE in an aquatic ecosystem, but the congeners were detected in biota from zooplanckton to fish species (Burreau et al. 2006).

2.2.2.3. Bioaccumulation from sediment exposures

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 ± 0.016 $days^{-1}$.

2.2.2.4. Toxicokinetics and relevance of metabolisms

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.

Stapleton et al. (2004) in a dietary study on carps found depuration rates of 0.051± 0.036 $days^{-1}$ and assimilation efficiencies of 4% ± 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. I*n vitro* studies have demonstrated the microsomal debromination in fish (Stapleton et al. (2006).

Tomy et al (2004) exposed juvenile lake trout (*Salvelinus namaycush*) to three dietary concentrations of 13 BDE congeners (3-10 Br atoms) in the laboratory for 56 days, followed by 112 days of clean food. Half-lives (t1/2's) for some BDE congeners (e.g., BDE-85 and -190) were much lower than expected based on their Kow, whereas t1/2's of other BDE congeners (e.g., BDE-66, -77, -153, and -154) were much longer than anticipated based on Kow. This was explained by debromination. The detection of three BDE congeners (an unknown PentaBDE, BDE-140, and an unknown HexaBDE) in the fish that were not present in the food or in the control fish provided further evidence for the debromination of BDEs.

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).

The debromination of PBDEs has also indicated in mammals, e.g. for a c-PentaBDE formulation in mice (Qiu et al., 2007) and for DecaBDE in cows (Kierkegaard et al., 2007).

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 in a 36 days study.

Van den Steen et al. (2007) used silastic implants to expose European starlings (Sturnus vulgaris) to DecaBDE209 and found Octa- (BDE196, BDE197) and NonaBDEs (BDE206, BDE207, BDE208) in muscle and liver in addition to DecaBDE209, resulting in the first indications of debromination in birds.

2.2.2.5 Integrated assessment of the bioaccumulation potential.

A high potential for bioaccumulation (including a moderate potential for bioconcentration) and food-web biomagnification has been demonstrated for HexaBDE; and it is fully in line with the reported elimination rates.

The food-web biomagnification has been also demonstrated for HeptaBDE, although at a lower extend than expected from the Kow; this fact can be explained by metabolism resulting in a relatively short half-life (experimentally demonstrated and explained by the authors by debromination).

The presence of Octa and NonaBDE in biota is well document but its potential for bioaccumulation from water and food is much lower than expected from their Kow. Reduced availability, metabolisms or both can justify this fact. The number of scientific papers demonstrating debromination of Deca-, Nona, and Octa- BDE to other PBDEs is continuously increasing; this is critical for the assessment as would indicate that the supposed low bioaccumulation potential could be in reality the consequence of metabolism to bioaccumulative PBDEs. A quantitative estimation cannot be presented yet, but the debromination process has been already reported for aquatic organisms, mammals and birds.

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, Japan info) is considered the best demonstration for the potential for long range transport of these chemicals. As debromination in 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, since Hexa to Deca congeners have similar atmospheric transport characteristics in terms of gas-partitioning and reactivity it is evidence of long range transport for DecaBDE and is indirect evidence of long range transport 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, 2003). 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

2.3.1 Atmosphere

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.

Bergander et al. (1995) analyzed air samples from two areas of Sweden remote from industry, HexaBDE and HeptaBDE were found in the particulate phase samples.

In a monitoring study carried out in coastal areas of Korea over one-year period, twenty individual PBDE congeners were found in atmospheric samples collected from urban, suburban and rural sites. DecaBDE (BDE 209) was the predominant congener (<93%) The depositional fluxes ranged from 10.1 to 89.0 µg/m2/year (Moon et al., 2007a). In northwest China, the measurements of total PBDEs (8.3 ± 4.0 pg/m3) in the samples collected at the Waliguan Baseline Observatory (April to May, 2005) were at comparable concentration levels with other remote areas (Cheng et al., 2007).

PBDEs have also been detected over the Indian Ocean (mean concentration of 2.5 pg/m3) and along the coastal line of Java, Indonesia (values of 15 pg/m3). Air back trajectory analysis is suggested in relation to the potential of PBDEs for long-range atmospheric transport from remote regions of areas more industrialized (Wurl et al. 2006).

Wang et al. (2005,) report atmospheric concentrations for c-OctaBDE components for a large number of remote locations, and additional information about the presence of Penta to HeptaBDE congeners in air at several locations can be found in the review paper by de Wit et al. (2006).

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 EC (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.

There is additional information on concentrations of c-OctaBDE components (HexaBDEs 153 and 154) in the dissolved phase in water in a study by Law et al. (2006).

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; Environment Agency UK, 1997)). The highest levels were in sediments downstream from a warehouse where c-DecaBDE was stored. 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.

Historical trends of PBDEs in sediments have been determined in the Lake of Ellasjøen, Norwegian Arctic, where contamination is due to both atmospheric and biological transport. Maximum level of PBDEs was detected in 2001 (0.73 ng/g dw) (Evenset et al., 2007). Marvin et al. 2007, have reported temporal trends in PBDEs in Niagara river suspended sediments from 1988 to 2004. Prior to 1988, PBDEs (sum of 16 congeners including DecaBDE) were generally detected at low-ppb concentrations, but showed a trend toward increasing concentrations over the period 1980–1988. After 1988, PBDE concentrations in the Niagara River showed a more rapidly increasing trend (maximum of approximately 35 ng/g in 1995). DecaBDE was the predominant congener detected, and a similar situation has been observed in Europe (Eljarrat et al., 2005), and Asia (Moon et al. 2007b).

The study by Law et al. (2006) provides additional information on concentrations of c-OctaBDE components (HexaBDEs 153 and 154) for sediments at a background location.

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.

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.

Temporal, spatial, and interspecific trends in PBDEs were determined in eggs of marine and freshwater bird species from the province of British Columbia, Canada. Temporal trends in the Fraser River estuary, 1983-2002, were examined by analysis of eggs of great blue herons (Ardea herodias) and from the Strait of Georgia marine ecosystem, 1979-2002, in eggs of double-crested cormorants (Phalacrocorax auritus). PBDEs increased exponentially with a doubling time of 5.7 years in eggs of both herons and cormorants. The PBDE pattern was relatively consistent in most years and sites, with BDEs 47 > 100 > 99 > 153 > 154 > 28 > 183. This was interpreted as evidence of technical PentaBDE formulations as primary sources of the contamination, with the OctaBDE formulations as secondary. Higher resolution analysis of a subsample of the eggs revealed the presence of up to nine other congeners, including BDE209 (range: 0.9-1.8 microg/kg), indicating exposure and uptake of DecaBDE sourced congeners in North American foodchains (Elliot et al., 2005)

A recent study (Burreau et al., 2006) has demonstrated the presence of Hexa to NonaBDE in biota (zooplankton, sprat, herring and salmon) from the Baltic Sea and Northern Atlantic.

2.3.7 Humans

EC (2003) presents some information on the levels of components of c-OctaBDE measured in human samples including human milk, blood, and adipose tissue. 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 (18 congeners from BDE17 to BDE-183) found in Australian human milk were lower than those reported from North America but higher than those reported from Europe and Asia

Thomsen et al., 2007, investigated the levels of PBDEs in 21 pooled serum samples archived from the general Norwegian population (from 1977 to 2003). In serum from men (age 40–50 years) the sum of seven PBDE congeners (28, 47, 99, 100, 153, 154 and 183) increased from 1977 (0.5 ng/g lipids) to 1998 (4.8 ng/g lipids). From 1999 to 2003 the concentration of PBDEs seems to have stabilised.

Fernandez et al., 2007, have reported a study of the detection of PBDEs in the adipose tissue of women from Spain. Mean ∑PBDE (BDE 28, 75, 71, 47, 66, 77, 100, 119, 99, 85, 154, 153, 138, and 183) levels were 3.85 and 0.36 ng/g of lipid, respectively. Among PBDEs, congeners 153, 47, 183, 99, and 100 were the most frequent and abundant and together constituted 96% of the total amount of PBDEs in adipose tissue. Concentrations of PBDEs in this population were similar to those reported in other parts of Spain and in Swedish and Belgium populations but lower than those found in other Western countries.

PBDEs were measured in samples of human blood serum taken from 23 donors in Wellington, New Zealand. Concentrations expressed as the sum of congeners 47, 99, 100, 153, 154, and 183 (∑PBDE) were – at an average of 7.17 ng ∑PBDE g $(\text{lipid})^{-1}$ – within the range reported for human tissues in Europe, but lower than in Australia and North America (Harrad et al., 2007).

Based on the measured PBDE levels detected in various meat, fish and dairy food products, an average daily dietary intake estimate of PBDEs was calculated in a study carried out in Belgium. PBDE intake calculations were estimated between 23 and 48 ng/day of total PBDEs. Fish is the major contributor to the total daily PBDE-intake (around 40%) due to the high PBDE levels in this type of food, although it is only a minor constituent of the Belgian diet. Meat products account for around 30% of the total dietary intake of PBDEs. Dairy products and eggs contribute to a lesser degree (less than 30%, Voorspoels et al., 2007).

Schuhmacher et al., 2007 have carried out an study to compare levels of PBDEs due to dietary intake and population living near a hazardous waste incinerator (HWI), in Spain. This study suggests that dietary intake is more relevant for human exposure to PBDEs than living near the HWI. Dietary intake of PBDEs for standard adult women were 72 and 63 ng/day for PBDEs, for residents in urban and industrials areas, respectively. Mean PBDE concentrations were 2.2 and 2.5 ng/g fat for women living in urban and industrial zones, respectively. Similar results have also been reported in a study carried out in Korea (Lee et al., 2007)

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 including biota. There is no sufficient information for assessing these processes in quantified terms. The exposure route is mainly via food. 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.

Measured levels of Hexa and Hepta 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) 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. The presence of Hexa and HeptaBDE in fish from remote alpine lakes in Switzerland (Schmid et al., 2007) reported to be related to atmospheric deposition confirms the potential for atmospheric long-range transport. Hexa to NonaBDE have been found in salmon in the Atlantic Ocean west of Iceland (Burreau et al. 2006).

Despite its large molecular size, the evidence demonstrates the capability of c-OctaBDE components to cross the cellular membranes and to accumulate in biota. 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

2.4.1. Experimental studies

2.4.1.1. Aquatic Organisms

The EU Risk Assessment report (EC, 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.

2.4.1.2. 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.

2.4.1.3. 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).

2.4.1.4. 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 pentoxyresorufin *O*-deethylase (PROD) activities in rats receiving 10 or more mg/kg bw/day of commercial OctaBDE (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.,2001)
- Eriksson et al. (2002) 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 PentaBDE congeners-47, -99, -100, and -153 dissolved in safflower oil (18.7 µg 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).

Although these studies do not allow a quantitative assessment, they indicate the need for addressing long-term and delayed effects, as well as specific mechanisms of action, in the evaluation of potential health and ecosystem adverse effects.

2.4.2. 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

A quantitative evaluation of the specific risks of c-OctaBDE is not possible due to the presence of its components in commercial Penta- and Deca mixtures, and the lack of information; this include the absence of information for supporting quantitative assessments of the role on debromination and 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. Australia and Canada have reported quantitative risk assessments for health and for the environment based on risk quotients and margins of safety suggesting a potential risk. The evaluations do not cover expected conditions in remote areas but are useful in the overall assessment (Environment Canada, 2006; NICNAS, 2007).

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, including those present in c-PentaBDE and c-DecaBDE..

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 depends on the level of bromination. HexaBDE shows a significant potential for bioconcentration and biomagnification; HeptaBDE biomagnifies through the food web but at a lower extend than that expected from the Kow. Octa and NonaBDE have been found in biota but no food-web biomagnification has been observed. Metabolisms and/or reduced bioavailability explain the divergences between observations and Kow predictions. The contribution of metabolism through debromination into other BDEs is supported by and increasingly amount of scientific evidence.

Biota monitoring data in remote areas cover Hexa and HeptaBDE and offer the best demonstration on the potential for long range transport of c-OctaBDE components. Theoretically this presence could also be explained by the transport of DecaBDE and its subsequent debromination. However, it is not realistic to assume that DecaBDE debromination may explain the process without additional transport from other congeners. The role of atmospheric transport is confirmed 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; but 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,.

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 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 summed. McDonald (2005) estimated a critical body burden for HexaBDE 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 HexaBDE 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.

The degradation of PBDEs in the environment and biota is a key issue as higher congeners are converted to lower, and possibly more toxic, congeners. This possibility has been demonstrated for debromination of DecaBDE and several c-OctaBDE components (see references above) but the extent to which different PBDEs can be degraded under various conditions, the role of metabolism in addressing the bioaccumulation potential, and the identity of any lower congeners that may be produced, is an active research field. New results will need to be assessed by the POPRC as they appear in refereed literature.

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

4. Concluding statement

The evaluation of the human and environmental risk of commercial OctaBDE associated to its potential for long range transport must consider that 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. 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, it is concluded that the Hexa and HeptaBDE components of the commercial octabromodiphenyl ether are likely, as a result of LRET, to lead to significant adverse human health and/or environmental effects, such that global action is warranted.

The increasing evidence related to debromination of Octa and Nona BDE into BDEs with POPs properties and considering that under Article 8, paragraph 7(a) of the Convention states that the lack of full scientific certainty shall not prevent a proposal from proceeding, it is concluded that the Octa and NonaBDE components of the commercial octabromodiphenyl ether 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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