Draft Screening Assessment

Cyclododecane, 1,2,5,6,9,10-hexabromo-

Chemical Abstracts Service Registry Number 3194-55-6

Environment Canada
Health Canada

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SYNOPSIS

Pursuant to section 74 of the *Canadian Environmental Protection Act, 1999* (CEPA 1999), the Ministers of the Environment and of Health have conducted a screening assessment of Cyclododecane, 1,2,5,6,9,10-hexabromo-, more commonly referred to as hexabromocyclododecane (HBCD), Chemical Abstracts Service Registry Number 3194-55-6. HBCD 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. Therefore, the focus of this assessment relates principally to ecological risk.

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.

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 in 2000. Amounts imported into the country in that year were in the range of 100 000 to 1 000 000 kg.

Environment

Monitoring studies document the presence of HBCD in many environmental media, sometimes at high concentrations. 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 25 to 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; however, ultimate degradation in the environment has not been definitively established. Laboratory studies conducted using water, sediment, soil and sludge confirm the presence of primary degradation products, including 1,5,9-cyclododecatriene, a substance that is not readily biodegradable and may be stable in the environment. Available evidence indicates that 1,5,9-cyclododecatriene is potentially very toxic to aquatic life (with measured and predicted median lethal concentrations (LC_{50}s) < 1 mg/L) and is potentially highly bioaccumulative in aquatic organisms.

Considered together, the lines of evidence from degradation studies and monitoring data establish that HBCD can remain stable in the environment for a period exceeding one year. The substance therefore meets the criteria for *persistence* 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—bioaccumulation (BAF) or bioconcentration factors (BCF) of 5000 or more—and is likely to have significant bioaccumulation potential in the environment. Bioconcentration factors 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. In mammals, sublethal exposures have been associated with potential toxicological effects on the liver and thyroid system, including cellular damage, significantly increased hepatic enzyme activity, significant reductions in circulating thyroid hormone levels and increases in thyroid weight.

Combustion of HBCD under certain 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 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 HBCD in the environment warrants concern in light of strong evidence that the substance is environmentally persistent and bioaccumulative.

While recent detailed production and use information are not available for HBCD, monitoring studies suggest that North American and global use of the substance may be on the rise. As well, there is evidence that HBCD may be replacing some polybrominated diphenyl ether (PBDE) flame retardants (notably the commercial Decabromodiphenyl Ether formulation).

Based on the information in this draft screening assessment, it is proposed 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, sediment), household dust, indoor air, human breast milk, and HBCD-treated consumer products. HBCD may be released from the matrix of a product over time through abrasion and usage, as it is not covalently bound. As HBCD has a low vapour pressure, it will not volatilize or off-gas from a product.

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 critical effect for the characterization of risk to human health is reproductive toxicity, with reported effects including decreased fertility and effects upon the thyroid. The highest upper-bounding estimated intake of HBCD is expected to be in infants from ingestion of human breast 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 results in margins of exposure that are considered adequately protective of human health. Based on the available information it is proposed 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.

Proposed Conclusion

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

In addition, it is proposed 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.

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 brominated flame retardant Cyclododecane, 1,2,5,6,9,10-hexabromo (hexabromocyclododecane; HBCD; CAS RN 3194-55-6) was identified in a pilot project list of 123 substances for screening assessment under CEPA 1999, based on chemical attributes which suggest it may be persistent, bioaccumulative and inherently toxic to non-human organisms. It was subsequently confirmed to meet these categorization criteria.

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 draft 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 assessement, information obtained as of March 2009 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.[1] 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 draft 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 fromToxicology, Excellence for Risk Assessment. Although external comments were taken into consideration, the content and conclusions of the screening risk assessment remain the responsibility of Health Canada and Environment Canada.

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

[1] 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 the substances in the Chemicals Management Plan (CMP) 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.

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 1. 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 6 diastereomeric pairs of enantomers and 4 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.

Physical and Chemical Properties

Table 2 contains experimental and modelled physical and chemical properties of HBCD that are relevant to its environmental fate.

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/annum (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). 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).

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 Dow Chemicals Canada Inc. to 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 XPS 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 roof
- 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 the polymer matrix. A number of factors act to constrain migration of HBCD within polymers, including the low vapour pressure, low water solubility and high predicted organic carbon/water partition coefficient (K_{oc}) of the substance (2007 email from Albemarle Corporation to Environment Canada; unreferenced). 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 synergistic 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 Dow Chemicals Canada Inc. to 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. In addition, releases from processing are expected to be much lower than those associated with the application of HBCD-containing backcoat to textiles (2007 email from Albemarle Corporation to Environment Canada; unreferenced). Whether present in air as dust particles or sorbed to particulates, the relatively high density of HBCD (2.1–2.37 g/mL, see Table 2) suggests that 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. 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 exposured to the weather once building construction is complete. 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 can be expected to undergo some degree of disintegration over time, with 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.

Products and materials containing HBCD in landfill sites will be subject to weathering, releasing HBCD particulates primarily to soil and, to a lesser extent, to water and air. 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, for example. 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 solubility 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 probably 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) 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 will 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.A recent 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 2.

Releases of HBCD to the Canadian environment due to the substance's use as a flame retardant are expected to be diffuse 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 12–14.

Low water solubility (3.4×10^{-3} mg/L at 25°C; see Table 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 unlikely to partition into air and water, moving instead into 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 support the expectation that HBCD predominantly resides in soil or sediment, depending on the compartment of release. EPIWIN Suite III fugacity modelling predicted the following partitioning to air, water, soil and sediment: air 0.0007%, water 2.1%, soil 40%, and sediment 58% (EPIsuite 2007).

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).

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%) comprised of 6.0% α-isomer, 8.5% β-isomer and 79.1% γ-isomer (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 3). 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 may exceed 182 days, making the substance persistent in this medium. As well—as noted below—there is evidence for the formation of a stable and potentially persistent 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, 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 the γ-isomer, and no information is available on the fate of α- and β-HBCD. This is particularly significant given the evidence for a predominance of the α-isomer 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 presented in terms of disappearance times rather than biodegradation (EU RAR 2008).

In a high-quality study, EBFRIP (2004b) and Davis et al. (2006) 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. Beta- 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 contrasted with those of EBFRIP (2004b), which determined there were no differences in the transformation behaviour of the three isomers.

No information could be found on the degradation properties and toxicities of tetrabromocyclododecene and dibromocyclododecadiene; however, some limited data are available for 1,5,9-cyclododecatriene, the final debromination product. The substance is classified as not readily biodegradable, with only 1% biodegradation observed in standard 28-day ready biodegradation testing (du Pont 2003). 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, Appendix A). Data from NITE (2002) further indicate that the substance has a high bioconcentration potential, with measured 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 BAF values range from 66 360 (corrected for metabolism) to 177 828 (no metabolism) (Appendix A). 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 2006) 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 needed 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 remain stable under some study conditions. Based on the available information, 1,5,9-cyclododecatriene is considered to be potentially persistent in the environment.

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; Kohler et al. 2008; Tanabe 2008). For example, Rembergeret 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 for sediment, data for HBCD suggest that the substance is persistent in sediment. Primary degradation half-lives are relatively long, but likely less than 365 days. However, ultimate degradation half-lives are likely much longer than 365 days based on an extrapolation ratio of 1:4 for a water:sediment biodegradation half-life (Boethling et al. 1995). Furthermore, sediment core measurements suggest that degradation in the environment may be on the order of years to decades. Information gathered to date on the HBCD degradation products suggests that these products are expected to be bioaccumluative and toxic, like HBCD itself.

ACCBFRIP (2003c) 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. EU RAR (2008) noted that, as with the water/sediment microcosm study described above, only the γ-isomer 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 markedly 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 dosing was higher in EBFRIP (2004b) and the test substance contained a higher proportion of γ-isomer, making it closer in composition to the current commercial product. 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 OECD 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 ACCBFRIP (2003f) 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).

In summary for soils, 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 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 on empirical and modelled 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 also possible 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 4).

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 also 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 bioconcentration factor (BCF) of 18 100 in fathead minnow, *Pimephales promelas*, exposed to 0.0062 mg/L HBCD for 32 days, while CMABFRIP (2000) calculated bioconcentration factor 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) 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 the γ-isomer. 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 the α-isomer. 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.

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 the α-isomer and 0.2 to 9.9 for γ-HBCD. A BMF for the β-isomer 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 biomagnifaction factors 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).

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.

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 per day for up to 90 days. Concentrations of

the α-isomer 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 the β-isomer was present at from 9% to 15%. This contrasted markedly with proportions present in the test substance, which contained 84.5% γ-isomer, 8.9% α-isomer and 6.6% β–isomer. 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 shown in Table 5. Model estimates range from approximately 275 400 to 6 457 000 for the BAF and from 20 400 to 24 000 for the BCF.

Based on empirical and modelled data, HBCD meets the criteria for bioaccumulation (bioaccumulation and bioconcentration factors 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 6 and 7).

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).

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 tend to be 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 (4.88×10^4 ng/L; see Table 2) relative to that of the β- (1.47×10^4 ng/L) and γ- (2.08×10^3 ng/L) isomers. Surficial sediment grab samples from the same region contained a mean concentration of 0.05 ng/g dry weight 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 dry weight for α-HBCD). The results were consistent with the γ-isomer being the most hydrophobic of the three isomers.

In a draft study, 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 g/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 dry weight, 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 the γ-isomer, while the remaining samples were dominated by the α-isomer. 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 dry weight 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 dry weight for α-HBCD). The researchers commented that the results were consistent with the γ-isomer being the most hydrophobic of the three isomers.

Concentrations of less than 1.7 ng/g to 1680 ng/g dry weight 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 dry weight 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 the α-isomer (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).

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 dry weight HBCD (mean concentration 62 800 ng/g dry weight) (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 dry weight, and decreased with increasing distance from the plant.

Waste Effluent and By-products

No North American data on concentrations in waste treatment products were found in the literature.

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 dry weight for the particulate phase; de Boer et al. 2002). However, concentrations of 2.5 ng/g to 36 000 ng/g dry weight (mean 5906 ng/g dry weight) were measured in the samples collected in the Netherlands. The substance occurred only in the particulate phase, and the γ-isomer 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 dry weight.

Concentrations of up to 29.4 ng/g dry weight (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 dry weight) in the effluents, but was present at 531 ng/g to 2683 ng/g dry weight (mean 1401 ng/g dry weight) 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 dry weight (mean 45 ng/g dry weight; 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 dry weight, with a mean value of 149 ng/g dry weight and a median of 123 ng/g dry weight.

Zennegg et al. (2005) reported concentrations of 19 to 170 ng/g dry weight (mean 85 ng/g dry weight) 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.

Biota

HBCD has been detected in North American organisms, as well as organisms from other parts of the world.

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 wet weight) 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 wet weight), 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 wet weight) and γ-HBCD (1.4 ng/g to 6.5 ng/g lipid weight; 0.23 ng/g to 0.77 ng/g wet weight). 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 (see below).

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 wet weight). Concentrations of α-HBCD were consistently higher than those of γ-HBCD, while the β-isomer was below the method detection limit (estimated at 0.03 ng/g wet weight) 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 wet weight 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 wet weight) 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 wet weight (19 ng/g to 380 ng/g lipid weight) and 0.051 ng/g to 3.6 ng/g wet weight (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 wet weight for both isomers). No significant trend in concentration over time was evident in the samples.

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 the γ-isomer 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 substantially 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 wet weight in yellow eel (*Anguilla anguilla*) and < 1.2–6758 ng/g wet weight in brown trout (*Salmo trutta*) collected from eight locations along the rivers Skerne and Tees in the United Kingdom.

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 wet weight. Concentrations in the blubber increased sharply from about 2001 onward, 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 wet weight (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, with the downward trend continuing between 2004 and 2006. The researchers attributed this to possibly being the result of the closure in 2003 of an HBCD manufacturing plant in northeastern England and 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).

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.

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.

Concentrations of 0.43 ng/g dry weight (α-isomer) and 3.88 ng/g dry weight (γ-isomer) 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 dry weight).

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 wet weight), PBDE-99 (1.56 ng/g wet weight) and PBDE-100 (6.16 ng/g wet weight), as well as total PBDEs (17.2 ng/g wet weight; 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 wet weight 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 wet weight. 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 wet weight). 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 wet weight 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 wet weight and a maximum measured value of 63.9 ng/g wet weight. 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 wet weight 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 wet weight (mean of 117 ng/g wet weight) with feathers, and from 38.4 ng/g to 194 ng/g wet weight (mean 91.0 ng/g wet weight) 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 wet weight 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 diastereoisomer predominance were attributed, at least in part, to the differing environmental fates and behaviours of the isomers, with the least water-soluble γ-isomer 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 the γ-isomer 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).

Temporal Trends

Remberger et al. (2004) reported concentrations of 0.8–1.5 ng/g dry weight 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 dry weight. 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 1950s and 1960s.

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 dry weight 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 dry weight in samples from prior to 1980 and increasing rapidly to a maximum concentration of around 0.7 ng/g dry weight 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 dry weight, 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. 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. Concentrations decreased from a highest 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.

Stapleton et al. (2006) measured 0.71–11.85 ng/g wet weight 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 the α-isomer 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 (Stapleton et al. 2006).

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).

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 wet weight (56.9 ng/g lipid weight), while those collected from the same region in 1998 contained up to 2.70 ng/g wet weight (22.67 ng/g lipid weight; Bytingsvik et al. 2004). This represents a more than six-fold increase when considered on a wet weight basis (a more than 2.5-times increase in terms of lipid weight).

Diastereomeric Differences

Studies providing a breakdown of the individual diastereomers commonly report a predominance of α-HBCD in biota samples, with the γ- and β- isomers present at lower levels or below detection limits. This congener profile contrasts markedly with that seen in commercial formulations and sediment samples, in which the γ-isomer 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-isomer than that present in the original technical formulation. This may increase the potential for organism exposure to α-HBCD during product use and disposal. As well, α-HBCD has higher water solubility (see Table 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 the α-isomer compared with those of γ-HBCD in the livers of several fish species, and considered this a possible indication that the γ-isomer 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 the α-isomer relative to that of the other two isomers.

Research by Law et al. (2006b) demonstrated 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 the α-isomer, 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.

Ecological Effects Assessment

The ecotoxicity database 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 (*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*) 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 diasteriomer found in the commercial product. Inferences about which diastereoisomer is responsible for the observed effects are not possible, since organisms would be exposed to varying HBCD diasteriomers concurrently.

No information was found on a possible mode of toxic action for HBCD. 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).

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 body weight (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.

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.

Increased microsomal enzyme activity and oxidative stress were observed in mature (4-6 months) Chinese rare minnow (*Gobiocypris rarus*) exposed to water concentrations of up to 500 µg/L of HBCD for duration periods of 28 and 42 days (Zhang et al. 2008). The researchers concluded that increasing the duration of HBCD exposure induced microsomal enzymes such as ethoxyresorufin-*O*-deethylase and pentaoxyresorufin-*O*-dependentylase, and caused the formation of excess reactive oxygen

species, finally resulting in oxidative damage to lipids, proteins and DNA, and decreased antioxidant capacities in the fish.

Kuiper et al. (2007) reported that immature European flounder, *Platichthys flesus*, exposed for 78 days to a wide range of concentrations in sediment and food (up to 800 µg/g total organic carbon (TOC) and 3000 µg/g lipid in sediment/food test systems and 8000 µg/kg TOC in sediment-only systems) exhibited no signs of hepatic microsomal enzyme induction, no alterations to thyroid gland activity or thyroid hormone levels, and no indications of endocrine effects as measured through production of the yolk precursor protein vitellogenin.

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 dry weight 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 dry weight 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 dry weight 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 dry weight 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 dry weight 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 dry weight of soil.

There are no published reports describing potential effects to wildlife species; however, a number of studies have examined toxicity in rodents. Theses 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 cultured 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.

Summaries of key toxicity studies used in the effects assessment of HBCD are provided in Table 16.

Potential to Cause Ecological Harm

The approach taken in this ecological screening assessment was to examine various pieces of supporting information and to develop conclusions based on a weight-of-evidence approach, as required under section 76.1 of CEPA 1999. The screening assessment is a conservative assessment, intended to represent reasonable worst-case conditions. It integrates known or potential exposure to the target 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. Evidence that a substance is persistent and bioaccumulative, together with evidence of commercial activity provides a significant indication of its potential to enter the environment under conditions that may have harmful long-term ecological effects (Environment Canada 2006). 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 will exist 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 toxicity at relatively low concentrations, and evidence of uses and releases) 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 3). 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*. under CEPA 1999.

Measured bioconcentration factors of up to 18 100 are reported in the published literature. Based on these data, HBCD meets CEPA 1999 bioaccumulation criteria (bioaccumulation and bioconcentration factors of 5000 or more; see Table 4).

HBCD has demonstrated toxicity 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; Zhang et al. 2008) 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 isomer, 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 combustion of flame-retarded polystyrene materials containing HBCD.

North American and global demand for HBCD may be on the rise. 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 Dow Chemicals Canada Inc. to Environment Canada; unreferenced). There is also evidence that HBCD may be replacing 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 from polybrominated diphenyl ether (PBDE) products to HBCD (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).

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

Quantitative risk estimation methods are also used to evaluate potential to cause ecological harm. A summary of data used in the risk quotient analysis of HBCD is presented in Table 17. Exposure data used in the determination of predicted exposure concentrations can be found in Tables 6 and 7. 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 exposure and determining predicted exposure concentrations (PECs) in water and sediments (see Appendix B). The database of soil HBCD concentrations was also considered inadequate, and so the soil predicted exposure concentration was derived using a simple calculation procedure involving the application of sewage sludge to agricultural soil and pastureland. Toxicity data used to determine critical toxicity values and predicted no effect concentrations are summarized in Table 16.

For pelagic organisms, risk quotients exceeded 1, indicating a current potential for risk, in surface water scenarios associated with 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. Since no Canadian or North American sewage sludge data were available, a European value was selected to represent possible levels in populated regions of Canada, such as southern Ontario. The risk quotient results suggested that current estimated exposure concentrations in Canadian soils are unlikely to exceed those leading to adverse effects in organisms.

The risk quotient derived for wildlife species highlights the potential for intake arising from the uptake of HBCD in food. In this analysis, the critical toxicity value is based on significant reductions in the levels of circulating thyroid hormones in rats receiving oral doses of 1×10^5 µg/kg to 1×10^6 µg/kg-bw per day over a 90-day period (CMABFRIP 2001). It should be noted that this level represents the lowest effect level and not the lowest adverse effect level, since no adverse effects were apparent in the affected animals. However, the endpoint is considered relevant to potential impacts in wildlife populations, since disruptions in thyroid hormone homeostasis may alter critical metabolic processes such as development of the central nervous system and cell metabolic rates (Dorland 2006). Allometric scaling was used to extrapolate data obtained from laboratory feeding studies with rats to a surrogate wildlife species, American mink. The results 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 diastereoisomeric 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 life, as well as biomagnification in terrestrial wildlife.

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.

Clarification of toxicity to sediment and soil organisms is also required. Markedly divergent outcomes were reported in 28-day *Lumbriculus* testing (i.e., NOECs of 5 and ≥ 1000 mg/kg sediment dry weight), 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

A comparison of North American and European levels of HBCD in human breast milk, blood serum (maternal and cord blood), food, adipose tissue and dust is presented in Tables 8–14. According to these data, in Canada and North America, HBCD levels in human breast milk, maternal blood/cord blood, and food, as well as dietary intakes of HBCD, either fall within the rangesof, or are lower than, those found in Europe. This

would be expected given the global distribution of HBCD usage in manufacturing consumer and industrial end-use products. Consequently, it is expected that Canadian exposures are less than European exposures to HBCD. Scenarios reported by the European Union include those listed in Table 15 (EU RAR 2008).

Upper-bounding estimates of the general population of Canada are presented in Appendix D.

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* enantioselective metabolism / excretion rather than dust ingestion or diet (Roosens et al. 2009).

The highest reported Canadian human breast milk concentration was 28 µg/kg lipid weight obtained from Canadian women from the Hamilton area in 2005 based on n=35 with 23 measured samples containing HBCD. This study reported low HBCD levels (ppb) found in North American human milk lipid. HBCD values were 20 to 100 times less than BDE47 (a congener of tetrabromodiphenyl ether used as a marker of exposure to this class of brominated flame retardants) in the same samples. HBCD global data suggest that human exposure is relatively uniform. This was the first report of isomeric content of HBCD α- and not β- or γ-HBCD in human samples and also of potential chiral selectivity of HBCD in humans (Ryan et al. 2006a). As percent lipid content of human breast milk is < 6% wt/wt and often around 3%; 3% lipid content will be used to derive an estimate of intake.

Concentrations of HBCD in representative food commodities for 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 across 310 composite samples of 31 food types were measured. Total HBCD varied in and across food groups. The HBCD intake was estimated at 16 ng/day primarily from meat consumption. Limits of detection values were used for instances of non-detects. Upper-bounding intakes from food were as follows: meat, 0.86 µg/kg wet weight (ww); dairy, 0.261 µg/kg ww; eggs 0.01 µg/kg ww; fish products 1.46 µg/kg ww; fats 0.810 µg/kg ww; cereals 0.180 µg/kg ww; fruits 0.022 µg/kg ww; and vegetables 0.018 µg/kg ww.

Consumption of fish from a contaminated lake 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. For this reason, consumption of fish with HBCD concentrations of 4.6 µg/kg wet weight (lake trout consumption in Lake Ontario, Canada) was incorporated into the derivation of upper-bounding estimates of exposure for the general population of Canada (Alaee et al. 2004). Additional data on concentrations in food are presented in Table 7.

The Canadian and Russian arctic outdoor ambient air concentration of HBCD that was selected was 1.8 pg/m^3 or 1.8 x 10^{-6} µg/m^3 for Alert, Tagish and Dunai (PWGSC-INAC-

NCP 2003). As no Canadian indoor air concentrations of HBCD were identified, measured HBCD in indoor air from residences of the United Kingdom was used as a surrogate (median HBCD concentration of 180 pg/m^3 or 0.0002 µg/m^3) (Abdallah et al. 2008a).

The highest dust concentration data for indoor air in Canadian homes reported by Abdallah et al. (2008b) of 1300 µg/kg dry weight were used to derive the upper-bounding estimates of exposure for the general population of Canada.

As concentrations of HBCD in Canadian drinking water were not found, the concentration of HBCD in lakes in the U.K. of 270 pg/L or 2.7 x 10^{-4} µg/L was used (Harrad et al 2009).

In Canada, the highest exposures were for breast-fed infants 0–6 months of age, with estimated exposures of 1.1 x 10^{-1} µg HBCD/kg-bw per day (Health Canada 2008). Concentrations in human breast milk are presented in Table 8. As HBCD is bioaccumulative, it was considerd appropriate to also derive estimates of daily intake based on measured levels of HBCD in human blood of the general population of Canada. Based on first-order kinetics the estimate of daily intake derived from the highest mean Canadian maternal/cord blood level of 2.4 µg/kg lipid weight and a half-life of 64 days with 100% absorption from the oral route was 0.01 µg/kg-bw per day. This value is very similar to the deterministic exposure estimate derived for food sources, confirming the use of the daily dietary intakes as an appropriate measure of exposure.

Consumer Products

HBCD is a brominated flame retardant that may be released from the matrix of consumer products as it is not covalently bound; it is only mixed or dissolved in the material. For this reason, HBCD may migrate from the product over time due to abrasion and usage. As HBCD has a low vapour pressure, it will not volatilize or off-gas from the product.

Upper-bounding estimates of potential oral exposure to HBCD from the mouthing of cushion or upholstered furniture were derived based on the model scenario (Environ 2003a, 2003b). These estimates are presented in Appendix E. Estimates of exposure for infants aged 0–6 months was 5.6 x 10^{-5} µg/kg-bw per day, while the oral exposure estimate for toddlers aged 6 months to 4 years of age was 2.7 x 10^{-5} µg/kg-bw per day. These estimates for infants and toddlers were derived using an established exposure algorirthm, one formally applied for the chlorinated organo-phosphate flame retardant Tris-2-chloroethyl phosphate (TCEP) (Canada 2009; Environ 2003a, 2003b). The algorithm uses a release rate of 84 mg HBCD/m^2 of fabric surface area to model wear of unaged or UV-aged fabrics, and was considered an appropriate surrogate to estimate exposure to children from mouthing cushions or upholstery. In comparison, the European Union used a release rate of 2000 mg/m^2 (a rate used for general textile release of HBCD) to model a similar exposure scenario. The approach taken in the current screening assessment is also consistent with the approach used by the U.S. Environmental Protection Agency's Voluntary Children's Chemical Evaluation Program (Environ 2003a, 2003b).

A preliminary health risk assessment for HBCD emitted into indoor air by drawing the curtain was carried out with an exposure calculation tool (US EPA MCCEM; Miyake et al. 2009). The lifetime average daily dose was calculated to be 2.67×10^{-4} µg/kg-bw per day. Parameters used by Miyake et al (2009) included an average indoor air peak concentration of 8.6 ng/m^3 for HBCD, with input parameters for room size, room volume, and air exchange rate set to be 5.25 m × 3.80 m × 2.70 m, 53.9 m^3, and 0.45 h^{-1}, respectively. Miyake et al. (2009) derived a margin of exposure of 2.1×10^5, which they reported indicated low concern for this exposure scenario.

Exposures by the dermal route are considered to be negligible based on the findings of Roper et al. (2007) that the *stratum corneum* is an efficient barrier to radiolabelled ^{14}C-HBCD penetration via the dermal route. Exposure estimates for all routes of exposure to consumer products except for mouthing (i.e., dermal and inhalation) were considered negligible and thus were not carried forward in the risk characterization of the European risk assessment (EU RAR 2008).

Health Effects Assessment

One carcinogenicity bioassay was identified (Kurokawa et al. 1984). B6C3F1 mice, 50 per sex per group, were exposed via the diet for 18 months, resulting in intakes of approximately 0, 13, 130 or 1300 mg/kg-bw per day. There were no overt signs of toxicity. The exposed animals had hepatic changes (hepatocyte swelling, degeneration, necrosis, vacuole formation, fatty infiltration), but this was not well correlated to dose. Incidences of total liver tumours were within the normal range for this strain of mouse.

The European Union reported that consistently negative results had been 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, HPCD 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 did identify the liver and thyroid as target organs for HBCD toxicity (EU RAR 2008).

Chengelis (1997) exposed male and female Sprague-Dawley rats for 28 days by gavage, at doses of 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 lowest-observed-adverse-effect level (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).

Van der Ven et al. (2006) exposed 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 typically examined in OECD guideline repeated-dose studies, which could explain why those effects were undetected in other 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.

Chengelis (2001) exposed 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 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) weights were noted. Minimal hepatocellular vacuolization was observed in all exposed animals. The LOAEL was 100 mg/kg-bw per day, based upon increased relative liver weight in both sexes. The European Union reported that control animals may also have been inadvertently exposed (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 weight was increased significantly at

the highest dose (750 mg/kg-bw per day). There were no significant changes in 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 dose 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 initiated 10 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 dose-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; see also Lilienthall et al. 2009a) conducted a one-generation 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 the 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 effects appeared to be induced during development and therefore were probably persistent. Similarly, retinoids regulate the transcription of numerous genes and can affect developmental programming, skeletal morphogenesis, embryonic growth, sex differentiation, vascularisation and reproduction. Modulation of the retinoid concentrations was proposed to be related to 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 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). 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. 2009a) 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) exposed 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 animals exposed to HBCD were hypoactive during the first part of the 60-minute period, while toward the end of the test period they became hyperactive.

Associative learning and memory were assessed by a Morris swim maze. 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 find 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 exposed to 0.9 mg/kg-bw ($p < 0.05$). The mice in the lower dose group did not differ significantly from controls. On the fifth day, the mice exposed to 13.5 mg/kg-bw took significantly longer ($p < 0.05$) to find the new position of the platform. The EU RAR (2008) considered that the study was well performed and 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 serum T3 and 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. With regard to the effect on brain development, HBCD showed evidence of affecting oligodendroglial development at a dose of 10 000 ppm, probably as a result of developmental hypothyroidism. The authors concluded that, based on the developmental brain effect, 100 ppm was the NOAEL for HCBD from changes in thyroid parameters (8.1–21.3 mg/kg-bw per day by maternal exposure level). The LOAEL would therefore be 1000 ppm, or 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

HBCD has low acute toxicity. In the one chronic bioassay identified, dietary exposure to mice for 18 months did not result in increased incidence of any tumours. The results of a limited database indicate that HBCD does not have significant 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 had selected the LOAEL of 29.9 mg/kg bw per day as one of two critical effect levels upon which to characterize risk (EU RAR 2008).

The European Union had selected a NOAEL (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 LOAEL was 101 mg/kg-bw per day, based upon a 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) exposed 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, and 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 noted 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. 2009a). The authors stated that “the most sensitive” effects were decreased mineral density in trabecular bone in F1 females (critical effective dose, 0.18 mg/kg-bw per day), decreased concentration of apolar retinoids in liver in F1 females (critical effect dose, 5.1 mg/kg-bw per day) and increased immune response in F1 males (critical effective dose, 1.45 mg/kg-bw per day). For each of these three endpoints, a 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). Due to the limitations of this study, it was not considered further for risk characterization.

HBCD has been detected in human blood, cord serum, human breast milk, dust and ambient air in Canada. Data were identified for concentrations in foods in the United States. The most relevant study for human health risk characterization was determined to be the two-generation reproductive assay with rats (Ema et al. 2008).

To assess the risk from exposure of the general population of Canada to HBCD over a lifetime, a conservative NOAEL of 10 mg/kg-bw/day was selected from the Ema et al. (2008) two-generation reproductive toxicity study. Additionally, it was considered appropriate to charactize the magnitude of the margin between potential exposures to infants and children and the LOAEL of 0.9 mg/kg-bw derived from the Eriksson (2006) study.

The most highly exposed subpopulation of the general population of Canada was breast fed infants of 0–6 months age at 1.1×10^{-1} µg/kg-bw per day derived from combined intakes from food, soil, dust and other environmental media (Health Canada 2008). This finding correlates well with the estimate derived by Eljarrat et al. (2009) for nursing infants in A. Corûna, northwestern Spain, of 1.75×10^{-1} µg ∑HBCD/kg-bw per day. The exposures for formula-fed infants and non-formula-fed infants 0–6 months of age were 1.0×10^{-2} µg ∑HBCD/kg-bw per day and 4.0×10^{-2} µg ∑HBCD/kg-bw per day, respectively. The upper-bounding estimates of exposure for the general population of Canada, as reported in Appendix D, considers levels of HBCD in household dust and food.

Based on product scenario modelling, the highest consumer product exposure estimate was 5.6×10^{-5} µg/kg-bw per day for an infant (0–6 months of age) from mouthing of flame-retarded textile or upholstered furniture.

A comparison between the critical effect level identified for the general population (10 mg/kg-bw per day) and the upper-bounding estimates of exposure for the general population (0.047 µg HBCD/kg-bw per day) results in a margin of exposure of 213 000. Additionally, the margin between upper-bounding exposures (1.1×10^{-1} µg HBCD/kg-bw per day) for breast-fed infants and the LOEL of 0.9 mg/kg-bw per day results is 8200. These margins of exposure are considered adequate to address uncertaintities in the exposure and health effects databases. Since the intakes of the rats in the two-generation study spanned *in utero* exposure, lactation and feed, the margins between the estimated intakes from human milk and the critical effect level in the animal database are considered protective of human health. This is consistent with the conclusions in the EU RAR (2008), which derived a margin of safety (MOS) of 7.0×10^{5} and concluded no concern for reproductive/fertility toxicity for breast-fed infants and no concern for breast-fed infants for repeated-dose toxicity with a MOS of 1.5×10^{6}.

The margin of exposure between the estimated intake of 5.6×10^{-5} µg/kg-bw per day by infant mouthing of textiles and the most conservative LOAEL in the animal database of 0.9 mg/kg-bw per day is 1.8×10^{7}.

Uncertainties in Evaluation of Risk to Human Health

There is moderate confidence in the database of toxicity studies for HBCD. Although the only carcinogenicity bioassay was inadequately reported, the available genotoxicity studies were negative. The critical two-generation reproduction study was reported to be compliant with the OECD guideline and conducted in accordance with good laboratory practices. Furthermore, consistent effects at similar levels of exposure were observed among the studies.

Canadian environmental media data were available for several media including levels in Canadian human breast milk, outdoor air, food biota (lake trout) and dust. Uncertainty exists in the use of the Canadian maximum level of HBCD in dust of 1300 µg/kg. There is uncertainty in the use of a surrogate median level of 0.0002 µg/m^3 from the United Kingdom (Abdallah et al. 2008a) and in the use of total HBCD levels in food from a market basket survey conducted in the United States (Schecter et al. 2009). In regard to the latter, the uncertainty is probably low given that similar food commodities are available in Canada, and U.S. levels would likely be representative of levels found in Canada. Worst-case exposure estimates are achieved through the inclusion of limit of detection values for non-detects. A level of HBCD in lakes in the U.K. was used as surrogate data in the absence of data on HBCD in Canadian drinking water. Uncertainties also are associated with the assumptions incorporated into the consumer product scenario model. The mouthing scenario estimates for infants and toddlers were similar to the ones conducted for the TCEP screening assessment (Canada 2009) and appear to be underestimates when compared to the EU derived sub-scenario; however, when the EU RAR value of 84 mg/m^2 fabric is substituted in the EU scenario, rather than the 2 mg/m^2 value, the estimates are comparable. An intake derived from a back calculation of blood level was similar to that of the intake from food sources, and a comparable estimate of exposure was derived by Eljarrat et al. (2009) for nursing infants. For these reasons, there is high confidence in the environmental media and consumer product derived estimates of exposure for the general population of Canada and the resulting derived margins of exposure.

Proposed Conclusion

Based on the information presented in this screening assessment, it is proposed 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 estimated exposures to HBCD and critical effect levels, it is proposed 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 proposed 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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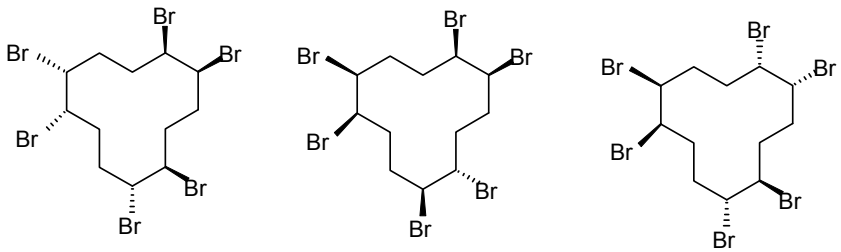
Table 1. Substance identity for HBCD

Chemical Abstracts Service Registry Number	**3194-55-6 (contains, predominantly mixed isomers α, β, γ)**
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 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	167−168 (low melt) 195−196 (high melt)		Buckingham 1982
		180−185		Albemarle Corporation 2000a, 2000b
		175−195		ACCBFRIP 2005
		180−197		Great Lakes Chemical Corporation 2005a, 2005b
	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
	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; VP/Wsol method)[2]	25	HENRYWIN 2000
Water solubility[3] (mg/L)	Experimental	3.4×10^{-3}	25	CMABFRIP 1997c
		4.88×10^{-2} (α-isomer) 1.47×10^{-2} (β-isomer) 2.08×10^{-3} (γ-isomer)	20	EBFRIP 2004a

Table 2. Physical and chemical properties of HBCD

Property	Type	Value	Temperature (°C)	Reference
	Modelled	2.09×10^{-5} 3.99×10^{-3} (calculated)	25 25	WSKOWWIN 2000 ECOSAR 2004
Log K_{ow} (Octanol-water partition coefficient; dimensionless)	Experimental	5.81	25	Veith et al. 1979
		5.625	25	CMABFRIP 1997a
	Calculated	5.07 ± 0.09 (α-isomer) 5.12 ± 0.09 (β-isomer) 5.47 ± 0.10 (γ-isomer)	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 3. Modelled data for degradation of HBCD

Fate process	Model and model basis	Model output	Expected half-life (days)[1]
AIR			
Atmospheric oxidation	AOPWIN 2000	$t_{1/2}$ = 2.133 days	> 2
Ozone reaction	AOPWIN 2000	n/a[2]	n/a
WATER			
Hydrolysis	HYDROWIN 2000	$t_{1/2} = 1.9 \times 10^5$ days (pH7) $t_{1/2} = 1.9 \times 10^5$ days (pH8)	n/a
Biodegradation (aerobic)	BIOWIN 2000 Sub-model 3: Expert Survey (ultimate biodegradation)	2.0	> 182
Biodegradation (aerobic)	BIOWIN 2000 Sub-model 4: Expert Survey (primary biodegradation)	3.1	≤ 182
Biodegradation (aerobic)	BIOWIN 2000 Sub-model 5: MITI linear probability	-0.4	> 182
Biodegradation (aerobic)	BIOWIN 2000 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] Expected half-lives for BIOWIN and CPOPs models are determined based on Environment Canada 2009.
[2] Model does not provide an estimate for this type of structure.

Table 4. 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 5. Modelled bioaccumulation data for HBCD

Test organism	Endpoint	Value wet weight (L/kg)	Reference
Fish	BAF	(assuming no metabolic transformation) 6 456 542[1]; 275 423[2]	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003)
Fish	BCF	(assuming no metabolic transformation) 20 417[1]; 23 988[2] 6211	Gobas BAF/BCF Middle Trophic Level (Arnot and Gobas 2003) BCFWIN 2000

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

Table 6. 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	United States; 2002–2003	< 0.00007–0.011 ng/m^3	in 120 of 156	Hoh and Hites 2005
Air	The Netherlands; 1999	280 ng/m^3	ns[1]	Waindzioch 2000
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	0.00002–0.00061 pg/m^3	14	Covaci et al. 2006
Air	Alert, Tagish (Canadian Arctic), Dunai (Russian Arctic)	< 0.0018 pg/m^3	12	PWGSC-INAC-NCP 2003
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	0.08–2.7 ng/L	27	Harrad et al. 2009
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. 2009
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
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

Table 6. Concentrations measured in the ambient environment and waste treatment products (continued)

Medium	Location; year	Concentration	Samples	Reference
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	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	7.91×10^7–8.61×10^7 ng/L 8850–8.17×10^7 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 effluent	United Kingdom; 1999	 1.72×10^5–1.89×10^6 ng/L 3030–46 400 ng/L	3	Dames and Moore 2000a
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] Dry weight
[7] Wet weight

Table 7. 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 wet weight			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 7. 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
Chesapeake Bay, USA; 2003	American eel (*Anguilla rostrata*)	2.2, 5.9	2	Larsen et al. 2005
	Bluegill (*Lepomis macrochirus*)	4.8	1	
	Brown bullhead (*Ameiurus nebulosus*)	25.4	1	
	Brown trout (*Salmo trutta*)	7.5	1	
	Channel catfish (*Ictalurus punctatus*)	2.2–73.9	9	
	Largemouth bass (*Micropterus salmoides*)	8.7	1	
	Pumpkinseed sunfish (*Lepomis gibbosus*)	5.3	1	
	Redbreast sunfish (*Lepomis auritus*)	4.5–9.1	4	
	Rock bass (*Ambloplites rupestris*)	1.7–6.0	3	
	Smallmouth bass (*Micropterus dolomieu*)	7.1, 15.9	2	
	Striped bass (*Morone saxatilis*)	nd–59.1	9	
	White perch (*Morone americana*)	1.0–21.0	11	
	White sucker (*Catostomus commersoni*)	3.9–19.1	3	
	Yellow bullhead (*Ameiurus natalis*)	6.9, 18.9	2	
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*)	39.9–10 275 ng/g wet weight	ns	Allchin and Morris 2003
	Brown trout (*Salmo trutta*)	< 1.2–6758 ng/g wet weight		
United Kingdom; no year	Peregrine falcon (*Falco peregrinus*)	nd–1200	in 12 of 51	de Boer et al. 2004
	Sparrow hawk (*Accipiter nisus*)	nd–19 000	in 9 of 65	
United Kingdom; 1998	Harbour porpoise (*Phocoena phocoena*)	< 5–1019	5	Morris et al. 2004
United Kingdom; 1999–2000	Cormorant (*Phalacrocorax carbo*)	138–1320	5	
United Kingdom; 2001	Sea star (*Asterias rubens*)	769	1	
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 wet weight	in 137 of 138	Law et al. 2008

Table 7. 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 dry weight	ns	Bouma et al. 2000
	Sprat (*Sprattus sprattus*)	65.5 ng/g dry weight	1	
	Bass (*species not known)*	124 ng/g dry weight	1	
	Tern (*Sterna hirundo*) egg	533–844 ng/g dry weight	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 wet weight)**	**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	
Sweden; 1999	Salmon (*Salmo salar*)	21–180	60	
		51	5	
Sweden; 2002	Herring (*Clupea harengus*)	1.5–31	ns	Asplund et al. 2004

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

Table 7. 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 7. Concentrations measured in biota (continued)

Location; year	Organism	Concentration (ng/g lipid weight)	Samples	Reference
China; 2006	Mandarin fish (*Siniperca chuatsi*) Snakehead (*Channa argus*)	**α-HBCD** 80, 120 **β-HBCD** 2.8, 3.6 **γ-HBCD** 150, 200 **ΣHBCD** 240, 330 **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 wet weight	in 4 of 66	Watanabe and Tatsukawa 1990
Japan; 1999	Minke whale (*Balaenoptera acutorostrata*) Striped dolphin (*Stenella coeruleoalba*)	57 90	1 1	Marsh et al. 2004
Japan; 2001–2006	Racoon dog (*Nyctereutes procyonoides*)	**α-HBCD** < 0.005–10 **β-HBCD** < 0.005–3.7 **γ-HBCD** < 0.005–20 **ΣHBCD** < 0.005–29	39	Kunisue et al. 2008

[1] Not detected; detection limit not specified
[2] Not specified

Table 8. Human milk lipid concentrations of individual HBCD isomers and total (∑) HBCD

Location	Human milk (µg /kg lipid weight)	N=	Reference
Canada, Province of Ontario 2003, 2005 United States of America, Austin, State of Texas 2002, 2004	Median α-HBCD 0.41 Range α-HBCD 0.2–8.8	n=27 (+13)	Ryan et al. 2006a
	Median α-HBCD 0.54 Range α-HBCD 0.2–28	n=35 (+23)	
	Median α-HBCD 0.40 Range α-HBCD 0.2–0.9	n=24 (+21)	
	Median α-HBCD 0.49 Range α-HBCD 0.2–1.2	n=25 (+20)	
Sweden 2000–2001	Median α-HBCD 0.30 Range α-HBCD 0.2–2.4	n=30 (+24)	
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=197 women between 18 and 30 years old distributed over all Belgian provinces. n=178 pooled	Coles et al. 2008
A Corûna (northwestern Spain) 2006, 2007	Median 27 Range 3–188	n= 33 (+30) Diastereoisomer levels were determined and body burden of mothers and infant exposure reported. Nursing infant dietary intake of 0.175 µg/kg-bw per day.	Eljarrat et al. 2009

Table 9. Human blood serum and cord plasma for individual isomers and ∑HBCD

Location	Human blood serum (µg /kg lipid weight)	N=	Cord plasma	N=	Reference
Canada, Arctic Nunavut and NWT regions 1994–1999	Median α-HBCD 0.7 Range α-HBCD 0.5–0.9	n=10 (+3)	Median α-HBCD 2.4 Range α-HBCD 2.4–2.4	n=10 (+1)	Walker et al.2003 as cited in Ryan et al. 2005
Canada, Arctic	HBCD at quantities < 1 Median 0.7 Range 0.5–0.9	n=10 (+3) Lipid 0.63%	Non-detect Lipid 0.17%	n=10 (0)	Muckle et al. 2001
Netherlands	Mean 1.1 Range < 0.16–4.2	n=78 weeks 20 and 35 of pregnancy			Weiss et al. 2004 as cited in Antignac et al. 2008
Netherlands	Range n.d–7	n=90	Means of 1.1 and 1.7 at weeks 20 and 35 of pregnancy		Weiss et al. 2004
Netherlands	Median 0.7 Range nd–7.4	n=69 (+68)	Median 0.2 Range 0.2–4.3	n=12 (+5)	Meijer et al. 2008
Netherlands	Median of 1.1 Range < 0.2–7.0	n=78			Meijer et al. 2008
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=2 (workers) Gamma-HBCD was high at 39% nd > 1 in a control group having no work-related exposure			Thomsen et al. 2007
Sweden	∑HBCDs Median 0.5 Range < 0.24–3.4	n=50 Gamma at 13%			Weiss et al. 2006a
Belgium	∑HBCDs Median of 1.7 Range of < 0.5–11.3	n=16(+7)			Roosens et al. 2009

Table 10. 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
-	Skin	HBCD remained on surface of skin and stratum corneum was an efficient barrier to ^{14}C - HBCD penetration.	Roper et al. 2007

Note: In Europe, the calculated margin of safety (MOS) for HBCD was $5.1 \times 10^{+3}$ to $2.0 \times 10^{+5}$, exceeding the MOS reference of $5.3 \times 10^{+2}$ (Weiss and Bergman 2006b). The 2006 level of HBCD in European humans was not considered to be of concern. It was also determined that the HBCD data were too weak for any assessment in the U.S. at that time.

Table 11. Food concentrations and dietary intakes for ∑HBCD

Location	Food (ng/g wet weight) and dietary intakes (ng/day)	Reference
United States	n=31 food commodities, 310 samples Intake mainly from meat 16 ng/day (n.d. at 60 pg/g wet weight; measured values from 23 to 192 pg/g wet weight) Dairy and Eggs (n.d. range from 4 to 128 pg/g wet weight) Fats (n.d. range from 35 to 393 pg/g wet weight; measured value for peanut butter of 300 pg/g wet weight) Cereals (n.d. of 180 pg/g wet weight) Fruit (apples) (n.d. of 22 pg/g wet weight) Potatoes (n.d. of 18 pg/g wet weight) Fish (n.d. range from 29 to 59; measured values from 113 to 593 pg/g wet weight)	Schecter et al. 2009
Belgium	n=165(+13) Median 0.10 Mean 0.13 ± 0.11 Range < 0.01–0.35 (duplicate diets) Intake median 5.5 Intake mean 7.2+/-5.2 Intake range 1.2–20	Roosens et al. 2009
Sweden	Range < 0.8–4.9 (various items)	Remberger et al. 2004
United Kingdom	Range 0.02–0.30 (market basket survey) Intake Range 354–474	Driffield et al. 2008
Norway	Range 0.12–5 (fish) Range 0.03–0.15 (meat) Range 0.2–6 (egg) Intake median 16 Intake mean 18 Intake range 4–81	Knutsen et al. 2008
Netherlands	(Market basket survey) Intake range 174	De Winter-Sorkina et al. 2003

Note: Roosens et al.'s (2009) dietary estimates of 0–20 ng ∑HBCD/day are lower than those previously reported. They are based on a short snapshot of time of exposure for a small number of individuals; the diets consumed consisted of lean meats and vegetables with low or no HBCD content; there were low detection frequencies of HBCD in the market survey; and LOQ or half LOQ concentrations were used.

Table12. Dust concentrations for individual isomers and ∑HBCD (Roosens et al. 2009)

Location	Level ng/g dry weight	n=	Reference
Canada	∑HBCD Median 640 Mean 670+/- 390 Range 64–1300	n=8	Abdallah et al. 2008b
United States	∑HBCD Median 390 Mean 810+/- 1100 Range 110–4000	n=13	Abdallah et al. 2008b
United States	∑HBCD Median 230 Mean (geo) 354 Range <4.5–130 200	n=16	Stapleton et al. 2008
Belgium	∑HBCD Median 114 Mean 160+/- 169 Range 33–758	n=16	Roosens et al. 2009
United Kingdom	∑HBCD Median 1300 Mean 8300+/- 26 000 Range 140–140 000	n=45	Abdallah et al. 2008a
United Kingdom	∑HBCD Median 730 Mean 6000+/- 20 000 Range 140–110 000	n=31	Abdallah et al. 2008b

Table 13. Mean +/- SD Exposure Factors of α, β, γ-HBCD in food, dust, serum (Roosens et al. 2009)

Compound	Food (n=12)	Dust (n=9)	Serum (n=9)
α-HBCD	0.49 ± 0.04	0.52 ± 0.02	0.28 ± 0.02
β-HBCD	0.52 ± 0.02	0.48 ± 0.03	ND
γ-HBCD	0.51 ± 0.03	0.50 ± 0.02	ND

Note: Chiral signature of all detected isomers in food and dust was racemic or close to it in all samples above LOQ. The (-)α-HBCD was the dominating enantiomer in human serum. Comparison of exposure factors with other studies is not possible as this is the first study to suggest a racemic chiral signature of HBCD in duplicate diets (Roosens et al. 2009).

Table 14. Measured total HBCDs environmental media levels

Media	Level	Reference
Indoor air (occupational)	Median 2.1 µg/m^3 Range 2–150 µg/m^3	Thomsen et al. 2007
	n=33 homes Median=180 pg/m^3 n=25 offices Median=170 pg/m^3 n=4 micro-environments Median=900 pg/m^3	Abdallah et al. 2008a
	1.8 pg/m^3 for Alert, Tagish (Canadian Arctic) and Dunai, (Russian Arctic)	PWGSC-INAC-NCP 2003
	n=9 Range 880–4800 pg/g dry weight	Harrad et al. (pending)
Dust	n=45 homes Median 1300 ng/g n=28 offices Median 760 ng/g n=20 cars Median 13 000 ng/g n=4 public micro-environments Median 2700 ng/g p< 0.05 total cars >>> total HBCDs in homes and offices	Abdallah et al. 2008a
	n=31 homes Median 730 ng/g United Kingdom, Birmingham n=13 homes Median 390 ng/g Amarillo/Austin Texas n=8 homes Median 640 ng/g Toronto, Canada n=6 offices United Kingdom, Birmingham Median 650 ng/g Highest U.K. house dust level was 110 000 ng/g	Abdallah et al. 2008b
	Median 230 ng/g Range <4.5–130 200 ng/g dry weight	Stapleton et al. 2008

Table 15. European Union Risk Assessment on HBCD

Exposure estimates of the HBCD EU 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 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 16. Summary of key toxicity studies used in the 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 $CaCO_3$, • USEPA 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, rainbow trout juvenile	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 16. Summary of key toxicity studies used in the 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 dry weight (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%, 573.4–595.5 lux	• 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] • 56-day EC_{50}	ACCBFRIP 2003a

		• USEPA 1996d; OECD 1984b, 2000	(reproduction) = 771 mg/kg soil dw	

Table 16. Summary of key toxicity studies used in the 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
Rat	99.99% purity	• 90-day treatment period, 28-day recovery period • nominal doses: 0, 100, 300 and 1000 mg/kg-bw per day by gavage • 15 female and 15 male rats per treatment • Endpoints measured: survival, clinical observations, functional operational battery, locomotor activity, clinical pathology, ophthalmic examination, reproductive function, anatomic pathology • US EPA 1998; OECD 1998b	• 90-day LOEL (decreased serum thyroid hormone) = 100 mg/kg bw per day • 90-day NOEL < 100 mg/kg bw per day	CMABFRIP 2001

[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 17. Summary of data used in the risk quotient analysis of HBCD

	Pelagic organisms	Benthic organisms	Soil organisms	Wildlife consumers
PEC	0.00004–0.015 mg/L[1]	0.33–108.2 mg/kg dry weight (dw)[1]	0.021–0.041 mg/kg soil dw[6]	4.51 mg/kg wet weight (ww)[9]
CTV	0.0056 mg/L[2]	29.25 mg/kg sediment dw[4]	235 mg/kg soil dw[7]	395 mg/kg food ww[10]
Assessment factor	10[3]	10[3]	10[3]	10[11]
PNEC	0.00056 mg/L	6.5 mg/kg sediment dw[5]	10.9 mg/kg soil dw[8]	39.5 mg/kg food ww
Risk quotient (PEC/PNEC)	0.071–10.7	0.05–7.11	0.002–0.004	0.114

[1] Due to the lack of adequate measured data, PECs were estimated using a fugacity Level III (steady-state) box model described in Appendix B, and in Environment Canada (2009).
[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) and the approach by Bonnell Environmental Consulting (2001):
$PEC_{soil} = (C_{sludge} \times AR_{sludge}) / (D_{soil} \times BD_{soil})$
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 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. A sludge concentration of 1.401 mg/kg dw reported by Morris et al. (2004) was used as C_{sludge} in the calculation. As the organic carbon content of the sludge was not specified, a standard OC level of 2% (European Communities 2003) was assumed.
[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 (LOEL) of 100 mg/kg–bw per day, based on significantly reduced levels of circulating thyroid hormones in rats (CMABFRIP 2001), was selected as the CTV for the evaluation of potential effects in wildlife. This endpoint was considered relevant as disruptions in thyroid hormone homeostasis may alter critical metabolic processes such as development of the central nervous system and cell metabolic rates. 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 395 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 A. Modelled Degradation, Bioaccumulation and Aquatic Toxicity Data for 1,5,9-Cyclododecatriene

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

Test organism	Endpoint	Value wet weight (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)

Table 1-2. 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

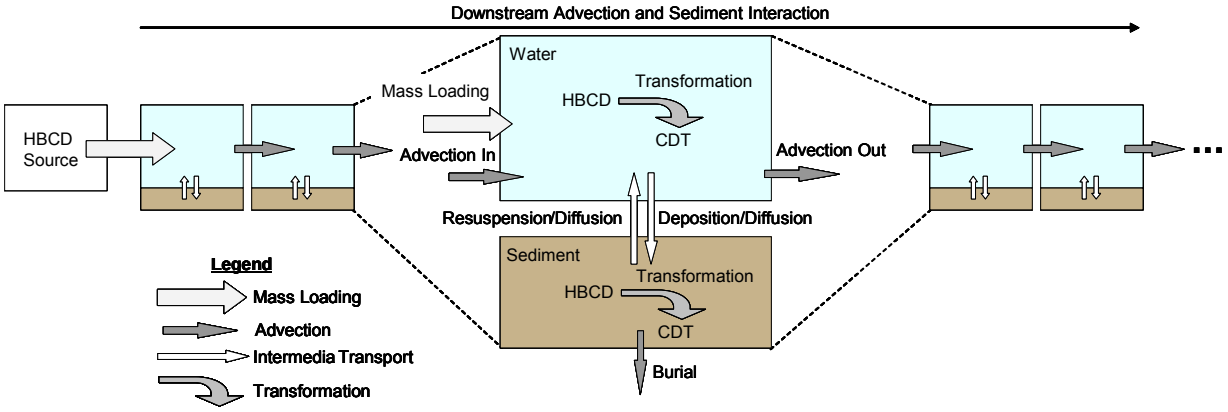
[1] Used measured log Kow of 5.5 (Howard et al. 1996)

APPENDIX B. 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 2-1 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 2-1. 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. For a detailed review of equations of

this model, the reader is referred to the supporting working document for this assessment (Environment Canada 2009).

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 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 (based on Table B2.8 of Appendix I of the TGD). 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 2-1.

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. Scenario 1a resulted in a maximum predicted HBCD concentration in the water column which exceeded the measured water solubility for HBCD (refer to Table 1). Furthermore, 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 2-2 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 2-3 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 2-1. 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 2-2. 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[5]	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[6]	10.7	1.79	NA[6]	3.75	0.893	NA[6]	1.07	0.179	NA[6]	0.339	0.071
Distance (m) with Q > 1											
NA[6]	> 5000	> 5000	NA[6]	> 5000	NA[7]	NA[6]	> 5000	NA[7]	NA[6]	NA[7]	NA[7]

[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] Predicted dissolved HBCD concentration exceeds measured water solubility (refer to Table 1).
[6] Risk quotient not calculated because the "no treatment" scenarios were considered unrealistic.
[7] Not applicable as the predicted exposure concentration was less than the estimated no effect level.

Table 2-3. 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/L)											
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/L)[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/L)[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 C. 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, 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	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, but 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 sum 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 dry weight; 28-day LOEC (no. of worms) = 29.25 mg/kg sediment dry weight; 28-day NOEC (large vs. small worms, mean biomass) = 29.25 mg/kg sediment dry weight; 28-day LOEC (large vs. small worms, mean biomass) = 311.35 mg/kg sediment dry weight.		
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): ABC 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 dry weight; 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) $\geq$ 4190 mg/kg soil dry weight; 56-day NOEC (reproduction) = 128 mg/kg soil dry weight; 56-day LOEC (reproduction) = 235 mg/kg soil dry weight; 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 D. 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]	6.0×10^{-8}	6.0×10^{-8}	6.0×10^{-8}	1.4×10^{-7}	1.1×10^{-7}	6.0×10^{-8}	5.0×10^{-8}	4.0×10^{-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]	1.0×10^{-1}	nil	2.6×10^{-2}	3.9×10^{-2}	2.8×10^{-2}	1.6×10^{-2}	1.4×10^{-2}	9.4×10^{-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	1.05×10^{-1}	5.3×10^{-3}	3.1×10^{-2}	4.7×10^{-2}	3.1×10^{-2}	1.7×10^{-2}	1.5×10^{-2}	1.0×10^{-2}

[1] Human breast milk: 28 µg/kg lipid or 0.084 or 0.1 µg/kg-bw per day based on 3% lipid or human breast milk fat content, body weight of 7.5 kg and 750 g milk consumed per day (Health Canada 2008).

[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 was 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] 1.8 pg or 1.8×10^{-6} µg/m^3 from the Canadian and Russian Arctic (Alert, Tagish and Dunai) was selected. (PWGSC-INAC-NCP 2003). Canadians are assumed to spend three hours outdoors each day (Health Canada 1998). Several ambient air measurements were taken across these northern regions of Canada. Other data considered included Hoh 2005, Hites and Hoh 2004, Backus et al 2005, Remberger et al 2004, PWGSC-INAC-NCP 2003, and Abdallah et al 2008a.

[10] The median indoor air concentration of 180 pg/m^3 or 0.0002 µg/m^3 from the United Kingdom was used as surrogate indoor air data for Canadians, n=33 (Abdallah et al. 2008a).

[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. For

fish, a value of 4.62 µg/kg will be used to capture levels of HBCD in lake trout from Lake Ontario, Canada (Alaee et al. 2004).

[13] Highest Canadian dust level in Canadian homes reported by Abdallah et al. 2008b, of 1300 µg/kg dry weight was selected.

Appendix E. Upper-bounding estimates of oral exposure to HBCD from mouthing flame-retarded cushion or upholstered furniture

Consumer product scenario	Assumptions	Estimated exposure
Oral mouthing of HBCD flame-retarded cushion or upholstered furniture	**Infants 0–6 months old** Default values from Environ (2003a, 2003b) for ingestion from mouthing. Water solubility of HBCD is 8.6 µg /L, salivary flow rate in a child's mouth (Vs) is 0.22 mL/min, fractional rate of extraction by saliva (FR) is 0.038, absorption factor by the oral route (AF_0) is 0.5, exposure frequency mouthing behaviour (EF_{mouth}) is 9 min/day (Environ 2003a, 2003b), infant body weight of 7.5 kg (Health Canada 1998). Dose rate = [WS x V_s x FR x AF_0 x EF_{mouth} x 1]/bw = [8.6 µg/L x 0.22 ml/min x 0.001 L/mL x 0.05 x 0.5 x 9min/day x 1]/7.5 kg = 5.6 x 10^{-5} µg/kg-bw per day	5.6 x 10^{-5} µg/kg-bw per day
Oral mouthing of HBCD flame-retarded cushion or upholstered furniture	**Toddlers 6 months to 4 years of age** Default values from Environ (2003a, 2003b) for ingestion from mouthing. Water solubility of HBCD is 8.6 µg /L, salivary flow rate in a child's mouth (Vs) is 0.22 mL/min, fractional rate of extraction by saliva (FR) is 0.038, absorption factor by the oral route (AF_0) is 0.5, exposure frequency mouthing behaviour (EF_{mouth}) is 9 min/day (Environ 2003a, 2003b), toddler body weight of 15.5 kg (Health Canada 1998). Dose rate = [WS x V_s x FR x AF_0 x EF_{mouth} x 1]/bw = [8.6 µg/L x 0.22 mL/min x 0.001 L/mL x 0.05 x 0.5 x 9min/day x 1]/15.5 kg = 2.7 x 10^{-5} µg/kg-bw per day	2.7 x 10^{-5} µg/kg-bw per day