Screening Assessment Report on

Hexabromocyclododecane

Chemical Abstracts Service Registry Number
3194-55-6

Environment Canada
Health Canada

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SYNOPSIS

Pursuant to section 74 and 68(b) of the *Canadian Environmental Protection Act, 1999* (CEPA 1999), the Ministers of the Environment and of Health have conducted a screening assessment of hexabromocyclododecane (HBCD). HBCD having Chemical Abstracts Service Registry Number[1] 3194-55-6 was one of the substances on the Domestic Substances List (DSL) selected for a pilot project for screening assessments. During the categorization of the DSL, the substance was identified as a high priority for screening assessment as it met the criteria for persistence, bioaccumulation and inherent toxicity to aquatic life. It is recognized that the CAS Registry contains more than one number for HBCD (e.g., CAS RN 25637-99-4 refers to HBCD where the bromine substituents are not numbered). In this assessment, all available relevant data and studies that are of reliable quality were considered equally, and thus the assessment findings and scope apply to HBCD in general and are irrespective of the CAS RN.

The primary application of HBCD is as a flame retardant in polystyrene foams that are used as thermal insulation materials in the construction industry. A second application is the flame retarding of textiles for usage in residential and commercial upholstered furniture, transportation seating, wall coverings and draperies. Minor uses include addition to latex binders, adhesives, and paints and to high-impact polystyrene and styrene-acrylontrile resins for electrical and electronic equipment.

For the years and continents having available data since 2000, increases in the demand for HBCD have been reported. Global demand for HBCD was estimated at 16 700 tonnes in 2001, representing 8.2% of total demand for brominated flame retardants that year. Results from a section 71 *Notice with Respect to Certain Substances on the Domestic Substances List (DSL)* conducted for the year 2000 indicated that HBCD was not manufactured in Canada at that time. Amounts imported into the country in that yearwere in the range of
100 000 to 1 000 000 kg.

Environment
Monitoring studies document the presence of HBCD in many environmental media, with highest concentrations reported near urban/industrial sources. Analyses of sediment core samples show a clear trend of increasing concentrations of HBCD since the 1970s, confirming stability in deep sediments for periods of more than 30 years. As well, there is evidence of increasing HBCD levels in North American and European biota, both within species and along food chains.

Measured and modelled data indicate that HBCD will undergo primary degradation under some conditions; however, ultimate degradation in the environment is a slow process.

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Laboratory studies conducted using water, sediment, soil and sludge confirm the presence of primary degradation products, including 1,5,9-cyclododecatriene, a substance that, based on laboratory testing, is not readily biodegradable, is potentially toxic to aquatic life (with measured and predicted median lethal concentrations (LC_{50}s) < 1 mg/L) and is potentially bioaccumulative in aquatic organisms.

Considered together, the lines of evidence from degradation studies and sediment monitoring data establish that HBCD can remain stable in the environment for relatively long periods. The substance therefore meets the criteria for persistence in water, soil, and sediment as outlined in the *Persistence and Bioaccumulation Regulations* under CEPA 1999 (i.e., half-life in water and soil of 182 days or more, and half-life in sediment of 365 days or more). Additionally, HBCD meets the criteria for persistence in air set out in the same regulations (i.e., half-life of two days or more, or being subject to atmospheric transport from the source to a remote area), based on a predicted atmospheric half-life of 2.13 days and evidence of occurrence in regions considered remote from potential sources, including the Arctic.

The weight of experimental and predicted data indicate that HBCD meets the criteria for bioaccumulation as specified in the *Persistence and Bioaccumulation Regulations* under CEPA 1999 (i.e., bioaccumulation factors [BAFs] or bioconcentration factors [BCFs] of 5000 or more) and is likely to have significant bioaccumulation potential in the environment. BCFs of 18 100 (rainbow trout) and 12 866 (steady state, fathead minnow) were obtained in laboratory studies. Field studies show evidence that bioaccumulation and biomagnification are occurring within food webs.

HBCD has demonstrated toxicity in both aquatic and terrestrial species, with significant adverse effects on survival, reproduction and development reported in algae, daphnids and annelid worms. Recent studies indicate potential impacts on the normal functioning of liver enzymes and thyroid hormones in fish.

Combustion of HBCD under certain uncontrolled conditions may lead to production of polybrominated dibenzo-*p*-dioxins (PBDDs) and dibenzofurans (PBDFs). Trace levels of these compounds and their precursors have been measured during combustion of flame-retarded polystyrene materials containing HBCD. These transformation products are brominated analogues of the Toxic Substances Management Policy Track 1 polychlorinated dibenzofurans and dibenzo-*p*-dioxins.

The widespread presence of HBCD in the environment warrants concern in light of strong evidence that the substance is environmentally persistent and bioaccumulative. In addition, the analysis of risk quotients determined that current HBCD concentrations in the Canadian environment have the potential to cause adverse effects in populations of pelagic and benthic organisms but are currently unlikely to result in direct adverse effects to soil organisms and wildlife.

Based on the information in this screening assessment, it is concluded that HBCD is entering the environment in a quantity or concentration or under conditions that have or

may have an immediate or long-term harmful effect on the environment or its biological diversity.

Human Health

Exposures of the general population of Canada to HBCD may occur through oral and inhalation routes. Known sources of human exposure to HBCD include environmental media (ambient air, water, soil), household dust, indoor air, human milk, and HBCD-treated consumer products. HBCD may potentially be released from the matrix of a product over time through abrasion and usage, as it is not covalently bound.

The human health hazard risk characterization for HBCD was based primarily upon the assessment of the European Union, with more recent data taken into consideration. The results of a limited database indicate that HBCD does not have genotoxic potential *in vitro* or *in vivo*, was not carcinogenic, and did not cause systemic toxicity in a chronic oral feeding study. The critical study for the characterization of risk to human health was a two-generation reproductive toxicity assay, with reported effects including decreased fertility and a weak hypothyroidism in pregnant dams and, at high doses, reversible hyperthyroidism in offspring from weaning to adulthood. Additionally, because of potential developmental effects, it was considered appropriate to consider a behavioural lowest-observed-adverse-effect level (LOAEL) for infants and children. The highest upper-bounding estimated intake of HBCD is expected to be in infants from ingestion of human milk and the mouthing of consumer products. A comparison of these exposure estimates with the critical effect levels identified in the two-generation reproductive toxicity assay and the behavioural LOAEL results in margins of exposure that are considered adequate to address uncertainties in the health effects and exposure databases. Based on the available information, it is concluded that HBCD is not entering the environment in a quantity or concentration or under conditions that constitute or may constitute a danger in Canada to human life or health.

Conclusion

Based on the information available for environment and human health considerations, it is concluded that HBCD meets one or more of the criteria set out in section 64 of CEPA 1999.

In addition, it is concluded that HBCD meets the criteria for persistence and bioaccumulation potential as set out in the *Persistence and Bioaccumulation Regulations*, and its presence in the environment results primarily from human activity, and it is not a naturally occurring radionuclide or a naturally occurring inorganic substance; therefore, it meets the criteria set out in subsection 77(4) of CEPA and is proposed for the implementation of virtual elimination under subsection 65(3) of CEPA 1999.

Where relevant, research and monitoring will support verification of assumptions used during the screening assessment and, where appropriate, the performance of potential control measures identified during the risk management phase.

Introduction

This screening assessment was conducted pursuant to section 74 of the *Canadian Environmental Protection Act, 1999* (CEPA 1999) (Canada 1999). This section of the Act requires that the Minister of the Environment and the Minister of Health conduct screening assessments of substances that satisfy the the categorization criteria set out in section 73 of the Act, in order to determine whether they meet or may meet the criteria set out in section 64 of the Act.

Based on the information obtained through the categorization process, the Ministers identified a number of substances as high priorities for action. These include substances that:

- met all of the ecological categorization criteria, including persistence (P), bioaccumulation potential (B) and inherent toxicity to aquatic organisms (iT), and were believed to be in commerce; and/or
- met the categorization criteria for greatest potential for exposure (GPE) or presented an intermediate potential for exposure (IPE) and had been identified as posing a high hazard to human health based on classifications by other national or international agencies for carcinogenicity, genotoxicity, developmental toxicity or reproductive toxicity.

The substance, hexabromocyclododecane; HBCD; Chemical Abstracts Service Registry Number [CAS RN] 3194-55-6), was one of 123 substances on the Domestic Substances List (DSL) selected for a pilot project for screening assessments. During the categorization of the DSL, the substance was identified as a high priority for screening assessment, as it met the criteria for persistence, bioaccumulation and inherent toxicity to aquatic life.

Although HBCD was determined to be a high priority for assessment with respect to the environment, it did not meet the criteria for GPE or IPE and high hazard to human health based on classifications by other national or international agencies for carcinogenicity, genotoxicity, developmental toxicity or reproductive toxicity. Therefore, this assessment focuses principally on information relevant to the evaluation of ecological risks.

Screening assessments focus on information critical to determining whether a substance meets the criteria as set out in section 64 of CEPA 1999. Screening assessments examine scientific information and develop conclusions by incorporating a weight-of-evidence approach and precaution.

This final screening assessment includes consideration of information on chemical properties, hazards, uses and exposure. Data relevant to the screening assessment of this substance were identified in original literature, review and assessment documents, stakeholder research reports and from recent literature searches. For the ecological assessment, information obtained as of July 2010 was considered for inclusion in this document, and literature searches up to January 2010 were considered for the human

health assessment. Key studies were critically evaluated; modelling results may have been used to reach conclusions. In addition, an industry survey on HBCD was conducted in 2000 through a *Canada Gazette* notice issued under section 71 of CEPA 1999. This survey collected data on the Canadian manufacture, import, uses, and releases of HBCD (Environment Canada 2001).

Evaluation of risk to human health involves consideration of data relevant to estimation of exposure (non-occupational) of the general population, as well as information on health hazards (based principally on the weight-of-evidence assessments of other agencies that were used for prioritizing the substance). Decisions for human health are based on the nature of the critical effect and/or margins between conservative effect levels and estimates of exposure, taking into account confidence in the completeness of the identified databases on both exposure and effects, within a screening context.[2] The screening assessment does not represent an exhaustive or critical review of all available data. Rather, it presents a summary of the critical information upon which the conclusion is based.

This final screening assessment was prepared by staff in the Existing Substances Programs at Health Canada and Environment Canada and incorporates input from other programs within these departments. The ecological component of this assessment has undergone external written scientific peer review/consultation, and comments received were considered in the production of this report. Comments on the technical portions relevant to human health were received from Dr. Bernard Gadagbui, Toxicology Excellence for Risk Assessment, Dr. Michael Jayjock, The LifeLine Group, and Dr. Susan Griffin, U.S. Environmental Protection Agency (EPA). Additionally, the draft of this screening assessment was subject to a 60-day public comment period. Although external comments were taken into consideration, the content and conclusions of the screening assessment remain the responsibility of Health Canada and Environment Canada.

The critical information and considerations upon which this assessment is based are summarized below.

[2] A determination of whether one or more of the criteria of section 64 are met is based upon an assessment of potential risks to the environment and/or to human health associated with exposures in the general environment. For humans, this includes, but is not limited to, exposures from ambient and indoor air, drinking water, foodstuffs, and the use of consumer products. A conclusion under CEPA 1999 on this substance is not relevant to, nor does it preclude, an assessment against the hazard criteria specified in the *Controlled Products Regulations*, which is part of the regulatory framework for the Workplace Hazardous Materials Information System [WHMIS] for products intended for workplace use. Similarly, a conclusion based on the criteria contained in section 64 of CEPA 1999 does not preclude actions being taken under other sections of CEPA or other Acts.

Substance Identity

For the purposes of this document, this substance will be referred to as HBCD, which has been derived from the chemical name hexabromocyclododecane.

The chemical structures of HBCD are shown in Table A-1 (Appendix A). HBCD is a cyclo-aliphatic bromide produced by the bromination of cyclododecatriene (CAS RN 27070-59-3: Mack 2004). The resulting technical product is primarily a mixture of three diastereomers (stereoisomers), designated alpha (α), beta (β) and gamma (γ) and defined according to their order of elution from a reverse-phase high-performance liquid chromatography column. Trace amounts of two other diastereomers—delta (δ) and epsilon (ε)—have also been reported, and in principle up to 16 stereoisomers, including six diastereomeric pairs of enantiomers and four meso forms, are possible based on the structural characteristics of the substance (Heeb et al. 2004; Law et al. 2005). The α-, β- and γ-isomers have been observed in chiral pairs, while no optical rotation was detected for the δ- and ε-stereoisomers; therefore, these have been tentatively assigned as meso forms (Law et al. 2005).

Commercial HBCD is typically composed of approximately 80–85% γ-isomer, 8–9% α-isomer and 6% β–isomer (ACCBFRIP 2005). Four commercial grades are available—low melt, medium range, high melt and thermally stabilized—with each containing different proportions of the three stereoisomers (Tomy et al. 2004a). Final use determines the grade of HBCD selected.

It is recognized that the CAS Registry contains more than one number for HBCD (e.g., CAS RN 25637-99-4). In this assessment, all available relevant data and studies that are of reliable quality were considered equally, and thus the assessment findings and scope apply to HBCD in general and are irrespective of the CAS RN.

Physical and Chemical Properties

Table A-2 contains experimental and modelled physical and chemical properties of HBCD that are relevant to its environmental fate. HBCD is characterized by a high octanol-water partition coefficient (log K_{ow}) and a high organic carbon-water partition coefficient (log K_{oc}) (specifically, a log K_{ow} of 5.625–5.81, and an extrapolated log K_{oc} of 5.097), low water solubility (65.6 µg/L, sum of solubilities of the three major diastereomers), and low vapour pressure (6.27×10^{-5} Pa at 21°C).

Sources

There is no reference in the published literature to the natural occurrence of HBCD in the environment. Sources of exposure to HBCD are anthropogenic.

Results from an industry survey, as reported under section 71 of CEPA 1999, show that HBCD was not manufactured above reporting thresholds in Canada in 2000, although amounts in the range of 100 000–1 000 000 kg of the substance were imported into Canada in that year (Environment Canada 2001).

Globally, HBCD is a U.S. high production volume chemical (HPV) and is produced in quantities above 16 700 tonnes/year (Heeb et al. 2005). Annual U.S. production/import volumes were between 10 and 50 million pounds (4535–22 679 tonnes) for the reporting years 1994, 1998 and 2002 (US EPA 2002). Global demand for HBCD was estimated at 16 700 tonnes in 2001, representing 8.2% of the total brominated flame retardant demand for that year and placing HBCD third in global production after tetrabromobisphenol A and decabromodiphenyl ether (BSEF 2005). Global demand for HBCD increased the following two years, and was estimated at 21 400 tonnes/annum in 2002, and at 22 000 tonnes/annum in 2003 (BSEF 2006). Major markets in 2001 were Europe (9500 tonnes), where HBCD is classified as a high production volume chemical, Asia (3900 tonnes) and the Americas (2800 tonnes). Estimates for 2007 show European annual consumption had increased to 11 000 tonnes (ECHA 2008).

Uses

HBCD is used primarily as a flame retardant in expanded (EPS) and extruded (XPS) polystyrene foams that are used as thermal insulation materials in the construction industry (ACCBFRIP 2005). EPS and XPS are incorporated into materials such as boardstock for insulation of industrial and residential buildings (Great Lakes Chemical Corporation 2005a). EPS is also used to insulate coolers and as a packaging material (2007 email from an Environmental Quality Manager of the importing company to Existing Substances Branch, Environment Canada; unreferenced). Foam HBCD levels in Europe are higher than used in Canada to meet European fire performance standards. For European foams, typical HBCD levels are around 0.67% in EPS and 1–3% in XPS (EU RAR 2008). HBCD levels in polystyrene foams in Canada are typically from 0.5 to 1% (EPSMA et al. 2009).

A second application is the flame retarding of textiles, in which HBCD is applied in a typical concentration of 6–15% to the back of upholstery fabric encapsulated in a polymer (ACCBFRIP 2005). Common end products from this application include residential and commercial furniture, upholstery seating in vehicles, draperies and wall coverings (FRCA 1998). HBCD may be added to latex binders, adhesives and paints to make them flame retardant (Albemarle Corporation 2000a; Great Lakes Chemical Corporation 2005a). It may also be added to high-impact polystyrene used in electrical and electronic equipment, such as audiovisual equipment, although this application is not common (BSEF 2003). HBCD is not used in electronic housings in products such as television set and computers, which are required to meet higher flame retardancy standards than other products (ACCBFRIP 2005).

The primary uses of HBCD in Canada (i.e., in EPS, XPS and textiles) are consistent with the above-noted global and European use patterns. The European Union Risk Assessment Report on HBCD (EU RAR 2008) indicates some examples of end-use products containing HBCD:

- flat and pile upholstered furniture (residential and commercial furniture)
- upholstery seating in transportation, draperies and wall coverings
- bed mattress ticking
- interior textiles, e.g., roller blinds
- automobile interior textiles
- car cushions
- insulation boards used in building construction, e.g., used in walls, cellars, indoor ceilings, inverted roofs
- insulation boards used to prevent frost heaving of roads and railway embankments
- packaging material
- electrical and electronic equipment, e.g., distribution boxes for electrical lines
- video cassette housings
- polyvinyl chloride wire, cable and textile coating
- protective paints

HBCD is an additive-type flame retardant. Additive flame retardants are physically combined with the material being treated rather than being chemically bonded as is the case with reactive flame retardants; therefore, there is potential for migration, at least to some extent, within and from the polymer matrix. A number of factors act to constrain migration of HBCD within polymers, including the low vapour pressure, low water solubility and a high predicted K_{oc} of the substance (2007 email from an Environmental Quality Manager of the importing company to Existing Substances Branch, Environment Canada; unreferenced). Nevertheless, some HBCD at the surface of a polymer or product could be released into the environment during use or disposal of the product.

Small quantities of organic peroxides are commonly added to HBCD to enhance performance efficiency (US NRC 2000), and thermally stabilized grades of HBCD are required for processing temperatures above 200°C. Dicumyl peroxide can be used in expanded polystyrene as a synergist with HBCD to enhance the flame retardant activity (2007 email from an Environmental Quality Manager of the importing company to Existing Substances Branch, Environment Canada; unreferenced).

Sources of Release

Release of HBCD into the environment may occur during production and manufacturing, processing, transportation, use, improper handling, improper storage or containment, point-source discharges, migratory releases from manufactured product usage and from

disposal of the substance or products containing the substance. HBCD may be released to air, water, soil and sediment.

Since production of HBCD is not known to be occurring in Canada, potential releases from this source were not considered further in this assessment. HBCD released during processing activities may enter the air or be discharged to wastewater. As major uses are associated with polymers for the construction industry and with textiles, most releases would likely be to urban and industrial areas. Releases from processing of polystyrene foams are expected to be much lower than those associated with the application of HBCD-containing backcoat to textiles (2007 email from an Environmental Quality Manager of the importing company to Existing Substances Branch, Environment Canada; unreferenced). In Europe, for example, although polystyrene foam applications represent the large majority of HBCD use, with regard to the total mass of HBCD released from processing and application service life, polystyrene foam applications represent a smaller source than textile applications (EU RAR 2008). Furthermore, releases of HBCD are more likely to be associated with processing activities involving the direct use and handling of HBCD compared to those activities simply employing HBCD-containing materials (2010 letter from EPSMA to Government of Canada; unreferenced). Whether present in air as dust particles or sorbed to particulates, the substance can be removed from air by settling. HBCD released to wastewater would likely be transported to a treatment facility. High octanol-water and organic carbon-water partition coefficients (log K_{ow} of 5.625–5.81, estimated log K_{oc} of 5.097) suggest that most HBCD entering a treatment plant sequesters into sludge; however, small amounts (e.g., 1260 ng/L, Deuchar 2002) have been measured in final effluents discharged to receiving waters. Most HBCD entering surface waters would be expected to partition into bed sediments, after sorption to suspended particulates in the water and subsequent settling. Release into the soil could occur during the application of biosolids to agricultural and pasture lands.

Over the service life of end products, HBCD may be released in vapour or particulates to air or by leaching to water. Releases are expected to be initially to air; however, settling and removal of particulates would result ultimately in losses to soil or water. Losses through abrasion and degradation of polymer end products may also occur. HBCD present in foam insulation is unlikely to be exposed to the weather once building construction is complete (e.g., polystyrene foam products in an installed state). However, prior to and during construction, as well as during demolition, the insulation may be subject to weathering, physical disintegration and wear, leading to the potential release of particulates containing HBCD. Once enclosed, these construction materials may undergo a small degree of disintegration over time, with the potential for subsequent release of HBCD. However, it is expected that release from encapsulated materials would be low, since dust and fragmentation would likely be minimal and volatilization of HBCD from products would be low. HBCD encapsulated within textile backcoating materials will have more opportunity for weathering and wear throughout the lifetime of the polymer product, including being washed and chemically cleaned. Losses will likely be primarily to solid waste and wastewater. In the case of construction materials, however, releases to the soil, with subsequent transport by air or runoff, could also occur. These losses apply

to HBCD in products manufactured in Canada, as well as to HBCD in finished and semi-finished products imported into the country.

A European industry survey (EBFRIP 2009) determined that there is potential for loss to land due to disposal practices for packaging waste. The survey included a selection of HBCD producers, warehouses and first-line direct users of HBCD in Europe, representing the first stages in the HBCD life cycle. Packaging waste was the main contributor to potential releases to soil, due to uncontrolled landfill or compost, recycling of empty paper packaging, substances going to unknown destinations, and the protected storage of packaging. The survey also found that implementing best practices in handling noticeably reduced the total releases, from 2017 kg/year in 2008 to 309 kg/year in 2009 (EBFRIP 2009), with annual losses to soil reduced from 1857 HBCD kg/year (2008) to 196 kg/yr (2009).

Products and materials containing HBCD in landfill sites will be subject to weathering, releasing HBCD primarily to soil and, to a lesser extent, to water and air. Most of the HBCD released to soil during landfill operations would be expected to sorb to particles and organic matter, remaining largely immobile. Some limited surface transport in water may occur, due to scavenging in rainfall and runoff. However, the low vapour pressure of the substance suggests that volatilization from the surface of the landfill is unlikely. There is little information on the quantity of HBCD in landfill leachate; however, given the low water solubility of the substance, it is expected that leaching from the surfaces of polymer products in the landfill is limited. Low levels (maximum 9 ng/L; Remberger et al. 2004) were measured in two leachate samples collected from a Swedish landfill used for construction and demolition waste. Much higher concentrations (maximum 36 000 ng/g dry weight [dw]) were present in the particulate phase of leachate water from the Netherlands (Morris et al. 2004); however, these samples were taken from leachate water before treatment for release to surface water. The tendency of HBCD to sorb to particulates, its limited solubility in water, and evidence that it may undergo anaerobic biodegradation all suggest that the risk of groundwater contamination from HBCD-containing products in landfills is probably low.

HBCD is unstable at temperatures above 200°C (Albemarle Corporation 2000a) and will, therefore, decompose during burning. Experimental evidence confirms that under some conditions HBCD and products containing HBCD may release small amounts of polybrominated dibenzo-*p*-dioxins and dibenzofurans during burning. Trace levels of these compounds have been measured during combustion of flame-retarded polystyrene materials containing HBCD (Dumler et al. 1989; Desmet et al. 2005). PBDDs and PBDFs present in HBCD waste will likely be destroyed by the very high operating temperatures employed in well-functioning incinerators. However, there is potential for the release of these substances from uncontrolled burns and accidental fires, as well as from incinerators that are not functioning well (Weber and Kuch 2003). A study by Desmet et al. (2005) documented the formation of bromophenols, known precursors of polybrominated dibenzodioxins and dibenzofurans, during combustion of flame-retarded extruded polystryrene containing HBCD; however, this study did not find that dioxins and furans were themselves formed.

Environmental Fate

A summary of selected measured and predicted physical and chemical properties for HBCD is presented in Table A-2.

Releases of HBCD to the Canadian environment due to the substance's use as a flame retardant are expected to be diffuse, with some point sources (e.g., from processing facilities), and primarily to wastewater. Release to the soil could also occur through the application of sewage sludge as biosolids to agricultural and pasture lands. Releases may occur in both indoor and outdoor environments. Dust, food, serum and indoor air concentrations are presented in Tables A-9 to A-12, and are discussed in the Potential to Cause Harm to Human Health section.

Low water solubility (65.6 μg/L, sum of the individual solubilities of the three diastereomers [γ-HBCD: 2.0 μg/L, α-HBCD: 48.8 μg/L, and β-HBCD: 14.7 μg/L at 20°C]; see Table A-2), low vapour pressure (6.27×10^{-5} Pa at 21°C) and high partition coefficients (log K_{ow} of 5.625–5.81, estimated log K_{oc} of 5.097) suggest that HBCD released into the environment will be less likely to partition into air and/or to remain in water, moving instead to the sediment and soil compartments. The high partition coefficients indicate that HBCD that is released into water is expected to adsorb to the organic fraction of suspended solids and sediments. If released to soil, HBCD is expected to be minimally mobile based on its estimated log K_{oc}. Based on its low vapour pressure, the substance is not expected to volatilize from dry soil surfaces. The results of Level III fugacity modelling (Table A-3) support the expectation that HBCD predominantly resides in soil or sediment, depending on the compartment of release (EQC 2003). The model predicted the following partitioning:

- when HBCD is released 100% to air: 0.002% partitions to air, 2.1% to water, 87.3% to soil, 10.6% to sediment;
- when HBCD is released 100% to water: 0% partitions to air, 17% to water, 0% to soil, 83% to sediment; and
- when HBCD is released 100% to soil: 0% partitions to air, 0% to water, 100% to soil, 0.04% to sediment (EQC 2003).

Persistence and Bioaccumulation Potential

Environmental Persistence

The predicted half-life for atmospheric degradation of HBCD due to reaction with the hydroxyl radical is 2.13 days (AOPWIN 2000).

An experiment by Harrad et al. (2009a) determined a primary degradation half-life of 12.2 weeks (85.4 days) for HBCD in indoor dust exposed to natural light, and 26 weeks (182 days) for HBCD in dust held in dark controls. The study found a significant shift

from γ-HBCD to α-HBCD within 1 week under natural light exposure, with no significant changes in enantiomer fractions. Decreasing concentrations of HBCD with a concurrent increase in pentabromocyclododecanes (PBCD – degradation products of HBCD) was also observed, with the process enhanced under the light exposure. The study concluded that a rapid, photolytically mediated shift in diastereomer profile and a concurrent, slower degradative loss of HBCD via elimination of hydrogen bromide took place.

HBCD is not expected to undergo hydrolysis in the environment, due to a lack of hydrolyzable functional groups and low water solubility (Harris 1990; ACC 2002). Velsicol Chemical Corporation (1979) conducted a hydrolysis experiment using the commercial product, Firemaster 100. No significant hydrolysis occurred over the 39-day test period.

MITI (1992) observed only 1% biodegradation over 28 days in a ready biodegradation test for HBCD. The results indicate that the ultimate degradation half-life in water is likely to be much longer than 182 days (more than 5 years assuming first-order degradation kinetics), and that the substance is therefore likely to persist in this environmental compartment. Similarly no biodegradation was reported in 28-day ready biodegradation testing conducted using a composite sample of HBCD (purity 93.6%) composed of 6.0% α-HBCD, 8.5% β-HBCD and 79.1% γ-HBCD (CMABFRIP 1996; ACC 2002).

Although experimental data on the biodegradation of HBCD in water are available, model estimates derived from quantitative structure-activity relationships (QSARs) were also considered (Environment Canada 2007; see Table A-4). BIOWIN (2000) sub-model 4 predicts that HBCD is amenable to primary degradation (estimated half-life of $\leq$ 182 days). However, with respect to ultimate degradation, sub-model 3 predicts that HBCD biodegrades slowly. Both BIOWIN (2000) sub-models 5 and 6 (both ultimate biodegradation models) also predict a low probability of rapid biodegradation. CPOPs (2008), which predicts ultimate biodegradation, estimates a biochemical oxygen demand (BOD) of only 0.1%, which further suggests very slow biodegradation. When results of the empirical ready biodegradation tests are considered together with the model data, it appears likely that HBCD will undergo some primary biodegradation in water but that the time to ultimate biodegradation is likely to exceed 182 days, making the substance persistent in this medium. As well, as noted below, there is evidence for the formation of a potentially stable transformation product, 1,5,9-cyclododecatriene.

ACCBFRIP (2003b) and Davis et al. (2005) examined the degradability of HBCD using aerobic and anaerobic water/sediment microcosms and soils. Disappearance half-lives were 11 and 32 days in the aerobic microcosms, 1.1 and 1.5 days in the anaerobic microcosms and 6.9 days for anaerobic soil. No degradation products were detected in the sediment, overlying water or headspace of the microcosms. In their analysis of the study, the EU RAR (2008) noted that recoveries of HBCD in the test vessels varied from 33 to 125%, with most recoveries below 70%. An interfering chromatographic peak with characteristics identical to that of γ-HBCD was also present in one of the two river

sediment samples, indicating possible contamination of the sample with HBCD. In addition, the very low initial HBCD concentration resulted in levels of the α- and β-diastereomers being below detection limits by the completion of the test. For this reason, quantification was only possible for γ-HBCD, and no information is available on the fate of α- and β-HBCD. This is particularly significant given the evidence for a predominance of α-HBCD in biota, suggesting that this isomer may have greater environmental stability (see Bioaccumulation section below). As no degradation products, including carbon dioxide, were identified in the study, biotic processes could not be conclusively linked to the observed rapid disappearance of HBCD, and the results are therefore interpreted in terms of disappearance times rather than biodegradation (EU RAR 2008).

In a high-quality study, EBFRIP (2004b) and Davis et al. (2006b) investigated biodegradation of HBCD in activated and digester sludge, river sediment, and surface soil. The study objectives emphasized identification of degradation pathways and products, and transformation half-lives were not reported for the various test media. Substantial transformation occurred in the anaerobic digester sludge and in freshwater aerobic and anaerobic sediment microcosms. Degradation rates were slower in the activated sludge samples, and no degradation of HBCD was observed in the aerobic soil microcosms. Tetrabromocyclododecene, dibromocyclododecadiene and 1,5,9-cyclododecatriene were identified as primary biotransformation products, providing evidence that degradation of HBCD in the environment may occur through a process of sequential debromination.

Gerecke et al. (2006) reported a degradation half-life of 0.66 days for technical HBCD incubated with digested sewage sludge under anaerobic conditions. Although all three diastereomers degraded rapidly, β- and γ-HBCD degraded more rapidly than α-HBCD, leading the researchers to propose that differential degradation rates may contribute to the relative enrichment of α-HBCD observed in biota samples. Findings from the study differed from those of Davis et al. (2006b), which determined there was little difference in the transformation behaviour of the three isomers, although β-HBCD was found to transform significantly faster than α- and γ-HBCD when incubated within the digester sludge (Davis et al. 2006b).

No information could be found on the degradation properties and inherent toxicities of tetrabromocyclododecene and dibromocyclododecadiene; however, some limited data are available for 1,5,9-cyclododecatriene (CDT), the final debromination product. Bridié et al. (1979a, 1979b) measured a BOD of 0.02 g/g and a 24-h LC_{50} (median lethal dose) for goldfish (*Carassius auratus*) of 4 mg/L, suggesting that 1,5,9-cyclododecatriene is resistant to microbial oxidation processes and is potentially toxic to aquatic species. Other measured and estimated data support the finding that the substance presents high hazard to aquatic organisms. For instance, NITE (2002) reports a 48-hour LC_{50} of 0.166 mg/L for rice fish (*Oryzias latipes*), and ECOSAR (2009) predicts acute toxicity to aquatic organisms below 1 mg/L (i.e., fish 96-hour LC_{50} = 0.104 mg/L; daphnid 48-hour LC_{50}=0.098 mg/L; and green algae 96-hour EC_{50} = 0.214 mg/L, see Appendix B: Table B-1). Data from NITE (2002) further indicate that the substance has a high

bioconcentration potential, with measured bioconcentration factors (BCFs) for carp of 2360 to 12 500 and 1920 to 14 800, resulting from 10-week exposures to 0.01 and 0.001 mg/L, respectively. Using the Arnot and Gobas (2003) bioaccumulation model, calculated BCF values for 1,5,9-cyclododecatriene range from 9813 (corrected for metabolic transformation) to 18 620 L/kg (no metabolism), and bioaccumulaton factor (BAF) values range from 66 360 (corrected for metabolism) to 177 828 (no metabolism) (Table B-2). However, it is noted that the likelihood of CDT bioaccumulation will depend on the stability of CDT within a given compartment; the substance must be stable long enough in order to bioaccumulate. The substance is classified as not readily biodegradable, with only 1% biodegradation observed in standard 28-day ready biodegradation testing (du Pont 2003). Enhanced aerobic ready biodegradation testing conducted using the isomer trans, trans, trans-1,5,9-cyclododecatriene determined that although the substance is not readily biodegradable, it will undergo primary biodegradation following a lag phase of approximately 14 days (EBFRIP 2006). Conclusive results with respect to complete mineralization were not possible from the study. A subsequent study conducted under similar conditions and using lower test concentrations (Davis et al. 2006a) documented the formation of carbon dioxide over the course of the 77-day test period, indicating that mineralization of the substance was occurring under the conditions of the study. While this study provides evidence that 1,5,9-cyclododecatriene will biodegrade under the conditions of enhanced aerobic ready biodegradation testing, information is lacking on the potential for biodegradation under low oxygen conditions, as these are most likely to prevail in subsurface layers of the soil and sediment compartments to which HBCD preferentially partitions. Additionally, complete mineralization of HBCD has not yet been demonstrated, an indication that degradation products such as 1,5,9-cyclododecatriene may remain stable under some study conditions.

Sediment core studies in Europe and Japan have reported HBCD concentrations in sediment layers that date back to the 1960s and 1970s (Remberger et al. 2004; Minh et al. 2007; Bogdal et al. 2008; Kohler et al. 2008; Tanabe 2008). For example, Remberger et al. (2004) measured concentrations of HBCD in sediment layers approximately 30 and 40 years old in cores from the Stockholm archipelago; these concentrations were 25–33% of HBCD concentrations found in the top layer of the cores. Such studies suggest that degradation half-lives under field conditions are not as fast as simulation degradation studies (e.g., ACCBFRIP 2003b) might indicate (EU RAR 2008).

In summary, data for HBCD suggest that the substance is persistent in sediment. Results from laboratory studies suggest that primary degradation half-lives may be less than 365 days. However, ultimate degradation half-lives are likely much longer than 365 days, based on results of ready biodegradation testing and an extrapolation ratio of 1:4 for a water:sediment biodegradation half-life (Boethling et al. 1995). Furthermore, sediment core measurements suggest that primary degradation half-lives in the environment are likely to be much longer than 365 days. Information gathered to date on the HBCD degradation products suggests that certain of these products (e.g., 1,5,9-cyclododecatriene) are potentially bioaccumulative and highly hazardous, like HBCD itself.

ACCBFRIP (2003c) and Davis et al. (2005) also investigated the degradation of HBCD in aerobic and anaerobic soil microcosms. An average HBCD decrease of 75% was observed in the aerobic soil microcosms over the 119-day test period. In the anaerobic test system, HBCD decreased by 92% over 21 days in the test microcosms. Based on the results of the study, disappearance half-lives of 63 and 6.9 days were determined in the aerobic and anaerobic soils, respectively. No degradation products were detected in the soil or headspace of the microcosms. The EU RAR (2008) noted that, as with the water/sediment microcosm study described above, only γ-HBCD was quantified and therefore this study provides no information on the fate of α- and β-HBCD in soil. As well, only one soil type was tested, making it difficult to evaluate the representativeness of the determined half-lives to conditions in the environment. Finally, in the absence of identified transformation products, the mechanism behind the observed disappearance of HBCD remains unclear and may in part be due to adsorption to soil, given the large differences observed between measured and nominal HBCD concentrations in the soil at test initiation (EU RAR 2008).

The absence of observable degradation in the aerobic soil microcosms of EBFRIP (2004b) contrasted with results obtained by ACCBFRIP (2003c), which reported a disappearance half-life of 63 days in aerobic soils. The test substances used in the two studies were comparable in composition, although the EBFRIP (2004b) test substance contained a higher proportion of γ-HBCD, making it closer in composition to the current commercial product. Additionally, EBFRIP (2004b) testing was conducted at greater HBCD concentrations than testing by ACCBFRIP (2003c). The test soils were collected at different times of year (April for ACCBFRIP 2003c and November for EBFRIP 2004b) from the same site in North Dakota (EBRIP 2004b), and exposure periods were of comparable duration (119 vs. 112 days). The longer pre-stabilization period of 35 days used in the ACCBFRIP (2003c) study may have produced a more stable microbial population at test initiation; however, the 15-day period employed by EBFRIP (2004b) was well within the Organisation for Economic Co-operation and Development's (OECD's) Guideline's recommended range of 2 days to 4 weeks (OECD 2002). A key difference was the addition of activated sludge to the microcosms of ACCBFRIP (2003c), a procedure designed to investigate possible degradation outcomes following the addition of biosolids containing HBCD to surface soils during land treatment. While Schaefer and Siddiqui (2003) reported an almost 30% inhibition of activated sludge micro-organisms following treatment with HBCD, it is likely that the presence of these organisms in the soil microcosms of ACCBFRIP (2003c) significantly enhanced degradation rates relative to those of EBFRIP (2004b). EBFRIP (2004b) testing was conducted at much greater HBCD concentrations than testing by ACCBFRIP (2003c), and the authors indicated that degradation kinetics may have been limited by the mass transfer of the substance to the microbes.

A study isolating the soil bacterial strain *Pseudomonas* sp. found 1 mM γ-HBCD was degraded by 81% within five days (Yamada et al. 2009), with no metabolites measured during the assays. When the same bacterial strain was tested with six related alkanes,

including α-HBCD, it failed to degrade α-HBCD, and therefore the authors proposed that any γ-HBCD debrominating enzymes must possess substrate specificity.

In summary, despite uncertainties, existing data for HBCD suggest that the substance is persistent in soil. The ultimate degradation half-life in soil is likely much longer than 182 days, based on results of ready biodegradation testing and an extrapolation ratio of 1:1 for a water:soil biodegradation half-life (Boethling et al. 1995). Primary degradation rates appear to be variable, but may also be longer than 182 days (EBFRIP 2004b).

Based primarily on empirical data, HBCD meets the persistence criteria in air, water, soil and sediment (half-life in air ≥ 2 days, half-lives in soil and water ≥ 182 days, and half-life in sediment ≥ 365 days) as set out in the *Persistence and Bioaccumulation Regulations* (Canada 2000).

Wania (2003) used a modelling approach to evaluate the potential for long-range atmospheric transport of HBCD and concluded that, based on physical and chemical properties, the substance should have low potential to reach remote areas. In a subsequent study, Brown and Wania (2008) identified HBCD as a potential Arctic contaminant based on an atmospheric oxidation half-life of greater than two days and structural similarities to known Arctic contaminants. The low volatility of HBCD likely results in significant sorption to atmospheric particulates and for this reason, the long-range transport potential of HBCD may depend upon the transport behaviour of the atmospheric particulates to which it sorbs. HBCD has been measured in air, sediment and biota samples collected from remote sites such as the Arctic (e.g., Remberger et al. 2004; Verreault et al. 2005, 2007a, 2007b; Muir et al. 2006; Evenset et al. 2007; Svendsen et al. 2007; Tomy et al. 2008). As there is no evidence for the natural production of HBCD, these data are indicative of contamination from anthropogenic sources. While this contamination may be local in origin, it is likely that the findings represent evidence that under some circumstances HBCD may be capable of atmospheric transport over long distances and to remote locations. Based on the available information, it is considered that HBCD meets the persistence criterion of being subject to atmospheric transport from its source to a remote area, as specified in CEPA 1999 (see Table A-5).

Additional evidence for the persistence of HBCD is its potential for biomagnification (see section below: studies by Morris et al. 2004; Tomy et al. 2004a; and Law et al. 2006a). The occurance of biomagnification is indicative of environmental persistence and/or a lack of significant metabolism, for in order to biomagnify significantly, a substance must persist long enough to be transferred successively from lower to higher trophic levels and/or not be subject to metabolic transformation.

Potential for Bioaccumulation

Veith et al. (1979) measured a BCF of 18 100 in fathead minnow, *Pimephales promelas*, exposed to 0.0062 mg/L HBCD for 32 days, while CMABFRIP (2000) reported BCF values ranging from 4650 to 12 866 in rainbow trout, *Oncorhynchus mykiss*, exposed for 35 days to 0.0034 mg/L HBCD.

Law et al. (2006b) and Law (2006) measured biomagnification factors (BMFs), derived on a lipid weight, of 9.2, 4.3 and 7.2 for α-, β- and γ-HBCD, respectively, by exposing juvenile rainbow trout, *Oncorhynchus mykiss*, to single isomer concentrations ranging from 12 ng/g to 29 ng/g lipid weight in the diet. Bioaccumulation of γ-HBCD was linear, while that of α- and β-HBCD increased exponentially with respective doubling times of 8.2 and 17.1 days. Both β- and γ-HBCD followed first-order depuration kinetics, with depuration rate constants (k_d) of 0.44 x 10^{-2} and 0.48 x 10^{-2} d^{-1} and calculated half-lives of 157 (±71) and 144 (±60) days, respectively. A k_d value and half-life could not be calculated for α-HBCD, since depuration out of the muscle tissue did not obey a first-order rate process. Assimilation efficiencies, calculated by comparing concentrations measured in the fish with those in the food, were determined to be 31.1, 41.4 and 46.3% for α-, β- and γ-HBCD, respectively. Bioisomerization of HBCD was also reported in the study, with statistically significant amounts of α-HBCD measured in the muscle tissue of trout exposed exclusively to γ-HBCD. Similarly, both α- and γ-HBCD were present in statistically significant quantities in fish exposed only to β-HBCD. The results suggested that juvenile rainbow trout were able to bioisomerize the β- and γ-isomers of HBCD, with preferential formation of α-HBCD. The α-isomer appeared recalcitrant to bioisomerization in this fish species. Selective bioisomerization of HBCD has the potential to contribute appreciably to determining isomer distributions within organisms.

Haukas et al. (2009) conducted oral exposure experiments with juvenile rainbow trout to assess the role of selective uptake on diastereomer-specific accumulation of HBCD to liver, brain and muscle tissues. Exposed fish were fed 10 mg/kg body weight technical HBCD followed by 21 days of food deprivation. Results showed HBCD was effectively accumulated from diet and distributed throughout the fish to the tissues sampled. HBCD accumulation peaked 4 to 8 days after exposure, and the relative change in HBCD accumulation pattern within days 0 to 8 suggested selective diastereomer uptake of α and β-HBCD. Assimilation efficiencies suggested α- and β-HBCD were assimilated more efficiently through the gut than the γ diastereomer. A 70% reduction in total HBCD levels after 21 days indicated elimination from liver and brain, but not muscle. The authors reported that this suggested organ-specific diastereomer accumulation.

Tomy et al. (2004a) reported a strong positive linear correlation between tissue concentrations of HBCD and trophic level in a Lake Ontario pelagic food web, evidence that bioaccumulation and biomagnification was occurring within the web. Species examined in the study included a top predator—lake trout (*Salvelinus namaycush*)—and prey species such as alewife (*Alosa pseudoharengus*), rainbow smelt (*Osmerus mordax*), slimy sculpin (*Cottus cognatus*), mysid (*Mysis relicta*), amphipod (*Diporeia hoyi*) and zooplankton, such as copepods and cladocerans. Lipid-normalized BMFs exceeded 1 for most feeding relationships, and ranged from 0.4 to 10.8 for α-HBCD and 0.2 to 9.9 for γ-HBCD. A BMF for β-HBCD was not determined from the study. A trophic magnification factor was calculated for HBCD in the food web by comparing HBCD concentrations with those of the stable nitrogen 15 isotope (δ15N). Trophic magnification factors of around 0 suggest that a chemical moves through the food web without being biomagnified, while those exceeding 1 indicate that biomagnification is occurring (Broman et al. 1992; Fisk et al. 2001). A trophic magnification factor of 6.3 was

calculated for HBCD, comparable to that of known biomagnifying substances, such as the persistent organochlorines *p,p'*-DDE (6.1) and polychlorinated biphenyls (PCBs) (5.7).

Law et al. (2006a) calculated trophic magnification factor values for a Lake Winnipeg pelagic food web, using zooplankton, mussels (*Lampsilis radiate*), walleye (*Stizostedion vitreum*), whitefish (*Coregonus commersoni*), emerald shiner (*Notropis atherinoides*), burbot (*Lota lota*), white sucker (*Catostomus commersoni*) and goldeye (*Hiodon alosoides*). The trophic magnifcation factors were 2.3, 2.3 and 4.8 for α-, β- and γ-HBCD, respectively, while that for total HBCD was 3.1. The highest individual BMFs were associated with the predator/prey pairs of goldeye/mussel (8.2), burbot/emerald shiner (6.3), walleye/whitefish (5.3), burbot/mussel (5.0) and emerald shiner/plankton (5.0). The results indicated that biomagnification was occurring, but at a lesser rate than it was taking place in a comparable Lake Ontario food web (Tomy et al. 2004a).

Tomy et al. (2009) examined the trophodynamics of HBCD throughout a western Canadian Arctic and marine food web. Individual lipid-adjusted predator-prey relationships (BMFs) were < 1 for α-HBCD, except for the beluga whale (*Delphinapterus leucas*): pacific herring (*Clupea pallasi*) feeding relationship (BMF = 1.7). The authors concluded that metabolic depletion of HBCD (and other brominated flame retardants) is likely taking place in higher-trophic-level animals, whereas it is accumulated/stored faster than metabolized/eliminated in lower-trophic-level animals.

Biomagnification of HBCD in a North Sea food web was evaluated by comparing concentrations in species from various trophic levels (Morris et al. 2004). Amounts in top predators, such as harbour porpoise (*Phocoena phocoena*) and harbour seal (*Phoca vitulina*) were several orders of magnitude higher than those measured in aquatic macro-invertebrates such as sea star (*Asterias rubens*) and common whelk (*Buccinium undatum*) collected from the same area. Similarly, high concentrations were detected in liver samples from cormorant (*Phalacrocorax carbo*), a top predator bird, and in eggs of the common tern (*Sterna hirundo*). Intermediate amounts were found in cod (*Gadus morhua*) and yellow eel (*Anguilla anguilla*). Results from the study were considered to indicate bioaccumulation and biomagnification up the aquatic food chain.

In their study of bioaccumulation and biotransformation in East Greenland, Letcher et al. (2009) reported an α-HBCD BMF (derived for lipid weight) for Greenland ringed seal (*Pusa hispida*) blubber: polar bear (*Ursus maritimus*) adipose tissue of 1.7 ± 0.6. *In vitro* 90-minute assays using hepatic microsomes from polar bears found a significant depletion of 24% for α-HBCD. No oxidative metabolites were detected *in situ*. The authors speculated that biotransformation of α-HBCD via CYP2B enzymes could explain depletion of α-HBCD in the assay, as polar bears have relatively high liver CYP2B content. The authors concluded that polar bears in East Greenland were potentially exposed to, and accumulated, substantial levels of HBCD.

Velsicol Chemical Corporation (1980) reported rapid metabolism of HBCD in the blood, muscle, liver and kidneys of rats given a single oral dose of radiolabelled substance.

Elimination occurred primarily via the feces (70%) and urine (16%), with 86% of the radiocarbon removed over the three days following dosing. The test substance distributed throughout the body, with the highest amounts in the fatty tissue, followed by the liver, kidney, lung and gonads. HBCD remained mostly unchanged in fatty tissue. The study concluded that HBCD was capable of accumulating in the fatty tissue of rats following repeated exposure.

CMABFRIP (2001) examined the presence of individual diastereomers in adipose tissue of rats dosed with 1000 mg/kg body weight (mg/kg-bw) per day for up to 90 days. Concentrations of α-HBCD exceeded those of β- and γ-HBCD, accounting for 65% to 70% of the total HBCD present. Gamma-HBCD accounted for 14% to 20% of the total, while β-HBCD was present at 9–15%. This contrasted markedly with proportions present in the test substance, which contained 84.5% γ-HBCD, 8.9% α-HBCD and 6.6% β–HBCD. The highest tissue concentrations were measured on study day 89, and the amounts were consistently higher in female rats as compared with males.

Although empirical bioaccumulation data are available for HBCD, QSARs were also applied (Environment Canada 2007) using the predictive models, with their results shown in Table A-6. Model estimates range from approximately 158 500 (estimate considers metabolic transformation) to 6 457 000 (without metabolic transformation) for the BAF, and from 4300 (estimate considers metabolic transformation) to 24 000 (no metabolic transformation considered) for the BCF.

Based on empirical and modelled data, HBCD meets the criteria for bioaccumulation (BAFs and BCFs of 5000 or more) as set out in the *Persistence and Bioaccumulation Regulations* (Canada 2000).

Ecological Exposure Assessment

While Canadian and North American exposure data are limited, HBCD has been detected in all environmental media in many parts of the world, with highest levels occurring near urban and industrial areas (see Tables A-7 and A-8).

Air

Concentrations of up to 0.011 ng/m^3 were measured in the particle phase of air samples collected in 2002 and 2003 at five sites from Lake Michigan through the U.S. Midwest to the Gulf of Mexico (Hoh and Hites 2005). Based on similarities in spatial concentration patterns of HBCD and the brominated diphenyl ether flame retardant PBDE-209 (decabromodiphenyl ether), the researchers speculated that the brominated flame retardant market may be shifting from diphenyl ether products to HBCD (Hites and Hoh 2005).

HBCD was detected (0.001 to 0.003 ng/m^3 HBCD; data read from graph) during continuous high-volume air measurements collected at Alert, Nunavut, in the Canadian Arctic between 2006 and 2007 (Xiao et al. 2010).

Precipitation samples collected from the Great Lakes basin contained up to 35 ng/L (Backus et al. 2005). All three major diastereomers were detected, with an average distribution of 77%, 15% and 8% for α-, β- and γ-HBCD, respectively.

European concentrations are often higher than those measured in North America. Remberger et al. (2004) analyzed HBCD in air and rainfall samples collected in 2000 and 2001 from various locations in Sweden. Air concentrations near potential sources (e.g., an extruded polystyrene manufacturing facility, landfill for construction and demolition waste, textile industry facility) ranged from 0.013 ng/m^3 to 1070 ng/m^3 while those at urban stations in Stockholm were 0.076 ng/m^3 to 0.61 ng/m^3. The highest concentration, 1070 ng/m^3, was recorded close to the exhaust of an air ventilation system at an extruded polystyrene manufacturing facility.

Surface Waters

Law et al. (2006a) reported a mean dissolved phase concentration of 0.011 ng/L for α-HBCD in surface water samples collected from the south basin of Lake Winnipeg in 2004. Beta- and γ-HBCD were not detected (detection limit: 0.003 ng/L). The researchers commented that detection of only α-HBCD in the samples was consistent with its much greater aqueous solubility (4.88 x 10^4 ng/L; see Table A-2) relative to that of the β- (1.47 x 10^4 ng/L) and γ- (2.08 x 10^3 ng/L) isomers. Surficial sediment grab samples from the same region contained a mean concentration of 0.05 ng/g dw of γ-HBCD. Alpha- and β-HBCD were not detected in the samples (detection limit: 0.04 for β- and γ-HBCD to 0.08 ng/g dw for α-HBCD). The results were consistent with γ-HBCD being the most hydrophobic of the three isomers.

Filtered surface water and suspended solids samples were collected upstream of a sewage treatment plant in the United Kingdom (U.K.). Filtered water samples contained 57 ng/L to 1520 ng/L; HBCD was not detected (detection limit: 50 ng/L) in a single sample taken approximately one kilometre downstream of the plant (Deuchar 2002). Concentrations in the suspended solids of the upstream samples were up to 1310 ng/L, while the single downstream sample contained 215 ng/L. Two U.K. locations considered remote from industrial activity contained from less than 50 ng/L to 210 ng/L.

Sediment

Marvin et al. (2004, 2006) measured HBCD in suspended sediments collected along the Detroit River from Lake St. Clair to the outflow to Lake Erie, and determined that occurrence of the substance was strongly associated with urban and industrial activities. Annual mean concentrations ranged from 0.012 ng/g to 1.14 ng/g dw, with the highest levels being found downstream of the urban region surrounding the city of Detroit. About two thirds of the samples had isomeric profiles similar to those found in commercial technical mixtures, with a predominance of γ-HBCD, while the remaining samples were dominated by α-HBCD. The β-isomer was present at substantially lower levels, consistent with its lower prevalence in commercial mixtures. The researchers concluded that distribution of HBCD in the Detroit River appeared to be heavily influenced by HBCD associated with shoreline-based urban and industrial activities. In addition, the

widespread occurrence of relatively low concentrations suggested that large urban areas may act as diffuse sources of HBCD.

Four surficial sediment grab samples collected in 2003 from four sites in the south basin of Lake Winnipeg contained a mean concentration of 0.05 ng/g dw γ-HBCD (Law et al. 2006a). Alpha- and β-HBCD were not detected in the samples (detection limit: 0.04 ng/g for β- and γ-HBCD to 0.08 ng/g dw for α-HBCD). The researchers commented that the results were consistent with γ-HBCD being the most hydrophobic of the three isomers.

Concentrations of less than 1.7 ng/g to 1680 ng/g dw were measured in river and estuarine sediments collected from 2000 to 2002 at various locations throughout the U.K. (Morris et al. 2004). The highest concentration occurred close to a brominated fire retardant manufacturing plant in northeast England that closed in 2003 and was demolished in 2004 (EU RAR 2008). The same study examined sediments from the region surrounding the Western Scheldt (the Netherlands) and Scheldt Basin (Belgium). Concentrations of up to 950 ng/g dw were measured in the samples, with highest levels occurring near areas of industrial activity. Most samples contained isomeric patterns closely resembling that of the commercial formulations, with a predominance of γ-HBCD. In some instances, however, sediments contained higher percentages of α- and β-HBCD. Thermal rearrangement of HBCD isomers at temperatures greater than 160°C has been documented, resulting in the conversion of γ-HBCD into α-HBCD (Peled et al. 1995). As these temperatures are commonly employed in processes to incorporate HBCD into a polymer matrix, the presence of higher proportions of α- and β-isomers in the sediment samples was considered to indicate use of HBCD in processing operations such as polymer and textile applications (Morris et al. 2004).

A study in Spain detected no HBCD in Cinca River sediment upstream of Mozon, a heavily industrialized town, yet sediment concentrations downstream measured up to 2430 ng/g dw (Guerra et al. 2009). The isomeric profile of the sediment samples showed that γ-HBCD represented 94–99% of the total HBCD. The authors reported that sediment concentrations had increased at this site from 2002 to 2005, then remained similar for 2006. The pattern mirrored that observed in fish tissue sampled in the area. Additionally, a caged fish (barbel) survey conducted at the site found neglibile HBCD accumulation upstream but very high accumulated levels (i.e., 15 518 ng/g lipid weight or 1337 ng/g wet weight [ww]) at the downstream site.

Soil

The existing literature contains few references to soil concentrations of HBCD. Four shallow soil samples (actual depth not provided) taken from the vicinity of a U.K. flame retardant coating manufacturing facility in 1999 contained 18 700 to 89 600 ng/g dw HBCD (mean concentration 62 800 ng/g dw) (Dames and Moore 2000a). Remberger et al. (2004) analyzed soil samples collected in 2000 at distances of 300 m, 500 m and 700 m from a Swedish facility known to manufacture extruded polystyrene with HBCD. Concentrations of HBCD in the samples ranged from 140 ng/g to 1300 ng/g dw, and decreased with increasing distance from the plant.

Waste Effluent and By-products

Limited North American data on levels of HBCD in waste treatment products were found in the literature. One study examined processed sewage sludge from a wastewater treatment facility (activated sludge) in the mid-Atlantic United States that treats domestic and industrial waste (La Guardia et al. 2010). Samples were collected in 2002, 2005, 2007 and 2008. HBCD was one of 23 flame retardants detected, and in 2005 represented 86% of total flame retardant concentration in the 2005 sample. HBCD concentrations were 1160 ng/g total organic carbon (TOC) in 2002 (sludge sample was 28% TOC), 1 600 000 ng/g TOC in 2005 (sample was 25% TOC), 92 500 ng/g TOC in 2007 (sample was 25% TOC), and 45 300 ng/g TOC in 2008 (sample was 7% TOC). Gamma-HBCD was typically the dominant isomer in samples representing 40–69% of total HBCD; α-HBCD followed, representing 9–50% of total HBCD; β-HBCD was the smallest fraction, representing 4–22% of total HBCD (< 9% in 3 of 4 samples).

Morris et al. (2004) sampled landfill leachates in 2002 from sites in southeast England, Ireland and the Netherlands. HBCD was not detected in the U.K. samples (detection limits: 15 ng/L for the dissolved phase and 3.9 ng/g dw for the particulate phase; de Boer et al. 2002). However, concentrations of 2.5 ng/g to 36 000 ng/g dw (mean 5906 ng/g dw) were measured in the samples collected in the Netherlands. The substance occurred only in the particulate phase, and γ-HBCD predominated in the samples.

Concentrations of 3 ng/L and 9 ng/L were measured in two leachate samples collected in 2000 at a landfill site for construction and demolition waste near Stockholm (Remberger et al. 2004). Sediment from the leachate sedimentation basin contained less than the detection limit of 0.1 ng/g dw.

Concentrations of up to 29.4 ng/g dw (particulates) and 24 ng/L (dissolved phase) were measured in influent samples collected in 2002 from five sewage treatment plants in southeast England (Morris et al. 2004). The substance was not detected (detection limit: 3.9 ng/g dw) in the effluents, but was present at 531 ng/g to 2683 ng/g dw (mean 1401 ng/g dw) in sludge samples taken from the sites. The γ-isomer predominated in the samples, with α- and β-HBCD present in smaller and almost equal quantities. The researchers proposed that release of HBCD from contaminated dust, such as office dust containing brominated flame retardants, may account, at least in part, for the presence of the substance in sewage treatment plant influents and sludge.

Sludge sampled in 2000 from 50 sewage treatment plants throughout Sweden contained from 3.8 ng/g to 650 ng/g dw (mean 45 ng/g dw; Law et al. 2006c). Higher concentrations occurred in samples collected near known or suspected sources, such as textile industries, producers of extruded polystyrene and a company that upholstered cars.

HBCD was present in all of 19 samples collected from 16 Swiss wastewater treatment plants from May to July 2003 and in January 2005 (Kupper et al. 2008). Concentrations in the samples ranged from 39 ng/g to 597 ng/g dw, with a mean value of 149 ng/g dw and a median of 123 ng/g dw.

Zennegg et al. (2005) reported concentrations of 19 to 170 ng/g dw (mean 85 ng/g dw) in urban compost collected from six composting facilities in Switzerland. The study also evaluated levels of several other brominated flame retardants, including polybrominated diphenyl ethers (PBDE congeners 28, 47, 99, 100, 153, 154, 183 and 209) and tetrabromobisphenol A. HBCD was the most prominent brominated flame retardant in the samples.

Due to the limited HBCD data for surface water and sediment concentrations in Canada, a fugacity modelling approach was also applied in the assessment to estimate aquatic exposure to HBCD in the pelagic and benthic compartments, and to support risk analyses for water and sediments (see Appendix C).

Biota

HBCD has been detected in North American organisms, as well as organisms from other parts of the world (Table A-8).

Archived samples of Lake Ontario lake trout, *Salvelinus namaycush*, contained from 16 ng/g to 33 ng/g lipid weight (2 ng/g to 4 ng/g ww) total HBCD, with the amounts decreasing significantly between 1979 and 2004 (Ismail et al. 2009). The α-isomer predominated in the samples (15 ng/g to 27 ng/g lipid weight; 1.7 ng/g to 3.4 ng/g ww), with lower levels of β- (0.16 ng/g to 0.94 ng/g lipid weight; 0.03 ng/g to 0.11 ng/g ww) and γ-HBCD (1.4 ng/g to 6.5 ng/g lipid weight; 0.23 ng/g to 0.77 ng/g ww). The researchers proposed that alterations to food web processes in the lake, such as changes to the lake trout diet and/or changes at the base of the food web, as well as possible temporal variations in contaminant loadings and voluntary emission-limiting measures undertaken by industry, may be factors in the downward trend in concentration. However, the need for further research was emphasized, given the conflicting evidence of increasing temporal trends reported in other studies.

Mean concentrations ranging from 3 ng/g to 65 ng/g lipid weight were measured in fish, mussels and zooplankton collected from the south basin of Lake Winnipeg between 2000 and 2002 (Law et al. 2006a). The β-isomer was consistently detected at much lower levels than were the α- and γ-isomers, while the proportions of α- and γ-HBCD varied between species.

Tomy et al. (2004a) examined bioaccumulation and biomagnification of HBCD in a Lake Ontario pelagic food web by measuring concentrations in lake trout (*Salvelinus namaycush*, a top predator) and several of its major prey. Alpha- and γ-HBCD were detected at all trophic levels, with the highest concentrations present in lake trout (mean total HBCD 1.68 ng/g ww). Concentrations of α-HBCD were consistently higher than those of γ-HBCD, while β-HBCD was below the method detection limit (estimated at 0.03 ng/g ww) in all the species tested.

Pooled homogenates of herring gull (*Larus argentatus*) eggs collected from six colonies around the Great Lakes contained from 2.1 ng/g to 20 ng/g ww α-HBCD (Gauthier et al. 2007). Highest levels were measured at Gull Island on northern Lake Michigan, likely a

result of this lake being the most urbanized and industrialized of the Great Lakes (Norstrom et al. 2002). Beta-HBCD was not detected in the samples; however, low levels of γ-HBCD were present in two of the six. It should be noted, however, that the southern portions of the lake are more heavily industrialized as compared to the areas from which the samples were taken. The findings confirm the presence of HBCD in the aquatic food web associated with herring gulls in the Great Lakes, with mother gulls exposed via their diet and subsequent *in vivo* transfer to the eggs (Gauthier et al. 2007).

HBCD was not detected (detection limit: 0.01 ng/g ww) in 29 blood samples collected from 2001 to 2003 from nestling bald eagles (*Haliaeetus leucocephalus*) in British Columbia and southern California (McKinney et al. 2006). Sampling was conducted at four locations in southwestern British Columbia (Barkley Sound, Nanaimo/Crofton, Delta/Richmond, Abbotsford/Chilliwack), one location in northern B.C. (Fort St. James) and one southern California site (Santa Catalina Island).

Blubber and liver samples collected from Atlantic white-sided dolphin (*Lagenorhynchus acutus*) stranded on the east coast of the United States between 1993 and 2004 contained from 14 ng/g to 280 ng/g ww (19 ng/g to 380 ng/g lipid weight) and 0.051 ng/g to 3.6 ng/g ww (2.9 ng/g to 140 ng/g lipid weight), respectively (Peck et al. 2008). The α-isomer was present in all samples, while β- and γ-HBCD were not detected (detection limit: 0.4 ng/g ww for both isomers). No significant trend in concentration over time was evident in the samples.

HBCD was detected in 87% of fish samples representing major prey of Atlantic harbour seals (*Phoca vitulina concolor*): Atlantic herring (*Clupea harengus)*, alewife (*Alosa pseudoharengus*), and Atlantic mackerel (*Scomber scombrus*), collected off the coast of Maine in the northwestern Atlantic (Shaw et al. 2009). Concentrations ranged from 2.4 to 38.1 ng/g lipid weight (mean 17.1 ng/g lipid weight).

Almost all (50 out of 52) fish samples collected in 2003 from Chesapeake Bay of the northeastern United States contained at least one stereoisomer of HBCD (Larsen et al. 2005). Total HBCD concentrations ranged from 1.0 ng/g lipid weight (white perch) to 73.9 ng/g lipid weight (channel catfish), with the highest levels measured in samples collected from historically contaminated areas. Isomer distributions differed significantly between benthic fish (e.g., catfish, eel), which had a predominance of α-HBCD, and pelagic species (e.g., striped bass), in which γ-HBCD dominated.

Johnson-Restrepo et al. (2008) measured concentrations in the blubber of bottlenose dolphin (*Tursiops truncates*) and the muscle tissue of bull shark (*Carcharhinus leucas*) and Atlantic sharpnose shark (*Rhizoprionodon terraenovae*) collected from the coastal waters of Florida from 1991 to 2004. HBCD was present in all samples at concentrations ranging from 0.460 ng/g to 72.6 ng/g lipid weight in bottlenose dolphin, 9.15 ng/g to 413 ng/g lipid weight in bull shark, and 1.83 ng/g to 156 ng/g lipid weight in Atlantic sharpnose shark. The α-isomer predominated in the samples, although most also contained smaller amounts of both β- and γ-HBCD.

Concentrations in European biota tend to be higher than those measured in North America, likely reflecting the higher market demand for HBCD in Europe and possibly the higher human population density. Allchin and Morris (2003) reported concentrations of 39.9–75 ng/g ww in yellow eel (*Anguilla anguilla*) and < 1.2–6758 ng/g ww in brown trout (*Salmo trutta*) collected from eight locations along the rivers Skerne and Tees in the U.K.

Morris et al. (2004) examined biomagnification in the North Sea food web by comparing concentrations present in species from various trophic levels from 1998 to 2001. The highest levels were found in top predator species, such as harbour porpoise (*Phocoena phocoena*; 440–6800 ng/g lipid weight), harbour seal (*Phoca vitulina*; 63–2055 ng/g lipid weight) and cormorant (*Phalacrocorax carbo*; 138–1320 ng/g lipid weight) and in the eggs of the common tern (*Sterna hirundo*; 330–7100 ng/g lipid weight). HBCD was also present in cod (*Gadus morhua*; maximum 50 ng/g lipid weight), yellow eel (*Anguilla anguilla;* maximum 690 ng/g lipid weight), sea star (*Asterias rubens*; maximum 84 ng/g lipid weight) and common whelk (*Buccinium undatum*; maximum 47 ng/g lipid weight). The α-isomer strongly dominated the diastereomeric profile, particularly in top predator species such as fish.

HBCD was detected in all of 85 samples of harbour porpoise blubber collected from 1994 to 2003 from animals stranded or caught in waters off the U.K. coast (Law et al. 2006d). The α-isomer predominated in the samples, with concentrations ranging from 10 ng/g to 19 200 ng/g ww. Concentrations in the blubber increased sharply from about 2001 to the end of the study in 2003, suggesting changing patterns in the use of HBCD. The researchers postulated that limitations on production and use of two commercial polybrominated diphenyl ether (PBDE) formulations (i.e., commercial pentaBDE and octaBDE) may have been driving the increase, since HBCD may be being used as a substitute for these formulations in some applications.

In a subsequent study, analyses were conducted of an additional 138 samples collected from the same region from 2003 to 2006 (Law et al. 2008). Concentrations of total HBCD in the samples ranged from less than 10 ng/g to 11 500 ng/g ww (up to 12 800 ng/g lipid weight), with the maximum value determined for an animal stranded or caught in 2003. A statistically significant decrease in levels was seen between 2003 and 2004. The researchers attributed this to possibly being the result of the closure in 2003 of an HBCD manufacturing plant in northeastern England, and noted two voluntary schemes to reduce emissions to the environment that took effect in 2006.

Lindberg et al. (2004) analyzed peregrine falcon (*Falco peregrinus*) eggs collected from 1991 to 1999 from wild and captive breeding populations in Sweden. Eggs from a northern wild breeding population contained 34–590 ng/g lipid weight, while those from the south contained 79–2400 ng/g lipid weight. HBCD was not detected in eggs collected from the captive breeding population (detection limits: 4–8 ng/g lipid weight). Dietary differences were considered primarily responsible for the observed range in HBCD levels. Birds from the northern wild population prey mainly on aquatic species, such as waders and ducks, while those in the south feed on birds in the terrestrial food web

(Lindberg and Odsjö 1983). The captive breeding population received a controlled diet of domestic chickens. These samples were later re-examined alongside eggs collected from the same regions from 1987 to 1999. These tests confirmed higher concentrations of HBCD in the two wild populations compared with the concentrations in the captive population (Johansson et al. 2009).

Studies from Asia indicate that HBCD is widely distributed among aquatic species in the Asia-Pacific region. Ueno et al. (2006) reported a maximum concentration of 45 ng/g lipid weight in muscle samples of skipjack tuna (*Katsuwonus pelamis*) collected from 1997 to 2001 in offshore waters near Japan, Taiwan, the Philippines, Indonesia, the Seychelles and Brazil, as well as various locations in the Japan Sea, East and South China seas, Indian Ocean and North Pacific Ocean. The presence of HBCD in all but three of the 65 samples, including those taken from remote regions in the mid-Pacific Ocean, was considered evidence of widespread contamination in the global marine environment. Similar concentrations were observed in tuna collected from remote regions of the North Pacific Ocean (up to 29 ng/g lipid weight) and those from coastal Asian areas (28-45 ng/g lipid weight in samples from off the coast of Japan and East China Sea). This was considered indicative of an unknown local pollution source in the North Pacific or evidence of long-range atmospheric transport of HBCD with subsequent deposition in cold-water regions through the process of global distillation, or both. Other recent studies report the presence of HBCD in aquatic invertebrates (Ramu et al. 2007), fish (Xian et al. 2008) and marine mammals (Isobe et al. 2008) collected from coastal areas of Korea and China, as well as terrestrial vertebrates in Japan (Kunisue et al. 2008).

A recent study of bivalves (oysters and mussels) from Japanese coastal waters found that HBCD was ubiquitous in the area and was the dominant organohalogen measured in study organisms (Ueno et al. 2010). HBCD concentrations ranged from 12–5200 ng/g lipid weight (sample lipid weight 1–3.5%), with the highest levels found near urban and industrial areas, within which the highest levels were near the Osaka area. HBCD levels in Japanese bivalves measured in this study are reported to be among the highest in Asia and comparable to the highest levels in Europe, and were greater than PBDE concentrations measured in this study by 1 to 2 orders of magnitude. Alpha-HBCD was predominant in the bivalves sampled, followed by γ-HBCD.

Presence in Remote Regions

HBCD has been measured in air, sediment and biota collected in regions considered to be remote from potential sources, including the Arctic.

Continuous air measurements taken between 2006 and 2007 at Alert, Nunavut, in the Canadian Arctic show that HBCD was sometimes detected (> 0.001 ng/m^3 HBCD) (Xiao et al. 2010). The author noted that a lack of seasonal variability in flame retardant concentrations may indicate that measurements are reflective of global background contamination and long-range atmospheric transport.

Remberger et al. (2004) reported concentrations of up to 0.28 ng/m^3 in air samples collected at remote sampling locations in Sweden and the Arctic areas of Finland.

Air samples collected near Ny Ålesund, Svalbard in the Norwegian Arctic found mean air concentrations of 0.0065–0.0071 ng/m^3 HBCD, and found γ-HBCD was predominant (Manø et al. 2008).

Concentrations of 0.43 ng/g dw (α-HBCD) and 3.88 ng/g dw (γ-HBCD) were measured in sediment collected from Lake Ellasjøen on Bjornoya (Bear Island) in the Norwegian Arctic (Evenset et al. 2007). The β-isomer was not detected in the samples (detection limit: 0.06 ng/g dw). The authors estimated the sediment layer represented the years 1973 to 1987.

Yolk of newly hatched European shag (*Phalacrocorax aristotelis*), a fish-eating top predator related to the cormorant, contained a mean concentration of 417 ng/g lipid weight of HBCD (Murvoll et al. 2006a). The samples were collected in 2002 from a Norwegian island considered remote and free from pollution. HBCD was present in all of 30 samples. The samples were also analyzed for several of the more persistent and bioaccumulative PBDE congeners. The mean concentration of HBCD in the yolk samples exceeded that of any PBDE congener measured, including PBDE-47 (mean concentration of 5.59 ng/g ww), PBDE-99 (1.56 ng/g ww) and PBDE-100 (6.16 ng/g ww), as well as total PBDEs (17.2 ng/g ww; sum of seven tri- to hexaBDE congeners).

A similar study was conducted on North Atlantic kittiwake (*Rissa tridactyla*) collected from an island off Norway and at Svalbard in the Norwegian Arctic (Murvoll et al. 2006b). Yolk sacs collected from newly hatched chicks contained mean concentrations of 260 ng/g lipid weight (island location) and 118 ng/g lipid weight (Arctic location). The presence of HBCD in Arctic kittiwake hatchlings provides further evidence of possible transport of the substance to regions remote from its source.

Muir et al. (2006) reported total HBCD in adipose tissue of polar bears (*Ursus maritimus*) from Alaska, Eastern Greenland and Svalbard in the Norwegian Arctic. Concentrations of up to 35.1 ng/g lipid weight were measured in two of eight female bears collected from 1994 to 2002 in the Bering-Chukchi Sea of Alaska. Male bears in the region contained no detectable HBCD (detection limit: 0.01 ng/g lipid weight). HBCD was present in all 11 samples collected from 1999 to 2001 from female polar bears in Eastern Greenland. Concentrations ranged from 32.4 ng/g to 58.6 ng/g lipid weight in the samples. HBCD was also present in all 15 samples collected in 2002 from female bears in the Svalbard area, with concentrations of 18.2–109 ng/g lipid weight.

Concentrations of 0.07–1.24 ng/g ww were measured in the blood plasma of adult glaucous gulls (*Larus hyperboreus*) collected in the Norwegian Arctic during May and June 2004 (Verreault et al. 2005). Plasma collected from female polar bears (*Ursus maritimus*) living in the same region contained up to 0.85 ng/g ww. While HBCD was present in all 27 gull samples, only 2 of the 15 polar bear plasma samples contained levels above the detection limit (0.03 ng/g ww). The researchers hypothesized that the lower occurrence in the bears may indicate a superior ability to detoxify and eliminate HBCD. Alternatively, the lower levels may reflect differences in diet and feeding rate

between the two species. Plasma levels averaged 1.73–2.07 ng/g ww in gulls collected from the same region in May and June of 2006 (Verreault et al. 2007a). HBCD was found in around 60% of the 49 plasma samples; however, the substance was present in all 31 gull eggs sampled in the study, with an average concentration in the yolk of 19.8 ng/g ww and a maximum measured value of 63.9 ng/g ww. The results provide evidence of potential maternal transfer of HBCD to the eggs of glaucous gulls.

An earlier study by Verreault et al. (2007b) measured average concentrations of 3.29 ng/g and 75.6 ng/g ww in blood and liver, respectively, collected from Norwegian Arctic glaucous gulls in early July 2002. Whole body concentrations ranged from 52.6 ng/g to 270 ng/g ww (mean of 117 ng/g ww) with feathers, and from 38.4 ng/g to 194 ng/g ww (mean 91.0 ng/g ww) when content in the feathers was not included.

Sørmo et al. (2006) analyzed representative species from various trophic levels of the polar bear food chain, using samples collected from 2002 to 2003 at Svalbard in the Norwegian Arctic. HBCD was below detection limits (minimum 0.012 ng/g lipid weight) in the amphipod, *Gammarus wilkitzkii*. Concentrations increased from polar cod (*Boreogadus saida*; 1.38 ng/g to 2.87 ng/g lipid weight) to ringed seal (*Phoca hispida*; 14.6 ng/g to 34.5 ng/g lipid weight), but decreased in the top predator, polar bear (*Ursus maritimus*, 5.31 ng/g to 16.51 ng/g lipid weight). The results suggested that substantial biomagnification was occurring from polar cod to ringed seal but none from ringed seal to polar bear. The lower levels in the polar bear samples were considered to indicate possible enhanced metabolic capability in the bears.

Gebbink et al. (2008) measured a mean concentration of 41 ng/g ww in adipose tissue collected from 10 adult male and 10 adult female polar bears in central East Greenland between 1999 and 2001. The substance was not detected in blood, brain and liver samples from the bears (detection limit not specified). Morris et al. (2007) reported a concentration of 0.38 ng/g lipid weight in the blubber of ringed seal (*Phoca hispida*) from the Barrow Strait, Nunavut.

Tomy et al. (2008) investigated isomer-specific accumulation of HBCD at several trophic levels of an eastern Canadian Arctic marine food web. Alpha- and γ-HBCD were present in all species examined (beluga whale, *Delphinapterus leucas*; walrus, *Odobenus rosmarus*; narwhal, *Monodon monoceros*; arctic cod, *Boreogadus saida*; deepwater redfish, *Sebastes mentella*; shrimp, *Pandalus borealis* and *Hymenodora glacialis*; clam, *Mya truncata* and *Serripes groenlandica*; and mixed zooplankton) with total HBCD concentrations ranging from 0.6 ng/g (geometric mean) to 3.9 ng/g lipid weight. The β-isomer was below detection limits (0.0004–0.0059 ng/g lipid weight) in all samples. No clear trend was evident in the diastereomeric profile of the animals; however α-HBCD contributed greater than 70% of the total HBCD burden in shrimp, redfish, arctic cod, narwhal and beluga, while zooplankton, clams and walrus contained more than 60% γ-HBCD. The observed differences in diastereomer predominance were attributed, at least in part, to the differing environmental fates and behaviours of the isomers, with the least water-soluble γ-HBCD more likely to diffuse passively from the water into zooplankton, which have proportionately high lipid content. Similarly, as benthic filter

feeders, clams may be more likely to absorb a large proportion of γ-HBCD from the surrounding sediment, where this isomeric form has been shown to predominate. The presence of large proportions of α-HBCD, such as in the beluga and narwhal, may indicate enhanced metabolic capability based on evidence of stereoisomer-specific biotransformation of the γ-isomer into the α- form (see, for example, Zegers et al. 2005; Law et al. 2006b). The researchers reported a significant positive relationship of α-HBCD with trophic level, indicative of biomagnification throughout the food web, while a significant negative relationship was observed between concentrations of γ-HBCD and trophic level (i.e., trophic dilution).

Tomy et al. (2009) examined a western Canadian Arctic marine food web and found that median HBCD concentrations in pelagic fish ranged from 0.9 ng/g lipid weight in Arctic cisco (*Coregonus autumnalis*) to 11.8 ng/g lipid weight in Arctic cod (*Boreogadus saida*). In higher-trophic-level organisms, median HBCD concentrations ranged from 1.1 ng/g lipid weight in ringed seal (*Phoca hispida*) to 1.9 ng/g lipid weight in beluga whales (*Delphinapterus leucas*).

Temporal Trends

Remberger et al. (2004) reported concentrations of 0.8–1.5 ng/g dw in surface sediments (2–4 cm in depth) collected in 1996 and 1997 from three locations in Stockholm. Deeper core samples (20–32 cm in depth) from the same sites contained 0.2-0.5 ng/g dw. Higher concentrations in the surface sediments were considered to indicate increasing deposition with time. Based on radioactive dating, the surface sediments were estimated to originate in the mid 1990s, while those in the deeper layers represented deposition from the 1960s and possibly earlier. Given that the chemical was introduced to the market during the 1960s, the authors describe the occurrence of HBCD in the oldest sediments as surprising, and speculate about whether mixing by bioturbation and/or uncertainty in the dating models influenced these sediment results.

Kohler et al. (2008) reported a rapid and linear increase in HBCD levels present in successive layers of a sediment core collected in 2003 from the deepest point of a shallow suburban lake in Switzerland. HBCD first appeared in a sediment layer corresponding to approximately the mid 1970s and reached a maximum concentration of 2.5 ng/g dw at the surface layer of the core, estimated to be from approximately 2001. A similar trend was evident in a sediment core collected from a deep pre-alpine Swiss lake, with levels of less than 0.1 ng/g dw in samples from prior to 1980 and increasing rapidly to a maximum concentration of around 0.7 ng/g dw in the surface layer, corresponding to the early 2000s (Kohler et al. 2007).

HBCD was present in all three sediment cores and six surface sediment samples collected in 2002 from Tokyo Bay (Minh et al. 2007). Concentrations ranged from 0.056 ng/g to 2.3 ng/g dw, with the highest levels found near densely populated and industrialized areas. HBCD first appeared in the sediment cores at depths of 20–25 cm, estimated to date from the late 1960s and early 1970s, with the concentration increasing steadily to the

highest levels at the surface. Based on the data, Tanabe (2008) estimated concentration doubling times of 7.1–12 years for HBCD in the sediment.

A number of studies examine HBCD concentrations in biota over time as a means of identifying possible trends in contamination levels. However, there are few studies that analyze temporal trends of HBCD in Canadian and North American biota. Braune et al. (2007) reported mean concentrations of 2.1–3.8 ng/g lipid weight in pooled samples of eggs of the ivory gull (*Pagophila eburnea*) collected from the Canadian Arctic from 1976 to 2004. Over 28 years, concentrations decreased from a value of 3.8 ng/g lipid weight in 1976 to 3.0 ng/g lipid weight in 1987 and 2.1 ng/g lipid weight in 2004.

Archived samples of Lake Ontario lake trout, *Salvelinus namaycush*, contained from 16–33 ng/g lipid weight (2–4 ng/g ww) total HBCD, with total HBCD decreasing significantly over the 25 years between 1979 and 2004 (Ismail et al. 2009). Although α-HBCD predominated in the samples, the decreasing temporal trend for this isomer was not significant. The need for further research was emphasized, given the conflicting evidence of increasing temporal trends reported in other studies.

Stapleton et al. (2006) measured 0.71–11.85 ng/g ww in blubber samples collected from male California sea lions (*Zalopus californianus*) stranded along the California coast between 1993 and 2003. HBCD was present in 80% of the samples analyzed, with α-HBCD predominant in all samples. Levels increased almost exponentially over the 10-year study period and, while the researchers cautioned that the sample size of 26 might have been too limited to allow accurate estimation of accumulation rates, the doubling time in the sea lion blubber over the study period was approximately two years, if the increase is assumed to be exponential as the data suggest (Stapleton et al. 2006).

As described above, Law et al. (2006d, 2008) detected HBCD in samples of harbour porpoise blubber collected from 1994 to 2003 from animals stranded or caught in waters off the U.K. coast (Law et al. 2006d), and found concentrations in the blubber increased sharply from approximately 2001 to 2003 (ranging from 10–19 200 ng/g ww). The authors subsequently found, in samples collected at the same location between 2003 and 2006, a statistically significant decrease in levels between 2003 and 2004 (concentrations ranged from less than 10 ng/g to 11 500 ng/g ww).

A marked increase was evident in blubber concentrations of juvenile male grey seals (*Halicoerus grypus*) collected in the Baltic Sea from 1980 to 2000 (Roos et al. 2001). Concentrations ranged from 16 ng/g to 177 ng/g lipid weight, with lowest levels in seals collected during the early 1980s.

Atlantic cod (*Gadus morhua*) collected in 2003 from the southern industrialized region of Norway, near Oslo, contained up to 16.9 ng/g ww (56.9 ng/g lipid weight), while those collected from the same region in 1998 contained up to 2.70 ng/g ww (22.67 ng/g lipid weight; Bytingsvik et al. 2004). This represents a more than six-fold increase when considered on a ww basis (a more than 2.5-times increase in terms of lipid weight).

Sellström et al. (2003) observed a steady and significant ($p < 0.001$) increase in concentrations present in the eggs of guillemot (*Uria aalgae*) collected from the Baltic Sea from 1969 to 2001. The observed increase was attributed to increasing use of HBCD, although this was difficult to substantiate due to a lack of industrial production and use information. The presence of HBCD in the eggs was considered to indicate possible biomagnification of the substance (Kierkegaard et al. 1999).

Diastereomeric Differences

Studies providing a breakdown of the individual diastereomers commonly report a predominance of α-HBCD in biota samples, with γ- and β-HBCD being present at lower levels or below detection limits. This congener profile contrasts markedly with that seen in commercial formulations and sediment samples, in which γ-HBCD most often dominates. The isomeric pattern observed in biota may reflect differences in exposure potential, uptake, metabolism or depuration of the three isomers. There is evidence that conversion of γ-HBCD to α-HBCD occurs at temperatures above 160°C (Peled et al. 1995), suggesting that finished products subjected to high temperatures during processing may carry a much higher proportion of α-HBCD than that present in the original technical formulation. For instance, Kajiwara et al. (2009) observed the proportion of γ-HBCD decrease and that of α-HBCD increase at the boiling point of toluene (110.6°C), and concluded that HBCD isomer profiles in textiles will change during the heating process. A detailed study of HBCD isomerization at 160°C demonstrated a "cascade of interconversions" where all stereoisomers develop from any given one, although at different rates (Koeppen et al. 2008). The authors suggested that this cascade process is likely typically obscured by the predominating, faster transformation of γ-HBCD to α-HBCD in the alteration of technical HBCD mixture. Processes resulting in a higher proportion of α-HBCD in finished products may increase the potential for organism exposure to α-HBCD during product use and disposal.

A recent soil assay study isolated a soil bacterial strain, *Pseudomonas* sp., that degraded γ-HBCD by 81% within five days, but failed to degrade α-HBCD, suggesting that any γ-HBCD debrominating enzymes possess substrate specificity (Yamada et al. 2009).

Alpha-HBCD has higher water solubility (see Table A-2), suggesting that it may more readily enter organisms through preferential transfer from particles through water (Morris et al. 2004).

Janák et al. (2005) reported consistently higher levels of α-HBCD compared with those of γ-HBCD in the livers of several fish species, and considered this a possible indication that γ-HBCD was more easily metabolized. Further evidence for differential rates of biotransformation was provided by *in vitro* assays in which β- and γ-HBCD were significantly metabolized by rat and harbour seal liver microsomes, while α-levels remained mostly unchanged (Zegers et al. 2005). The net result was accumulation of α-HBCD relative to that of the other two isomers.

Research by Law et al. (2006b) suggests that bioformation or bioisomerization of HBCD appeared to occur in some species. Statistically significant amounts of α-HBCD were

measured in the muscle tissue of rainbow trout (*Oncorhynchus mykiss*) exposed exclusively to γ-HBCD via the diet. Similarly, both α- and γ-HBCD were present in statistically significant quantities in fish exposed only to β-HBCD. The results suggested that selective bioisomerization of HBCD, with preferential formation of α-HBCD, may contribute appreciably to determining isomer distributions in the environment. The α-isomer appeared recalcitrant to bioisomerization in the fish, a factor that may also contribute to its proportionately higher tissue levels in biota samples.

Tomy et al. (2009) found differences in HBCD diastereomer concentration profiles throughout a western Canadian Arctic and marine food web. Alpha-HBCD accounted for 95% of the total HBCD body burden in Beaufort Sea beluga whales (*Delphinapterus leucas*) (γ-HBCD < 5%), whereas α-HBCD and γ-HBCD accounted for 20% and 77%, repectively, of the total HBCD body burden in its primary prey, Arctic cod (*Boreogadus saida*). The authors suggested that this was evidence of biotransformation of γ-HBCD to α-HBCD in higher-trophic-level organisms.

Ecological Effects Assessment

The available data set for HBCD includes endpoint values from several pelagic trophic levels (i.e., fish, invertebrates, algae), as well as data for benthic and terrestrial species. Most data were derived using standard methods and species, although results from novel studies are also reported in the literature. Acute or chronic (partial life cycle) toxicity testing results (or both) are available for rainbow trout (*Oncorhynchus mykiss*), bluegill sunfish (*Lepomis macrochirus*), water flea (*Daphnia magna*), green algae (*Pseudokirchneriella subcapita , Chlorella* sp., formerly *Selenastrum capricornutum*, *Chlorella* sp.) and diatoms (*Skeletonema costatum, Thalassiosira pseudonana*). Toxicity data are also available for benthic organisms (*Lumbriculus variegates*, *Hyalella azteca*), earthworm (*Eisenia fetida*), cultured chicken (*Gallus domesticus*), American kestral eggs (*Falco sparverius*) and six terrestrial plant species. While most studies failed to determine a numerical endpoint value, indicating only that minimum effect levels can be expected to exceed that of the highest concentration tested, the quantity and quality of the available studies make HBCD a rich source of data compared to most brominated flame retardants.

It should be noted that toxicity studies generally utilize the commercial HBCD mixture; thus, organisms would be exposed to various amounts of each diastereomer found in the commercial product. Inferences about which diastereomer is responsible for the observed effects are thus not possible. It should also be noted that testing the toxicity of HBCD to aquatic organisms may be complicated by its low water solubility. In some studies, test organisms are exposed to water concentrations that exceed the reported water solubility limits for HBCD, making interpretation of effects difficult (OECD 2000; Arnot et al. 2009). For this assessment, toxicity studies in which calculated-effect concentrations exceeded the reported water solubility were reviewed on a case-by-case basis (OECD 2000, ECB 2003). However, preference was given to aquatic studies with measured treatment concentrations within the range of water solubility.

ECOSAR (2004) classifies the substance as a neutral organic, based on its chemical structure. As a neutral organic, HBCD is expected to exhibit effects through nonpolar narcosis (i.e., through non-specific disruption of cellular membrane integrity or function, or both). Recent studies have provided information on possible sites of toxic action for HBCD. Studies indicate sublethal exposures of HBCD may affect the thyroid and liver systems of fish (Ronisz et al. 2004; Lower and Moore 2007; Palace et al. 2008; Palace et al. 2010) and mammals (Legler 2008). Embryo toxicity studies demonstrate HBCD effects on hepatic gene expression in bird embryos (Crump et al. 2010) and oxidative stress and cell apoptosis in fish embryos (Deng et al. 2009).

HBCD has demonstrated toxicity in both aquatic and terrestrial organisms, with significant adverse effects on survival, reproduction and development reported in algae, aquatic invertebrates, fish and terrestrial annelid worms. In aquatic species, a 21-day no-observed-effect concentration (NOEC) and lowest-observed-effect concentration (LOEC) of 3.1 µg/L and 5.6 µg/L, respectively, were determined for the water flea, *Daphnia magna*, based on significantly reduced growth (CMABFRIP 1998). Daphnids exposed to the highest test concentration of 11 µg/L exhibited statistically significant reductions in length, dry weight, and number of young.

Walsh et al. (1987) examined the effect of HBCD on population density in two unicellular marine algae, *Skeletonema costatum* and *Thalassiosira pseudonana*, using six nutrient media. Depending on the nutrient medium used, the 72-hour median effective concentration (EC_{50}) values based on reduced population density ranged from 9.3 µg/L to 12.0 µg/L in *S. costatum* and from 50 µg/L to 370 µg/L in *T. pseudonana*.

Ronisz et al. (2004) injected juvenile rainbow trout, *Oncorhynchus mykiss*, with HBCD dissolved in peanut oil and observed the effects on several biomarkers relating to liver enzyme function and hormonal activity. Ethoxyresorufin-*O*-deethylase activity was significantly inhibited in fish receiving approximately 5×10^5 µg/kg-bw for a period of 28 days, while fish dosed at 5×10^4 and 5×10^5 µg/kg-bw for 5 days displayed significantly increased catalase activity. Significant increases in the liver somatic index (LSI; liver weight as a percentage of whole body weight) were evident in high-dose fish following an exposure period of 28 days. The induction of catalase at 5 days, together with increased LSI in exposed fish after 28 days, suggested that HBCD may be a peroxisome proliferator, a negative hormonal response. Further investigation into this possibility by the researchers yielded inconclusive results. Peroxisome proliferators are considered to be tumor promoters through a non-genotoxic mechanism (Waxman 1999; Vanden Heuvel 1999) and have been associated with hepatocarcinogenesis (Ackers et al. 2000).

Altered thyroid status, including changes to circulating plasma thyroid hormone levels and hepatic metabolic enzyme activity, were reported in juvenile rainbow trout fed lipid-corrected concentrations of 29.14 µg/kg, 11.84 µg/kg and 22.84 µg/kg of α-, β- or γ-HBCD, respectively (approximately 10 µg/kg to 30 µg/kg-bw) for 56 days followed by a clearance period of 112 days (Palace et al. 2008). The results provided evidence that

HBCD exposure can affect the thyroid system in fish, with effects increasing at higher concentrations.

To examine the potential effects of individual HBCD diastereomers, Palace et al. (2010) fed juvenile rainbow trout environmentally relevant concentrations (5 ng/g in diet) of α-, β- and γ-HBCD (or control) for 32 days, followed by oral dosing with [^{125}I]-T4 (thyroid hormone) to examine tissue disposition over 14 days. Although [^{125}I]-T4 uptake was low (< 10%), measurements indicated that tissue disposition and elimination of the hormone were affected by HBCD. The authors concluded that HBCD may affect the thyroid system of fish through lower iodide uptake by the gland, or through elevating thyroid hormone turnover rate. They also reported that fish exposed to β- and γ-HBCD had significantly higher activity of type II outer ring deiodinase enzyme in their livers relative to reference fish.

Atlantic salmon, *Salmo salar* L., exposed to low levels of HBCD (0.011 µg/L) in freshwater for 30 days over the peak natural smoltification period, and then transferred to clean seawater for 20 days, exhibited significant alterations in the levels and patterns of circulating thyroid hormones (Lower and Moore 2007). These hormones play a key role in smoltification and are critical to the imprinting of olfactory memory, which allows the fish to return to their natal river for spawning. Thyroid hormone (T4, T3) levels were significantly higher in control fish following transfer to seawater, peaking at the time of transfer. In contrast, the levels in HBCD-exposed fish did not show this increase at transfer, peaking earlier, at the end of the freshwater exposure period. Olfactory sensitivity was also significantly decreased in the HBCD-exposed fish. The researchers concluded that while all fish appeared to complete the parr-smolt transformation successfully and were able to survive and osmoregulate in saline conditions for a period of 20 days, the HBCD-exposed fish displayed evidence of disruption to thyroid hormone homeostasis during development, which may ultimately affect imprinting and other behaviour in the adult fish.

Deng et al. (2009) conducted 96-h post-fertilization-toxicity waterborne experiments on zebrafish embryos, with HBCD concentrations of 0, 50, 100, 500 and 1000 µg/L (not specified if nominal or measured concentrations). The results showed embryo malformations in a concentration-dependent manner (significant effects initiated at 100 µg/L), and significantly reduced survival in larvae initiated at 50 µg/L and above. Heart rate (all exposures) and larval growth (100 µg/L and above) were also significantly reduced compared to the control group. In examining reactive oxygen species and gene expression at the various HBCD concentrations, the authors suggest the mechanism of developmental toxicity appears to be the generation of oxidative stress and consequent triggering of apoptosis genes.

Sediment testing with the freshwater oligochaete, *Lumbriculus variegates*, yielded 28-day NOEC and LOEC values of 3.25×10^3 and 2.93×10^4 µg/kg dw of sediment, respectively, based on significant reductions in total worm numbers (Oetken et al. 2001). The researchers concluded that the sediment-bound fraction of HBCD is bioavailable and causes effects. ACCBFRIP (2003d, 2003e) conducted 28-day tests using the same

species, as well as the amphipod, *Hyalella azteca*, and chironomid, *Chironomus riparius*, but found no dose-responsive, statistically significant effects in any of the three species up to concentrations of 1×10^6 µg/kg dw of sediment.

The effects of HBCD on terrestrial plant seedling emergence and growth were evaluated in a 21-day study using corn (*Zea mays*), onion (*Allium cepa*), ryegrass (*Lolium perenne*), cucumber (*Cucumis sativa*), soybean (*Glycine max*) and tomato (*Lycopersicon esculentum*) (ACCBFRIP 2002). No apparent adverse treatment-related effects were observed on seedling emergence, survival or growth for any of the six species tested, and the 21-day NOEC for the study was equal to or greater than the highest test concentration of 5×10^6 µg/kg dw of soil.

A toxicity study using the earthworm, *Eisenia fetida*, determined a 56-day NOEC and LOEC of 1.28×10^5 and 2.35×10^5 µg/kg dw of soil, respectively, based on significantly reduced reproduction (ACCBFRIP 2003a). The 56-day EC_{10} (10% inhibition) and EC_{50} (50% inhibition) for reproduction were 2.16×10^4 and 7.71×10^5 µg/kg dw of soil, respectively. As the calculated EC_{10} value was less than the lowest concentration tested, it was considered an estimate only. There was no significant effect on adult worm survival, and the 28-day NOEC for survival was equal to or greater than the highest test concentration of 4.19×10^6 µg/kg dw of soil.

There are limited reports describing potential effects on wildlife species. A number of studies have examined toxicity in rodents; these studies are summarized in the Human Health portion of this assessment.

Crump et al. (2008) reported significant up-regulation of enzymes involved with the metabolism of xenobiotics (CYP enzymes and uridine 5'-diphospho-glucuronosyltransferase) in the domestic chicken, *Gallus domesticus*, hepatocytes following 24- and 36-hour exposures to concentrations of 1 µM to 30 µM α-HBCD or technical HBCD. Significant down-regulation of proteins associated with the thyroid hormone pathway and lipid regulation also occurred in this concentration range.

Crump et al. (2010) reported HBCD effects on embryo toxicity, isomer-specific accumulation in liver and cerebral cortex, and hepatic gene expression in the domestic chicken at environmentally relevant concentrations for avian species. Eggs were injected with the following doses: control (0 mg/ml), 50 ng/g egg (actual HBCD-technical mixture concentration: 0.22 mg/ml), 100 ng/g egg (actual concentration: 0.43 mg/ml), 300 ng/g egg (nominal concentration: 1.5 mg/ml, actual not quantified), 1000 ng/g egg (actual concentration: 4.98 mg/ml), and 10 000 ng/g egg (nominal concentration: 50 mg/ml, actual not quantified). Pipping success was reduced at the 100 ng/g egg dose (0.43 mg/ml) to 70.9% compared to control. Pipping success was reduced to 35% at the 10 000 ng/g dose, but no significant pipping effects were found at 50, 300 or 1000 ng/g. Further analysis indicated that the toxic effects of HBCD occurred after initial embryogenesis. Liver tissue and cerebral cortex tissue concentrations of embryos increased in a dose-dependent manner, with maximum concentrations reaching 1170 g/g ww in liver and 102 ng/g ww in cerebral cortex tissue at the 10 000 ng/g dose.

Isomer-specific processes occurred in the egg or embryos 19–20 days after exposure *in ovo*. In liver tissue, the proportion of α-HBCD increased compared to HBCD-technical mixture, while γ-HBCD decreased and no effect on β-HBCD was found. No significant change in the proportion of isomers in cerebral cortex was detected. Genes associated with phase I and II metabolism, thyroid hormone homeostasis, lipid regulation, and hormones associated with growth were altered by HBCD.

Fernie et al. (2009) exposed captive American kestrals (*Falco sparverius*) by diet to environmentally relevant levels of pentaBDE technical formulation (DE-71), which unintentionally included HBCD, for three weeks prior to pairing through to first hatching. The birds exposed to the pentaBDE and HBCD mixture showed delayed egg laying, and laid smaller eggs with thinner eggshells. Further effects included differential weight loss during embryonic development, reduced fertility, and reduced reproductive success. Alpha-HBCD levels in exposed eggs were 3.27 ± 0.68 ng/g ww (low exposure) and 15.61 ± 2.43 ng/g ww (high exposure), comparable to levels found in wild herring gulls and peregrine falcons. Correlation analysis among tissue concentrations and reproductive parameters indicated that thickness of shell significantly declined with increasing α-HBCD concentrations, and laying time for the entire clutch increased. Mass of eggshell was not significantly affected by α-HBCD level.

In a related study, Marteinson et al. (2010) found that accidental exposure of male *in ovo* Amercan kestral (*Falco sparverius*) to small concentrations of HBCD during exposure to pentaBDE technical formulation (DE-71) may have contributed to synergistic/additive effects. HBCD levels in male offspring of kestrals were measured at 3.27 ± 0.68 ng/g ww (low exposure), and 15.61 ± 2.63 ng/g ww (high exposure) based on sibling eggs. HBCD levels were significantly negatively correlated with reproductive success parameters of the male offspring: clutch size, fertility, copulation and behaviour. However, because PBDE levels were also significantly correlated to these parameters, the authors determined that it was difficult to separate the influences of HBCD from those of PBDE.

Brandsma et al. (2009) studied hydroxylated metabolites of HBCD in tern egg, harbour seal, flounder, and 28-day-exposed Wistar rats (30 and 100 mg HBCD per kg-bw per day). The authors found four groups of hydroxylated HBCD metabolites in various rat tissues: monohydroxy-metabolites of HBCD, pentabromocyclododecene (PBCDe), tetrabromocyclododecene (TBCDe) and dihydroxy-HBCD. Debromination of HBCD to PBCDe was also identified as a pathway in rat tissue. In the wildlife samples, the presence of monohydroxy-HBCD was found in tern eggs, monohydroxy-metabolites of HBCD and PBCDe were found in blubber of harbour seal, but no metabolites were detected in flounder muscle tissue.

Summaries of key toxicity studies used in the ecological effects assessment of HBCD are provided in Table A-15. Some key studies were critically reviewed for vailidity. These reviews (Robust Study Summaries) are found in Appendix D.

Potential to Cause Ecological Harm

The approach taken has been to examine various pieces of scientific information, and to develop conclusions based on a weight-of-evidence approach and application of precaution, as required under CEPA 1999. The screening assessment is a conservative assessment, intended to represent reasonable worst-case conditions. It integrates known or potential exposure to the substance with known or potential effects on the environment.

The potential for HBCD to persist in the environment and accumulate within organisms formed primary lines of evidence in support of a decision relating to ecological harm. Substances that are persistent remain in the environment for a long time after being released, increasing the potential magnitude and duration of exposure. Substances that have long half-lives in mobile media (air and water) and that are present within these media have the potential to cause widespread contamination. Releases of small amounts of bioaccumulative substances may lead to high internal concentrations in exposed organisms. Highly bioaccumulative and persistent substances are of special concern, since they may biomagnify in food webs, resulting in very high internal exposures, especially for top predators. Evidence that a substance is both persistent and bioaccumulative, when taken together with other information (such as evidence of harmful effects at relatively low concentrations, and evidence of uses and releases within Canada) may therefore be sufficient to indicate that the substance has the potential to cause ecological harm.

HBCD has been detected in all environmental media, and there is evidence that the substance meets CEPA 1999 persistence criteria (half-life in air of 2 days or more, half-lives in soil and water of 182 days or more, and half-life in sediment of 365 days or more; see Table A-5). In addition, the substance is present in samples collected from regions considered remote from potential sources, including the Arctic, indicating that it is sufficiently stable in the environment to allow long-range transport in air or water, or both. Atmospheric transport of a substance to an area remote from its source is a criterion for persistence in air, as defined by the *Persistence and Bioaccumulation Regulations*..

Measured BCFs of up to 18 100 are reported in the published literature. Based on these data, HBCD meets bioaccumulation criteria defined by the *Persistence and Bioaccumulation Regulations* (bioaccumulation and bioconcentration factors of 5000 or more; see Table A-5).

HBCD has demonstrated ecotoxicity in both aquatic and terrestrial species (21-day LOEC of 5.6 µg/L for reduced growth in *Daphnia magna*, for example; CMABFRIP 1998), with significant adverse effects on survival, reproduction and development reported in algae, daphnids and annelid worms. Recent studies indicate a potential link to altered hormonal status in fish, with reported impacts on the activity and normal functioning of liver enzymes (Ronisz et al. 2004) and thyroid hormones (Lower and Moore 2007; Palace et al. 2008). The α-isomer has displayed a greater capacity to disrupt

hormonal function i*n vitro* and this apparent higher potency is of concern, given the higher prevalence of this diastereomer, compared to the other two, in biota samples.

As mentioned previously, combustion of HBCD under certain conditions may lead to the formation of polybrominated dibenzo-*p*-dioxins and polybrominated dibenzofurans, brominated analogues of the Toxic Substances Management Policy Track 1 polychlorinated dibenzo-*p*-dioxins and dibenzofurans. Trace levels of these compounds and their precursors have been measured during uncontrolled combustion of flame-retarded polystyrene materials containing HBCD.

HBCD levels in the environment are generally increasing (UNEP-POPs 2010). For the years and continents having available data since 2000, increases in the demand for HBCD have been reported (BSEF 2006, ECHA 2008). Higher concentrations are reported in surficial layers of sediment cores as compared with those in deeper layers, an indication of increasing deposition with time (Remberger et al. 2004; Minh et al. 2007; Kohler et al. 2008). As well, time-trend analyses conducted using birds (Sellström et al. 2003) and marine mammals (Roos et al. 2001; Stapleton et al. 2006; Law et al. 2006d) document nearly exponential increases in biota levels beginning in the early 1990s. While HBCD was first commercially introduced to the brominated flame retardant market in the 1960s, its application in extruded polystyrene did not commence until the 1980s (2007 email from an Environmental Quality Manager of the importing company to Existing Substances Branch, Environment Canada; unreferenced). There is also evidence that HBCD concentrations in the environment may be increasing to levels that are similar to those of PBDE flame retardants, some of which are no longer in production. Spatial concentration patterns of HBCD in U.S. air samples were similar to those of PBDE-209, possibly signalling a shift in the dominant products (Hoh and Hites 2005). This is further supported by comparison studies that report levels approaching or exceeding those of PBDEs in compost (Zennegg et al. 2005) and bird yolk (Murvoll et al. 2006a, 2006b). It has also been reported that, as for PBDEs, HBCD is ubiquitous in the Arctic (de Wit et al. 2010).

The available information on the persistence, bioaccumulation potential, ecotoxicity and use and potential release of HBCD in Canada therefore suggests that this substance has the potential to cause ecological harm in Canada.

Quantitative risk estimation methods, integrating conservative estimates of exposure with effects information, are also used to evaluate potential to cause ecological harm. Due to the general paucity of HBCD surface water and sediment concentrations in Canada, a fugacity modelling approach, based on principles described by Cahill et al. (2003) and, more generally, Mackay (1991), was applied for estimating local aquatic exposure to HBCD in the pelagic and benthic compartments and determining predicted exposure concentrations (PECs) in water and sediments (see Appendix C for model description). The database of soil HBCD concentrations was also considered inadequate, and so the soil PEC was derived using a simple calculation procedure involving the application of sewage sludge to agricultural soil and pastureland. A summary of data used in the risk quotient analysis of HBCD is presented in Table A-16. Exposure data used in the

determination of PECs can be found in Tables A-7 and A-8. Toxicity data used to determine critical toxicity values and predicted no-effect concentrations (PNECs) are summarized in Table A-15.

The aquatic exposure scenario yielded a PEC of 0.00004–0.006 mg/L for water and 10.33–46.2 mg/kg dw for sediment (Table A-16). A pelagic organism PNEC was derived from the chronic toxicity value of 0.0056 mg/L (as the most sensitive valid experimental value) for *Daphnia magna*, by dividing this value by an assessment factor of 10 (to account for interspecies and intraspecies variability in sensitivity and extrapolation from laboratory to field conditions) to give a value of 0.00056 mg/L. The resulting risk quotient (PEC/PNEC) = 0.071–10.7. Using a similar approach, the aquatic exposure scenario tool predicted a sediment PNEC for HBCD of 6.5 mg/kg sediment dw, based on a chronic toxicity value of 29.25 mg/kg sediment for *Lumbriculus variegates* (dividing this value by an assessment factor of 10 to account for interspecies and intraspecies variability in sensitivity and extrapolation from laboratory to field conditions, and standardizing to 4% organic carbon). The resulting risk quotient (PEC/PNEC) = 0.05–7.11.

For pelagic organisms, risk quotients exceeded 1, indicating a current potential for risk, in surface water scenarios associated with releases from facilities handling raw materials and compounding HBCD. Application of secondary treatment processes greatly reduced the potential for risk; however, predicted exposure values still exceeded minimum effect levels for scenarios associated with large production quantities (e.g., 100 000 kg per year) or use of only primary wastewater treatment, or both. Similar trends were observed in the benthic compartment, in which predicted bulk sediment concentrations of HBCD exceeded minimum effect levels for facilities handling large volumes of raw materials (e.g., 100 000 kg per year) and for smaller volume facilities (e.g., 10 000 kg per year) using only primary wastewater treatment. Predicted bulk sediment concentrations were less than 1 for scenarios associated with compounding facilities, suggesting that current estimated HBCD exposure concentrations derived from compounding activities in Canada are unlikely to exceed minimum effects levels in organisms.

Risk quotients for the soil compartment were determined using exposure values calculated from concentrations measured in sewage sludge. This approach was used because the application of sewage sludge to agricultural soils and pasturelands is considered to represent a direct pathway for HBCD into soil (see Table A-16 for methods). As no Canadian and very limited North American sewage sludge data were available, a mid-Atlantic United States value (La Guardia et al. 2010) was selected to represent possible levels in populated regions of Canada, such as southern Ontario. A geometric mean value was selected, because HBCD sludge data in the study were few and the distribution was skewed by one value. The geometric mean sludge value of 10.04 mg/kg dw was considered to be conservative; it is greater than most concentrations reported for European sewage sludge (Table A-7). The resulting PEC for soil was 0.15–0.30 mg/kg soil dw. A soil organism PNEC was derived from the chronic toxicity value of 235 mg/kg soil dw for *Eisenia fetida* (dividing this value by an assessment factor of 10 to account for interspecies and intraspecies variability in sensitivity and extrapolation

from laboratory to field conditions, and standardizing for organic carbon content) to yield a PNEC of 10.9 mg/kg soil dw. The resulting risk quotient (PEC/PNEC) = 0.014–0.027. The risk quotient results suggest that current estimated exposure concentrations in Canadian soils are unlikely to exceed those leading to adverse effects in organisms. However, a very elevated HBCD concentration was reported for one year (2005) at the mid-Atlantic U.S. wastewater treatment plant, suggesting that if HBCD usage were to increase in Canada, risks to the soil compartment could be more of a concern than at present.

The risk quotient derived for wildlife species highlights the potential for intake arising from the uptake of HBCD in food. As identified in the Health Effects Assessment section, the data set of reproductive studies suggests potential effects at a dose of 101 mg/kg-bw per day and higher (study details provided in Health Effects Assessment section). The lowest-observed-adverse-effect level (LOAEL) of 101 mg/kg-bw per day is selected as an estimate for critical toxicity value in wildlife, based on a two-generation reproductive toxicity study in rats. Allometric scaling was used to extrapolate data obtained from laboratory feeding studies with rats to a surrogate wildlife species, American mink (*Mustela vison*) (Sample and Arenal 1999), in order to account for observed higher sensitivities in larger animals (i.e., mink) when compared with smaller ones (i.e., rats). An assessment factor of 10 was applied to account for extrapolation from laboratory to field conditions. The resulting PNEC was 39.8 mg/kg food ww (see Table A-16 for details). A PEC of 4.51 mg/kg ww was selected based on the Lake Ontario study by Tomy et al. (2004a), which found this maximum concentration in lake trout (*Salvelinus namaycush*), an important prey species in the diet of carnivorous semi-aquatic mammals such as mink and river otter (*Lutra Canadensis*). The resulting risk quotient results (PEC/PNEC) = 0.113 indicated that current HBCD concentrations in Canadian biota are unlikely to exceed minimum effects levels.

The analysis of risk quotients determined that HBCD concentrations in the Canadian environment have the potential to cause adverse effects in populations of pelagic and benthic organisms, but are unlikely to result in direct adverse effects to soil organisms and wildlife. However, it must be considered that the presence of even small amounts of HBCD in the environment warrants concern in light of strong evidence that the substance may be environmentally persistent and bioaccumulative.

Uncertainties in Evaluation of Risk to the Environment

There is some uncertainty regarding physical and chemical properties of the individual HBCD diastereomers and how these relate to persistence, bioavailability, bioaccumulation potential and toxicity of HBCD in the environment.

The assessment finds that HBCD may biodegrade based on laboratory studies. While there may be some lack of understanding respecting diastereomeric transformations in the environment (including biota), when modelled and monitoring data are considered together, the data on HBCD indicate a significant level of persistence in the environment as well as transportability to remote locations. HBCD is highly bioaccumulative in

aquatic biota; however, there is some uncertainty respecting the potential to bioaccumulate in sediment and soil organisms, as well as regarding biomagnification in terrestrial wildlife.

Although HBCD has been detected in air in remote regions, the role of partitioning to atmospheric particulates and the potential for long-range atmospheric transport of particle-bound HBCD warrants further consideration.

There is a general lack of data on HBCD concentrations in the Canadian environment, particularly in sediments, soils, sewage sludge and biota.

Data on toxicity to sediment and soil organisms are also limited. Markedly divergent outcomes were reported in 28-day *Lumbriculus* testing (i.e., NOECs of 5 and ≥ 1000 mg/kg sediment dw), suggesting that effects in soil and sediment tests may be significantly influenced by procedures used to incorporate the test substance, such as the use of a carrier substance. Uncertainties are also associated with toxicity to wildlife, including possible metabolic pathways and products, and effects on pelagic, benthic, soil and wildlife species resulting from prolonged (e.g., lifetime and multigenerational) exposure.

Potential to Cause Harm to Human Health

Exposure Assessment

HBCD is primarily used as an additive flame retardant in expanded and extruded polystyrene foams and in textiles in Canada, which is consistent with the global use pattern. Canadians may be exposed to HBCD in air, water, food and dust arising from releases to the environment and losses during the life cycle of products, as well as from direct use of some consumer products containing HBCD.

HBCD concentrations in air, water, dust, food, biota, human milk, blood and adipose tissue are presented in Tables A-7 through A-13. Although commercial HBCD is typically composed of approximately 80–85% γ-isomer, concentrations measured in environmental media, biota and humans are primarily the α-isomer. According to these data, HBCD levels in human milk, maternal blood/cord blood, and food, as well as dietary intakes of HBCD in Canada and North America, are within the ranges of those found in Europe. This is expected, given the global distribution of HBCD usage in manufacturing consumer and industrial end-use products. Consequently, it is expected that Canadian exposures to HBCD are similar to European exposures. Relevant scenarios conducted by the European Union are presented in Table A-14 (EU RAR 2008).

Environmental Media and Food

Multimedia intake estimates were derived primarily from available North American and European data. Upper-bounding estimates of daily intake of HBCD from environmental

media and food for the general population of Canada are summarized in Appendix E. The total estimates ranged from 5.3 x 10^{-3} µg HBCD/kg-bw per day for formula-fed infants (0–6 months old) to 8.9 x 10^{-2} µg HBCD/kg-bw per day for breastfed infants. Food, followed by dust, was estimated to be the highest contributor to total intake for most age groups.

Levels of HBCD have been measured in various environmental media, including ambient and indoor air, water and food. Levels of HBCD measured in ambient air in North America and Europe ranged from 0.0002–0.61 ng/m^3; elevated levels were measured around manufacturing facilities in Europe (from 280–1070 ng/m^3). Ambient air concentration data are presented in Table A-7. An ambient air concentration of 0.002 ng/m^3 from the Canadian Arctic was selected for use in the intake assessment, as this was the maximum value reported in Canadian ambient air. For intake from indoor air, a median value of 0.18 ng/m^3 from homes in the U.K. was utilized, as no Canadian data were available; refer to Table A-9 for details. No data on HBCD levels in Canadian drinking water were found. Concentrations in water ranged from 6 x 10^{-6} µg/L in Lake Winnipeg to 15.8 µg/L at a European manufacturing plant; refer to Table A-7 for details. A concentration of 2.7 x 10^{-4} µg HBCD/L was used to estimate exposure to the Canadian public from drinking water, based on measured levels in lakes in the U.K. (Harrad et al. 2009b).

Concentrations of HBCD in representative food commodities for North America were obtained from a U.S. food market basket survey (Schecter et al. 2009) and are presented in Table A-10. Total HBCD across 310 composite samples of 31 food types was measured, and varied in and across food groups. HBCD concentrations measured in composite samples were 0.86 µg/kg ww in meat; 0.261 µg/kg ww in dairy; 0.01 µg/kg ww in eggs; 1.46 µg/kg ww in fish and fish products; 0.810 µg/kg ww in fat; 0.180 µg/kg ww in cereal; 0.022 µg/kg ww in apples; and 0.018 µg/kg ww in potatoes. Limit of detection values were conservatively used for instances of non-detects. HBCD intake in this study was estimated at 15.4 ng/day, primarily from meat consumption. HBCD concentrations from the U.S. food market basket survey were considered to be representative of Canadian diets and were used in the estimation of dietary intake for Canadians, with the exception of fish. An HBCD concentration of 4.6 µg/kg ww (~ 35 ng/g lipid) was incorporated into the dietary intake estimates and is considered to be representative of high-end HBCD levels in northern and southern Canadian fish species; refer to Appendix E for further details. Consumption of fish from a contaminated lake in Norway has been found to correlate with HBCD serum levels (Thomsen et al. 2008). High HBCD serum levels in Norwegians also correlated with dietary exposure to HBCD from seafood consumption. Additional data on concentrations in biota, including Arctic species, are presented in Table A-8. Dietary intake estimates for Canadians ranged from 7.9 x 10^{-3} µg/kg/day (60 years of age and over) to 3.3 x 10^{-2} µg/kg/day (6 months to 4 years of age).

Concentrations in human milk are presented in Table A-11. Alpha-HBCD concentrations in Canadian human milk ranged from 0.1–28 µg/kg lipid weight. HBCD values were 20–100 times less than PDE47 (a congener of tetrabromodiphenyl ether used as a marker of

exposure to this class of brominated flame retardants) in the same samples. North American and European HBCD human milk data suggest that exposure is relatively uniform. This was the first report of isomeric content of α-HBCD, and not β- or γ-HBCD in human samples, and also of potential chiral selectivity of HBCD in humans (Ryan et al. 2006). The maximum α-HBCD value of 28 µg/kg lipid in human milk, obtained from Canadian women in the Hamilton area in 2005, along with a lipid content of 3% as measured in the study, was used to derive an intake estimate of 8.4×10^{-2} µg/kg-bw per day for the highest-exposed breastfed infants.

HBCD has been quantified in dust in several indoor locations, including homes, offices, cars and public microenvironments; data are presented in Table A-9. There is high variability in the North American and European HBCD dust concentrations, ranging from < 4.5 to 1.4×10^{5} µg/kg dw. Intake estimates for the general Canadian population from dust were based upon the maximum dust concentration measured in Canadian homes, which is 1300 µg/kg dw (Abdallah et al. 2008b).

Consumer Products

HBCD is an additive-type flame retardant; it is not chemically altered when used as a flame retardant. Thus, there is potential for release from some consumer products over time due to abrasion and usage, as it is not covalently bound within the polymer matrix. As outlined in the uses section, HBCD is applied to the back of textiles, such as upholstery fabric, and is encapsulated in a polymer. Common end-use products include furniture, vehicle upholstery, draperies and wall coverings.

Estimates of potential oral exposure to HBCD from mouthing upholstered furniture were derived for infants 6–24 months of age, when mouthing behaviour is most prevalent, and are presented in Appendix F. There is uncertainty in the estimates of exposure from this scenario, resulting from limited empirical data on the quantity of HBCD available for exposure through mouthing textiles and the variability in mouthing behaviour patterns for infants. One exposure estimate of 1.2×10^{-3} µg/kg-bw per day was generated using water solubility as a surrogate for the surface concentration of HBCD on upholstered furniture, coupled with salivary flow rate, saliva extraction rate, oral absorption, mouthing duration and body weight, similar to an approach presented by Environ International Corporation to the Voluntary Children's Chemical Evaluation Program for polybrominated flame retardants (Environ 2003a, 2003b). This algorithm only accounts for the soluble component of HBCD, and may underestimate exposure from this scenario. Another estimate for this scenario was generated based on a textile surface loading of HBCD of 2 mg/cm^2, the area of fabric mouthed by a child, an empirically derived saliva extraction factor, duration spent mouthing, and body weight based on an algorithm from the U.S. National Research Council (US NRC 2000), similar to the approach used in the HBCD European Union risk assessment (EU RAR 2008). Exposure was estimated at 4.0 µg/kg-bw per day, which is considered to be a significant overestimate when taking into consideration the inherently low water solubility of HBCD (refer to Table A-2).

Exposure through the dermal and inhalation routes from consumer products is considered in the EU RAR (EU RAR 2008) to be negligible when compared with oral exposure. The exposures estimated for Canadians are presented in Appendix F. The *stratum corneum* is an efficient barrier to radio-labelled ^{14}C-HBCD penetration (Roper et al. 2007), and off-gassing from products is not expected due to the low vapour pressure of HBCD. A preliminary health risk assessment for HBCD emitted into indoor air by drawing a curtain was carried out by Miyake et al. (2009) using a Multi-Chamber Concentration and Exposure Model (MCCEM), a U.S. EPA exposure calculation tool. Input parameters included the average measured peak indoor concentration of 8.6 ng/m^3, room size (5.25 m × 3.80 m × 2.70 m), room volume (53.9 m^3) and air exchange rate (0.45 h^{-1}). Lifetime average daily dose was calculated from these data to be 2.67×10^{-4} µg/kg-bw per day, with a margin of exposure of 2.1×10^5; Miyake et al. indicated low concern for this exposure scenario.

Biomonitoring Data

In addition to human milk, HBCD has been measured in human serum and adipose tissue. Levels in human serum ranged from non-detectable to 52 ng/g lipid in the general population, and up to 856 ng/g lipid in individuals occupationally exposed. Limited Canadian blood data exist; however, HBCD was measured in pooled maternal serum and cord blood from individuals in Canadian Arctic communities (Ryan et al. 2005). In maternal blood, a maximum concentration of 0.9 ng/g lipid was measured and levels were not detectable in cord blood of infants; refer to Table A-12 for more details. Levels in adipose tissue (Table A-13) ranged from non-detectable to 12 ng/g lipid. In a study conducted by Roosens et al. (2009), serum concentrations of HBCDs were correlated with dust exposures but not with dietary exposure. Authors reported that the enrichment of the α-HBCD enantiomer in humans appears to be due to *in vivo* enantiomer selective metabolism/excretion rather than dust ingestion or diet (Roosens et al. 2009).

HBCD levels in blood are a measure of steady-state exposure from all sources. As HBCD has a half-life in humans of approximately 64 days, it was considered appropriate to estimate daily intake for adults from blood levels for comparison with exposure estimates from environmental media. Assuming steady-state exposure and first-order kinetics, intake was estimated using the following values: 0.9 ng HBCD/g lipid from Canadian maternal serum (pooled serum from Nunavut and the Northwest Territories), a half-life of 64 days, a lipid concentration of 0.75 kg lipid/kg-bw in adults, and 100% absorption from the oral route. Daily intake, determined with these inputs, was estimated at 7.3×10^{-3} µg/kg-bw per day. This estimate is very similar to the deterministic dietary and total exposure estimates derived for adults, which is 1.2×10^{-2} µg/kg-bw per day, and suggests that exposure estimates are representative of the general population, including vulnerable subpopulations and those living in northern Canada.

Health Effects Assessment

The hazard potential of HBCD has been extensively documented in several reports (EU RAR 2008, ECHA 2008, UNEP-POPs 2010). The current assessment focuses on the key studies identified in the available database.

For evaluating hazards resulting from chronic exposures, a carcinogenicity bioassay was identified in which B6C3F1 mice, 50 per sex per group, were fed diets containing 0, 100, 1000 or 10 000 ppm (equivalent to about 0, 13, 130 or 1300 mg/kg per day) for 18 months. There were no overt signs of toxicity. In the treated mice, male mice had hypertrophic and vacuolized/fatty changes in the liver at the 1000- and 10 000-ppm dosing level, and an increase in altered liver foci was seen in the 1000-ppm group but not at the 10 000-ppm level (highest dose tested). No changes were observed in the female mice. As there was no consistency or dose trend observed between treatment and incidence of neoplastic changes in the liver of male mice, the study authors concluded that there was no evidence of carcinogenicity, because the incidences of total liver tumours were within the normal range for this mouse strain (Kurokawa et al. 1984[3]; EU RAR 2008).

The European Union reported that consistently negative results were observed for HBCD in a range of mutagenicity assays with *Salmonella typhimurium* (Simmon et al. 1976; Baskin and Phillips 1977; GSRI 1979; Zeiger et al. 1987; Ogaswara and Hanafusa 1993; Hossack et al.1978; US EPA 1990a), in an *in vitro* cytogenetic test for chromosomal aberrations with human peripheral blood lymphocytes (Guid and Schadly 1996) and in an *in vivo* assay for clastogenicity in the mouse micronucleus test (Engelhardt and Hoffman 2000). In a non-standard assay with two Chinese hamster cell lines containing duplication mutations in the hypoxanthine-guanine phosphoribosyltransferase (HPRT) gene, a small but significant increase of somatic recombinations was observed (Helleday et al. 1999). The European Union concluded that HBCD lacks significant genotoxic potential both *in vitro* and *in vivo*, and suggested that "there is no reason to explore this endpoint further" (EU RAR 2008). Accordingly, HBCD is not considered to have genotoxic potential.

Zeller and Kirsch (1969) exposed male and female Sprague-Dawley rats for 28 days to dietary concentrations equivalent to 0, 940, 2400 or 4700 mg/kg-bw per day. This study was considered insufficient to assign effect levels, but the data indicate that the liver and thyroid are target organs in HBCD-dosed rats (EU RAR 2008).

Chengelis (1997) dosed male and female Sprague-Dawley rats for 28 days by gavage, at 0, 125, 350 or 1000 mg/kg-bw per day. No significant histopathological lesions were observed. The protocol did not include measurement of thyroid gland weight or serum concentrations of thyroid-stimulating hormone (TSH), T3 or T4. Relative liver weight was significantly increased at the two highest doses in males. The LOAEL was 125 mg/kg-bw per day, based upon significantly increased relative liver weight in all groups of exposed females. The European Union noted a potential issue of contamination of controls in a 90-day study carried out at the same laboratory (Chengelis 2001 as cited in EU RAR 2008). Although the concentrations found in the fat of the "untreated" animals

[3] Kurokawa Y, Inoue T, Uchida Y, Momma J. 1984. Carcinogenesis test of flame retarder hexabromocyclododecane in mice (unpublished, translated into English). M. Hardy, Albermarle Corporation, personal communication. Department of Toxicology. National Public Health Research Institute. Biological Safety Test and Research Centre.

were within historical control ranges, the uncertainties introduced by the potential contamination precluded use of this study in risk characterization.

Van der Ven et al. (2006) dosed five Wistar rats of each sex by gavage for 28 days to 0, 0.3, 1, 3, 10, 30, 100 or 200 mg/kg-bw per day. The protocol focused upon immune and endocrine effects, including the thyroid hormone axis, hematology, bone size and mineralization and retinoid parameters. Such endpoints are not required to be examined in OECD Guideline repeated-dose studies, which would explain why those effects were undetected in OECD Guideline studies. The "most remarkable" findings were dose-related decreased total thyroxin, increased pituitary weight, increased immunostaining of TSH in the pituitary, increased thyroid weight and thyroid follicle cell activation. These effects were restricted to females. In females, liver weight increases were noted at a dose of 29.9 mg/kg-bw per day (BMDL, 22.9 mg/kg-bw per day), while pituitary weight increases were noted at a dose of 50.6 mg/kg-bw per day (BMDL, 29.9 mg/kg-bw per day). The thyroid weight increase occurred at 3.4 mg/kg-bw per day (BMDL, 1.6 mg/kg-bw per day). In a follow-up report, Germer et al. (2006) studied hepatic cytochrome P450 levels and CYP 450 activity. Induction of CYP 3A4 was observed in females while induction of CYP 2B was reported for males, suggesting that sex-specific metabolism could explain the thyroid toxicity noted in females only. These data support the view that HBCD does not exert the toxic action on the thyroid directly, it acts most probably through liver enzyme induction and consequent metabolism of thyroxin.

Chengelis (2001) dosed Sprague-Dawley rats (15/sex/group) by gavage (in corn oil) for 90 days, at dose levels of 0, 100, 300 or 1000 mg/kg-bw per day. Five animals per sex per group were maintained for a 28-day recovery period. Increases in the weights of liver (all dose groups), thyroid (mid- and high-dose groups, females only) and prostate (dose-dependant increase with statistical significance in the high-dose group) were noted. Minimal hepatocellular vacuolization was observed in animals from all dose groups. The LOAEL was 100 mg/kg-bw per day, based upon increased relative liver weight in both sexes. As noted above, the European Union reported that the study control animals may have also been inadvertently dosed (EU RAR 2008).

Zeller and Kirsch (1970) exposed rats via the diet for 90 days, at concentrations that were equivalent to doses of 0, 120, 240, 470 or 950 mg/kg-bw per day. The European Union had noted that the study identified the liver as a target organ, but that effect levels could not be deduced (EU RAR 2008).

Murai et al. (1985) fed pregnant Wistar rats (20 per group) diets that delivered approximate doses of 0, 7.5, 75 or 750 mg/kg-bw per day from days 0–20 of gestation. Six animals per group were allowed to deliver and the pups were maintained until 7 weeks. The absolute and relative maternal liver weights were increased significantly at the highest dose (750 mg/kg-bw per day). There were no significant changes in the number of implants, resorptions, live or dead fetuses, or external, visceral or skeletal anomalies observed in the pups (fetal NOAEL, 750 mg/kg-bw per day).

Stump (1999) dosed 25 Charles River rats by gavage on days 6–19 of gestation, at levels of 0, 500 or 1000 mg/kg-bw per day. There were no indications of maternal or fetal toxicity reported in this study.

Ema et al. (2008) conducted a two-generation reproductive assay with Crl:CD(SD) rats. The F0 animals consisted of 24 rats per sex per group. Dietary administration resulted in dose levels of 0, 10, 101 and 1008 mg/kg-bw per day for males and 0, 14, 141 and 1363 mg/kg-bw per day for females. Diet preparations were formulated by mixing HBCD particles into an appropriate amount of powdered diet for each dose group. Administration was initiated10 weeks prior to mating to capture the full spermatigenic cycle, throughout mating, gestation and lactation. The mid dose was the LOAEL (101 mg/kg-bw per day), based upon a treatment-related decrease in fertility index in the F0 generation, a significant decrease in the number of primordial follicles in the ovary and a significant increased incidence of animals with decreased size of thyroid follicles in the two highest dose groups in both sexes in the F0 generation and the highest dose group of females in the F1 generation. Neurotoxicity parameters were measured. The only significant effect was a lower completion rate of mid-air righting reflex in F2 female pups at the highest dose (1363 mg/kg-bw per day). The NOAEL for this study was 10 mg/kg-bw per day. The European Union had noted that this study was carried out according to OECD guideline 416 and was in accordance with the principles for good laboratory practice (EU RAR 2008).

Subsequent to the European Union's assessment, van der Ven et al. (2009) conducted a one-generation reproduction dietary study with Wistar rats, with targeted exposures of 0, 0 (corn oil solvent control), 0.1, 0.3, 1, 3, 10, 30 or 100 mg/kg-bw per day. Exposure was throughout premating (10 weeks for males, 2 weeks for females), mating, gestation and lactation. Each F0 group consisted of 10 males and 10 females. All F1 litters were maintained. Offspring were further exposed from weaning until 11 weeks of age. The authors considered "the most sensitive effects" to be decreased trabecular bone mineral density and decreased concentration of apolar retinoids in the liver of F1 females and an increased immune response in F1 males. They noted that the immunological effect appeared to be induced during development and was therefore probably persistent; however, there were no actual data to support their contention. It is known that retinoids regulate the transcription of numerous genes and can influence developmental programming, skeletal morphogenesis, embryonic growth, sex differentiation, vascularisation and reproduction. Modulation of the retinoid concentrations was proposed to be related to the effect on the immune response. Retinoid signalling is also implicated in the development of the testis and bone tissue, both of which were affected in F1 animals. The lowest critical effective doses, as determined by the authors, were 0.18 mg/kg-bw per day (BMDL, 0.056 mg/kg-bw per day) for decreased tibia trabecular bone mineral density in F1 females, 1.45 mg/kg-bw per day (BMDL, 0.46 mg/kg-bw per day) for increased immune response (immunoglobulin G, sheep red blood cells) in F1 males and 5.1 mg/kg-bw per day (BMDL, 1.3 mg/kg-bw per day) for decreased sum of apolar retinoids in liver of F1 females. Concurrently, the offspring were assessed for dopamine-dependent behaviour and hearing function, by haloperidol-induced catalepsy and brainstem auditory evoked potentials (BAEPs); this was reported separately in Lilienthal

et al. 2009. Reduced latencies to movement onset were observed mainly in females. The overall pattern of BAEP alterations (increased thresholds and prolonged latencies of early waves) suggested a predominant cochlear effect. Although the authors (Lilienthal et al. 2009) reported that the lower bounds of benchmark doses were between ≤1 and 10 mg/kg-bw per day for catalepsy and BAEP thresholds, no supplementary data were available, as were for the previous endpoints described.

Eriksson et al. (2006) dosed neonatal (day 10) male NMRI mouse pups to HBCD by gavage once, at a dose of 0, 0.9 or 13.5 mg/kg-bw. At the age of three months, the mice were assessed for spontaneous behaviour and learning and memory capability.

Ten male mice per group were tested for spontaneous behaviour by measuring locomotion (horizontal movement, detected by infrared beams), rearing and total activity (all movements, e.g., grooming). The activities were measured for three 20-minute periods. Quantitative data were not presented. For all variables, the control animals became habituated, i.e., activity in response to the novelty of the test chamber diminished over time. The HBCD-dosed animals were hypoactive during the first part of the 60-minute period, while toward the end of the test period they became hyperactive, in contrast to the habituation seen in the controls.

Associative learning and memory were assessed using the Morris swim maze challenge. Groups of 12-17 male mice were tested for the ability to locate a submerged platform in a pool for four consecutive days, and on the fifth day, were tested to problem solve to locate the platform in a changed location in the pool. Five trials were carried out each day. During the acquisition period (days 1–4), both exposed and control mice improved their ability to locate the platform. On the fourth day, the mean latencies of the mice exposed to 13.5 mg/kg-bw were significantly longer than controls ($p < 0.01$) and the group dosed with 0.9 mg HBCD / kg-bw ($p < 0.05$). The mice in the lower-dose group did not differ significantly from controls. On the fifth day, the mice dosed with 13.5 mg HBCD / kg-bw took significantly longer ($p < 0.05$) to find the new position of the platform. The EU RAR (2008) considered the study to have been performed well and concurred with the authors that the LOAEL (based upon significantly altered spontaneous behaviour including hyperactive condition and reduced habituation) was 0.9 mg/kg-bw, the lowest dose tested in the study.

A developmental assay with Sprague-Dawley rats was published subsequent to the European Union assessment. Saegusa et al. (2009) exposed pregnant Sprague-Dawley rats to 0, 100, 1000 or 10 000 ppm HBCD via the diet, from gestational day 10 until day 20 after delivery (the day of weaning). On day 20 post-delivery, dosing was terminated and all dams sacrificed. Histopathological assessment was carried out on 10 male and 10 female offspring from each group. The remaining offspring were maintained on regular diet until 11 weeks of age and then sacrificed for histological assessment. The authors reported that maternal exposure resulted in a weak hypothyroidism effect, with weight and histopathological changes of the thyroid, and decreased serum T3 and increased TSH concentrations in offspring receiving 10 000 ppm until weaning. An increase of thyroid weight and decrease of serum T3 concentration continued until the adult stage in groups

receiving at least 1000 ppm in the diet. With regard to the effect on brain development, HBCD showed evidence of affecting oligodendroglial development at the high dose of 10 000 ppm, which the authors indicated was probably as a result of developmental hypothyroidism. The authors concluded that, based on the developmental brain effect seen, 100 ppm was the NOAEL for HBCD based on changes in thyroid parameters (8.1–21.3 mg/kg-bw per day by maternal exposure level). The LOAEL would therefore be the next-highest dietary level of 1000 ppm (80.7–212.9 mg/kg-bw per day), based upon decreased triiodothyronine and increased relative thyroid weight in male offspring at week 11.

Characterization of Risk to Human Health

In a carcinogenicity bioassay, dietary exposure to mice for 18 months did not result in increased incidence of total liver tumours; the incidence seen was within that of the historical control. The overall negative results of the available genotoxicity studies indicate that HBCD does not have genotoxic potential *in vitro* or *in vivo*.

Short-term repeated-dose toxicity studies have identified effects upon the liver and the thyroid with adverse effect levels ranging from 29.9 to 125 mg/kg-bw per day (Chengelis 1997; van der Ven et al. 2006). The European Union selected an NOAEL of 22.9 mg/kg per day for liver weight increase, thyroid weight increase (and decreased serum T4 levels), and increased pituitary weight as one of two critical effect levels upon which to characterize risk (EU RAR 2008). In addition, the European Union selected an NOAEL of 10 mg/kg-bw per day from the Ema et al. (2008) two-generation reproductive assay with Crl:CD(SD) rats for assessing risk to susceptible populations for long-term exposure (EU RAR 2008). The study LOAEL was 101 mg/kg-bw per day, based upon a treatment and dose-related decrease in fertility index in the F0 generation, a significant decrease in the number of primordial follicles in the ovary and an increased incidence of animals with decreased size of thyroid follicles in the two highest dose groups in both the F0 and F1 generations.

One study identified an endpoint of potential concern for susceptible subpopulations (i.e., infants and children). Eriksson et al. (2006) dosed neonatal (day 10) male NMRI mouse pups to HBCD by gavage once, at either 0, 0.9 or 13.5 mg/kg-bw. At the age of three months, the mice were assessed for spontaneous behaviour, learning and memory capability. The lowest dose was the LOAEL, 0.9 mg/kg-bw, based upon significantly altered spontaneous behaviour (hyperactive condition and reduced habituation). While changes in spontaneous behaviour have not been reported in other animal studies, this endpoint was taken into consideration in risk characterization.

Low adverse effect levels were noted in a recent one-generation study with rats (van der Ven et al. 2009; Lilienthal et al. 2009). The authors stated that “the most sensitive” effects were decreased mineral density in trabecular bone in F1 females, decreased concentration of apolar retinoids in liver in F1 females, and increased immune response in F1 males. The study had several limitations, such as the exclusion of animals from the

dose-response analysis, insufficient information to define the significance of the bone mineral density effects in relation to these three endpoints, and the fact that the dose-response is either not clear (e.g., trabecular bone mineral content, increased immune response) or is evident only at the higher levels of exposure (apolar liver retinoids). These limitations precluded the use of this study in hazard identification and, consequently, risk characterization.

Based on the studies examined in this assessment, the most relevant study for human health risk characterization was determined to be the two-generation reproductive assay with rats (Ema et al. 2008).

For the exposure characterization across age groups, there are Canadian data quantifying HBCD in human blood, cord serum, human milk, biota, dust and ambient air. Concentrations of HBCD in foods were identified in studies carried out in the United States.

To assess the potential risk from exposure of the general population of Canada to HBCD over a lifetime, a conservative NOAEL of 10 mg/kg-bw per day was selected from the two-generation reproductive toxicity study (Ema et al. 2008). Additionally, it was considered appropriate to characterize the magnitude of the margin between potential exposures to infants and children and the behavioural effect LOAEL of 0.9 mg/kg-bw, because of altered behaviour seen in mice 90 days after treatment with a single HBCD dose on postnatal day 10 (Eriksson et al. 2006),

The most highly exposed subpopulation of the general population of Canada was breastfed infants aged 0–6 months, at 8.9×10^{-2} µg/kg-bw per day derived from combined intakes from milk, dust and other environmental media (Health Canada 1998). This finding correlates well with that derived by Eljarrat et al. (2009) for nursing infants in A Coruna, northwestern Spain, of 1.75×10^{-1} µg ∑HBCD/kg-bw per day. Exposure estimates for formula-fed infants and non-formula-fed infants (aged 0–6 months) were 5.3×10^{-3} µg/kg-bw per day and 3.1×10^{-2} µg/kg-bw per day, respectively. The upper-bounding estimates of exposure for the general population of Canada, as reported in Appendix E, incorporate levels of HBCD in household dust and food.

A comparison between the critical effect level NOAEL identified for the general population (10 mg/kg-bw per day) and the upper-bounding estimates of exposure for the general population (0.042 µg HBCD/kg-bw per day) results in a margin of exposure of 2.4×10^{5}. Additionally, the margin between upper-bounding exposures (8.9×10^{-2} µg HBCD/kg-bw per day) for breastfed infants and the LOAEL of 0.9 mg/kg-bw per day results in a margin of exposure of 1.0×10^{4}. These margins between the estimated intakes from human milk for breastfed infants and the critical effects are considered to be protective of this vulnerable subgroup. These margins of exposure are considered adequate to address uncertainties in the exposure and health effects database.

Based on product scenario modelling using two different exposure algorithms, consumer product exposure estimates for infants 6–24 months from mouthing of flame-retarded

textiles or upholstered furniture were 1.2 x 10^{-3} and 4.0 µg/kg-bw per day, respectively. The larger intake estimate of 4.0 µg/kg-bw per day is considered to significantly overestimate intake when taking into consideration the inherently low water solubility of HBCD. The respective margins of exposure between these two intake estimates and the most conservative behavioural LOAEL in the animal database of 0.9 mg/kg-bw per day are 225 and 7.3 x 10^5, respectively. Because the higher exposure estimate is considered to overestimate exposure by several orders of magnitude, when the inherently low water solubility of HBCD is taken into account, these margins of exposure are considered adequate to address uncertainties in the exposure and health effects database. This is consistent with the conclusions in the EU RAR (2008).

Uncertainties in Evaluation of Risk to Human Health

There is moderate confidence in the database of toxicity studies for HBCD. The two-generation reproduction study, identified as the critical study for risk assessment, was reported to be compliant with the relevant OECD guideline and conducted in accordance with good laboratory practices. Furthermore, consistent effects at similar levels of exposure were observed across the studies examined.

Effects on the thyroid, though apparently inconsistent between sexes, have been observed in treated animals in both sexes but have also been limited. The mechanism for the thyroid effects in females only is not clear, but a mode of action has been proposed that HBCD induction of liver enzymes increases thyroid hormone metabolism, resulting in decreased blood thyroid levels. This is supported by the findings of sex-specific cytochrome isoenzyme stimulation by HBCD *in vivo* (Germer et al. 2006). The decrease in thyroid hormone levels subsequently triggers a compensatory increase in TSH levels, accounting for the reported changes in thyroid weight. Further research to fully elucidate the precise mechanism of action would be helpful in understanding whether these properties are shared by both the γ-isomer and the α-isomer. There is emerging information from a Health Canada rodent development and residue depletion study suggesting that the Fisher rat appears to be more sensitive than the Wistar and Sprague-Dawley to HBCD (Curran et al. 2009[4]). Therefore, there is some additional uncertainty as to which of the laboratory animal strains reflect human sensitivity to HBCD.

Canadian environmental media data were available for several media, including levels in human milk, ambient air, fish and dust, and these data were incorporated into the exposure estimates. Due to the variability of HBCD in dust and the small sample size in the Canadian study, uncertainty exists regarding the use of the Canadian maximum level of HBCD in dust, which is 1300 µg/kg. The use of indoor air and water data from the U.K. as a surrogate for Canada introduces the potential for overestimating intakes. HBCD

[4] Curran I, Bondy G, Liston V, Gurofsky S, Rawn T, Pantazopoulos P. 2009. Preliminary Data on the Toxicological Evaluation of Hexabromocyclododecane (HBCD): Rodent Development and Residue Depletion Study. Abstract from the Eleventh International Workshop on Brominated Flame Retardants, Ottawa (ON). May 19–20, 2009.

levels in food commodities (except fish) from a U.S. market basket survey were utilized in the dietary assessment. However, both countries share similar food commodities and thus HBCD levels in U.S. food commodities are likely representative of levels found in Canada. Upper-bounding food exposure estimates were achieved by using the limit of detection values for non-detects in the food database, thus conservatively overestimating dietary exposure.

The intake based on HBCD blood levels in Canadian women from Nunavut and the Northwest Territories is similar to estimated intake from environmental media. The intake as shown is driven predominantly by total food intake. Overall, there is no significant difference in exposure determined by either method, irrespective of any potential uncertainty introduced from the measured environmental media concentrations. For these reasons, there is high confidence in the environmental-media-derived estimates of exposure for the general population of Canada and the resulting derived margins of exposure.

There is uncertainty with the approach used to generate exposure estimates for infants mouthing textiles, due to limited empirical data on the quantity of HBCD available for exposure (through mouthing textiles) and the variability in mouthing behaviour patterns for infants. As such, exposure estimates were generated for this scenario based on two different exposure algorithms. Exposure estimates based on the use of these algorithms differed by several orders of magnitude. There is high confidence that exposure from consumer products will not exceed the upper-bounding exposure estimates, because the higher value is considered to overestimate exposure by several orders of magnitude when the inherently low water solubility of HBCD is taken into account and therefore the derived margins of exposure are protective.

Conclusion

The available information on persistence, bioaccumulation and toxicity, as well as the risk quotient analysis for pelagic and benthic organisms, indicate that HBCD has the potential to cause ecological harm in Canada. The widespread presence of HBCD in the environment warrants concern in light of strong evidence that the substance is environmentally persistent and bioaccumulative. Based on the available information, it is concluded that HBCD is entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity

Based on the adequacies of the margins between upper-bounding estimates of exposure to HBCD and critical effect levels, it is concluded that HBCD is not entering the environment in a quantity or concentration or under conditions that constitute or may constitute a danger in Canada to human life or health.

It is therefore concluded that HBCD meets one or more of the criteria set out in section 64 of CEPA 1999. In addition, HBCD meets the criteria for persistence and bioaccumulation potential as set out in the *Persistence and Bioaccumulation Regulations* (Canada 2000).

Where relevant, research and monitoring will support verification of assumptions used during the screening assessment and, where appropriate, the performance of potential control measures identified during the risk management phase.

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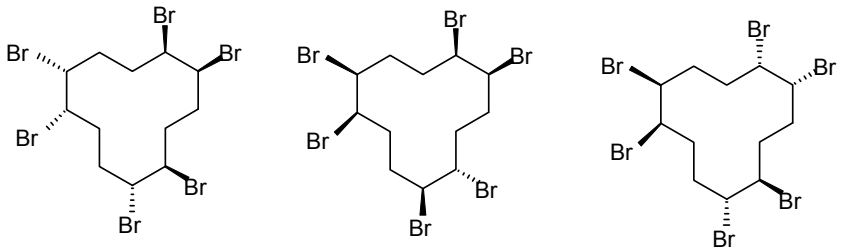
Appendix A. Data Tables for HBCD Assessment.

Table A-1. Substance identity for HBCD

Chemical Abstracts Service Registry Number	**3194-55-6**
DSL name	**Cyclododecane, 1,2,5,6,9,10-hexabromo-**
National Chemical Inventories (NCI) names[1]	*Cyclododecane, 1,2,5,6,9,10-hexabromo- (TSCA, ENCS, AICS, PICCS, ASIA-PAC, NZIoC)* *1,2,5,6,9,10-Hexabromocyclodecane (EINECS)* *1,2,5,6,9,10-Hexabromocyclododecane (ENCS, ECL, PICCS)* *Hexabromocyclododecane (ECL)* *1,2,5,6,9,10- HEXABROMOCYCLODODECANE (PICCS)* *CYCLODODECANE, 12,5,6,9,10-HEXABROMO- (PICCS*
Other names	*Hexabromocyclododecane (HBCD); 1,2,5,6,9,10-Hexabromocyclododecane hbcd* *Bromkal 73-6D* *FR 1206* *FR 1206HT* *Hexabromocyclododecane (HBCD)* *Pyroguard SR 104* *SR 104* *YM 88A*
Chemical group	Brominated flame retardant
Chemical subgroup	Brominated cyclic alkane
Chemical formula	$C_{12}H_{18}Br_6$
Chemical structures	**Dominant Isomer Structures of Hexabromocyclododecane (HBCD)**  alpha-HBCD 10─13% beta-HBCD 1─12% gamma-HBCD 75─89% Ratios of dominant isomers in technical product. Each isomer is a pair of enantiomers or mirror-images.
SMILES[2]	BrC(C(Br)CCC(Br)C(Br)CCC(Br)C(Br)C1)C1
Molecular mass	641.69 g/mol (ACC 2002)
Physical state	White powder at 25°C

[1] National Chemical Inventories (NCI). 2009: AICS (Australian Inventory of Chemical Substances); ASIA-PAC (Asia-Pacific Substances Lists); ECL (Korean Existing Chemicals List); EINECS (European Inventory of Existing Commercial Chemical Substances); ENCS (Japanese Existing and New Chemical Substances); NZIoC (New Zealand Inventory of Chemicals); PICCS (Philippine Inventory of Chemicals and Chemical Substances); and TSCA (Toxic Substances Control Act Chemical Substance Inventory).

[2] Simplified Molecular Input Line Entry System.

Table A-2. Physical and chemical properties of HBCD

Property	Type	Value	Temperature (°C)	Reference
Molecular mass (g/mol)	Experimental	641.7		Sigma Aldrich 2004
Melting point (ºC)	Experimental	180−185		Albemarle Corporation 2000a, 2000b
		175−195		ACCBFRIP 2005
		180−197		Great Lakes Chemical Corporation 2005a, 2005b
		172–184 (crude product) 201–205 (highest melting version)		ECHA 2008
		179–181 α-HBCD 170–172 β-HBCD 207–209 γ-HBCD		ECHA 2008
	Modelled	180 (weighted value)		MPBPWIN 2000
Boiling point (ºC)	Experimental	Decomposition starts at 200		Albemarle Corporation 2000a
		Decomposes at > 445		Great Lakes Chemical Corporation 2005a
		Decomposes at > 190		ECHA 2008
	Modelled	462 (Adapted Stein and Brown method)		MPBPWIN 2000
Density (g/mL)	Experimental	2.36−2.37	Not provided	Albemarle Corporation 2000a, 2000b
		2.1	25	Great Lakes Chemical Corporation 2005a, 2005b
Vapour pressure (Pa)	Experimental	6.27×10^{-5}	21	CMABFRIP 1997b
	Modelled	2.24×10^{-6} (1.68×10^{-8} mm Hg; Modified Grain method)	25	MPBPWIN 2000
Henry's Law constant (Pa m^3/mol)	Modelled	0.174 (1.72×10^{-6} atm·m^3/mole; Bond method) 6.52×10^{-6} (6.43×10^{-11} atm·m^3/mole; Group method) 11.8 (1.167×10^{-4} atm·m^3/mole; VP/Wsol method)[1] 68.8 (6.79×10^{-4} atm·m^3/mole;	25	HENRYWIN 2000

Table A-2. Physical and chemical properties of HBCD

Property	Type	Value	Temperature (°C)	Reference
		VP/Wsol method)[2]		
Water solubility[3](mg/L)	Experimental	3.4×10^{-3} (γ-HBCD) 4.88×10^{-2} (α-HBCD) 1.47×10^{-2} (β-HBCD) 2.08×10^{-3} (γ-HBCD) Total: 6.56×10^{-2}	25 20	CMABFRIP 1997c EBFRIP 2004a
	Modelled	2.09×10^{-5} 3.99×10^{-3} (calculated)	25 25	WSKOWWIN 2000 ECOSAR 2004
	Saltwater (Marine)	3.43×10^{-2} (α-HBCD) 1.02×10^{-2} (β-HBCD) 1.76×10^{-3} (γ-HBCD)		ECHA 2008
Log K_{ow} (Octanol-water partition coefficient; dimensionless)	Experimental	5.81	25	Veith et al. 1979
	Experimental	5.625	25	CMABFRIP 1997a
	Calculated	5.07 ± 0.09 (α-HBCD) 5.12 ± 0.09 (β-HBCD) 5.47 ± 0.10 (γ-HBCD)	25	Hayward et al. 2006
	Modelled	7.74	25	KOWWIN 2000
Log K_{oc} (Organic carbon-water partition coefficient; dimensionless)	Modelled	5.10 (corrected value)	25	PCKOCWIN 2000

[1] Estimate was derived using user-entered values for water solubility of 0.0034 mg/L (for the gamma isomer) and vapour pressure of 6.27×10^{-5} Pa (for the commercial product).
[2] Estimate was derived using model-entered values for water solubility of 2.089×10^{-5} mg/L (WSKOWWIN 2000) and vapour pressure of 2.24×10^{-6} Pa (MPBPWIN 2000).
[3] Water solubility is a function of isomer content.

Table A-3. Results of Level III fugacity modelling for HBCD (EQC 2003)[1]

	Percentage of substance partitioning into each compartment			
Substance released to:	**Air**	**Water**	**Soil**	**Sediment**
Air (100%)	0.002	2.1	87.3	10.6
Water (100%)	0.0	17.0	0.0	83.0
Soil (100%)	0.0	0.0	100.0	0.04

[1] Model inputs are listed in Appendix G.

Table A-4. Modelled data for degradation of HBCD

Fate process	Model and model basis	Model output	Extrapolated half-life (days)
AIR			
Atmospheric oxidation	AOPWIN 2000[1]	$t_{1/2}$ = 2.133 days	> 2
Ozone reaction	AOPWIN 2000[1]	n/a[2]	n/a
WATER			
Hydrolysis	HYDROWIN 2000[1]	$t_{1/2} = 1.9 \times 10^5$ days (pH7) $t_{1/2} = 1.9 \times 10^5$ days (pH8)	n/a
Biodegradation (aerobic)	BIOWIN 2000[1] Sub-model 3: Expert Survey (ultimate biodegradation)	2.0	> 182
Biodegradation (aerobic)	BIOWIN 2000[1] Sub-model 4: Expert Survey (primary biodegradation)	3.1	≤ 182
Biodegradation (aerobic)	BIOWIN 2000[1] Sub-model 5: MITI linear probability	-0.4	> 182
Biodegradation (aerobic)	BIOWIN 2000[1] Sub-model 6: MITI non-linear probability	0.0	> 182
Biodegradation (aerobic)	CPOPs 2008; Mekenyan et al. 2005 % BOD (biological oxygen demand)	0.1	> 182

[1] EPIWIN (2000).
[2] Model does not provide an estimate for this type of structure.

Table A-5. Persistence and bioaccumulation criteria as defined in CEPA 1999 *Persistence and Bioaccumulation Regulations* (Canada 2000)

Persistence[1]		Bioaccumulation[2]
Medium	Half-life	
Air	≥ 2 days or is subject to atmospheric transport from its source to a remote area	BAF≥ 5000; BCF ≥ 5000; log K_{ow}≥ 5
Water	≥ 182 days (≥ 6 months)	
Sediment	≥ 365 days (≥ 12 months)	
Soil	≥ 182 days (≥ 6 months)	

[1] A substance is persistent when at least one criterion is met in any one medium.
[2] When the bioaccumulation factor (BAF) of a substance cannot be determined in accordance with generally recognized methods, then the bioconcentration factor (BCF) of a substance will be considered; however, if neither its BAF nor its BCF can be determined with recognized methods, then the log K_{ow} will be considered.

Table A-6. Modelled bioaccumulation data for HBCD

Test organism	Endpoint	Value ww (L/kg)	Reference
Fish	BAF	$k_M = 5.89 \times 10^{-3}$ d^{-1}: 1 819 701[1]; 158 489[2] $k_M = 0$ d^{-1}: 6 456 542[1]; 275 423[2]	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003)
Fish	BCF	$k_M = 5.89 \times 10^{-3}$ d^{-1}: 4 266[1]; 17 378[2] $k_M = 0$ d^{-1}: 20 417[1]; 23 988[2]	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003)
		6211	BCFWIN 2000

[1] Log K_{ow} 7.74 (KOWWIN 2000) used
[2] Log K_{ow} 5.625 (CMABFRIP 1997a), primarily for γ-HBCD, used

Table A-7. Concentrations measured in the ambient environment and waste treatment products

Medium	Location; year	Concentration	Samples	Reference
Air	Canadian and Russian Arctic; 1994–1995	< 0.0018 ng/m^3	12	Alaee et al. 2003
Air	Alert, Canadian Arctic; 2006– 2007	0.001–0.002 ng/m^3, peak at ~ 0.003 ng/m^3	High volume continuous for 1 year	Xiao et al. 2010[3]
Air	United States; 2002–2003	< 0.0002–0.011 ng/m^3	In 120 of 156	Hoh and Hites 2005
Air	United Kingdom; 2007	0.002–0.04 ng/m^3	5	Abdallah et al. 2008a
Air	The Netherlands; 1999	280 ng/m^3	ns[1]	Waindzioch 2000
Air	Svalbard, Norwegian Arctic; 2006–2007	0.0065 ng/m^3(2006) 0.0071 ng/m^3 (2007)	Mean values	Manø et al. 2008, as cited by de Wit et al. 2010
Air	Sweden; 1990–1991	0.0053–0.0061 ng/m^3	2	Bergander et al. 1995
Air	Sweden; 2000–2001	< 0.001–1070 ng/m^3	11	Remberger et al. 2004
Air	Finland; 2000–2001	0.002, 0.003 ng/m^3	2	Remberger et al. 2004
Air	China; 2006	0.0012–0.0018 ng/m^3	4	Yu et al. 2008a
Air	China; 2006	0.00069–0.00309 ng/m^3	4	Yu et al. 2008b
Air	Sweden urban and rural	0.002–0.61 ng/m^3	14	Covaci et al. 2006
Precipitation	Great Lakes; no year	nd[2]–35 ng/L	ns	Backus et al. 2005
Precipitation	The Netherlands; 2003	1835 ng/L	in 1 of 50	Peters 2003
Precipitation	Sweden; 2000–2001	0.02–366 ng/m^2·d	4	Remberger et al. 2004
Precipitation	Finland; 2000–2001	5.1, 13 ng/m^2·d	2	Remberger et al. 2004
Water	United Kingdom lakes	0.08–0.27 ng/L	27	Harrad et al. 2009b
Water	Lake Winnipeg, Canada; 2004	α-HBCD: 0.006–0.013 ng/L β-HBCD: < 0.003 ng/L γ-HBCD: < 0.003–0.005 ng/L	3	Law et al. 2006a
Water	United Kingdom; no year	< 50–1520 ng/L	6	Deuchar 2002
Water	United Kingdom; 1999	4810–15 800 ng/L	ns	Dames and Moore 2000b
Water	The Netherlands; no year	73.6–472 ng/g dw[6] (solid phase)	ns	Bouma et al. 2000
Water	Japan; 1987	< 200 ng/L	75	Watanabe and Tatsukawa 1990
Water (solid phase)	Detroit River, Canada - United States; 2001	< 0.025–3.65 ng/g dw	63	Marvin et al. 2004, 2006
Sediment	United Kingdom lakes	0.88–4.80 ng/g dw	9	Harrad et al. 2009b
Sediment	Lake Winnipeg, Canada; 2003	α-HBCD: < 0.08 ng/g dw β-HBCD: < 0.04 ng/g dw γ-HBCD: < 0.04–0.10 ng/g dw	4	Law et al. 2006a
Sediment	Norwegian Arctic; 2001	α-HBCD: 0.43 ng/g dw β-HBCD: < 0.06 ng/g dw γ-HBCD: 3.88 ng/g dw	4	Evenset et al. 2007
Sediment	United Kingdom; no year	1131 ng/g dw	1	Deuchar 2002
Sediment	England; 2000–2002	< 2.4–1680 ng/g dw	22	Morris et al. 2004
Sediment	Ireland; 2000–2002	< 1.7–12 ng/g dw	8	Morris et al. 2004
Sediment	Belgium; 2001	< 0.2–950 ng/g dw	20	Morris et al. 2004
Sediment	The Netherlands; no year	25.4–151 ng/g dw	ns	Bouma et al. 2000
Sediment	The Netherlands; 2000	< 0.6–99 ng/g dw	28	Morris et al. 2004
Sediment	The Netherlands; 2001	14–71 ng/g dw	ns	Verslycke et al. 2005
Sediment	Dutch North Sea; 2000	< 0.20–6.9 ng/g dw	in 9 of 10	Klamer et al. 2005
Sediment	Switzerland; no year	< 0.1–0.7 ng/g dw[3]	1	Kohler et al. 2007
Sediment	Switzerland; 2003	0.40–2.5 ng/g dw	1	Kohler et al. 2008
Sediment	Sweden; 1995	nd–1600 ng/g dw	18	Sellström et al. 1998

Table A-7. Concentrations measured in the ambient environment and waste treatment products (continued)

Medium	Location; year	Concentration	Samples	Reference
Sediment	Sweden; 1996–1999	0.2–2.1 ng/g dw	9	Remberger et al. 2004
Sediment	Sweden; 2000	< 0.1–25 ng/g dw	6	Remberger et al. 2004
Sediment	Norway; 2003	α-HBCD: < 0.03–10.15 ng/g dw β-HBCD: < 0.08–7.91 ng/g dw γ-HBCD: < 0.12–3.34 ng/g dw	26	Schlabach et al. 2004a, 2004b
Sediment	Spain; 2002	0.006–513.6 ng/g dw	4	Eljarrat et al. 2004
Sediment	Spain; no year	< 0.0003–2658 ng/g dw	4	Guerra et al. 2008
Sediment	Spain; 2002–2006	nd–2430 ng/g dw	13	Guerra et al. 2009
Sediment	Japan; 1987	nd–90 ng/g dw	in 3 of 69	Watanabe and Tatsukawa 1990
Sediment	Japan; 2002	0.056–2.3 ng/g dw	in 9 of 9	Minh et al. 2007
Soil	United Kingdom; 1999	18 700–89 600 ng/g dw	4	Dames and Moore 2000a
Soil	Sweden; 2000	140–1300 ng/g dw	3	Remberger et al. 2004
Soil	China; 2006	1.7–5.6 ng/g dw	3	Yu et al. 2008a
Landfill leachate	England; 2002	nd	3	Morris et al. 2004
Landfill leachate	Ireland; 2002	nd	3	Morris et al. 2004
Landfill leachate	The Netherlands; 2002	2.5–36 000 ng/g dw (solid phase)	11	Morris et al. 2004
Landfill leachate	Sweden; 2000	3, 9 ng/L	2	Remberger et al. 2004
Landfill leachate	Norway; no year	α-HBCD: nd–0.0091 ng/g ww[7] β-HBCD: nd–0.0038 ng/g ww γ-HBCD: nd–0.079 ng/g ww	ns	Schlabach et al. 2002
STP[4] influent STP effluent Receiving water	United Kingdom; 1999	7.91 x 10^7–8.61 x 10^7 ng/L 8850–8.17 x 10^7 ng/L 528–744 ng/L	3 9 3	Dames and Moore 2000b
STP influent STP effluent STP sludge	United Kingdom; no year	934 ng/L (dissolved phase) 216 000 ng/g dw (solid phase) nd (dissolved phase) 1260 ng/g dw (solid phase) 9547 ng/g dw	ns	Deuchar 2002
STP influent STP effluent STP sludge	England; 2002	nd–24 ng/L (dissolved phase) < 0.4–29.4 ng/g dw (solid phase) < 3.9 ng/L 531–2683 ng/g dw	5 5 5 5	Morris et al. 2004
STP sludge	Ireland; 2002	153–9120 ng/g dw	6	Morris et al. 2004
STP effluent Activated sludge	The Netherlands; 1999–2000	10 800–24 300 ng/L 728 000–942 000 ng/g dw	ns 3	Institut Fresenius 2000a, 2000b
STP influent STP effluent STP sludge	The Netherlands; 2002	< 330–3800 ng/g dw (solid phase) < 1–18 ng/g dw (solid phase) < 0.6–1300 ng/g dw	5 5 8	Morris et al. 2004
STP sludge	Sweden; 1997–1998	11–120 ng/g dw	4	Sellström 1999; Sellström et al. 1999
STP sludge	Sweden; 2000	30, 33 ng/g dw	2	Remberger et al. 2004
STP primary sludge STP digested sludge	Sweden; 2000	6.9 ng/g dw < 1 ng/g dw	1 3	Remberger et al. 2004
STP sludge	Sweden; 2000	3.8–650 ng/g dw	ns	Law et al. 2006c
Plant WWTP[5] influent	United Kingdom; 1999	1.72 x 10^5–1.89 x 10^6 ng/L	3	Dames and Moore 2000a

effluent		3030–46 400 ng/L		
WWTP-(domestic/ industrial waste) secondary sludge	Mid-Atlantic United States; 2002–2008	1160–1 600 000 ng/g TOC (320 –400 000 ng/g dw)	4	La Guardia et al. (2010)
Laundry effluent	Sweden; 2000	31 ng/L	1	Remberger et al. 2004
STP sludge	Switzerland; 2003 and 2005	39–597 ng/g dw	19	Kupper et al. 2008
Compost	Switzerland; no year	19–170 ng/g dw	ns	Zennegg et al. 2005

[1] Not specified
[2] Not detected; detection limit not specified
[3] Values estimated from graphical representation of data
[4] Sewage treatment plant
[5] Wastewater treatment plant
[6] Dw
[7] Ww

Table A-8. Concentrations measured in biota

Location; year	Organism	Concentration (ng/g lipid weight)			Samples	Reference
Canadian Arctic; 1976–2004	Ivory gull (*Pagophila eburnea*) egg	2.1–3.8			24	Braune et al. 2007
Canadian Arctic; 1996–2002		**α-HBCD**	**Dγ-HBCD**			Tomy et al. 2008
	Beluga (*Delphinapterus leucas*)	< 0.63–2.08	< 0.07–0.46		5	
	Walrus (*Odobenus rosmarus*)	nd–0.86	< 0.12–1.86		5	
	Narwhal (*Monodon monoceros*)	2.05–6.10	< 0.11–1.27		5	
	Arctic cod (*Boreogadus saida*)	nd–1.38	nd–0.07		8	
	Redfish (*Sebastes mentella*)	< 0.74–3.37	< 0.28–1.03		5	
	Shrimp (*Pandalus borealis*, *Hymenodora glacialis*)	0.91–2.60	0.23–1.24		5	
	Clam (*Mya truncate*, *Serripes groenlandica*)	nd–1.03	< 0.46–5.66		5	
	Zooplankton	nd–9.16	0.13–2.66		5	
Nunavut; 2007	Ringed seal (*Phoca hispida*)	0.38			10	Morris et al. 2007
Alaska; 1994–2002	Polar bear (*Ursus maritimus*)	< 0.01–35.1			in 2 of 15	Muir et al. 2006
Greenland; 1999–2001	Polar bear (*Ursus maritimus*)	32.4–58.6			11	Muir et al. 2006
Greenland; 1999–2001	Polar bear (*Ursus maritimus*)	41 ng/g ww			20	Gebbink et al. 2008
British Columbia, southern California; 2001–2003	Bald eagle (*Haliaeetus leucocephalus*)	< 0.01 ng/g			29	McKinney et al. 2006
Lake Winnipeg; 2000–2002		**α-HBCD**	**β-HBCD**	**γ-HBCD**		Law et al. 2006a
	Whitefish (*Coregonus commersoni*)	0.56–1.86	0.10–1.25	0.90–1.19	5	
	Walleye (*Stizostedion vitreum*)	2.02–13.07	0.66–2.36	1.65–6.59	5	
	Mussel (*Lampsilis radiate*)	6.15–10.09	< 0.04–2.37	6.69–23.04	5	
	Zooplankton	1.40–17.54	< 0.04–1.80	0.22–1.82	5 Pooled 5	
	Emerald shiner (*Notropis atherinoides*)	4.51–6.53	< 0.04–5.70	3.66–12.09	5	
	Goldeye (*Hiodon alosoides*)	7.39–10.06	< 0.04–2.08	3.23–6.95	5	
	White sucker (*Catostomus commersoni*)	2.30–5.98	0.27–0.90	1.53–10.34	5	
	Burbot (*Lota lota*)	10.6–25.47	2.29–10.29	24.4–47.90		
Great Lakes; 1987–2004	**(ng/g ww)**	**α-HBCD**	**β-HBCD**	**γ-HBCD**	41	Gauthier et al. 2006, 2007
	Herring gull (*Larus argentatus*) egg	nd–20	nd[1]	nd–0.67		
Lake Ontario; no year	Whitefish (*Coregonus commersoni*)	92			ns[2]	Tomy et al. 2004b
	Walleye (*Stizostedion vitreum*)	40				
Lake Ontario; 1979–2004	Lake trout (*Salvelinus namaycush*)	**α-HBCD** 15–27	**β-HBCD** 0.16–0.94		29	Ismail et al. 2009
		γ-HBCD 1.4–6.5	**ΣHBCD** 16–33			
Lake Ontario; 2002	**(ng/g ww)**	**α-HBCD**	**β-HBCD**	**γ-HBCD**		Tomy et al. 2004a
	Lake trout (*Salvelinus namaycush*)	0.37–3.78	< 0.030	0.07–0.73	5	
	Rainbow smelt (*Osmerus mordax*)	0.19–0.26	< 0.030	0.03–0.04	3	
	Slimy sculpin (*Cottus cognatus*)	0.15–0.46	< 0.030	0.02–0.17	3	
	Alewife (*Alosa pseudoharengus*)	0.08–0.15	< 0.030	0.01–0.02	3	
	Mysid (*Mysis relicta*)	0.04, 0.07	< 0.030	0.01, 0.02	2	
	Amphipod (*Diporeia hoyi*)	0.05, 0.06	< 0.030	0.02, 0.03	2	
	Plankton	0.02, 0.04	< 0.030	< 0.030, 0.03	2	

Table A-8. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)	Samples	Reference
Eastern U.S.; 1993–2004	Dolphin (*Lagenorhynchus acutus*)	2.9–380	73	Peck et al. 2008
Eastern U.S.; coast of Maine; 2006	Atlantic herring (*Clupea harengus*) Alewife (*Alosa pseudoharengus*) Atlantic Mackerel (*Scomber scombrus)*	23 7.6 14	6[3] 2[3] 4[3]	Shaw et al. 2009
Chesapeake Bay, USA; 2003	American eel (*Anguilla rostrata*) Bluegill (*Lepomis macrochirus*) Brown bullhead (*Ameiurus nebulosus*) Brown trout (*Salmo trutta*) Channel catfish (*Ictalurus punctatus*) Largemouth bass (*Micropterus salmoides*) Pumpkinseed sunfish (*Lepomis gibbosus*) Redbreast sunfish (*Lepomis auritus*) Rock bass (*Ambloplites rupestris*) Smallmouth bass (*Micropterus dolomieu*) Striped bass (*Morone saxatilis*) White perch (*Morone americana*) White sucker (*Catostomus commersoni*) Yellow bullhead (*Ameiurus natalis*)	2.2, 5.9 4.8 25.4 7.5 2.2–73.9 8.7 5.3 4.5–9.1 1.7–6.0 7.1, 15.9 nd–59.1 1.0–21.0 3.9–19.1 6.9, 18.9	2 1 1 1 9 1 1 4 3 2 9 11 3 2	Larsen et al. 2005
Florida; 1991–2004	Bottlenose dolphin (*Tursiops truncates*)	**α-HBCD** 1.29–7.87; **β-HBCD** 0.337–2.49; **γ-HBCD** 0.582–5.18; **ΣHBCD** 2.21–15.5	15	Johnson-Restrepo et al. 2008
	Bull shark (*Carcharhinus leucas*)	**α-HBCD** 8.01–14.5; **β-HBCD** 4.83–5.57; **γ-HBCD** 52.3–71.3; **ΣHBCD** 71.6–84.9	13	
	Sharpnose shark (*Rhizoprionodon terraenovae*)	**α-HBCD** 11; **β-HBCD** 3.78; **γ-HBCD** 39.7; **ΣHBCD** 54.5	3	
California; 1993–2000	California sea lion (*Zalopus californianus*)	0.71–11.85	26	Stapleton et al. 2006
United Kingdom; no year	Eel (*Anguilla anguilla*) Brown trout (*Salmo trutta*)	39.9–10 275 ng/g ww < 1.2–6758 ng/g ww	ns	Allchin and Morris 2003
United Kingdom; no year	Peregrine falcon (*Falco peregrinus*) Sparrow hawk (*Accipiter nisus*)	nd–1200 nd–19 000	in 12 of 51 in 9 of 65	de Boer et al. 2004
United Kingdom; 1998	Harbour porpoise (*Phocoena phocoena*)	< 5–1019	5 5	Morris et al. 2004
United Kingdom; 1999–2000	Cormorant (*Phalacrocorax carbo*) Sea star (*Asterias rubens*)	138–1320 769	1	
United Kingdom; 2001				
United Kingdom; 1994–2003	**(ng/g ww)** Harbour porpoise (*Phocoena phocoena*)	**α-HBCD** 10–19 200; **β-HBCD** < 3–54; **γ-HBCD** < 4–21	85	Law et al. 2006d
United Kingdom; 2003–2006	Harbour porpoise (*Phocoena phocoena*)	nd–11 500 ng/g ww	in 137 of 138	Law et al. 2008

Table A-8. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)			Samples	Reference
North Sea; no year	Harbour porpoise (*Phocoena phocoena*)	393–2593			24	Zegers et al. 2005
Scotland; no year	Harbour porpoise (*Phocoena phocoena*)	1009–9590			5	
Ireland; no year	Harbour porpoise (*Phocoena phocoena*)	466–8786			11	
France; no year	Dolphin (*Delphinus delphis*)	411–3416			6	
Spain; no year	Dolphin (*Delphinus delphis*)	97–898			31	
	Dolphin (*Delphinus delphis*)	51–454			27	
North Sea; 1999	Whelk (*Buccinium undatum*)	29–47			3	Morris et al. 2004
	Sea star (*Asterias rubens*)	< 30–84			3	
	Hermit crab (*Pagurus bernhardus*)	< 30			9	
	Whiting (*Merlangius merlangus*)	< 73			3	
	Cod (*Gadus morhua*)	< 0.7–50			2	
	Harbour seal (*Phoca vitulina*)	63–2055			2	
	Porpoise (*Phocoena phocoena*)	440–6800			4	
Belgium; 2000	Eel (*Anguilla anguilla*)	< 1–266			19	
Belgium; 1998–2000	Little owl (*Athene noctua*)	20, 40			in 2 of 40	Jaspers et al. 2005
The Netherlands; no year	Mussel (*species not known)*	125–177 ng/g dw			ns	Bouma et al. 2000
	Sprat (*Sprattus sprattus*)	65.5 ng/g dw			1	
	Bass (*species not known)*	124 ng/g dw			1	
	Tern (*Sterna hirundo*) egg	533–844 ng/g dw			ns	
The Netherlands; 2001		**α-HBCD**	**β-HBCD**	**γ-HBCD**		Janák et al. 2005
	Shrimp (*Crangon crangon*)	28, 38	nd	< 2, 18	2	
	Eel (*Anguilla anguilla*)	7, 27	nd, 3.4	2, 7	2	
	Sole (*Solea solea*)	100–1100	nd	< 1–17	4	
	Plaice (*Pleuronectus platessa*)	21–38	nd	< 2–8	3	
	Bib (*Trisopterus luscus*)	53–150	nd–2.2	< 3–43	3	
	Whiting (*Merlangius merlangus*)	16–240	nd	< 3–38	3	
The Netherlands; 1999–2001	Eel (*Anguilla anguilla*)	6–690			11	Morris et al. 2004
	Tern egg (*Sterna hirundo*)	330–7100			10	
The Netherlands; 2001	Mysid (*Neomysis integer*)	562–727			ns	Verslycke et al. 2005
The Netherlands; 2003	**(Median, maximum; ng/g ww)**	**α-HBCD**	**β-HBCD**	**γ-HBCD**		Van Leeuwen et al. 2004
	Eel (*species not known*)	12, 41	0.9, 1.6	3, 8.4	10	
Switzerland; no year	Whitefish (*Coregonus sp.*)	25–210			ns	Gerecke et al. 2003
Baltic Sea; 1969–2001	Guillemot (*Uria algae*) egg	34–300			10	Sellström et al. 2003
Baltic Sea; 1980–2000	Grey seal (*Halicoerus grypus*)	30–90			20	Roos et al. 2001
Sweden; 1995	Pike (*Esox lucius*)	< 50–8000			15	Sellström et al. 1998
Sweden; 1991–1999	Peregrine falcon (*Falco peregrinus*) egg	< 4–2400			21	Lindberg et al. 2004
Sweden; 1987–1999	Peregrine falcon (*Falco peregrinus*) egg	nd–1900			44	Johansson et al. 2009
Sweden; 2000	Pike (*species not known*)				Pooled:	Remberger et al. 2004
	Eel (*species not known*)	120–970			20	
Sweden; 1999–2000	Herring (*species not known*)	65–1800			20	
	Salmon (*Salmo salar*)	21–180			60	
Sweden; 1999		51			5	
Sweden; 2002	Herring (*Clupea harengus*)	1.5–31			ns	Asplund et al. 2004

Table A-8. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)			Samples	Reference
Norwegian Arctic; no year	Northern fulmar (*Fulmarus glacialis*)	3.8–61.6			14	Knudsen et al. 2007
Norwegian Arctic; 2002	Polar bear (*Ursus maritimus*)	18.2–109			15	Muir et al. 2006
Norwegian Arctic; 2002–2003	Amphipod (*Gammarus wilkitzkii*)	nd			5	Sørmo et al. 2006
	Polar cod (*Boreogadus saida*)	1.38–2.87			7	
	Ringed seal (*Phoca hispida*)	14.6–34.5			6	
	Polar bear (*Ursus maritimus*)	5.31–16.51			4	
Norwegian Arctic; 2002	North Atlantic kittiwake (*Rissa tridactyla*) yolk sac	Mean 118			18	Murvoll et al. 2006a, 2006b
Norway; 2002	North Atlantic kittiwake yolk sac	260			19	
	European shag (*Phalacrocorax aristotelis*) yolk sac	417			30	
Norwegian Arctic; 2002	Polar bear (*Ursus maritimus*)	< 0.03–0.85 ng/g ww			15	Verreault et al. 2005
Norwegian Arctic; 2004	Glaucous gull (*Larus hyperboreus*)	0.07–1.24 ng/g ww			27	
Norwegian Arctic; 2002	Glaucous gull (*Larus hyperboreus*)	0.51–292			57	Verreault et al. 2007b
Norwegian Arctic; 2006	Glaucous gull (*Larus hyperboreus*)	< 0.59–63.9			80	Verreault et al. 2007a
Norwegian Arctic; 2003	Polar cod (*Boreogadus saida*)	7.67–23.4			6	Bytingsvik et al. 2004
	Atlantic cod (*Gadus morhua*)	nd–56.9			41	
Norway; 1998–2003						
Norway; no year	**(ng/g ww)**	**α-HBCD**	**β-HBCD**	**γ-HBCD**		Schlabach et al. 2004a, 2004b
	Perch (*Perca fluviatilis*)	3.14–8.12	< 0.04	< 0.07–0.37	7–20 pooled	
	Pike (*Esox lucius*)	1.02–9.25	< 0.02	0.03–0.92		
	Smelt (*Osmerus eperlanus*)	2.1	0.03	0.25		
	Vendace (*Coregonus albula*)	3.15	0.4	0.62		
	Trout (*Salmo trutta*)	2.28–13.3	0.06–1.12	0.24–3.73		
Norway; 2003	Perch (*Perca fluviatilis*)	22.3	< 0.2	< 0.2	5–20 pooled	
	Orfe (*Leuciscus idus*)	14.8	< 0.2	< 0.2		
	Flounder (*Platichthys flesus*)	7.2	< 0.2	< 0.2		
	Cod (*Gadus morhua*)	9.3	< 0.2	< 0.2		
	Trout (*Salmo trutta*)	< 1.9	< 0.2	< 0.2		
	Eel (*Anguilla anguilla*)	4.7	< 0.2	< 0.2		
Northern Norway; no year	Blue mussel (*Mytilus edulis*)	3.6–11			ns	Fjeld et al. 2004
	Atlantic cod (*Gadus morhua*)	6.6, 7.7				
Norway; 2003	Blue mussel (*Mytilus edulis*)	< 0.17–0.87 ng/g ww			33	Bethune et al. 2005
	Herring (*Clupea harengus*)	< 0.63–2.75 ng/g ww			23	
	Mackerel (*Species not known*)	< 0.89–1.19 ng/g ww			24	
Norway; 1986–2004	Tawny owl (*Strix aluco*) egg	0.04–36.5			in 34 of 139	Bustnes et al. 2007
Spain; 2002	Barbell (*Barbus graellsi*)	nd–1172 ng/g ww			23	Eljarrat et al. 2004, 2005
	Bleak (*Alburnus alburnus*)	nd–1643 ng/g ww			22	

Table A-8. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)	Samples	Reference
South Africa; 2004–2005	African darter (*Anhinga rufa*) egg	< 0.2–11	14	Polder et al. 2008
	Reed cormorant (*Phalacrocorax africanus*) egg	< 0.2	3	
	Cattle egret (*Bubulcus ibis*) egg	< 0.2	20	
	Sacred ibis (*Threskiornis aethiopicus*) egg	4.8, 71	2	
	Crowned plover (*Vanellus coronatus*) egg	1.6	1	
	Little grebe (*Tachybaptus ruficollis*) egg	< 0.2	1	
	White-fronted plover (*Charadrius marginatus*) egg	< 0.2	1	
	Kelp gull (*Larus dominicanus*) egg	< 0.2	1	
Asia-Pacific; 1997–2001	Skipjack tuna (*Katsuwonus pelamis*)	**α-HBCD** < 0.1–45 **β-HBCD** < 0.1–0.75 **γ-HBCD** < 0.4–14 **ΣHBCD** nd–45	65	Ueno et al. 2006
South China Sea; 1990–2001	Finless porpoise (*Neophocaena phocaenoides*)	**α-HBCD** 4.4–55 **β-HBCD** < 0.006–4.0 **γ-HBCD** < 0.006–21 **ΣHBCD** 4.7–55	19	Isobe et al. 2008
	Humpback dolphin (*Sousa chinensis*)	**α-HBCD** 31–370 **β-HBCD** < 0.006–0.59 **γ-HBCD** < 0.006–4.6 **ΣHBCD** 31–380		
China; 2006	Silver carp (*Hypophthalmichthys molitrix*)	**α-HBCD** 15–29 **β-HBCD** < 0.005–1.2 **γ-HBCD** 5.5–8.9 **ΣHBCD** 23–38	17	Xian et al. 2008
	Bighead carp (*Aristichthys nobilis*)	**α-HBCD** 11–20 **β-HBCD** < 0.005–0.69 **γ-HBCD** 1.7–2.8 **ΣHBCD** 13–24		
	Grass carp (*Ctenopharyngodon idella*)	**α-HBCD** 7.2–75 **β-HBCD** < 0.005–2.8 **γ-HBCD** 4.3–13 **ΣHBCD** 12–91		
	Common carp (*Cyprinus carpio*)	**α-HBCD** 14–28 **β-HBC** 0.50–0.76 **γ-HBCD** 2.9–5.7 **ΣHBCD** 18–34		
	Crucian carp (*Carassius auratus*)	**α-HBCD** 12–130 **β-HBCD** 0.37–2.2 **γ-HBCD** 2.9–26 **ΣHBCD** 16–160		
	Brass gudgeon (*Coreius heterodon*)	**α-HBCD** 20–57 **β-HBCD** < 0.005–1.7 **γ-HBCD** 5.2–5.6 **ΣHBCD** 25–64		
	White amur bream (*Parabramis pekinensis*)	**α-HBCD** 8.1–74 **β-HBCD** 0.32–6.7 **γ-HBCD** 2.0–51 **ΣHBCD** 14–130		

Table A-8. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)		Samples	Reference
China; 2006	Mandarin fish (*Siniperca chuatsi*)	**α-HBCD** 80, 120	**β-HBCD** 2.8, 3.6		
		γ-HBCD 150, 200	**ΣHBCD** 240, 330		
	Snakehead (*Channa argus*)	**α-HBCD** 37	**β-HBCD** < 0.005		
		γ-HBCD 0.26	**ΣHBCD** 37		
Korea; 2005	Blue mussel (*Mytilus edulis*)	6.0–500		17	Ramu et al. 2007
Japan; 1987	Fish (*species not provided*)	10–23 ng/g ww		in 4 of 66	Watanabe and Tatsukawa 1990
Japan; 1999	Minke whale (*Balaenoptera acutorostrata*)	57		1	Marsh et al. 2004
	Striped dolphin (*Stenella coeruleoalba*)	90		1	
Japan; 2001–2006	Racoon dog (*Nyctereutes procyonoides*)	**α-HBCD** < 0.005–10	**β-HBCD** < 0.005–3.7	39	Kunisue et al. 2008
		γ-HBCD < 0.005–20	**ΣHBCD** < 0.005–29		
Japan; 2005	Oysters (*Crassostrea* sp.) Blue mussels (*Mytilus galloprovincialis*)	**α-HBCD** 7.5–3000	**β-HBCD** 0.77–210	26	Ueno et al. 2010
		γ-HBCD 3.6–2500	**ΣHBCD** 12–5200		

[1] Not detected; detection limit not specified.
[2] Not specified.
[3] 20 fished pooled as six composite samples, 10 fish pooled as two composite samples, 10 fish pooled as four composite samples.

Table A-9. Concentrations of total HBCD in indoor air and dust

Location	Level	n	Reference
Indoor air (pg/m^3)			
United Kingdom	Homes, median = 180	33	Abdallah et al. 2008a
	Offices, median = 170	25	Abdallah et al. 2008a
	Public microenvironments, median = 900	4	Abdallah et al. 2008a
Dust (ng/g dw)			
Canada	Homes, median 640, mean 670 ± 390, range 64–1300	8	Abdallah et al. 2008b
United States	Homes, median 390, mean 810 ± 1100, range 110–4000	13	Abdallah et al. 2008b
	Homes, median 230, geomean 354, range < 4.5–130 200	16	Stapleton et al. 2008
Belgium	Rooms, median 114, mean 160 ± 169, range 33–758	16	Roosens et al. 2009
United Kingdom	Homes, median 1300, mean 8300 ± 26 000, range 140–140 000	45	Abdallah et al. 2008a
	Homes, median 730, mean 6000 ± 20 000, range 140–110 000	31	Abdallah et al. 2008b
	Offices, median 760, mean 1600 ± 1700, range 90–6600	28	Abdallah et al. 2008a
	Offices, median 650, mean 1400 ± 1400, range 90–3600	6	Abdallah et al. 2008b
	Cars, median 13 000, mean 19 000 ± 19 000, range 190–69 000	20	Abdallah et al. 2008a
	Public microenvironments, median 2700, mean 2700 ± 390, range 2300–3200	4	Abdallah et al. 2008a
Scandinavia	Occupational-industrial processing plant (airborne dust), median 2.1 μg/m^3, range 2–150 μg/m^3	30	Thomsen et al. 2007

Table A-10. Food concentrations and dietary intakes for total HBCD

Location	Food concentration and dietary intakes (values > LOD)	Reference
United States	n = 31 food commodities, 310 samples Intake 15.4 ng/day (primarily from meat)	Schecter et al. 2009
	Meat: 23–192 pg/g ww, sum 860 pg/g ww	
	Dairy: n.d. < 4–128 pg/g ww, sum 261 pg/g ww	
	Eggs: n.d. < 11 pg/g ww	
	Fats: n.d. < 35–393 pg/g ww; sum 810 pg/g ww	
	Cereals: n.d. < 180 pg/g ww	
	Apples: n.d. < 22 pg/g ww	
	Potatoes: n.d. < 18 pg/g ww	
	Fish: n.d. < 29–593 pg/g ww, sum 1460 pg/g ww	
Belgium	n = 165 (13) Duplicate Diets: median 0.10, mean 0.13 ± 0.11, range < 0.01–0.35 Intake: median 5.5, mean 7.2 ± 5.2, range 1.2–20 ng/day	Roosens et al. 2009
Sweden	Range < 1–51 ng/g ww (various items)	Remberger et al. 2004
United Kingdom	Range 0.02–0.30 ng/g ww (market basket survey)	Driffield et al. 2008
Norway	Meat: range 0.03–0.15 ng/g ww Eggs: range 0.2–6 ng/g ww Fish: range 0.12–5 ng/g ww Intake: median 16, mean 18, range 4–81 ng/day	Knutsen et al. 2008
Netherlands	Market basket survey: Intake range 174 ng/day	De Winter-Sorkina et al. 2003

Table A-11. Human milk lipid concentrations of HBCD

Location	Human milk (µg/kg lipid weight)	n= (values > LOD)	Reference
Canada, Nunavik 1989–1991	Median α-HBCD 0.2 Range α-HBCD 0.1–0.6	n = 20 (16)	Ryan et al. 2005 (unpublished)
Canada, Nunavik 1996–2000	Median α-HBCD 0.9 Range α-HBCD 0.2–13.3	n = 20 (15)	
Canada, Ontario 2003	Median α-HBCD 0.60 Range α-HBCD 0.2–8.8	n = 27 (13)	Ryan et al. 2006 (unpublished)
Canada, Ontario 2005	Median α-HBCD 0.43 Range α-HBCD 0.2–28	n = 35 (23)	
U.S., Texas 2002	Median α-HBCD 0.40 Range α-HBCD 0.16–0.9	n = 21 (20)	
U.S., Texas 2004	Median α-HBCD 0.40 Range α-HBCD 0.16–1.2	n = 25 (20)	
Sweden 2000–2001	Median α-HBCD 0.30 Range α-HBCD 0.2–2.4	n = 30 (24)	Covaci et al. 2006
Sweden 2002–2003	Median α-HBDD 0.35 Range α-HBCD 0.2–1.5	n = 30 (24)	
Norway 2003–2004	Median α-HBCD 0.60 Range α-HBCD 0.4–20	n = 85 (49)	
Norway 1993–2001	Median 0.6 Range 0.3–20	n = 85 (49)	
Belgium 2006	∑HBCD 1.5	n = 178 pooled Women 18–30 yrs. old	Colles et al. 2008
A Corûna (northwestern Spain)	Median 27 Range 3–188	n = 33 (30) Diastereoisomer levels	Eljarrat et al. 2009

Location	Human milk (µg/kg lipid weight)	n= (values > LOD)	Reference
2006, 2007		were determined and body burden of mothers and infant exposure reported. Nursing infant dietary intake of 0.175 µg/kg-bw per day.	

Table A-12. Human blood and cord serum for HBCD

Location	Human blood serum (ng/g lipid weight)	n = (values > LOD)	Cord serum (ng/g lipid weight)	n = (values > LOD)	Reference
Canada, Arctic Nunavut and NWT regions 1994–1999	Median α-HBCD 0.7 Range α-HBCD 0.5–0.9 Pooled serum	n = 10 pools (3 pools) Total n = 560, 13–61 individuals per pool	Median α-HBCD < LOD (2.4)	n = 13 (0)	Ryan et al. 2005 (unpublished)
Netherlands	Mean 1.1 Median 1.3 Range < 0.16–7.0	n = 78 (77) weeks 20 and 35 of pregnancy	Mean 1.7 Median 0.32 Range < 0.16–4.2	n = 12 (5)	Weiss et al. 2004
Norway	∑HBCDs Median 4.1 Range < 1.0–52 ∑HBCDs Median 2.6 Range < 1.0–18	n = 41 men n = 25 women			Thomsen et al. 2008
Norway	∑HBCDs Median 101 Range 6–856	n =10 workers γ-HBCD 39% nd > 1 in a control group having no work-related exposure			Thomsen et al. 2007
Sweden	∑HBCDs Median 0.46 Range < 0.24–3.4	n = 50			Weiss et al. 2006a
Belgium	∑HBCDs Median of 1.7 Range of < 0.5–11.3	n = 16			Roosens et al. 2009

Note: Intake estimates (mg/kg/day) derived from serum concentrations based on
= [HBCD lipid concentration * bw * lipid concentration in blood * ln 2 / $t_{1/2}$] / bw * oral absorption
= [0.9 ug/kg lipid * 70.9 kg-bw * 0.75 kg lipid/kg-bw * ln 2/64 days] /70.9 kg-bw * 1
= 0.0073 ug/kg bw

Table A-13. Human tissue data for HBCD

Location	Tissue	Result	Reference
France	Adipose tissue	1–12 µg/kg lipid weight (l.d.) in 50% of samples from n = 26 mother-infant pairs	Antignac et al. 2008
Czech Republic	Adipose tissue	n = 98 Mean 1.2 ng/g l.d. Relative standard deviation (RSD)% 150 Median < 0.5 ng/g l.d. 5–95th percentile range 0.5–7.5 ng/g l.d.	Pulkrabova et al. 2009

Table A-14. Exposure estimates of the HBCDEuropean Union Risk Assessment Report[1,2] (EU RAR 2008)

Exposure scenario	EU RAR exposure estimate	Reference
Consumer products		
Oral exposure of children to HBCD from sucking a fabric (50 cm^2), one back-coated with HBCD daily for 2 years at 1 hr/day	Exposure estimate = 26 µg/kg-bw/day	US NRC 2000 as cited in EU RAR 2008
Dermal exposure that assumed exposure from furniture upholstery, back-coated with HBCD	Exposure estimated = 1.3 x 10^{-3} µg/kg-bw/day Exposure level was insignificant and not brought forward in the EU RAR risk characterization	
Inhalation exposure in a room, caused by wear of fabric upholstery and evaporation of HBCD from fabric upholstery treated with HBCD	$C_{indoors}$ of 3.9µg/m^3 Assume 60 kg adult, 24-hour exposure, inhalation rate of 20 m^3/day, 100% absorption Exposure estimate = 1.3 µg/kg-bw/day Exposure level was insignificant and not brought forward in the EU RAR risk characterization	
Textile in furniture and curtains	Concentration of HBCD in debris during wear testing (UV-aging and non-aging) was 0.47% HBCD by debris weight	EU RAR 2008
Sub-scenario: oral exposure to dust	Assume 10 kg child eating all dust generated from 2 sofas, 4 m^2 textile area, pica behaviour, thus 2.5 mg/day	

	Exposure estimate = 1.2 µg/kg-bw/day Exposure level was insignificant and not brought forward in the EU RAR risk characterization	
Sub-scenario: inhalation exposure	$C_{indoors}$= 4.4 µg/m^3 Assume 60 kg adult, 24-hour exposure, inhalation rate of 20 m^3/day, 100% absorption Exposure estimate = 1.5 µg/kg-bw/day Exposure level was insignificant and scenario construction was unrealistic, so it was not brought forward in the EU RAR risk characterization	
Sub-scenario: oral exposure by mouthing of textile	Assume daily mouthing of 50 cm^2 fabric back-coated with HBCD (2mg/cm^2), 0.9% release during 0.5 hours, 100% absorption, one mouthing every three days Exposure estimate = 30 µg/kg-bw/day If the back side is not available, exposure becomes 3 µg/kg-bw/day This sub-scenario estimate was carried forward for risk characterization	
Indoor air exposure from XPS construction boards	Exposure estimate = 0.19 or 0.002 µg/kg-bw/day Exposure level was insignificant and not brought forward in the EU RAR risk characterization	
Mattress ticking – lying down in a bed on a mattress with flame-retarded ticking	Exposure estimate of 0.01 µg/kg-bw/day Exposure level was insignificant and not brought forward in the EU RAR risk characterization	
Indirect exposure – regional intake	EUSES model prediction of ~ 5 µg/kg-bw/day	
Regional exposure of humans via the environment	Exposure estimate = 20 ng/kg-bw/day was derived from food basket studies	

[1] The EU RAR concluded that humans are primarily exposed to HBCD mainly by inhalation or ingestion of airborne dust or from direct contact with treated textiles and materials. Inhalation exposure to HBCD vapour is negligible due to HBCD's low vapour pressure. All these scenarios were found to typically result in insignificant exposures. Indirect exposure via the environment was estimated using EUSES modelling based on measured levels in biota and food. These estimates of exposures were attributed to food basket study data and the ingestion of fish and root crops contaminated with HBCD. Human exposures to HBCD from usage of consumer products or via the environment were concluded to be much lower than occupational exposures. Prenatal and neonatal exposures *in utero* or via breast feeding were also found to occur.

[2] The Scientific Committee on Health and Environmental Risks (SCHER) adopted an opinion on the final Human Health Part of the EU Risk Assessment Report (EU RAR) on HBCD. SCHER members felt that the health part of the EU RAR is of good quality, comprehensive and that the exposure and effects assessment adhere to the EU's Technical Guidance Document.

Table A-15. Summary of key toxicity studies used in the ecological assessment of HBCD

Species, life stage	Test material composition	Study design	Effect level	Reference
Daphnia magna, water flea < 24 hours old at test initiation	93.6% purity	• 21-day flow-through using well water • measured concentrations: 0, 0.87, 1.6, 3.1, 5.6 and 11 µg/L • 40 per treatment • 19.0–20.5°C, pH 8.1–8.4, dissolved oxygen 7.2–8.7 mg/L, hardness 128–132 mg/L as $CaCO_3$, • US EPA 1994; OECD 1984a; ASTM 1991	• 21-day NOEC (survival) ≥ 11 µg/L[1] • 21-day NOEC (reproduction) = 5.6 µg/L • 21-day LOEC (reproduction) = 11 µg/L • 21-day NOEC (growth) = 3.1 µg/L • 21-day LOEC (growth) = 5.6 µg/L	CMABFRIP 1998
Skeletonema costatum and *Thalassiosira pseudonana*, marine algae	composition and purity not provided	• 72-hour static test • concentration series not specified • six different nutrient media • pH 7.6–8.2, 30 ppt. • population density estimated by cell counts using a haemocytometer endpoint: survival (cell density)	• 72-hour EC_{50} = 9.3–12.0 µg/L for *S. costatum* • 72-hour EC_{50} = 50–370 µg/L for *T. pseudonana*	Walsh et al. 1987
Oncorhynchus mykiss, juvenile rainbow trout	composition and purity not provided	• 5- and 28-day flow-through tests using filtered fresh water • intraperitoneal injection using 0, 50 and "< 500"[2] mg/kg-bw doses • 1 replicate of 6–7 fish/treatment • 10°C • endpoints: hepatic detoxification and antioxidant enzymes, liver somatic index (LSI), blood plasma vitellogenin	• catalase activity significantly increased after 5 days at doses of 50 and "< 500" mg/kg-bw • EROD activity significantly inhibited after 28 days at "< 500" mg/kg-bw • LSI significantly increased after 28 days at "< 500" mg/kg-bw • no observed effects on blood plasma vitellogenin levels • no observed effect on formation of DNA adducts	Ronisz et al. 2004

Table A-15. Summary of key toxicity studies used in the ecological assessment of HBCD (continued)

Species, life stage	Test material composition	Study design	Effect level	Reference
Lumbriculus variegates, oligochaete	95% purity	• 28-day static test using dechlorinated tap water • measured concentrations: 0, nd[3], 0.25, 3.25, 29.25 and 311.35 mg/kg sediment dw • 40 per treatment • artificial sediment: 1.8% organic carbon, grain size 100–2000 µm • 20°C, pH 8.7 ± 0.15, dissolved oxygen. 7.5 ± 0.81 mg/L, conductivity 1026 ± 199 µs/cm • modified OECD 2004b	• 28-day NOEC (total number of worms) = 3.25 mg/kg sediment dw • 28-day LOEC (total number of worms) = 29.25 mg/kg sediment dw • 28-day NOEC (large vs. small worms, mean biomass) = 29.25 mg/kg sediment dw • 28-day LOEC (large vs. small worms, mean biomass) = 311.35 mg/kg sediment dw • no deformations observed	Oetken et al. 2001
Hyalella azteca, amphipod *Chironomus riparius*, chironomid *Lumbriculus variegates*, oligochaete	99.99% purity	• non-GLP (good laboratory practice) rangefinder testing with all three species using nominal test concentrations: 0, 50, 100, 500 and 1000 mg/kg sediment dw and 2% or 5% organic carbon (OC) • definitive 28-day flow-through test with *H. azteca* only using nominal concentrations: 0, 31, 63, 125, 250, 500 and 1000mg/kg sediment dw • definitive testing: 80 per treatment • two definitive trials using artificial sediment: (i) 2.3% OC; 22.4–23.5°C; pH 7.8–8.6, dissolved oxygen 5.6–8.6 mg/L (ii) 4.7% OC; 21.0–23.0°C, pH 7.8–8.4, D.O. 4.5–8.5 mg/L; aeration added to all test chambers on Day 22 • US EPA 1996a, 2000; ASTM 1995	• *Lumbriculus* and *Chironomus* rangefinder results not dose-responsive, statistical analyses not conducted on resulting data Results for definitive *Hyalella* test: • 28-day EC_{50} > 1000 mg/kg dw • 28-day NOEC ≥ 1000 mg/kg dw	ACCBFRIP 2003d, 2003e
Eisenia fetida, earthworm adult	99.99% purity	• 28-day survival and 56-day reproduction test using artificial soil with 4.3% OC • measured concentrations at 28 days: 0, 61.2, 145, 244, 578, 1150, 2180 and 4190 mg/kg soil dw • measured concentrations at 56 days: 0, 51.5, 128, 235, 543, 1070, 2020 and 3990 mg/kg soil dw • 80 per control, 40 per treatment • 19.4–22.7°C, pH 5.50–6.67, soil moisture 18.9-42.3%, 573.4–	• 28-day NOEC (survival) ≥ 4190 mg/kg soil dw • 28-day EC_{10}, EC_{50} (survival) > 4190 mg/kg soil dw • 56-day NOEC (reproduction) = 128 mg/kg soil dw • 56-day LOEC (reproduction) = 235 mg/kg soil dw • 56-day EC_{10} (reproduction) = 21.6 mg/kg soil dw[4]	ACCBFRIP 2003a

		595.5 lux • US EPA 1996d; OECD 1984b, 2000	• 56-day EC_{50} (reproduction) = 771 mg/kg soil dw	

Table A-15. Summary of key toxicity studies used in the ecological assessment of HBCD (continued)

Species, life stage	Test material composition	Study design	Effect level	Reference
Zea mays, corn *Cucumis sativa*, cucumber *Allium cepa*, onion *Lolium perenne*, ryegrass *Glycine max*, soybean *Lycopersicon esculentum*, tomato	99.99% purity	• 21-day test using artificial soil with 1.9% organic matter • nominal concentrations: 0, 40, 105, 276, 725, 1904 and 5000 mg/kg dw of soil • 40 seeds per treatment • 18.0–34.7°C, relative humidity 19–82%, 14:10 light:dark • US EPA 1996b, 1996c; OECD 1998a	• no apparent treatment-related effects on emergence, survival or growth • 21-day NOEC ≥ 5000 mg/kg soil dw	ACCBFRIP 2002

[1] Study identified that the highest concentration tested did not result in statistically significant results. Since the NOEC could be higher, the NOEC is described as being greater than or equal to the highest concentration tested.
[2] 500 mg/kg-bw dose could not be dissolved completely in peanut oil carrier, and residue was measured in the stomach cavity of test fish during analysis. Analysis confirmed that the fish had taken up most of the test substance; however, dose was considered to probably be less than 500 mg/kg-bw (i.e., < 500 mg/kg-bw).
[3] Not detected
[4] Value is less than the lowest test concentration used and is therefore considered to be an estimate only.

Table A-16. Summary of data used in the risk quotient analysis of HBCD

	Pelagic organisms	Benthic organisms	Soil organisms	Wildlife consumers
PEC	0.00004–0.006 mg/L[1]	0.33–46.2 mg/kg dw[1]	0.15–0.30 mg/kg soil dw[6]	4.51 mg/kg ww[9]
CTV	0.0056 mg/L[2]	29.25 mg/kg sediment dw[4]	235 mg/kg soil dw[7]	398 mg/kg food ww[10]
Assessment factor (AF)	10[3]	10[3]	10[3]	10[11]
PNEC (CTV/AF)	0.00056 mg/L	6.5 mg/kg sediment dw[5]	10.9 mg/kg soil dw[8]	39.8 mg/kg food ww
Risk quotient (PEC/PNEC)	0.071–10.7	0.05–7.11	0.014–0.027	0.113

[1] Due to the lack of adequate measured data, PECs were estimated using a Fugacity Level III (steady-state) box model described in Appendix C.
[2] CMABFRIP 1998.
[3] An assessment factor of 10 was applied to account for extrapolation from laboratory to field conditions and interspecies and intraspecies variations in sensitivity.
[4] Oetken et al. 2001.
[5] The critical toxicity value (CTV) of 29.25 mg/kg dw was obtained using sediments containing 1.8% organic carbon (OC). To allow comparison between the predicted no effects concentration (PNEC) and predicted environmental concentrations (PECs), the PNEC was standardized to represent sediment with 4% OC.
[6] Due to the lack of measured soil data, PECs were calculated for tilled agricultural soil and pastureland based on Equation 60 of the European Commission Technical Guidance Document (TGD; European Communities 2003),as follows:
$PEC_{soil} = (C_{sludge} \times AR_{sludge}) / (D_{soil} \times BD_{soil})$
where:

PEC_{soil} = PEC for soil (mg/kg)
C_{sludge} = concentration in sludge (mg/kg)
AR_{sludge} = application rate to sludge amended soils ($kg/m^2/yr$); default = 0.5 from Table A-11 of TGD
D_{soil} = depth of soil tillage (m); default = 0.2 m in agricultural soil and 0.1 m in pastureland from Table 11 of TGD
BD_{soil} = bulk density of soil (kg/m^3); default = 1700 kg/m^3 from Section 2.3.4 of TGD

The equation assumes no losses from transformation, degradation, volatilization, erosion or leaching to lower soil layers. Additionally, it is assumed there is no input of HBCD from atmospheric deposition and there are no background HBCD accumulations in the soil. To examine potential impacts from long-term application, an application time period of 10 consecutive years was considered. The geometric mean of sludge concentrations reported by La Guardia et al. (2010), 10.04 mg/kg dw, was used as C_{sludge} in the calculation. Data were converted from ng/g TOC to mg/kg dw using organic carbon content of the sludge specified in the study.
[7] ACCBFRIP 2003a.
[8] The CTV of 235 mg/kg dw was obtained using a soil with 4.3% OC. To allow comparison between the PNEC and PECs, the PNEC was standardized to represent a soil with 2% OC.
[9] Tomy et al. 2004a.
[10] Due to the lack of data for wildlife species, a lowest observed effect level (LOAEL) of 101 mg/kg–bw per day from a two-generation reproductive study in rats (see Health Effects Assessment Section; Ema et al. 2008), was selected as the CTV for the evaluation of potential effects in wildlife. Interspecies scaling was applied to extrapolate the total daily intake (TDI) in rats to a concentration of food in mink, *Mustela vison*, a surrogate wildlife species. The calculation used the typical adult body weight (bw; 0.6 kg) and daily food ingestion rate (DFI; 0.143 kg/d ww) of a female mink to estimate a CTV in mink based on exposure through food (CCME 1998). That is, $CTV_{food} = (CTV_{TDI\ in\ rats} \times bw_{mink}) / DFI_{mink}$ This equation assumes that all of the substance is exposed via food and that the substance is completely bioavailable for uptake by the organism. An allometric scaling factor of 0.94 (Sample and Arenal 1999) was then applied to this CTV value in order to account for observed higher sensitivities in larger animals (i.e., mink) when compared with smaller ones (i.e., rat). The final CTV, incorporating both interspecies and allometric scaling, is therefore 398 mg/kg food ww.
[11] An assessment factor of 10 was applied to account for extrapolation from laboratory to field conditions and from a rodent to a wildlife species.

APPENDIX B. Modelled aquatic toxicity and bioaccumulation data for the HBCD transformation product 1,5,9-Cyclododecatriene

Table B-1. Modelled data for aquatic toxicity for 1,5,9-Cyclododecatriene[1]

Test organism	Type of test	Endpoint	Value (mg/L)	Reference
Fish	Acute (96 hours)	LC_{50}	0.104	ECOSAR 2009
Fish	Chronic (14 day)	LC50	0.111	ECOSAR 2009
Daphnia	Acute (48 hours)	LC_{50}	0.098	ECOSAR 2009
Green algae	Acute (96 hours)	EC_{50}	0.214	ECOSAR 2009

[1] Used measured log Kow of 5.5 (Howard et al. 1996)

Table B-2. Modelled bioaccumulation data for 1,5,9-Cyclododecatriene[1]

Test organism	Endpoint	Value ww (L/kg)	Reference
Fish	BAF	$k_M = 0.01258\ d^{-1}$ [2]: 66 360 $k_M = 0\ d^{-1}$: 177 828	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003)
Fish	BCF	$k_M = 0.01258\ d^{-1}$ [2]: 9813 $k_M = 0\ d^{-1}$: 18 620	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003)

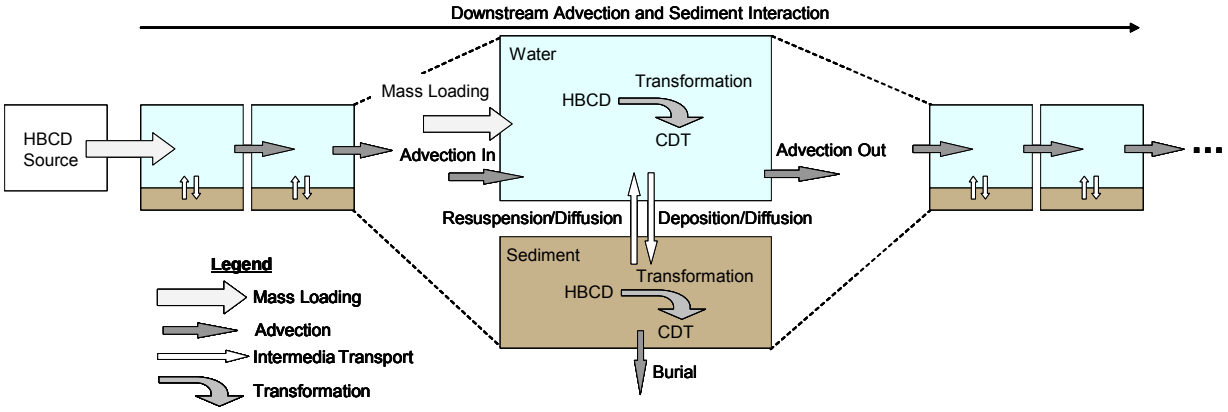
[1] Measured log K_{ow} 5.5 used (Howard et al. 1996)
[2] kM = 0.01258 (Arnot et al. 2008)

APPENDIX C. Derivation of Predicted Exposure Concentrations (PECs) for Pelagic and Benthic Organisms Using a Fugacity Level III Box Model

A Level III fugacity (steady-state) box model based on the Level IV multispecies model described by Cahill et al. (2003) was applied for estimating aquatic exposure to HBCD in the pelagic and benthic compartments. An important feature of the Cahill et al. model is its ability to model the fate of transformation products in addition to that of the parent chemical. For HBCD, degradation to 1,5,9-cyclododecatriene (CDT) is considered an important fate process and this degradation product was included in the model as an additional species. CDT was not included in the risk quotient analysis for HBCD but is considered with respect to the overall persistence of the parent substance.

Figure C-1 provides a conceptual overview of the fugacity model. The model is a mass balance system consisting of 10 downstream boxes each with water and sediment compartments. For modelling purposes, the river is assumed to be a straight channel of uniform and rectangular cross-section with little or no vegetation present in the watercourse or along the banks. Release from the outfall is considered to be continuous from a steady vertical point source.

Figure C-1. Conceptual overview of the fugacity box model used to estimate water and sediment concentrations of HBCD



For each box, the fugacity (f) of both HBCD and the potentially persistent degradation product, CDT, is modeled in each compartment (water, sediment). Fugacity, in units of Pascal (Pa) represents the "partial pressure" of a chemical species in a particular medium and is analogous to concentration, C (mol/m^3), normalized to the relative affinity of the chemical for a particular medium (also known as the "fugacity capacity", Z [mol/m^3.Pa]). Thus, f = C/Z (Mackay 1991).

Aside from mass loading (which is a known discharge rate [mol/h]), the mass transport associated with each process (mol/h) is represented as the product of a fugacity rate coefficient (D, in units of mol/h.Pa) and f (Pa) for other compartments/species (for input processes), or of the modeled compartment/species (for output processes). Transformation of HBCD to CDT is included in the reaction terms. A detailed review of equations for this model is available (Environment Canada 2011).

The main assumptions of the model:

1. chemical release to water only
2. volatilization or air/water intermedia transport is negligible
3. surface water consists of pure water, suspended sediment and biota phases
4. bottom sediment consists of pure water and sediment solids phases
5. first order reaction processes
6. complete instantaneous mixing within boxes
7. equilibrium between phases (pure water, sediment solids and biota) within a particular compartment

Model Parameters

The parameter inputs for the model include chemical properties (e.g., log K_{ow}, K_{oc}, degradation rates), substance release rates, receiving river conditions (e.g., river discharge and flow rates), and generic environmental parameters (e.g., organic carbon content of sediments and sediment deposition rates). Environmental parameters were chosen to represent rivers of southern Ontario based on parameters from ChemCan (Webster et al. 2004), the Cahill et al. (2003) model and plausible physical characteristics for similar river systems (considering values summarized in Chapra 1997 and Gobas et al. 1998). For this assessment, the model extended downstream 5000 m, split into 10 boxes. The length of the first and last boxes was set at 100 m each, and the length of the middle 8 boxes was set at 600 m each.

Loading Estimates and Model Scenarios

Loading estimates for the model were determined using quantities reported in the section 71 notice (Environment Canada 2001), default emission factors recommended by OECD (2004a) and default emission periods recommended in the European Communities Technical Guidance Document (TGD; European Communities 2003). Based on information provided in response to the section 71 notice, annual import volumes for the year 2000 were in the range of 100 000 to 1 000 000 kg. Furthermore, it was estimated that annual HBCD use at an individual facility in Canada would range from 10 000 kg/year to 100000kg/year. Two release scenario groups were developed to represent the types of HBCD-related activities most likely to be taking place in Canada: raw materials handling (Scenario Group 1), and compounding (Scenario Group 2). The OECD (2004a) defines raw materials handling as the handling of raw materials from their arrival on site to their addition to polymers, including manual handling of bags and sacks, conveyer belts and pneumatic or pumped transfer from bulk storage vessels. Compounding is then the process by which additives such as HBCD are incorporated into materials (e.g., plastics) during polymer production and includes processing and final conversion (OECD 2004a). The two activities of raw materials handling and compounding were separated in order to estimate the predicted incremental risk from each activity. HBCD is not produced in Canada and it is likely that any facility involved in compounding would also need to be involved with raw materials handling. For these facilities, the predicted incremental risks from raw materials handling and compounding would be additive.

Scenario Group 1 applied an emission factor of 0.6% based on OECD (2004a) and emission periods of 200 days for usage of 100 000 kg/year and 60 days for usage of 10 000 kg/year. For each usage rate, three possible levels of sewage treatment were applied (none, primary, and secondary) with removal rates estimated using EPIWIN (2000). The combination of two usage rates and three potential levels of sewage treatment yielded six possible emission scenarios for raw materials handling (Scenarios 1a–1f). Scenario Group 2 applied an emission factor of 0.055% based on OECD (2004a) and the same emission periods and levels of sewage treatment as Scenario Group 1, again resulting in six possible emission scenarios for compounding (Scenarios 2a–2f). Note that the OECD and TGD emission parameters were established by means of expert judgement and tend to the worst-case situation.

All release scenarios were assumed to describe industrial activities at a generic facility located in southern Ontario. Generic scenarios were employed to provide estimated release quantities in the absence of site-specific information. The generic facility was situated in southern Ontario as this region is associated with substantial industrial activity and might therefore be expected to have processing and production plants that utilize HBCD. The river dimension characteristics for these scenarios have been chosen to represent an average "medium-sized" river for the industrialized Lake Erie/lowland region of southern Ontario (i.e., the average of the middle 33% of rivers located in this region, based on Environment Canada's Hydat database). The river discharge rate was based on the 25th percentile discharge rate for these rivers.

The release scenarios were entered into the fugacity box model and the results obtained were used to estimate potential water column exposure concentrations for pelagic organisms. For each scenario, the dissolved concentration of HBCD predicted to occur in the first 100 m from the point of discharge, termed C_{max}, was considered to represent a reasonable and conservative exposure concentration in the river and was selected as the predicted environmental concentration (PEC). This concentration is equivalent to that which would result from instantaneous complete mixing of the substance in the first 100 m following discharge to the river.

The major characteristics and model input parameters for each scenario are summarized in Table C-1.

Model Results and Risk Analysis

Prior to calculation of risk quotients for the benthic and pelagic compartments, the scenarios and model-predicted concentrations were evaluated for their degree of "realism" with respect to expected actual HBCD release conditions in Canada. Upon review, it was judged that direct release of HBCD to watercourses without primary or secondary sewage treatment would not occur under normal operations of processing facilities. Based on these considerations, the scenarios with no sewage treatment (i.e., "none") were excluded from the risk characterization (i.e., risk quotients were not calculated).

Pelagic Organisms

Table C-2 summarizes the risk quotient results obtained for pelagic organisms under the retained scenarios. Risk quotients ranged from 0.071 to 3.75 for an annual usage quantity per facility of 10 000 kg/yr and from 0.179 to 10.7 for a use quantity of 100 000 kg/yr. Predicted dissolved water concentrations of HBCD exceeded the predicted no-effect concentration (PNEC) for all raw materials handling scenarios (Scenario Group 1), except for low-volume (10 000 kg/yr) facilities utilizing secondary wastewater treatment. For the compounding scenarios (Scenario Group 2), predicted dissolved water concentrations of HBCD were below the PNEC for all scenarios except for high-volume (100 000 kg/yr) facilities using primary treatment.

Based on the risk quotient results, it is concluded that concentrations of HBCD in surface waters resulting from activities associated with raw materials handling and compounding have the potential to cause adverse effects in populations of pelagic organisms in Canada. Application of secondary treatment processes to wastestreams originating from HBCD processing facilities greatly reduces the potential for risk; however, predicted exposure values still exceed those of minimum effects levels for scenarios associated with large production quantities (e.g., 100 000 kg/yr) and/or use of primary wastewater treatment. It should be noted that although HBCD concentrations are predicted to decrease with distance, the potential distance of impact downstream (i.e., distance with risk quotients greater than 1) is expected to be significant (> 5000 m).

Benthic Organisms

Table C-3 summarizes the risk quotient results obtained for benthic organisms under each retained scenario. Results for benthic organisms generally paralleled those for pelagic organisms. Risk quotients ranged from 0.051 to 2.37 for an annual usage quantity per facility of 10 000 kg/yr and from 0.152 to 7.11 for a use quantity of 100 000 kg/yr. Predicted bulk sediment concentrations of HBCD exceeded the PNEC for scenarios associated with large-volume raw materials handling (Scenarios 1b and 1c) and smaller-volume raw materials handling with only primary wastewater treatment (Scenario 1e). Predicted bulk sediment concentrations of HBCD were less than the PNEC for all compounding scenarios (Scenario Group 2), suggesting that current volume estimates for this activity should not result in bulk sediment concentrations that exceed minimum effects levels in organisms. It should be noted that although HBCD concentrations are predicted to decrease with distance, the potential distance of impact downstream (i.e., distance with risk quotients greater than 1) is expected to be significant (> 5000 m).

Table C-1. HBCD emission rates, river characteristics and release for fugacity modelling release scenarios

Industrial Activity												
Quantity used at facility (kg/yr)												
100 000	100 000	100 000	10 000	10 000	10 000	100 000	100 000	100 000	10 000	10 000	10 000
Raw materials handling scenarios						**Compounding scenarios**					
1a	**1b**	**1c**	**1d**	**1e**	**1f**	**2a**	**2b**	**2c**	**2d**	**2e**	**2f**
Emission factor (%)[2]											
0.6	0.6	0.6	0.6	0.6	0.6	0.055	0.055	0.055	0.055	0.055	0.055
Emission days[3]											
200	200	200	60	60	60	200	200	200	60	60	60
Quantity released from facility (kg/day)											
3	3	3	1	1	1	0.275	0.275	0.275	0.092	0.092	0.092
Wastewater treatment type											
None	1°[4]	2°[5]	None	1°	2°	None	1°	2°	None	1°	2°
Treatment removal rate (%)[6]											
0	57	90	0	57	90	0	57	90	0	57	90
Quantity of HBCD released to river (kg/day)											
3	1.28	0.3	1	0.43	0.1	0.28	0.12	0.028	0.092	0.039	0.0092
River discharge (m^3/s)[7]											
0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85
Mean flow depth (m)[8]											
0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33
River velocity (m/s)[8]											
0.31	0.31	0.31	0.31	0.31	0.31	0.31	0.31	0.31	0.31	0.31	0.31
River width (m)[8]											
8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5

[1] Environment Canada 2001
[2] OECD 2004a
[3] European Communities 2003
[4] Primary wastewater treatment
[5] Secondary wastewater treatment
[6] From STPWIN (EPIWIN 2000)
[7] Discharge estimates were made considering Southern Ontario streamflow data from the HYDAT streamflow database (National Water Data Archive, Environment Canada), and generally represent the 25th percentile of observed discharge rates.
[8] Channel geometry and hydraulic parameters were estimated using equations derived specifically for southern Ontario (Boivin 2005).

Table C-2. Model output and risk quotient analysis for pelagic organisms

Industrial Activity												
Quantity used at facility (kg/yr)												
100 000	100 000	100 000	10 000	10 000	10 000	100 000	100 000	100 000	10 000	10 000	10 000
Raw materials handling scenarios						**Compounding scenarios**					
1a	**1b**	**1c**	**1d**	**1e**	**1f**	**2a**	**2b**	**2c**	**2d**	**2e**	**2f**
Wastewater treatment type											
None	1°[1]	2°[2]	None	1°	2°	None	1°	2°	None	1°	2°
PNEC (mg/L)											
5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}
Maximum concentration (C_{max}, mg/L)[3]											
0.015	0.006	0.001	0.0049	0.0021	0.0005	0.0013	0.0006	0.0001	0.00045	0.00019	0.00004
Concentration 5 km downstream from discharge (C_{5000}, mg/L)[4]											
0.010	0.004	0.001	0.0034	0.0015	0.0003	0.0009	0.0004	0.0001	0.00032	0.00013	0.00003
Maximum risk quotient ($Q_{max} = C_{max}/PNEC$)											
NA[5]	10.7	1.79	NA[5]	3.75	0.893	NA[5]	1.07	0.179	NA[5]	0.339	0.071
Distance (m) with Q > 1											
NA[5]	> 5000	> 5000	NA[5]	> 5000	NA[6]	NA[5]	> 5000	NA[6]	NA[5]	NA[6]	NA[6]

[1] Primary wastewater treatment
[2] Secondary wastewater treatment
[3] C_{max} represents the dissolved HBCD concentration in the first 100 m of river downstream of the emission point.
[4] C_{5000} represents the dissolved HBCD concentration at a distance 4900–5000 m downstream of the emission point.
[5] Risk quotient not calculated because the "no treatment" scenarios were considered unrealistic.
[6] Not applicable as the predicted exposure concentration was less than the estimated no effect level.

Table C-3. Model output and risk quotient analysis for benthic organisms

Industrial Activity												
Quantity used at facility (kg/yr)												
100 000	100 000	100 000	10 000	10 000	10 000	100 000	100 000	100 000	10 000	10 000	10 000
Raw Materials Handling Scenarios						**Compounding Scenarios**					
1a	**1b**	**1c**	**1d**	**1e**	**1f**	**2a**	**2b**	**2c**	**2d**	**2e**	**2f**
Wastewater treatment type											
None	1°[1]	2°[2]	None	1°	2°	None	1°	2°	None	1°	2°
PNEC (mg/kg dw of sediment)											
6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5
Maximum concentration (C_{max}, mg/kg)[3]											
108.2	46.2	10.8	36.1	15.4	3.6	9.92	4.24	0.99	3.31	1.41	0.33
Concentration 5 km downstream from discharge (C_{5000}, mg/kg)[4]											
76.7	32.8	7.7	25.6	10.9	2.6	7.03	3.01	0.70	2.34	1.00	0.23
Maximum risk quotient ($Q_{max} = C_{max}/PNEC$)											
NA[5]	7.11	1.67	NA[5]	2.37	0.553	NA[5]	0.652	0.152	NA[5]	0.217	0.051
Distance (m) with Q > 1											
NA[5]	> 5000	> 5000	NA[5]	> 5000	NA[6]	NA[5]	NA[6]	NA[6]	NA[5]	NA[6]	NA[6]

[1] Primary wastewater treatment
[2] Secondary wastewater treatment
[3] C_{max} represents the sediment HBCD concentration in the first 100 m of river downstream of the emission point.
[4] C_{5000} represents the sediment HBCD concentration at a distance 4900–5000 m downstream of the emission point.
[5] Risk quotient not calculated because the "no treatment" scenarios were considered unrealistic.
[6] Not applicable as the predicted exposure concentration was less than the estimated no effect level.

Appendix D. Robust Study Summary Forms for Key HBCD Studies

ROBUST STUDY SUMMARY - Persistence

Item	Yes	No
***Reference*:** CMABFRIP. 1996. Hexabromocyclododecane (HBCD): Closed bottle test. Wildlife International Ltd. Project No. 439E-102. Easton (MD): Wildlife International Ltd., November 11, 1996.		
Test Substance (CAS RN and name): 3194-55-6, Cyclododecane, 1,2,5,6,9,10-hexabromo- (hexabromocyclododecane)		
Chemical composition of the substance (including purity, by-products)	X	
Method		
References	X	
OECD, EU, national, or other standard method?	X	
Justification of the method/protocol if a non-standard method was used		
***GLP* (good laboratory practice)**	X	
Test design/conditions		
Study type (photodegradation, hydrolysis, biodegradation, other –specify, do **not** assess): Biodegradation		
Test type (aerobic or anaerobic – specify, do **not** assess): Aerobic		
Test medium (air, water, soil, sediment – specify, do **not** assess): activated sludge		
Is information on stability of the substance in the media of concern reported?	X	
Controls (positive or negative): Negative and Positive (Reference)	X	
Number of replicates (including controls)	X	
Temperature	X	
Duration of the experiment	X	
For photodegradation only		
Light source (specify):		
Light spectrum and relative intensity based on sunlight intensity:		
For hydrolysis only		
Measured concentrations reported?		
Basic water properties (pH, hardness, etc.)		
For biodegradation only		
Ready or inherent biodegradation (specify): Ready	X	
Inoculum (concentration and source):	X	
Results		
Endpoints: Average oxygen uptake in controls, reference and treatments used to calculate biochemical oxygen demand (BOD) and percent degradation at each sampling interval. No degradation of the test substance was observed over the 28–day test period.		
Information on breakdown products (do **not** assess this item): No		
Overall score: 11**/11 = 100 %**		
***EC reliability code*:** 1		
Reliability category *(high, satisfactory, low)*: High		
Comments:		

ROBUST STUDY SUMMARY - Persistence

Item	Yes	No
***Reference*:** ACCBFRIP. 2003b. Evaluation of aerobic and anaerobic transformation of hexabromocyclododecane in aquatic sediment systems. Environmental Chemistry Research Laboratory Project Study ID 021081. Midland (MI): The Dow Chemical Company March 5, 2003.		
Test Substance (CAS RN and name): 3194-55-6, Cyclododecane, 1,2,5,6,9,10-hexabromo- (hexabromocyclododecane)		
Chemical composition of the substance (including purity, by-products)	X	
Method		
References	X	
OECD, EU, national, or other standard method?	X	
Justification of the method/protocol if a non-standard method was used		
***GLP* (good laboratory practice)**	X	
Test design/conditions		
Study type (photodegradation, hydrolysis, biodegradation, other –specify, do **not** assess): Biodegradation		
Test type (aerobic or anaerobic – specify, do **not** assess): Aerobic and anaerobic		
Test medium (air, water, soil, sediment – specify, do **not** assess): Sediment		
Is information on stability of the substance in the media of concern reported?	X	
Controls (positive or negative): Negative	X	
Number of replicates (including controls)	X	
Temperature	X	
Duration of the experiment	X	
For photodegradation only		
Light source (specify):		
Light spectrum and relative intensity based on sunlight intensity:		
For hydrolysis only		
Measured concentrations reported?		
Basic water properties (pH, hardness, etc.)		
For biodegradation only		
Ready or inherent biodegradation (specify): Ready	X	
Inoculum (concentration and source):	X	
Results		
Endpoints: Concentration of target substance at selected time intervals throughout exposure period used to calculate biotransformation half-lives. Biotransformation half-lives for HBCD determined as 11 and 32 days in the aerobic system and 1.1 and 1.5 days in the anaerobic system.		
Information on breakdown products (do **not** assess this item): Yes - not detected		
Overall score: 11**/11 = 100 %**		
***EC reliability code*:** 1		
Reliability category *(high, satisfactory, low)*: High		
Comments:		

ROBUST STUDY SUMMARY - Persistence

Item	Yes	No
***Reference*:** ACCBFRIP. 2003c. Evaluation of aerobic and anaerobic transformation of hexabromocyclododecane in soil. Environmental Chemistry Research Laboratory Project Study ID 021082. Midland (MI): The Dow Chemical Company March 5, 2003		
Test Substance (CAS RN and name): 3194-55-6, Cyclododecane, 1,2,5,6,9,10-hexabromo- (hexabromocyclododecane)		
Chemical composition of the substance (including purity, by-products)	X	
Method		
References	X	
OECD, EU, national, or other standard method?	X	
Justification of the method/protocol if a non-standard method was used		
***GLP* (good laboratory practice)**	X	
Test design/conditions		
Study type (photodegradation, hydrolysis, biodegradation, other –specify, do **not** assess): Biodegradation		
Test type (aerobic or anaerobic – specify, do **not** assess): Aerobic and anaerobic		
Test medium (air, water, soil, sediment – specify, do **not** assess): Soil		
Is information on stability of the substance in the media of concern reported?	X	
Controls (positive or negative): Negative	X	
Number of replicates (including controls)	X	
Temperature	X	
Duration of the experiment	X	
For photodegradation only		
Light source (specify):		
Light spectrum and relative intensity based on sunlight intensity:		
For hydrolysis only		
Measured concentrations reported?		
Basic water properties (pH, hardness, etc.)		
For biodegradation only		
Ready or inherent biodegradation (specify): Ready	X	
Inoculum (concentration and source):	X	
Results		
Endpoints: Concentration of target substance at selected time intervals throughout exposure period used to calculate biotransformation half-lives. Biotransformation half-lives for HBCD determined to be 63 and 6.9 days in the aerobic and anaerobic soils, respectively.		
Information on breakdown products (do **not** assess this item): Yes - not detected		
Overall score: 11**/11 = 100 %**		
***EC reliability code*:** 1		
Reliability category *(high, satisfactory, low)*: High		
Comments:		

ROBUST STUDY SUMMARY - Persistence

Item	Yes	No
***Reference*: EBFRIP. 2004. Investigation of the biodegradation of [14C]hexabromocyclododecane in sludge, sediment, and soil. Toxicology and Environmental Research and Consulting Laboratory Project Study ID 031178. Midland (MI): The Dow Chemical Company November 30, 2004.**		
Test Substance (CAS RN and name): 3194-55-6, Cyclododecane, 1,2,5,6,9,10-hexabromo- (hexabromocyclododecane)		
Chemical composition of the substance (including purity, by-products)	X	
Method		
References	X	
OECD, EU, national, or other standard method?	X	
Justification of the method/protocol if not a standard method was used		
***GLP* (good laboratory practice)**	X	
Test design / conditions		
Study type (photodegradation, hydrolysis, biodegradation, other –specify, do **not** assess): Biodegradation		
Test type (aerobic or anaerobic – specify, do **not** assess): Aerobic and anaerobic		
Test medium (air, water, soil, sediment – specify, do **not** assess): Soil, sediment and sludge		
Is information on stability of the substance in the media of concern reported?	X	
Controls (positive or negative): Negative	X	
Number of replicates (including controls)	X	
Temperature	X	
Duration of the experiment	X	
For photodegradation only		
Light source (specify):		
Light spectrum and relative intensity based on sunlight intensity:		
For hydrolysis only		
Measured concentrations reported?		
Basic water properties (pH, hardness, etc.)		
For biodegradation only		
Ready or inherent biodegradation (specify): Ready	X	
Inoculum (concentration and source):	X	
Results		
Endpoints: Numerical endpoints not determined as objective of study was to investigate pathways and major products formed during degradation.		
Information on breakdown products (do **not** assess this item): Yes		
Overall score: 11**/11 = 100 %**		
***EC reliability code*: 1**		
Reliability category *(high, satisfactory, low)*: High		
Comments:		

ROBUST STUDY SUMMARY - Persistence

Item	Yes	No
***Reference*:** Gerecke AC et al. 2006. Anaerobic degradation of brominated flame retardants in sewage sludge. Chemosphere 64:311–317.		
Test Substance (CAS RN and name): 3194-55-6, Cyclododecane, 1,2,5,6,9,10-hexabromo- (hexabromocyclododecane)		
Chemical composition of the substance (including purity, by-products): purity, not composition	X	
Method		
References		X
OECD, EU, national, or other standard method?		X
Justification of the method/protocol if a non-standard method was used	X	
***GLP* (good laboratory practice)**	not known	
Test design/conditions		
Study type (photodegradation, hydrolysis, biodegradation, other –specify, do **not** assess): Biodegradation		
Test type (aerobic or anaerobic – specify, do **not** assess): Anaerobic		
Test medium (air, water, soil, sediment – specify, do **not** assess): Sewage sludge		
Is information on stability of the substance in the media of concern reported?		X
Controls (positive or negative): Negative	X	
Number of replicates (including controls): Not specifically but range (see Comments)	X	
Temperature	X	
Duration of the experiment: Not specifically but upper limit (see Comments)	X	
For photodegradation only		
Light source (specify):		
Light spectrum and relative intensity based on sunlight intensity:		
For hydrolysis only		
Measured concentrations reported?		
Basic water properties (pH, hardness, etc.)		
For biodegradation only		
Ready or inherent biodegradation (specify): Ready	X	
Inoculum (concentration and source):	X	
Results		
Endpoints: Degradation rate constants and half-lives for technical mixture and individual isomers. Only values for technical mixture reported. Rate constant for technical HBCD was 1.1 ± 0.3 d^{-1}, corresponding to a half-life of 0.66 day.		
Information on breakdown products (do **not** assess this item): No		
Overall score: 8/**11 = 73 %**		
***EC reliability code*:** 2		
Reliability category *(high, satisfactory, low)*: Satisfactory		
Comments: Study is reported in a journal article and therefore not all details are included. Several brominated flame retardants were tested at the same time and the article reports overall methodology and results. While the method used is not standard, it appears to be scientifically sound and the study well conducted. Some important information (such as the number of replicates and exposure duration for the HBCD testing) is not provided.		

ROBUST STUDY SUMMARY - Bioaccumulation

Item	Yes	No
Reference: Veith et al. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. J Fish Res Board Can 36:1040–1048.		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**		**X**
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method?**	**X**	
Justification of the method/protocol if a non-standard method was used?	n/a	
***GLP (good laboratory practice)**	n/a	
Test organisms (specify common and Latin names): fathead minnow (*Pimephales promelas*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism	X	
Sex	n/a	
Length and weight of test organisms		X
Number of test organisms per replicate	X	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type (field, laboratory): laboratory	X	
Number of replicates (including controls) and concentrations	X	
***Measured concentrations reported?** Mean measured exposure concentration reported; description of test methodology specifies that concentration was measured each weekday	**X**	
***Was the chemical concentration in the water below the chemical's water solubility?** Mean measured concentration 6 µg/L; water solubility 3.4-8.6 µg/L	**X**	
***Experiment duration equal to or longer than the time required for the chemical concentration in the organism and water to reach steady state?** Exposure time 32 days; steady state BCF calculated from 32-day exposure.	**X**	
Exposure media conditions (temperature, pH, TOC, DOC, DO, other) reported? Temp., DO (saturation), hardness, alkalinity, pH of test water reported	X	
Photoperiod and light intensity: specifies that USEPA Methods (1975) used	X	
Stock and test solution preparation		X
Information on emulsifiers used for poorly soluble / unstable substances	X	
Statistical methods used	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Results		
Endpoints and values (BAF, BCF, or log $K_{ow;}$ do **not** assess this item): BCF = 18 100		
BAF or BCF either as: 1) the ratio of chemical concentration in the organism and in water, or 2) the ratio of the chemical uptake and elimination rate constants (**1** or **2** – specify; do **not** assess this item): 1		
Whether BAF/BCF was derived from a tissue sample or whole organism (do **not** assess this item)?	X	
Indication of whether average BAF/BCF was used (specify; do **not** assess this item)	X	
Indication of whether max BAF/BCF was used (specify; do **not** assess this item)		X
***BAF/BCF reported on a lipid-normalized basis, or was the lipid % reported?**	**X**	
Score: major items - **5/6**; overall score: **17/20 = 85%**		
***Reliability** (Klimisch) **code**:* 1		
***Reliability category (high, satisfactory, low):** High*		
Comments:		

ROBUST STUDY SUMMARY - Bioaccumulation

Item	Yes	No
Reference: CMABFRIP. 2000. Hexabromocyclododecane (HBCD): A flow-through bioconcentration test with the rainbow trout (*Oncorhychus mykiss*). Easton (MD): Wildlife International Ltd. Project No. 439A-11.		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**	**X**	
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method?**	**X**	
Justification of the method/protocol if a non-standard method was used?	n/a	
***GLP (good laboratory practice)**	**X**	
Test organisms (specify common and Latin names): rainbow trout (*Oncorhynchus mykiss*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism: same source and year class	X	
Sex	n/a	
Length and weight of test organisms	X	
Number of test organisms per replicate	X	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type (field, laboratory): laboratory	X	
Number of replicates (including controls) and concentrations	X	
***Measured concentrations reported?**	**X**	
***Was the chemical concentration in the water below the chemical's water solubility?**	**X**	
***Experiment duration equal to or longer than the time required for the chemical concentration in the organism and water to reach steady state?** Steady state achieved at highest test concentration, but not at lowest		**X**
Exposure media conditions (temperature, pH, TOC, DOC, DO, other) reported? Temp., DO, pH, hardness, alkalinity, conductivity, TOC reported	X	
Photoperiod and light intensity:	X	
Stock and test solution preparation	X	
Information on emulsifiers used for poorly soluble / unstable substances	X	
Statistical methods used	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Results		
Endpoints and values (BAF, BCF, or log $K_{ow;}$ do **not** assess this item): Day 35 BCF for 0.34 µg/L test concentration = 6531 (edible), 20 726 (nonedible), 13 085 (whole fish) NB. Steady-state not achieved at this concentration. Steady-state day 35 BCF at 3.4 µg/L test concentration = 4650 (edible), 12,866 (nonedible), 8974 (whole fish).		
BAF or BCF either as: 1) the ratio of chemical concentration in the organism and in water, or 2) the ratio of the chemical uptake and elimination rate constants (**1** or **2** – specify; do **not** assess this item): 1		
Whether BAF/BCF was derived from a tissue sample or whole organism (do **not** assess this item)?	X	
Indication of whether average BAF/BCF was used (specify; do **not** assess this item)	X	
Indication of whether max BAF/BCF was used (specify; do **not** assess this item)	X	
***BAF/BCF reported on a lipid-normalized basis, or was the lipid % reported?**	**X**	
Score: major items - **6/7**; overall score: **20/21 = 95%**		
***Reliability** (Klimisch) **code**:* 1		
***Reliability category (high, satisfactory, low):** High*		
Comments:		

ROBUST STUDY SUMMARY – Inherent toxicity

Item	Yes	No
***Reference*: CMABFRIP. 1988. Hexabromocyclododecane (HBCD): A flow-through life-cycle toxicity test with the cladoceran (*Daphnia magna*). Easton (MD): Wildlife International Ltd. Project No.439A-108.**		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**	**X**	
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method**?	**X**	
Justification of the method/protocol if a non-standard method was used		
***GLP (good laboratory practice)**	**X**	
Test organisms (specify common and Latin names): Water flea (*Daphnia magna*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism	X	
Sex	n/a	
Length and weight of test organisms	X	
Number of test organisms per replicate	X	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type – acute or chronic (specify; do **not** assess this item): chronic		
Experiment type (laboratory or field) specified?	X	
System type (static, semi-static, flow through)?	X	
Negative or positive controls (specify)? Negative and solvent controls	X	
Number of replicates (including controls) and concentrations	X	
Exposure pathways (food, water, both)	X	
Exposure duration	X	
***Measured concentrations reported?**	**X**	
Exposure media conditions (temperature, pH, electrical conductivity, hardness, TOC, DOC, DO, major cations and anions; other)	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Photoperiod and light intensity	X	
Stock and test solution preparation	X	
Use of emulgators/solubilizers (especially for poorly soluble / unstable substances)	X	
Analytical monitoring intervals	X	
Statistical methods used	X	
Results		
Toxicity values (LC_{50}, EC_{50}, or IC_{50} – specify; do **not** assess this item): 21-day LOEC (survival) > 11 µg/L, 21-day LOEC (reproduction) = 11 µg/L, 21-day LOEC (growth) = 5.6 µg/L, 21-day NOEC (overall study) = 3.1 µg/L		
Other endpoints reported – e.g., BCF/BAF (specify; do **not** assess this item): 21-day MATC = 4.2 µg/L		
***Was toxicity value below the chemical's water solubility**?	**X**	
Other adverse effects (carcinogenicity, mutagenicity, etc. (do **not** assess this item)		X
Score: major items – 5/**5**; overall score – 24/**25 (96%)**		
***EC Reliability code*: 1**		
Reliability category (high, satisfactory, low): high		
Comments: All major items reported "yes"; overall score 96%. Lowest toxicity value (5.6 µg/L) was slightly above the water solubility value of 3.4 µg/L (25°C) used by the study authors. However, a measured water solubility been reported by EBFRIP (2004a) in the range of 2.08 to 48.8 µg/L (20°C) for the individual diastereomers. Temperature 19.0–20.5°C. DO 7.2–8.8 mg/L. pH 8.1–8.4. Hardness 128–132 mg/L as $CaCO_3$. Alkalinity 176–178 mg/L as $CaCO_3$. Conductivity 310–320 µmhos/cm. Dimethylformamide solvent used. Good control performance, test concentrations well maintained throughout exposure period.		

ROBUST STUDY SUMMARY – Inherent toxicity

Item	Yes	No
***Reference*: EBFRIP. 2004b. Hexabromocyclododecane (HBCD): A 72-hour toxicity test with the marine diatom (*Skeletonema costatum*). Easton (MD): Wildlife International Ltd. Project No. 439A-125.**		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**	**X**	
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method**?	**X**	
Justification of the method/protocol if a non-standard method was used		
***GLP (good laboratory practice)**	**X**	
Test organisms (specify common and Latin names): marine alga (*Skeletonema costatum*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism	n/a	
Sex	n/a	
Length and weight of test organisms	n/a	
Number of test organisms per replicate	n/a	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type – acute or chronic (specify; do **not** assess this item): acute		
Experiment type (laboratory or field) specified?	X	
System type (static, semi-static, flow through)?	X	
Negative or positive controls (specify)? Negative and media controls	X	
Number of replicates (including controls) and concentrations	X	
Exposure pathways (food, water, both)	X	
Exposure duration	X	
***Measured concentrations reported?**	**X**	
Exposure media conditions (temperature, pH, electrical conductivity, hardness, TOC, DOC, DO, major cations and anions; other)	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Photoperiod and light intensity	X	
Stock and test solution preparation	X	
Use of emulgators/solubilizers (especially for poorly soluble / unstable substances)	X	
Analytical monitoring intervals	X	
Statistical methods used	X	
Results		
Toxicity values (LC_{50}, EC_{50}, or IC_{50} – specify; do **not** assess this item): 72-hour EC_{50} (cell density, area under growth curve, growth rate) > 41.0 µg/L		
Other endpoints reported - BCF/BAF, LOEC/NOEC (specify; do **not** assess this item): 72-hour NOEC (cell density, area under growth curve, growth rate) < 41.0 µg/L		
***Was toxicity value below the chemical's water solubility**?	**X**	
Other adverse effects (carcinogenicity, mutagenicity, etc. (do **not** assess this item)		X
Score: major items – 5/**5**; overall score – 22/**22 (100%)**		
***EC reliability code*: 1**		
Reliability category (high, satisfactory, low): high		
Comments: All major items reported "yes"; overall score 100%. Selected test concentration (41.0 µg/L) is well above reported water solubility of 3.4 µg/L (25°C) for HBCD; however, a recent study by EBFRIP (2004a) measured solubility values of 2.08 to 48.8 µg/L at 20°C for the individual diastereomers. Therefore, although a toxic endpoint was not determined in the present study, consider the reported results to be meaningful within the context of a rangefinder test. Temperature 18.0–22.0°C. pH 7.9–8.4. Light intensity 4130–4660 lux. Control growth over the 3-day test period was 10–11x, and less than the OECD recommended 16x for test validity. However, consider that the response between controls and test solution was sufficiently delineated to indicate that inhibition was occurring in the test substance flasks.		

ROBUST STUDY SUMMARY – Inherent toxicity

Item	Yes	No
***Reference*:** Oetken et al. 2001. Validation of the preliminary EU-concept of assessing the impact of chemicals to organisms in sediment by using selected substances. UBA-FB 299 67 411. Dresden (DE): Institute of Hydrobiology, Dresden University of Technology		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**		**X**
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method**?	**X**	
Justification of the method/protocol if a non -standard method was used		
***GLP (good laboratory practice)**	**Not reported**	
Test organisms (specify common and Latin names): Oligochaete (*Lumbriculus variegatus*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism	X	
Sex	n/a	
Length and weight of test organisms	X	
Number of test organisms per replicate	X	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type – acute or chronic (specify, but do **not** assess this item): chronic		
Experiment type (laboratory or field) specified?	X	
System type (static, semi-static, flow through)?	X	
Negative or positive controls (specify)? Negative and solvent controls	X	
Number of replicates (including controls) and concentrations	X	
Exposure pathways (food, water, both)	X	
Exposure duration	X	
***Measured concentrations reported?**	**X**	
Exposure media conditions (temperature, pH, electrical conductivity, hardness, TOC, DOC, DO, major cations and anions; other)	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Photoperiod and light intensity	n/a	
Stock and test solution preparation	X	
Use of emulgators/solubilizers (especially for poorly soluble / unstable substances)	X	
Analytical monitoring intervals	X	
Statistical methods used	X	
Results		
Toxicity values (LC_{50}, EC_{50}, or IC_{50} - specify, do **not** assess this item): 28-day NOEC (no. of worms) = 3.25 mg/kg sediment dw; 28-day LOEC (no. of worms) = 29.25 mg/kg sediment dw; 28-day NOEC (large vs. small worms, mean biomass) = 29.25 mg/kg sediment dw; 28-day LOEC (large vs. small worms, mean biomass) = 311.35 mg/kg sediment dw.		
Other endpoints reported - BCF/BAF (specify, do **not** assess this item):		
***Was toxicity value below the chemical's water solubility**?	**n/a**	
Other adverse effects (carcinogenicity, mutagenicity, etc. Do **not** assess this item) deformation (none)	X	
Score: major items – 2/**4**; overall score – 20/**22 (91%)**		
***EC Reliability code*:** 2		
Reliability category (high, satisfactory, low): satisfactory		
Comments: An OECD guideline (218) was used with modifications and while GLP has not been specified in the report, the description of the methodology is consistent with GLP. Consider that the study has met basic scientific principles, and that all necessary data and documentation have been presented. Temperature 20°C. DO 7.52 ± 0.81 mg/L. pH 8.7 ± 0.15. Conductivity 1026 ± 199 µs/cm.		

ROBUST STUDY SUMMARY – Inherent toxicity

Item	Yes	No
***Reference*: ACCBFRIP. 2003a. Effect of hexabromocyclododecane on the survival and reproduction of the earthworm, *Eisenia fetida*. Columbia (MI): *A*BC Laboratories Inc. Study No. 47222.		
Test Substance (CAS RN and name): 3194-55-6 (Hexabromocyclododecane)		
***Chemical composition of the substance (including purity, by-products)**	**X**	
Persistence/stability of test substance in test system	X	
Method		
References	X	
***OECD, EU, national, or other standard method**?	**X**	
Justification of the method/protocol if a non-standard method was used		
***GLP (good laboratory practice)**	**X**	
Test organisms (specify common and Latin names): Earthworm (*Eisenia fetida*)		
Latin or both Latin and common names reported?	X	
Life cycle age / stage of test organism	X	
Sex	n/a	
Length and weight of test organisms	X	
Number of test organisms per replicate	X	
Food type / feeding periods (acclimation/during test)	X	
Test design/conditions		
Test type – acute or chronic (specify, but do **not** assess this item): chronic		
Experiment type (laboratory or field) specified?	X	
System type (static, semi-static, flow through)?	X	
Negative or positive controls (specify)? Negative control	X	
Number of replicates (including controls) and concentrations	X	
Exposure pathways (food, water, both)	X	
Exposure duration	X	
***Measured concentrations reported?**	**X**	
Exposure media conditions (temperature, pH, electrical conductivity, hardness, TOC, DOC, DO, major cations and anions; other)	X	
Was pH within 6–9 range? (do **not** assess this item)	X	
Was temperature within 5–28°C range? (do **not** assess this item)	X	
Photoperiod and light intensity	X	
Stock and test solution preparation	X	
Use of emulgators/solubilizers (especially for poorly soluble / unstable substances)	n/a	
Analytical monitoring intervals	X	
Statistical methods used	X	
Results		
Toxicity values (LC_{50}, EC_{50}, or IC_{50} - specify, do **not** assess this item): 28-day EC_{10} and EC_{50} (survival) > 4190 mg/kg soil dw; 56-day EC_{10} (reproduction) = 21.6 mg/kg with 95% confidence limits of 0.000468 to 110 mg/kg; 56-day EC_{50} (reproduction) = 771 mg/kg with 95% confidence limits of 225 to 4900 mg/kg		
Other endpoints reported - BCF/BAF, LOEC/NOEC (specify, do **not** assess this item): 28-day NOEC (survival) ≥ 4190 mg/kg soil dw; 56-day NOEC (reproduction) = 128 mg/kg soil dw; 56-day LOEC (reproduction) = 235 mg/kg soil dw; BAFs ranging from 0.026 to 0.069.		
***Was toxicity value below the chemical's water solubility**?	**n/a**	
Other adverse effects (carcinogenicity, mutagenicity, etc. Do **not** assess this item)		X
Score: major items – 4/**4**; overall score – 22/**22 (100%)**		
***EC reliability code*:** 1		
Reliability category (high, satisfactory, low): high		
Comments: Good control performance. Temperature 19.4–22.7°C. pH 5.50–6.67. Soil moisture 18.9–42.3%. Light intensity 573.4–595.5 lux. Should note, however, that preparation of test soils differed from that suggested by ASTM and bioaccumulation factors were reported based on concentration in tissue (ppm) relative to average 28-day concentration in soil.		

Appendix E. Upper-bounding estimates of daily intake of HBCD by Canadians

Route of Exposure	Estimated intake (µg/kg-bw per day) of HBCD by various age groups							
	0–6 months[1, 2, 3]			0.5–4 years[4]	5–11 years[5]	12–19 years[6]	20–59 years[7]	60+ years[8]
	Breast fed	Formula fed	Not formula fed					
Ambient air[9]	7.0×10^{-8}	7.0×10^{-8}	7.0×10^{-8}	1.5×10^{-7}	1.2×10^{-7}	6.6×10^{-8}	5.7×10^{-8}	5.0×10^{-8}
Indoor air[10]	4.9×10^{-5}	4.9×10^{-5}	4.9×10^{-5}	1.1×10^{-4}	8.2×10^{-5}	4.7×10^{-5}	4.0×10^{-5}	3.5×10^{-5}
Drinking water[11]	nil	2.9×10^{-5}	1.1×10^{-5}	1.2×10^{-5}	9.6×10^{-6}	5.5×10^{-6}	5.7×10^{-6}	6.0×10^{-6}
Food [12]	8.4×10^{-2}	nil	2.6×10^{-2}	3.3×10^{-2}	2.4×10^{-2}	1.4×10^{-2}	1.2×10^{-2}	7.9×10^{-3}
Soil/ Dust[13]	5.2×10^{-3}	5.2×10^{-3}	5.2×10^{-3}	8.4×10^{-3}	2.7×10^{-3}	6.6×10^{-4}	5.5×10^{-4}	5.4×10^{-4}
Total intake	8.9×10^{-2}	5.3×10^{-3}	3.1×10^{-2}	4.2×10^{-2}	2.7×10^{-2}	1.5×10^{-2}	1.3×10^{-2}	8.5×10^{-3}

[1] Human milk: Based on 28 µg HBCD/kg lipid * 3% lipid human milk fat content as measured in the study, 750 g milk consumed per day and a body weight of 7.5 kg.

[2] Assumed to weigh 7.5 kg, to breathe 2.1 m^3 of air per day, to drink 0.8 L of water per day (formula fed) or 0.3 L/day (not formula fed) and to ingest 30 mg of soil per day (Health Canada 1998).

[3] For exclusively formula-fed infants, intake from water is synonymous with intake from food. The concentration of HBCD in water of 270 pg/L used to reconstitute formula was based on unpublished data. No data were identified on levels of HBCD in formula in Canada or elsewhere. Approximately 50% of not formula-fed infants are introduced to solid foods by 4 months of age and 90% by 6 months of age (NHW 1990 in Health Canada 1998.

[4] Assumed to weigh 15.5 kg, to breathe 9.3 m^3 of air per day, to drink 0.7 L of water per day and to ingest 100 mg of soil per day (Health Canada 1998).

[5] Assumed to weigh 31.0 kg, to breathe 14.5 m^3 of air per day, to drink 1.1 L of water per day and to ingest 65 mg of soil per day (Health Canada 1998).

[6] Assumed to weigh 59.4 kg, to breathe 15.8 m^3 of air per day, to drink 1.2 L of water per day and to ingest 30 mg of soil per day (Health Canada 1998).

[7] Assumed to weigh 70.9 kg, to breathe 16.2 m^3 of air per day, to drink 1.5 L of water per day and to ingest 30 mg of soil per day (Health Canada 1998).

[8] Assumed to weigh 72.0 kg, to breathe 14.3 m^3 of air per day, to drink 1.6 L of water per day and to ingest 30 mg of soil per day (Health Canada 1998).

[9] 2 pg or 2×10^{-6} µg/m^3 from the Canadian Arctic was selected (Xiao et al. 2010). All identified data for concentrations in ambient air presented in Table A-7 were considered, and this one was the maximum Canadian value. Canadians are assumed to spend three hours outdoors each day (Health Canada 1998).

[10] The median indoor air concentration of 180 pg/m^3 or 0.00018 µg /m^3 from the United Kingdom was used as surrogate indoor air data for Canadians, n = 33 (Abdallah et al. 2008a). No levels of HBCD in Canadian indoor air were identified. Canadians are assumed to spend 21 hours indoors each day (Health Canada 1998).

[11] No levels of HBCD in Canadian drinking water were identified. For this reason, unpublished data on HBCD in lakes of the United Kingdom have been used as a surrogate 270 pg/L or 2.7×10^{-4} µg/L. All identified data for concentrations in water were considered.

[12] Estimates of intake from food are based upon concentrations in foods indentified in a market basket survey of U.S. food commodities. Concentrations of HBCD in food commodities, those representative of North America were obtained from a U.S. food market basket survey (Schecter et al. 2009). In part I of this larger market basket study, total HBCD in composite samples of n=31 food

types and n=310 samples were measured. Limits of detection values were used for non-detects. Inputs were as follows: 0.86 µg/kg ww in meat; 0.261 µg/kg ww in dairy; 0.01 µg/kg ww in eggs; 0.810 µg/kg ww in fat; 0.180 µg/kg ww in cereal; 0.022 µg/kg ww in fruit; and 0.018 µg/kg ww in vegetables. For fish, a value of 4.6 µg/kg (∑HBCD; α-HBCD = 3.8 µg/kg, γ-HBCD = 0.8 µg/kg, β-HBCD = 0.03 µg/kg; approx. 35 ng/g lipid) from Lake Ontario lake trout was used as an estimate of HBCD in Canadian fish species (Tomy et al. 2004a). This is considered to be a reasonable high-end estimate of HBCD levels in northern and southern Canadian fish species.

[13] Highest Canadian dust level in Canadian homes reported by Abdallah et al. 2008b (1300 µg/kg dw) was selected. In North America and Europe, there is a large variation in HBCD levels in dust.

Appendix F. Estimates of oral exposure to HBCD for infants 6–24 months from mouthing flame-retarded cushion or upholstered furniture

Consumer product scenario	Algorithm and Assumptions	Estimated exposure
Oral mouthing of HBCD flame-retarded cushion or upholstered furniture Based on alorgithm from Environ International Corporation, 2003	Dose rate = [WS x V_s x FR x AF_0 x EF_{mouth} x 1]/bw Where: WS = Water solubility of α-HBCD is 48.8 µg/L Vs = Salivary flow rate in a child's mouth is 0.00022 L/min, from Watanabe et al. (1990) as cited in Environ (2003a, 2003b) FR = Fractional rate of extraction by saliva is 0.05, default value AF_0 = Absorption factor by the oral route is 1, default value EF_{mouth} = Exposure frequency mouthing, 23 min/d based on: Juberg et al. (2001) 22 min/day for children 0–18 months who mouthed objects (n = 46); Smith and Norris (2003) 24 min/day for children 6–9 months (n = 15); and 23 min/day for children 15–18 months (n = 14) who mouth other objects, as reported in the EPA Child-Specific Exposure Factors Handbook (US EPA 2008) BW = Body weight, assumed to be 10 kg for an infant 6–24 months when mouthing behaviour is most prevalent	1.2 x 10^{-3} µg/kg-bw per day
Oral exposure of children to HBCD from sucking a fabric Based on algorithm from U.S. National Research Council, 2000	$D = S_a \times A_f \times \mu_a \times f_{cc} / W_c$ Where: D = The dose rate of chemical (mass per unit body weight per unit time) S_a = Mass per unit surface area, application rate to the fabric or back-coating. 2 mg/cm^2 for HBCD as utilized in the EU HBCD risk assessment. A_f = The area of fabric sucked on each occasion, 50 cm^2. Default selected by U.S. NRC subcommittee for U.S. NRC HBCD assessment (US NRC 2000). μ_a = The fractional rate (per unit time) of FR extraction by saliva under the given conditions. Chemical-specific, 0.025/d, used by U.S. NRC in HBCD assessment. Based on extraction data for HBCD in polyester fiber in McIntyre et al. (1995) as cited in US NRC 2000. f_{cc} = The fraction (dimensionless) of the time a child sucks FR-treated fabric, 23 min/d based on: Juberg et al. (2001) 22 min/day for children 0–18 months who mouthed objects (n = 46); Smith and Norris (2003) 24 min/day for children 6–9 months (n = 15); and 23 min/day for children 15–18 months (n = 14) who mouth other objects as reported in the EPA Child-Specific Exposure Factors Handbook (2008) W_c = Body weight, assumed to be 10 kg for an infant 6–24 months when mouthing behaviour is most prevalent	4.0 µg/kg-bw/day

Note: The EU mouthing textile exposure scenario assumed daily mouthing of 50 cm^2 fabric back-coated with HBCD (2 mg/cm^2), 0.9% saliva extraction rate during 30 minutes, 100% absorption, one mouthing every three days; 10 kg 1-year-old infant. The resulting exposure estimate was 30 µg/kg-bw per day when both sides of textile were available for mouthing. If the back side is not available, exposure was 3 µg/kg-bw per day. Calculated margins of safety for these exposure estimates ranged from 330–7600.

Appendix G – PBT Model Inputs Summary Table

	Phys-Chem/Fate	Fate	Fate	Fate	Fate	Fate	Fate	PBT Profiling	Ecotoxicity
Model Input Parameters	EPISuite (all models, including: AOPWIN, KOCWIN, BCFBAF, BIOWIN and ECOSAR)	STP (1) ASTreat (2) SimpleTreat (3) (required inputs are different depending on model)	EQC (required inputs are different if Type I vs. Type II chemical)	TaPL3 (required inputs are different if Type I vs. Type II chemical)	OECD POPs Tool	Arnot-Gobas BCFBAF Model	Gobas Wolf BMF Model	Canadian-POPs (including: Catabol, BCF Mitigating Factors Model, OASIS Toxicity Model)	Artificial Intelligence Expert System (AIES)/ TOPKAT/ ASTER)
SMILES Code	BrC(C(Br)CCC(Br)C(Br)CCC(Br)C(Br)C1)C1							BrC(C(Br)CCC(Br)C(Br)CCC(Br)C(Br)C1)C1	x
Molecular weight (g/mol)		x (1, 2, 3)	641.7 (I,II)	x (I,II)	x				
Melting point (ºC)			x (I)	x (I)					
Boiling point (ºC)									
Data temperature (ºC)			25 (I,II)	x (I,II)					
Density (kg/m^3)		x (2)							
Vapour pressure (Pa)	6.27 × 10-5 Pa[5]	x (1, 3)	x (I)	x (I)					
Henry's Law constant (Pa·m^3/mol)		x (3)							
Log K_{aw} (air-water partition coefficient; dimensionless)		x (2)	2.63E-09 (II)	x (II)	x				
Log K_{ow} (octanol-water partition coefficient; dimensionless)		x (1)	x (I)	x (I)	x	7.74, 5.625 (γ-HBCD)	x		
K_{ow} (octanol-water partition coefficient; dimensionless)		x (2, 3)							
Log K_{oc} (organic carbon-water partition coefficient – L/kg)									
Water solubility (mg/L)	0.00345 mg/L[5] (γ-HBCD)	x (1, 3)	x (I)	x					

	Phys-Chem/Fate	Fate	Fate	Fate	Fate	Fate	Fate	PBT Profiling	Ecotoxicity
Model Input Parameters	EPISuite (all models, including: AOPWIN, KOCWIN, BCFBAF, BIOWIN and ECOSAR)	STP (1) ASTreat (2) SimpleTreat (3) (required inputs are different depending on model)	EQC (required inputs are different if Type I vs. Type II chemical)	TaPL3 (required inputs are different if Type I vs. Type II chemical)	OECD POPs Tool	Arnot-Gobas BCFBAF Model	Gobas Wolf BMF Model	Canadian-POPs (including: Catabol, BCF Mitigating Factors Model, OASIS Toxicity Model)	Artificial Intelligence Expert System (AIES)/ TOPKAT/ ASTER)
Log K_{oa} (octanol-air partition coefficient; dimensionless)							x		
Soil-water partition coefficient (L/kg)[1]			2502 (II)	x (II)					
Sediment-water partition coefficient (L/kg)[1]			5004 (II)	x (II)					
Suspended particles-water partition coefficient (L/kg)[1]		x (2)	25 020 (II)	x (II)					
Fish-water partition coefficient (L/kg)[2]			8974 (II)	x (II)					
Aerosol-water partition coefficient; dimensionless[3]			100 (II)	x (II)					
Vegetation-water partition coefficient; dimensionless[1]				x (II)					
Enthalpy (K_{ow})				-20 (3)					
Enthalpy (K_{aw})				55 (3)					
Half-life in air (days)			2.13 (I,II)	x (I,II)	x				
Half-life in water (days)			60 (I,II)	x (I,II)	x				
Half-life in sediment (days)			240 (I,II)	x (I,II)					
Half-life in soil (days)			60 (I,II)	x (I,II)	x				
Half-life in vegetation (days)[4]				x (I,II)					
Metabolic rate constant (1/days)						*	*		
Biodegradation rate constant (1/days) or (1/hr)–specify		x (3, 1/hr) (2, 1/days)							

	Phys-Chem/Fate	Fate	Fate	Fate	Fate	Fate	Fate	PBT Profiling	Ecotoxicity
Model Input Parameters	EPISuite (all models, including: AOPWIN, KOCWIN, BCFBAF, BIOWIN and ECOSAR)	STP (1) ASTreat (2) SimpleTreat (3) (required inputs are different depending on model)	EQC (required inputs are different if Type I vs. Type II chemical)	TaPL3 (required inputs are different if Type I vs. Type II chemical)	OECD POPs Tool	Arnot-Gobas BCFBAF Model	Gobas Wolf BMF Model	Canadian-POPs (including: Catabol, BCF Mitigating Factors Model, OASIS Toxicity Model)	Artificial Intelligence Expert System (AIES)/ TOPKAT/ ASTER)
Biodegradation half-life in primary clarifier ($t_{1/2\text{-}p}$) (hr)		x (1)							
Biodegradation half-life in aeration vessel ($t_{1/2\text{-}s}$) (hr)		x (1)							
Biodegradation half-life in settling tank ($t_{1/2\text{-}s}$) (hr)		x (1)							

[1] derived from $\log K_{oc}$
[2] derived from BCF data
[3] default value
[4] derived from half-life in water
[5] user-defined value used for determining Henry's Law constant only