Voluntary Children's Chemical Evaluation Program Pilot (VCCEPP)

Tier 1 Assessment of the Potential Health Risks to Children Associated With Exposure to the Commercial Pentabromodiphenyl Ether Product

CAS No. 32534-81-9

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Forward

Great Lakes Chemical Corporation is pleased to present to the U.S. Environmental Protection Agency (USEPA) this Tier 1 assessment report. This work, which was conducted by ENVIRON International Corporation, represents our commitment to participate in the Voluntary Children's Chemical Evaluation Program Pilot (VCCEPP) as sponsor of the children's health risk assessment of the commercial pentabromodiphenyl ether (pentaBDE) product.

Great Lakes Chemical Corporation's participation in the VCCEPP is consistent with our commitment to fulfill our responsibilities as an industry leader and corporate citizen. Great Lakes Chemical Corporation fully endorses the principles of Responsible Care® set forth by the American Chemistry Council and has translated these codes into a course of action and a way of life. The company and its employees work proactively to develop environmentally superior products and services, improve manufacturing operations and create safe working and living conditions for our employees and the communities that surround our facilities.

April 21, 2003

Great Lakes Chemical Corporation
West Lafayette, Indiana, USA

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Acronyms and Abbreviations

µg	microgram
µmol	micromole
Agency	U.S. Environmental Protection Agency
AOP	Atmospheric Oxidation Program
BDE	brominated diphenyl ether
BFR	brominated flame retardant
BMD	benchmark dose
BMDL	lower confidence limit of the benchmark dose
$BMDL_5$	benchmark dose associated with 5% increase in response over background
$BMDL_{10}$	benchmark dose associated with 10% increase in response over background
BW (or bw)	body weight
^{14}C	radioactive carbon
º C	degree Celsius
CALUX	chemical activated luciferase gene expression system
CDI	chronic daily intake
cm	centimeter
cm^3 $molecule^{-1}$ s^{-1}	cubic centimeter per molecule per second
cps	centipoises
CYP1A1	cytochrome P450 1A1
CYP2B	cytochrome P450 2B
d	day
dw	dry weight
ECB	European Chemicals Bureau
ENVIRON	ENVIRON International Corporation
EPIWIN	Estimation Program Interface for Windows
EQC	Equilibrium Concentration estimation program
EROD	ethoxyresorufin-odeethylase
EU	European Union
º F	degree Fahrenheit
FPUF	flexible polyurethane foam
fw	fresh weight
g	gram
GD	gestation day
GLCC	Great Lakes Chemical Corporation
GLP	good laboratory practice
HI	Hazard Index
HPV	high production volume
HQ	Hazard Quotient
hr	hour
IUCLID	International Uniform Chemical Information Database
kg	kilogram
kPa	kiloPascal
K_{oc}	organic carbon partition coefficient
K_{ow}	octanol-water partition coefficient
$K_{p(sed)}$	sediment partition coefficient
L	liter
LD_{50}	lethal dose at which 50% of animals die
lw	lipid weight

m^3	cubic meter
MAPK	mitogen activated protein kinase
mg	milligram
mg/day	milligram per day
mg/kg	milligram per kilogram
mg/kg-day	milligram per kilogram per day
mL	milliliter
mmHg	millimeter of mercury
molecules/cm^3	molecules per cubic centimeter
MSDS	Material Safety Data Sheet
N/A	not applicable
NCEA	National Center for Environmental Assessment
ng	nanogram
NKC	natural killer cell
nmol	nanomole
NRC	National Research Council
NOAEL	No Observable Adverse Effect Level
NOEL	No Observable Effect Level
NTP	National Toxicology Program
OPPTS	USEPA Office of Pesticide Programs and Toxic Substances
OECD	Organization of European Cooperation and Development
Pa	Pascal
PCB	polychlorinated biphenyl
pg	picogram
PLA_2	phospholipase 2
PND	post-natal day
ppb	part per billion
ppm	part per million
PROD	pentoxyresorufin-o-deethylase
RfD	Reference Dose
RME	Reasonable Maximum Exposure
SAMS model	Surface Area Modeling System model
SRBC	sheep red blood cell (erythrocyte)
T_3	3,3',5-triiodothyronine
T_4	3,3',5,5'-tetraiodothyronine, thyroxine
TBG	thyroid binding globulin
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TRα1	a thyroid hormone receptor
TRβ	a thyroid hormone receptor
TSH	thyroid stimulating hormone
UCL_{95}	95 percent upper confidence limit of the mean
UDPGT	uridinediphosphate-glucuronosyltransferase
U.S.	United States of America
USEPA	U.S. Environmental Protection Agency
VCCEPP	Voluntary Children's Chemical Evaluation Program Pilot
WHO	World Health Organization
ww	wet weight

Definition of Chemical Terms

The following acronyms are used throughout this document:

Commercial pentaBDE product	Refers only to the commercial pentabrominated diphenyl ether product
tetra-, penta-, hexa-, hepta-, octaBDE	Refers only to the homologue group containing either 4, 5, 6, 7, or 8 (respectively) bromine molecules on the diphenyl ether structure
BDE-###	Refers to one of 209 specific brominated diphenyl ether compounds using a numbering system similar to that used for polychlorinated biphenyls (see Appendix I)

Executive Summary

Introduction

A Tier 1 assessment of the potential health risks to children and prospective parents associated with exposure to the commercial pentabromodiphenyl ether (pentaBDE) product (CAS No. 32534-81-9) was conducted by Great Lakes Chemical Corporation (GLCC) in accordance with United States Environmental Protection Agency (USEPA or the Agency) Voluntary Children's Chemical Evaluation Program Pilot (VCCEPP; Federal Register Vol. 65, No. 248, pages. 81699-81718.). The commercial pentaBDE product is a brominated flame retardant produced by GLCC in the United States at one chemical plant located in the state of Arkansas. The commercial pentaBDE product is used almost exclusively by the furniture and upholstery industries as an additive in the manufacture of flexible polyurethane foam (FPUF) for cushions and mattresses.

This Tier 1 assessment is the first of three tiers in the VCCEPP program and includes four screening-level evaluations – hazard, exposure, risk, and data needs. Tiers 2 and 3 more fully characterize the potential for exposure and the associated health risks to children and prospective parents. The purpose of the VCCEPP program is to provide the USEPA with the means to evaluate and understand the potential health risks to children and prospective parents associated with the manufacture and use of industrial chemicals. The VCCEPP Tier 1 assessment of the commercial pentaBDE product addresses the following objectives:

1. Review and summarize the available animal bioassays and epidemiology studies reported in the scientific literature and identify potential human health endpoints and toxicity benchmarks relevant to children and prospective parents;

2. Understand how different uses of the commercial pentaBDE product in U.S. commerce may result in exposures to children and prospective parents;

3. Determine the plausible pathways by which children and adults might come into contact with the commercial pentaBDE product, and assess whether these pathways may result in potentially meaningful and relevant exposures to children and prospective parents;

4. Estimate exposures to the commercial pentaBDE product for potentially meaningful and relevant situations using available screening-level approaches such as the use of predictive models and/or direct environmental measurements, and including assessment of the potential for aggregate exposure (whereby an individual may be exposed simultaneously to the commercial pentaBDE product through more than one pathway);

5. Combine the findings from the hazard assessment with the results of the exposure assessment to derive qualitative and/or quantitative conclusions regarding the potential health risks to children and prospective parents; and,

6. Identify additional hazard and/or exposure information needed to reduce uncertainties surrounding the potential risks to children and, where relevant, to prospective parents.

Hazard Assessment

The commercial pentaBDE product manufactured by GLCC is composed of a mixture of primarily tetraBDE (approximately 24-38%), pentaBDE (approximately 50-62%), and hexaBDE (approximately 4-12%) congeners (IUCLID, see Appendix III). At present, GLCC is the sole manufacturer of the commercial pentaBDE product in the United States and currently produces four slightly different formulations. The four commercial pentaBDE product formulations are Great Lakes DE-60F Special™, DE-61™, DE-62™, and DE-71™, where the number indicates the percent total bromine by weight in the product.

The commercial pentaBDE product is a thick, sap-like, semi-solid material that typically is heated or mixed with diluent to improve the viscosity of the material when transferring from primary formulation vessels to product containers for sale primarily to manufacturers of FPUF. The commercial pentaBDE product is not readily biodegradable. Volatilization of the commercial pentaBDE product is minimal, and low vapor pressures have been estimated or measured for the primary constituents. The main BDE constituents of the commercial pentaBDE product are relatively immobile in soil and unlikely to leach into groundwater. There is limited data in the literature describing the bioavailability of commercial pentaBDE product or its constituents in either animals or humans. The limited animal data suggest that approximately 86% and 3% of the predicted exposure concentrations of the commercial pentaBDE product are absorbed by the oral and dermal routes of exposure, respectively. Inhalation absorption of the commercial mixture is highly unlikely; however, limited evidence suggests that friable or weathered FPUF may become airborne as fine particulate.

The available toxicology data suggest that the commercial pentaBDE product is not acutely toxic to humans or animals by the oral, dermal, or inhalation routes of exposure. It does not induce skin sensitization in guinea pigs, and has not been determined to be genotoxic in the Ames *Salmonella*, *Saccharomyces cerevisia* or in human lymphocytes *in vitro*. Among the chronic and subchronic toxicity studies available for the commercial pentaBDE product, the most prominently observed endpoints in animal bioassays are in the liver and include induction of enzymes that function in xenobiotic metabolism and microscopic observations characteristic of adaptive responses to enzyme induction, including increased liver weights and microscopic changes such as increased size of the hepatocytes and cytoplasm described as “ground glass” in appearance. Disruption of thyroid hormone (3,3’,5,5’-tetraiodothyronine, or T_4) levels have been reported (T_3 levels have not been affected). In reproductive/developmental toxicity studies in rats and mice, no effects on pregnancy or standard developmental endpoints have been reported. Decreased T_4 levels have been observed in treated dams and their offspring. Changes in neurobehavioral measures have been reported in rats and mice exposed to pentaBDE shortly after birth.

Toxicity values used in the Tier 1 assessment are summarized in Table ES-1. In the absence of data from laboratory animal models or humans, cancer was not included as an endpoint in this Tier 1 assessment. Theoretical exposures to children at different ages were evaluated by comparing the hypothetical daily intakes calculated using screening-level exposure models in the exposure assessment to toxicity threshold values for three potential health endpoints: (1) disruption of T_4 homeostasis, (2) thyroid hyperplasia, and (3) liver enzyme induction. For evaluating theoretical exposures to prospective parents, hypothetical daily intakes calculated using screening-level exposure models in the exposure assessment also were compared to toxicity threshold values. Liver enzyme induction was not considered a relevant health endpoint for adults, but was included to provide an upper-bound estimate of hazard to children and adults. There is no evidence to suggest that the alteration of liver enzyme function will result in an adverse effect on reproduction in humans.

Table ES-1. Human health endpoints and toxicity values used in the Tier 1 assessment

Human Health Endpoint	Toxicity Value	Relevant Study
Developmental Effects : Change in T_4 homeostasis	0.07 mg/kg/day	Zhou et al. (2002)
Systemic Effects: Thyroid hyperplasia	0.04 mg/kg/day	IRDC (1976)
Liver enzyme induction	0.002 mg/kg/day	USEPA Reference Dose (RfD) for the commercial pentaBDE product based on Carlson (1980b)

Exposure Assessment

The commercial pentaBDE product is produced by the direct bromination of diphenyl ether using a Friedel-Crafts catalyst. The mixture is a viscous liquid or semi-solid at ambient temperature and is supplied drummed as either the pure product or blended with a phosphate ester additive. In the United States, only one chemical manufacturing plant, owned and operated by GLCC and located in the state of Arkansas, manufactures and distributes the commercial pentaBDE product.

The commercial pentaBDE product is used almost exclusively to flame retard FPUF used in bed mattresses and cushioning in upholstered products manufactured by the furniture and textile industries. Approximately 7.5% of the more than 2.1 billion pounds of FPUF produced annually in the United States includes the commercial pentaBDE product as a flame retardant additive. The majority of FPUF products containing the commercial pentaBDE product are sold in California due to regulatory requirements in that state pertaining to flame retardant qualities of certain consumer products. According to the limited data provided by companies that purchase the commercial pentaBDE product, mattress FPUF contains 2-3% flame retardant and cushion FPUF contains 3-5% flame retardant. Scrap material from both industries have been used as

padding beneath carpets, which, as a result, likely contains 3-5% flame retardant similar to that for cushion FPUF. In both FPUF products and in carpet padding, typically only 75% of the flame retardant additive is the BDE portion of commercial pentaBDE product. The remaining 25% is aromatic phosphates.

Few environmental data describing levels of the commercial pentaBDE product in the United States are available in the literature. The available data reported in the scientific literature typically describe the occurrence and levels of one or more specific BDE congeners. A review of the scientific literature suggests that the predominant BDEs in ambient air, soil, sediment, and biota are BDE-28 (a triBDE), BDE-47 (a tetraBDE), BDE-66 (a tetraBDE), BDE-85 (a pentaBDE), BDE-99 (a pentaBDE), BDE-100 (a pentaBDE), BDE-153 (a hexaBDE), BDE-154 (a hexaBDE), BDE-183 (a hepta BDE), and BDE-209 (decaBDE). These BDE congeners occur in the commercial pentaBDE product. The current understanding of BDE levels in human breast milk is based primarily on the results of monitoring studies reported during the past decade in Sweden and Finland. At present, limited data on the levels of BDEs in breast milk from women in North America are available; among the few data reported, BDE levels in North American women are generally higher than BDE levels in breast milk reported in women from Sweden and Finland.

The populations that are potentially exposed to the commercial pentaBDE product include primary product manufacturers, FPUF manufacturers, end product manufacturers, and children and adults in home, school, and workplace environments. The entire population of workers engaged in primary manufacturing of the commercial pentaBDE product is fewer than 100 persons employed at one chemical plant located in Arkansas and are typically male workers ranging in age from 20 to 45 years old. The population of workers engaged in secondary, or chain-of-commerce, activities involving the use of the commercial pentaBDE product as an additive in the manufacture of FPUF or products containing FPUF is large, and cannot be estimated with any degree of accuracy. Similarly, the population of children and adult men and women potentially exposed to ambient levels of BDEs in the environment and through contact with consumer products containing the commercial pentaBDE product (primarily upholstered furniture, mattresses, and cushions) in the indoor home/school/office environments also is large, and cannot be estimated with any degree of accuracy.

Three scenarios were evaluated in the Tier 1 screening-level exposure assessment of children and prospective parents: exposures in the workplace, exposures in the indoor home/school/office environments, and exposures associated with ambient environmental levels (e.g., via the diet, direct and indirect contact with soil, and particulate in air). The major sources of worker exposures during the manufacture of FPUF are likely to be associated with handling of the commercial pentaBDE product prior to mixing with other ingredients; volatilization of lower BDEs from FPUF during the mixing and molding processes; and handling of residues, including FPUF scrap, during the cleaning of molding equipment. The potential exposure routes for workers engaged in FPUF manufacturing activities in the United States include inhalation of

vapors originating from primary product or foam, dermal contact with primary product or foam, and incidental ingestion via hand-to-mouth contact. In addition to direct exposures, two indirect exposure pathways involving dirty laundry and indoor floors in worker houses also were considered as plausible exposure scenarios. Workers' children are also evaluated for these last two pathways.

The potential exposure routes evaluated for children and adults in the indoor home/school/office environments in the United States included inhalation of respirable particulates originating from covered FPUF products, ingestion of particulates inhaled and swallowed, dermal contact with indoor dust that contains FPUF, incidental ingestion by young children mouthing cushions containing FPUF, and incidental ingestion of indoor dust via hand-to-mouth contact.

The potential exposure routes evaluated for children and adults associated with ambient levels in the U.S. environment included inhalation of respirable particulates in ambient air; ingestion of particulates inhaled and swallowed; dermal contact with soil; incidental ingestion of soil via hand-to-mouth contact; ingestion of meat, dairy, and vegetable food products; ingestion of recreationally caught fish; and ingestion of human breast milk by infants.

Risk Assessment

A summary of the exposure pathways in each of the three exposure scenarios for which theoretical exposures to children and prospective parents resulted in non-cancer hazard indices greater than or less than one is presented in Table ES-2. A hazard index greater than a value of one suggests that the theoretical level of exposure calculated in the screening-level Tier 1 model for a particular exposure pathway and receptor group exceeds one or more of the toxicity benchmark values for the different non-cancer human health endpoints indicated in Table ES-1.

The results of the screening-level Tier 1 exposure assessment shown in Table ES-2 are used to demonstrate that theoretical exposures to the commercial pentaBDE product by children did not result in levels that exceeded screening toxicity benchmark values. In other words, children's exposures to the commercial pentaBDE product in indoor home/school/office and ambient environments did not exceed screening toxicity benchmarks derived from animal studies for liver enzyme induction, thyroid hyperplasia, or disruption of T_4 homeostasis.

Likewise, theoretical exposure of prospective parents to the commercial pentaBDE product in the indoor home/school/office and ambient environments also did not result in hypothetical exposure levels above screening toxicity benchmark values. However, the liver enzyme induction benchmark was exceeded for prospective parents engaged in primary product production, FPUF production, and chain-of-commerce manufacturing activities. Screening toxicity benchmark values were not exceeded for thyroid hyperplasia or disruption of T_4 homeostasis. Liver enzyme induction is not considered a relevant endpoint for evaluation of exposure to the commercial pentaBDE product. It is included in this Tier 1 assessment only to provide an upper-bound estimate of hazard to children and adults. Review of the available hazard information suggests

that a higher toxicity benchmark for this endpoint is likely, which would result in hazard indices lower than those presented in this report.

Table ES-2. Summary of exposure pathways considered in the Tier 1 assessment associated with a theoretical non-cancer hazard index greater than one for one or more of the three non-cancer health endpoints included in the screening-level exposure assessment

Exposure Pathway	Hazard Index Above One?	
	Receptor Group	Toxicity Endpoint
Scenario #1 - Workplace Exposures for Adults		
Inhalation of Vapor	▪ Primary production workers ▪ Chain of commerce workers	Yes; liver enzyme induction, only
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Dermal Contact with Dirty Laundry	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact with Dirty Laundry	-none-	-none-
Dermal Contact with Dirty Floors	-none-	-none-
Incidental Ingestion of Dust from Dirty Floors	-none-	-none-
Scenario #1 – Exposures for Worker's Children		
Dermal Contact with Dirty Floors at Home	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact with Floor Surfaces	-none-	-none-
Scenario #2 – Home Exposure		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth contact	-none-	-none-
Ingestion from Mouthing a Cushion	-none-	-none-
Scenario #2 - School and Office Exposures		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Ingestion from Mouthing a Cushion	-none-	-none-
Scenario #3 – Ambient Environment Exposures		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Fish Consumption	-none-	-none-
Meat Consumption	-none-	-none-
Dairy Consumption	-none-	-none-
Fats & Oil Consumption	-none-	-none-
Egg Consumption	-none-	-none-
Vegetable Consumption	-none-	-none-
Breast Milk Consumption	-none-	-none-

Hazard indices for children based on aggregate exposures to the commercial pentaBDE product from all three exposure scenarios (i.e., workplace, home/school/office environments, and ambient environment) are summarized in Table ES-3. Aggregate exposures to children of different age groups were calculated by summing the theoretical total daily intakes associated with exposure to the commercial pentaBDE product in the indoor home (including workers' homes), school, and ambient environments. The theoretical daily intakes indicate that children in the age groups of <1 year, 2 years, and 3-5 years have the highest potential levels of exposure to the commercial pentaBDE product. For each age group, hazard indices were below a value of one.

Table ES-3. Total theoretical exposure to the commercial pentaBDE product by children with hypothetical exposures in workers' homes, indoor home/school/office environments, ambient environment, and the associated hazard indices

Receptor	Aggregate Total Theoretical Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg/day	unitless	unitless	unitless
<1 yr Child	8.90E-04	0.02	0.01	0.4
1-2 yr Child	7.41E-04	0.02	0.01	0.4
3-5 yr Child	6.75E-04	0.02	0.01	0.3
6-8 yr Child	5.73E-04	0.01	0.008	0.3
9-11 yr Male Child	5.22E-04	0.01	0.007	0.3
9-11 yr Female Child	5.18E-04	0.01	0.007	0.3
12-14 yr Male Child	4.86E-04	0.01	0.007	0.2
12-14 yr Female Child	4.68E-04	0.01	0.007	0.2
15-18 yr Male Child	4.09E-04	0.01	0.006	0.2
15-18 yr Female Child	4.21E-04	0.01	0.006	0.2

[1] Aggregate exposures to children were calculated by summing the theoretical total CDIs calculated for children hypothetically exposed to the commercial product in a worker home, in the indoor home/school/office environments, and ambient environment. This represents the highest level of total exposure developed in the Tier 1 screening-level assessment.

[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor.

[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of exposure and hazard for children and adults.

A similar approach was used to evaluate aggregate exposures to the commercial pentaBDE product by prospective parents. Hazard indices for prospective parents, based on aggregate exposures to the commercial pentaBDE product from all three exposure scenarios (i.e., workplace, indoor home/school/office environments, and ambient environment) are summarized in Table ES-4. Aggregate exposures to prospective adult male and female parents were calculated by summing the theoretical daily intakes associated with each exposure pathway evaluated in each of the three hypothetical exposure scenarios. Hazard indices were calculated in

a similar manner to that described for children in Table ES-3. The hazard indices associated with the aggregate total theoretical daily intake were calculated by summing the hazard indices associated with each of the three exposure scenarios included in the Tier 1 assessment.

The results shown in Table ES-4 demonstrate that exposure to the commercial pentaBDE product via all plausible pathways and exposure scenarios did not result in a level of exposure that exceeds relevant toxicity benchmark values for developmental or systemic effects. In the workplace, hypothetical exposures exceeded the USEPA (2003) RfD value based on liver enzyme induction. This endpoint (liver enzyme induction) is not considered applicable or relevant to the evaluation of children's health, but is provided in this Tier 1 assessment to represent an upper-bound estimate of the potential hazard to children and adults. The workplace scenario is the largest contributor to the aggregate hazard index, which indicates that good industrial hygiene practices are needed to control exposures.

Table ES-4. Total theoretical exposure to the commercial pentaBDE product by adults with hypothetical exposures in the workplace, indoor home/school/office environments, ambient environment, and the associated hazard indices

Hypothetical Exposure Scenario	Aggregate Total Theoretical Chronic Daily Intake	Hazard Index		
		Comparison to Thyroid Effects Benchmark [1]	Comparison to Developmental Effects Benchmark [2]	Comparison to Liver Enzyme Induction Benchmark [3]
	mg/kg-day	*unitless*	*unitless*	*unitless*
Prospective Parent, Female				
Workplace Scenario [4]	0.03	0.7	0.4	14
Home, School, and Office Scenario [5]	5.06E-07	0.00001	0.00001	0.0003
Ambient Environmental Scenario [6]	2.21E-05	0.0006	0.003	0.01
Recreational Freshwater Fishing Scenario [7]	1.68E-05	0.0004	0.0002	0.01
SUM	**0.03**	**0.7**	**0.4**	**14**
Prospective Parent, Male				
Workplace Scenario [4]	0.03	0.7	0.4	14
Home, School, and Office Scenario [5]	5.45E-07	0.00001	0.00001	0.0003
Ambient Environmental Scenario [6]	1.89E-05	0.0005	0.0003	0.009
Recreational Freshwater Fishing Scenario [7]	1.40E-05	0.0004	0.0002	0.007
SUM	**0.03**	**0.7**	**0.4**	**14**

[1] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor.

[2] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[3] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard for children and adults.

[4] The theoretical highest exposed adult in the workplace scenario is associated with cutting FPUF treated with the commercial pentaBDE product, as shown in Table 4-5.

[5] The theoretical exposures in the home, office, and school environments were calculated by summing the total CDIs for the prospective female parent in Table 4-3.

[6] The theoretical exposures in the ambient environment represent the prospective parent in Table 4-4.
[7] The theoretical exposures from recreational fish consumption represent the prospective fishing person in Table 4-4.

Data Needs Assessment

While the margins of safety found in this assessment for the most part indicate there is substantial room for uncertainty, both in terms of exposure and hazard assessment, the degree of uncertainty that is acceptable to stakeholders will vary considerably. There are existing gaps in the understanding of both exposure and hazard. Filling these gaps would clearly help reduce the degree of uncertainty, but would not necessarily change the current Tier 1 assessment results. Several sources of uncertainty have been identified in this Tier 1 assessment. One large source of uncertainty in the Tier 1 assessment includes the lack of source-specific exposure data on lower BDEs in the United States. Consequently, what are believed to be worst case exposure estimates were used in this evaluation. For example, the significance of mouthing behavior in young children as a source of exposure to the commercial pentaBDE product is poorly understood. The conservative screening-level approach used in this Tier 1 assessment likely over estimates by a large margin the level of exposure encountered by children less than 5 years old in the home and other indoor environments. Additional research could be done to further understand levels in FPUF furniture cushions and the mobility of the commercial pentaBDE product and/or its constituents from FPUF to the saliva of young children.

The levels of the commercial pentaBDE product and/or its constituents in different ambient environmental compartments represent additional sources of uncertainty in the Tier 1 assessment. At the present time, the pathways by which the commercial mixture or its primary constituents are released to the environment and accumulate in different biotic and abiotic compartments are the subject of debate and study in the United States and elsewhere. The data available describing environmental levels in the United States provide, at present, only partial understanding of the potential for exposures to the commercial mixture or its primary constituents in the ambient environment.

In addition to the uncertainties associated with U.S. data on environmental levels, there are data gaps in the available hazard studies that limit the understanding of the potential for adverse health effects associated with exposure to the commercial pentaBDE product and/or its constituents. In particular, the relevance to humans of effects on the thyroid, changes in body weight, and neurobehavioral effects observed in laboratory animals is not well understood. While it does not appear that the levels of lower brominated BDE's reported in the scientific literature exceed thresholds that potentially adversely affect human health, mechanistic studies could improve the understanding of the effects of BDE constituents in the commercial pentaBDE product on children.

Several of the VCCEPP Tier 1, 2, and 3 hazard studies specified by USEPA are available for commercial pentaBDE. One of the data gaps in the current understanding of the commercial pentaBDE product is the absence of a two-year chronic bioassay, which would also include

evaluation of carcinogenicity as a potential endpoint. A two-year chronic bioassay of the commercial pentaBDE product was identified as a possible data gap in the VCCEPP. This work has already been identified as a research goal by the National Toxicology Program. In addition, studies to improve the understanding of the effects of BDE constituents of the commercial product on thyroid function, neurobehavorial responses and developmental effects in laboratory animals could be performed to reduce uncertainties related to these endpoints. The lack of a multi-generation reproductive toxicity study is an additional data gap. Work planned by Health Canada is expected to address many, if not all, of these gaps. Consequently, it presently appears that current data gaps are likely to be filled within the next few years. It is anticipated that the results of various hazard and exposure studies currently underway or planned in the United States and Canada should improve the understanding of BDE levels in the environment and the potential risks posed to children and prospective parents.

1.0 Introduction

This Tier 1 assessment was conducted by ENVIRON International Corporation (ENVIRON) on behalf of Great Lakes Chemical Corporation (GLCC) to evaluate the potential health risks to children and prospective parents associated with exposure to the commercial pentabromodiphenyl ether (pentaBDE) product (CAS No. 32534-81-9). This work was performed in accordance with the framework for developing a Tier 1 assessment for the Voluntary Children's Chemical Evaluation Program Pilot (VCCEPP). The work described in this document meets both the specific policy and technical objectives of the U. S. Environmental Protection Agency Office of Pollution Prevention and Toxics (hereinafter referred to as USEPA or Agency) VCCEPP program, and reflects both a state-of-the science evaluation and practical judgment.

1.1 VCCEPP Program

In the December 26, 2000, Federal Register (Vol. 65, No. 248, pp. 81699-81718), USEPA announced the VCCEPP. The USEPA asked companies which manufacture and/or import 23 chemicals (including the commercial pentaBDE product) that have been found in human tissues or measured in the environment as part of various monitoring programs to volunteer to sponsor one or more of these chemicals in the first tier pilot of this Program.

The stated purpose of the VCCEPP is to provide the Agency and the public with the means to understand the potential health risks to children and prospective parents associated with potential exposures to chemicals. The VCCEPP is intended to ensure that human health effects and exposure data, which may include data not necessarily limited to assessment of children's health, are made available to USEPA and others for evaluation of the risks of these chemicals so in that respect, mitigation measures may be taken as appropriate. Additionally, the Agency also committed as part of the Program to consider animal welfare and to provide instructions on ways to reduce or, in some cases, eliminate animal testing, while at the same time ensuring that public health is adequately protected. The Agency views the VCCEPP as an important component of its overall Chemical Right-to-Know initiative.

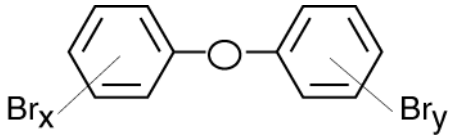
The Agency characterizes the VCCEPP as a pilot program focused only on 23 widely used industrial chemicals. The USEPA expects to refine the Program before its application to a larger range of chemicals in the future. Three tiers of assessment are envisioned in the VCCEPP. Tier 1 is characterized as a preliminary, screening evaluation (and is the focus of this report). If the results of the Tier 1 assessment indicate further evaluation is needed, then Tiers 2 and 3 will more fully characterize the risks that a chemical may pose to children, which would likely include laboratory toxicity bioassays and exposure studies to address data gaps or significant uncertainties in our understanding of the potential health risks.

In June 2001, GLCC agreed to participate in the VCCEPP and to sponsor a Tier 1 assessment of the commercial pentaBDE product. As part of Tier 1 sponsorship, GLCC committed to collect and/or develop information on health effects (hazard), exposure, and health risks. The hazard assessment requested by USEPA is nearly the same information requested in USEPA's High Production Volume (HPV) Challenge Program. GLCC also agreed to develop an exposure assessment and risk assessment. The assessment submitted as part of Tier 1 is intended to represent screening level information, with more detailed analyses submitted later as part of a future Tier 2 or Tier 3 commitment. In addition, GLCC agreed to perform a data needs assessment. This fourth item specifies whether additional environmental, exposure, and/or toxicology data are needed to address data gaps and reduce uncertainties either in the hazard assessment, exposure assessment, or risk assessment. The complete Tier 1 assessment (comprised of four components – hazard, exposure, risk, and data needs) will be submitted to USEPA's Peer Consultation Group for independent review.

1.2 Commercial Pentabromodiphenyl Ether (PentaBDE) Product

The focus of this VCCEPP Tier 1 assessment is the commercial pentaBDE product:

IUPAC name:	Pentabromodiphenyl ether (diphenyl ether, pentabromo derivative)
CAS Number:	32534-81-9
EINECS Number:	251-084-2
Molecular formula:	$C_{12}H_5Br_5O$
Molecular weight:	564.72
Structural formula:	where x + y = 5



The various synonyms and abbreviations used to denote individual BDE isomers, BDEs as a group, and the commercial products often lead unintentionally to mis-identification of the commercial pentaBDE product in the scientific literature. Synonyms for BDEs and their abbreviations include the following:

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

Unless otherwise stated, the term commercial pentaBDE product will be used in this assessment to refer specifically to the commercially available product. When referring to a BDE congener

group, the designation mono-, di-, tri-, tetra-, penta-, hexa-, hepta-, octa-, nona- and deca- will be added to the beginning of the abbreviation to indicate the degree of bromination. Reference to one of the 209 individual BDE congeners will be identified more specifically where appropriate using a numbering system similar to that used to distinguish the individual PCBs (see Appendix I).

According to Alaee and Wenning (2002), there are 175 different flame retardant chemicals, divided into four major groups: inorganic, halogenated organic, organophosphorus and nitrogen-based compounds and mixtures (EHC, 1997). Halogenated organic flame retardants are generally classified as either chlorinated or brominated flame retardants (BFRs). BFRs, which include the brominated diphenyl ethers (BDEs), are further classified as either reactive or additive materials. The three BDE flame retardants available commercially are additive materials and are referred to as the commercial penta-, octa-, and decabromodiphenyl ether products. Each of the commercial products is a mixture of BDEs with varying degrees of bromination.

This VCCEPP Tier 1 assessment addresses only the commercial pentaBDE product. The commercial pentaBDE product is composed of a mixture of triBDE to hexaBDE isomers. The predominate congeners in the commercial pentaBDE product are 2,2',4,4',5- pentaBDE (BDE-99) [1] and 2,2',4,4'-tetraBDE (BDE-47). Several chemical additives can be included in the commercial pentaBDE product and comprise roughly 20-30% by weight of the product formulation currently sold in the United States (i.e., Great Lakes DE-60F Special™). The relative proportions of the different BDE constituents in commercial pentaBDE products sold in the United States and elsewhere, both past and present, are presented in Tables 1-1 and 1-2.

[1] This acronym, as well as others, refers to a specific polybrominated diphenyl ether compound and is based on a similar numbering system to that developed for polychlorinated biphenyls (PCBs) (See Appendix I).

Table 1-1. Relative proportions by percent weight of different BDE homologue groups in the commercial pentaBDE, octaBDE, and decaBDE products

BDE Homologue Group	Commercial pentaBDE product			Commercial octaBDE product		Commercial decaBDE product
	2002	1985	late 1970's - early 1980's	2002	late 1970's - early 1980's	2002
	Great Lakes DE-71™ as currently produced [1]	Saytex 115 [2]	Great Lakes product [3]	Great Lakes DE-79™ as currently produced [4]	Great Lakes product [5]	Great Lakes DE-83R™ and DE-83™ as currently produced [6]
Decabromodiphenyl ether (1 congener)	--	--	0.8%	<0.7%	1.6%	>98%
Nonabromodiphenyl ether (3 congeners)	--	--	0.2%	<10%	13.0%	--
Octabromodiphenyl ether (12 congeners)	--	--	0.3%	<33%	30.7%	--
Heptabromodiphenyl ether (24 congeners)	--	--	2.6%	<45%	45.1%	--
Hexabromodiphenyl ether (42 congeners)	4-12%	--	13.3%	<12%	8.5%	--
Pentabromodiphenyl ether (46 congeners)	50-62%	--	58.1%	<0.50%	1.1%	--
Tetrabromodiphenyl ether (42 congeners)	24-38%	--	24.6%	--	--	--
Tribromodiphenyl ether (24 congeners)	0-1%	--	--	--	--	--

[1] Data reported in the IUCLID document (see Appendix III) and GLCC MSDS for this product (see Appendix II). Other Great Lakes pentaBDE products (DE-60F Special™, DE-61™ and DE-62™) contain, by weight, 75% BDEs and 25% aromatic phosphates. Great Lakes DE-79™ was used in the study by Zhou et al. (2001, 2002) to evaluate the potential for effects on the thyroid in rodents.

[2] Saytex 115 was used by Ethyl Corporation (1985a) to evaluate potential developmental effects in rodents. The composition of the product is unknown.

[3] The GLCC commercial pentaBDE product produced in the late 1970's and early 1980's was used by Carlson (1980a) to evaluate potential effects on the liver in rodents, which is the basis for USEPA's current non-cancer Reference Dose toxicity value reported in IRIS (USEPA, 2003).

[4] The composition of Great Lakes DE-79™ is reported in ECB (2000).

[5] The GLCC commercial octaBDE product produced in the late 1970's and early 1980's was used by Carlson (1980a) to evaluate potential liver effects in rodents, which is the basis for USEPA's current non-cancer Reference Dose toxicity value reported in IRIS (USEPA, 2003).

[6] MSDS (See Appendix II).

Table 1-2. Relative proportions by percent weight of different individual key BDE congeners in the commercial pentaBDE, octaBDE, and decaBDE product and occurrence in the manufacturing workplace, indoor buildings, and ambient environment

BDE Homologue Group	BDE Congener [1]	Formulations [2]				Indoor Buildings		Ambient Environment								
		Great Lakes DE-71 pentaBDE product	Great Lakes DE-79 octaBDE product	Bromkal 70-5DE pentaBDE product	Bromkal 79-8DE octaBDE product	Indoor Air [3]	Indoor Dust [4]	Ambient Air [5]	Soil [6]	Fish [7]	Meat [8]	Dairy [8]	Other Fats and Oils [8]	Eggs [8]	Vegetables [9]	Breast Milk [10]
Decabromodiphenyl Ether (1 congener)	**BDE-209**	**--**	**--**	**--**	**--**	11%	77%	1%	--	--	--	--	--	--	--	--
Nonabromodiphenyl Ether (3 congeners)	**--**	**--**	**--**	**--**	**--**	--	--	--	--	--	--	--	--	--	--	--
Octabromodiphenyl Ether (12 congeners)	**--**	**--**	**--**	**--**	**--**	--	--	--	--	--	--	--	--	--	--	--
Heptabromodiphenyl Ether (24 congeners)	**BDE-183**	**--**	**--**	**--**	**--**	1%	2%	--	--	--	--	--	--	--	--	✓
Hexabromodiphenyl Ether (42 congeners)	**BDE-154**	4%	2%	2%	4%	1%	1%	1%	--	✓	✓	✓	✓	✓	✓	✓
	BDE-153	6%	14%	3%	3%	2%	2%	1%	--	✓	✓	✓	✓	✓	✓	✓
Pentabromodiphenyl Ether (46 congeners)	**BDE-100**	8%	ND	7%	ND	4%	1%	4%	4%	✓	✓	✓	✓	✓	✓	✓
	BDE-99	43%	<1%	<1%	2%	27%	10%	31%	54%	✓	✓	✓	✓	✓	✓	✓
	BDE-85	--	--	--	--	1%	0%	--	--	--	--	--	--	--	--	--
Tetrabromodiphenyl Ether (42 congeners)	**BDE-66**	<1%	ND	<1%	ND	--	--	--	--	✓	--	--	--	--	--	**--**
	BDE-49	<1%	ND	<1%	ND	--	1%	--	--	--	--	--	--	--	--	**--**
	BDE-47	28%	<1%	30%	ND	53%	7%	63%	42%	✓	✓	✓	✓	✓	✓	✓
Tribromodiphenyl Ether (24 congeners)	**BDE-28/33**	**>1%**	**ND**	**>1%**	**ND**	--	--	--	--	--	--	--	--	--	--	✓

-- = data not reported

✓= congener reported relative percent occurrence not known

[1]. Numbering system adopted from PCBs (see Appendix I).

[2]. Data from Rayne and Ikonomou, 2002.

[3]. Sjödin et al., 2001a. Samples collected in a teaching hall with computers in Sweden.

[4]. Knoth et al., 2002. Samples collected from vacuum cleaner bags from German households.

[5]. Strandberg et al., 2001. Ambient air sampled in an urban location (Chicago) over three years (1997-1999).

[6]. Hale, 2002. Soil sampled near a foam production plant in mid-Atlantic region, USA.

[7]. Compilation of several studies. A total of 72 fish fillet samples from the USA.

[8]. Darnerud et al., 2000; unpublished. Swedish market basket study.

[9]. Ohta et al., 2002. Japanese market basket study.

[10]. Ryan et al., 2002. Twenty breast milk samples from Vancouver, Canada, collected in 2001-2002.

The commercial pentaBDE product is used almost exclusively to flame retard flexible polyurethane foam (FPUF) used in bed mattresses and cushioning in upholstered products (IPCS, 1994). According to the limited data provided by companies that purchase the commercial pentaBDE product from GLCC for use as a flame retardant additive in FPUF, mattress FPUF contains 2-3% flame retardant and cushion FPUF contains 3-5% flame retardant. Scrap material from both industries have been used as padding beneath carpets, and, as a result, likely contains 3-5% flame retardant similar to that for cushion FPUF. In both FPUF products and in carpet padding, typically only 75% of the flame retardant additive is the BDE portion of commercial pentaBDE product. The other 25% is aromatic phosphates.

BDEs were the first group of BFRs to be detected in the environment. In 1979, the presence of BDE-209 (decaBDE) was measured in soil and sludge samples collected from areas surrounding BDE manufacturing facilities in the United States (de Carlo, 1979). Jansson and Asplund (1987) reported certain BDEs in tissue samples of fish-eating birds and marine mammals collected from the Baltic Sea, North Sea, and Arctic Ocean. Similar reports confirmed the presence of lower BDE congeners in marine fish, shellfish, and sediments in the Pacific region and elsewhere (Watanabe et al., 1987). Stafford (1983) confirmed the presence of lower BDEs in North America, reporting several congeners in the eggs and tissues of fish-eating birds from six U.S. states and Canada.

In recent years, increasing attention has been given to the presence of BDEs in the environment. Nearly every environmental monitoring program conducted during the past decade has shown increasing levels of some BDE congeners in wildlife, particularly in Nordic countries where this trend contrasts with a general decline in the occurrence of dioxins, polychlorinated biphenyls (PCBs), and chlorinated pesticides in marine mammals and aquatic wildlife (Bergman, 2000; Hooper and MacDonald, 2000). More recently, evidence has begun to appear suggesting the presence of BDEs in humans. BDEs have been detected in human adipose tissue, blood serum, and in human breast milk (Stanley et. al., 1991; Klasson-Wehler et. al., 1997; Darnerud et al., 1998; Norén and Meironyté, 1998; Schroeter-Kermani et al., 2000; Päpke et. al., 2001; Petreas et. al., 2002; Ryan et. al., 2002; Thomsen et al., 2002; van Bavel et al., 2002; Choi et al., 2003; Meironyté-Guvenius et al., 2003; Petreas et al., 2003). The major route of human exposure appears to be through the diet, although data on this and other routes of exposure are limited at this time. The presence in humans requires that attention be given to exposure of infants and young children to BDEs and the potential for adverse health effects to the developing fetus and the breast-fed neonate.

To date, there have been few, if any, studies by scientists to measure the commercial BDE products in the environment or to correlate the occurrence of specific BDE isomers and congeners with the commercial pentaBDE products. Environmental monitoring data reported in the scientific literature typically describe the concentrations of total BDEs, congener groups, or the concentrations only of certain BDE congeners (see for example Alaee and Wenning, 2002; de Wit, 2002). Data reported in this manner hinder attempts to develop reliable exposure estimates

describing plausible human exposure to the commercial pentaBDE product. Interpretation of the available environmental data is further complicated by the fact that much more extensive analysis has been performed for only certain BDE congeners, some of which occur in the commercial pentaBDE product.

In general, the bioavailibility and toxicity of the BDE molecule declines with increasing levels of bromination. The risk assessment of the commercial pentaBDE product conducted by the European Chemicals Bureau (ECB) and other scientists acknowledge that there are relatively few data indicating potential adverse health effects in humans (Palm et al., 2002; Wenning, 2002; Darnerud et al., 2001; ECB, 2000). Available data indicate that the commercial pentaBDE product is not mutagenic *in vitro*, and probably not genotoxic *in vivo* (ECB, 2000; Hardy, 2002a, 2002b). There are no data supporting carcinogenicity of the commercial pentaBDE product. There is general agreement within the scientific community that the most sensitive end points for toxicity in animal bioassays appear to be associated with effects on thyroid function, and particularly disruption of T_4 homeostasis, induction of thyroid hyperplasia, and alteration of thyroid hormone production (Fowles et al., 1994).

1.3 Purpose & Objectives

Conceptually, the VCCEPP Tier 1 assessment of the commercial pentaBDE product is intended to address the following objectives:

1. Review and summarize the available animal bioassays and epidemiology studies reported in the scientific literature and identify potential human health endpoints and toxicity benchmarks relevant to children and prospective parents;

2. Understand how different uses of the commercial pentaBDE product in U.S. commerce may result in exposures to children and prospective parents;

3. Determine the plausible pathways by which children and prospective parents might come into contact with the commercial pentaBDE product, and assess whether these pathways may result in potentially meaningful and relevant exposures to children and prospective parents;

4. Estimate exposures to the commercial pentaBDE product for potentially meaningful and relevant situations using available screening-level approaches such as the use of predictive models and/or direct environmental measurements, and including assessment of the potential for aggregate exposure (whereby an individual may be exposed simultaneously to the commercial pentaBDE product through more than one pathway);

5. Combine the findings from the hazard assessment with the results of the exposure assessment to derive qualitative and/or quantitative conclusions regarding the potential health risks to children and prospective parents; and,

6. Identify additional hazard and/or exposure information needed to reduce uncertainties surrounding the potential risks to children and, where relevant, to prospective parents.

To achieve these objectives, the purpose of the VCCEPP Tier 1 assessment presented in this document is to report on the results of screening-level hazard, exposure, risk and data needs assessments based on the available environmental and toxicological information and using the current paradigms for children's health risk assessment. The information and modeling performed as part of this VCCEPP Tier 1 assessment have been fully documented and are consistent with federal guidance on risk assessment and children's health evaluation (USEPA, 1992, 1997, 2000a). The hazard, exposure, risk, and data needs assessments submitted to USEPA as part of this document are intended to be sufficient for independent third-party review by the Peer Consultation Group. This Tier 1 assessment represents the current state-of-the-science and addresses the adequacy of information associated with sources of exposure, populations at risk, environmental pathways, and routes of exposure to children.

1.4 Overall Tier 1 Approach & Scope

In accordance with available USEPA risk assessment guidance and recent guidance on preparing a VCCEPP Tier 1 assessment (Federal Register Notice, Vol. 65, No. 248, 81699-81718, Sections I through K), this Tier 1 assessment includes four major components: (1) hazard assessment, (2) exposure assessment, (3) risk assessment, and (4) data needs assessment. Recent USEPA guidance on risk assessment of chemical mixtures also was consulted during the preparation of this assessment (USEPA, 2000c).

Hazard Assessment – A toxicology profile of the commercial pentaBDE product, including abbreviated summaries of the available toxicology studies conducted to date, is presented in this section. A list of the chemical nomenclature used to define individual BDE congeners is included in Appendix I. A Material Safety Data Sheet (MSDS) for the commercial BDE products produced by GLCC is included in Appendix II. An International Uniform Chemical Information Database (IUCLID) data set containing hazard summaries describing each of the available relevant toxicology studies is included in Appendix III. In accordance with USEPA VCCEPP guidance, the IUCLID format is consistent with USEPA guidance for preparing Robust Summaries for the High Production Volume (HPV) Chemicals Challenge Program. This section concludes with a summary of available toxicity reference values used in the risk assessment for comparison to theoretical daily intake values and calculation of non-cancer hazard indices for various toxicity endpoints of concern.

Exposure Assessment – This section of the Tier 1 assessment includes four components. The first component describes the general substance information on the commercial pentaBDE product including physical/chemical properties, the environmental fate, and exposure limits. The second component is a summary of primary manufacturing methods and chain-of-commerce uses for the commercial pentaBDE product, including a description of primary manufacturing activities and highlighting those activities associated with the manufacture of the commercial pentaBDE

product that have the potential for significant worker exposures. Similarly, the predominant chain-of-commerce activities involving the use of the commercial pentaBDE product as part of the manufacture of FPUF and its subsequent use in consumer products also highlights the potential for significant worker exposures. A description of chain-of-commerce uses of consumer products containing FPUF that has been treated with the commercial pentaBDE product, along with an assessment of the potential for general population exposures, is included.

The third component of the exposure assessment is a summary of available data describing environmental levels in the United State, North America and elsewhere. During data compilation, preference was given to data relevant to the ambient environment, home environment, and workplace in the United States. In the absence of U.S. data, secondary preference was given to data from Canada; third preference was given to data from the European Union and elsewhere. The available U.S. data representing different environmental compartments (e.g., air, soil, sediment, fish, food products) are compared to data describing environmental levels in Europe and elsewhere. The data compiled from the literature are summarized in Appendix V. These comparisons provide a benchmark for establishing a general understanding of the occurrence of the commercial pentaBDE product in the U.S. environment. In some cases, the compiled data are used to support exposure point concentrations in the Tier 1 screening-level exposure models.

The fourth component of the exposure assessment is the Tier 1 conceptual exposure model describing the possible hypothetical routes of exposure to children and prospective parents. The conceptual exposure model for the commercial pentaBDE product illustrates three different scenarios: exposures in the workplace, exposures in the indoor home/school/office environments, and exposures associated with ambient environmental levels (e.g., via the diet, direct and indirect contact with soil, and particulates in air). For direct and indirect exposure routes illustrated in the conceptual model where exposures are plausible and quantifiable, screening-level exposure models developed in Microsoft® Excel are presented for children of different ages and adult men and women (prospective parents). The calculations used in the exposure assessment are presented in Appendix VI. Child exposures are evaluated for seven different age bins: <1 year, 1-2 years, 3-5 years, 6-8 years, 9-11 years, 12-14 years, and 15-18 years. Girls and boys are evaluated separately in the 9-11, 12-14, and 15-18 year age groups.

Consistent with the Tier 1 screening approach, exposure models rely on exposure point concentrations for different environmental compartments (e.g., air, soil, dust, food stuffs, and the workplace) representing the 95th upper confidence limit of the mean for robust data sets or the high end of the range of the available data, if the available data are limited. Exposure factors, based primarily on USEPA's *Exposure Factors Handbook* (USEPA, 1997) and *Child-Specific Exposure Factors Handbook* (USEPA, 2000a), are identified for each of the exposure scenarios and associated exposure pathways. All of the assumptions and parameter values used in the screening-level exposure calculations are specified and referenced to the relevant supporting technical information.

The exposure assessment section concludes with a summary of the results of the Tier 1 screening-level exposure models for the workplace, indoor home/school/office environments, and ambient environment scenarios. Summary tables are used to convey quantitative results relating sources of exposure to theoretical estimates of human exposure.

Risk Assessment – The results of the hazard and exposure assessments are used in the risk assessment section to convey both qualitative and quantitative conclusions about the likelihood that exposure to the commercial pentaBDE product may pose a specific hazard to children or, where relevant, to prospective parents. Chronic intakes estimated using screening-level exposure models are compared to toxicity benchmark values relevant to children and prospective parents. The oral reference dose (RfD) value reported in USEPA's Integrated Risk Information System (IRIS) database was considered not to be an appropriate value, but is included in the report to provide an upper-bound estimate of the potential hazard to children and adults. A key aspect of this effort is not to evaluate the toxicity of the individual chemical constituents in the commercial product, but, rather, to evaluate the toxicity of the commercial pentaBDE product as a whole. To this end, results are also compared to other toxicity reference values using endpoints identified in the literature associated with studies of the commercial pentaBDE product. Hazard information developed from studies of individual BDEs is not included in this Tier 1 risk assessment. The risk assessment section also includes a qualitative discussion of the uncertainties inherent in the screening-level exposure models and their potential impact on the results of the Tier 1 assessment.

Data Needs Assessment – This last section of the VCCEPP Tier 1 assessment addresses USEPA's specific requirement for chemical sponsors to specify the additional hazard and/or exposure information that may be needed to further evaluate and understand the potential risks to children and, where relevant, prospective parents.

2.0 Hazard Assessment

A toxicology profile of the commercial pentaBDE product, including abbreviated summaries of the available toxicology studies conducted to date, is presented in this section. An IUCLID data set prepared by GLCC containing hazard summaries describing each of the available relevant toxicology studies is included in Appendix III. The basis for USEPA's current non-cancer RfD for the commercial pentaBDE product is included in Appendix IV. This section concludes with a summary of available toxicity reference values used in the exposure assessment for comparison to theoretical daily intake values and calculation of non-cancer hazard indices for various toxicity endpoints of concern.

2.1 Toxicokinetic Profile

Absorption and Metabolism

The few experimental data addressing bioavailability of the commercial pentaBDE product in the scientific literature focus almost entirely on the oral route of exposure using rodents; inhalation and dermal bioavailability studies are not reported in the literature. Further, the majority of the limited available data focus on uptake, absorption rate, and distribution of specific congeners. With the exception of a single study of Great Lakes DE-71™ (a commercial pentaBDE product), information on the bioavailability of any commercial pentaBDE is lacking.

The available data on absorption, which are summarized in Tables 2-1 and 2-2, suggest significant differences based on the degree of bromination. Increasing the number of bromine atoms/molecule from 4 bromines in tetraBDE to 10 in decaBDE results in significantly reduced oral absorption, a shortened half-life, and a substantial increase in the percent of the total administered dose eliminated in feces and urine (Hardy, 2002a). Studies by Örn and Klasson-Wehler (1998) and Hakk et al. (1999) highlight the differences between lower and higher BDEs.

In a study conducted by Örn and Klasson-Wehler (1998), ^{14}C-labeled 2, 2', 4, 4'-tetraBDE (BDE-47) was used to measure absorption and elimination in rats and mice. Animals were dosed orally with 30 µmol/kg of the tetraBDE dissolved in corn oil and their urine and feces collected daily for five days. Thereafter, the animals were sacrificed and their organs analyzed for chemical content. Örn and Klasson-Wehler (1998) reported that the rat excreted 14% of the ^{14}C-labeled tetraBDE via feces and <0.5% via urine. Excretion rates in mice were higher; 20% of the ^{14}C-labeled tetraBDE was excreted via feces and 33% via urine. If the amount of parent compound excreted at the end of the first day is assumed to be the non-absorbed dose, the rats and mice excreted about 5% and 7%, respectively, of the total administered oral dose (Örn and Klasson-Wehler, 1998). Without taking into consideration metabolism and other biochemical changes in the parent compound, the percent of the parent compound retained in rat and mice tissue after five days was 86% and 46%, respectively (Tables 2-1 and 2-2).

A similar study was conducted in Sprague-Dawley rats using ^{14}C-labeled 2,2',4,4',5-pentaBDE (BDE-99; Hakk et al., 1999). Animals were administered an oral dose of 2.2 mg/rat dissolved in peanut oil and their urine and feces were collected daily for 3 days. The major excretory pathway for the pentaBDE was through the feces. At the end of three days, 43% of the administered dose was excreted (Tables 2-1 and 2-2).

Table 2-1. Percent of BDE-209 administered dose of a tetraBDE (BDE-47), a pentaBDE (BDE-99), decaBDE (BDE-209), the commercial pentaBDE product (Great Lakes DE-71™), and the commercial octaBDE product (Great Lakes DE-79™) excreted in the feces of rodents

Animal Species	PBDE Formulation	Route of Admini-stration	Dose (µmol/kg bw)	Time (days)	BDE Congener Evaluated	% Excretion of Parent	% of Dose Retained	Refe-rence
SD Rat	BDE-47 in corn oil	Oral	30	5	BDE-47	9	85.5	Klasson-Wehler et al. (2001)
SD Rat	BDE-47 in corn oil	Oral	30	1 2 3 4 5	BDE-47	5.7 5.4 1.2 0.9 0.5	94.3 88.9 87.7 86.8 86.3	Örn and Klasson-Wehler (1998)
SD Rat	BDE-99 in corn oil	Oral	15	1 3 12	BDE-99 (mean)	18 26 65	82 74 35	Klasson-Wehler et al. (2001)
SD Rat	BDE-99 in peanut oil	Oral	2.2 mg/rat	1 2 3	BDE-99	22.3 14.8 6	77.3 62.2 56**	Hakk et al. (1999)
SD Rat	DE-71 in peanut oil	Oral	672 ng/rat or 2.9 ppb	21	BDE-47 BDE-85 BDE-99 BDE-100 BDE-153 BDE-154	7.6 55.8 13.5 8.7 12.5 15.8	59.1 11.7 40.2 56.7 42.2 56.5	Hakk et al. (1999)
SD Rat	DE-79 in peanut oil	Oral	33 ng/day/rat (or 3 ppb)	21	BDE-153 BDE-154 BDE-183 BDE-190 Hepta Octa Octa Octa	15.9 4.9 27.1 20.9 31.5 38.2 16.7 44.3	19.6 76.1 37.2 52.1 31.4 39 83.3 39.5	Huwe et al. (2002a)
SD Rat	DecaBDE in Lutrol F-127, soya phospholipid, water	Oral	3	7	DecaBDE	22	78	Klasson-Wehler et al. (2001)
Mouse*	BDE-47 in corn oil	Oral	30	1 2 3 4 5	BDE-47	7.6 6.4 2.2 2.2 1.7	92.4 86 83.8 81.6 79.9	Örn and Klasson-Wehler (1998)

SD Rat = Male Sprague-Dawley Rat
* Male C57B1 Mice used
** This value was calculated based on the reported percentages excreted for days 1 and 2 in the study. However, the authors reported this value as 43% (Hakk et al., 1999)

Table 2-2. Percent of the administered dose of a tetraBDE (BDE-47), a pentaBDE (BDE-99) and decaBDE (BDE-209) excreted in the urine of rodents

Animal Species	PBDE Formulation	Route of Administration	Dose (µmol/kg bw)	Time (days)	BDE Congener Evaluated	% Excreted of Parent Compound	% of Dose Retained	Reference
SD Rat	BDE-47 in corn oil	Oral	30	--	BDE-47	<0.5	85.5	Klasson-Wehler et al. (2001)
SD Rat	BDE-47 in corn oil	Oral	30	5	BDE-47	<0.5	86.3	Örn and Klasson-Wehler (1998)
SD Rat	BDE-99 in corn oil	Oral	15	--	BDE-99	<0.9	--	Klasson-Wehler et al. (2001)
SD Rat	BDE-99 in peanut oil	Oral	2.2 mg/rat	1 2 3	BDE-99	0.4 0.3 0.2	77.3 62.2 56	Hakk et al. (1999)
SD Rat	DecaBDE in Lutrol F-127, soya phospholipid, water	Oral	3	--	DecaBDE	<0.5	--	Klasson-Wehler et al. (2001)
Mouse*	BDE-47 in corn oil	Oral	30	5	BDE-47	33	46.8	Örn and Klasson-Wehler (1998)

SD Rat = Male Sprague-Dawley Rat
* Male C57B1 Mice used
-- Not Reported

Hakk et al. (2001a) also reported the results of a feeding study in which male Sprague-Dawley rats were administered 672 ng/rat (rats weighed between 258-288 grams) of the commercial product Great Lakes DE-71™ in peanut oil for 21 days. The total concentration fed to rats was approximately 2.9 ppb, which Hakk et al. (2001a) believed to be representative of ambient environmental levels. Examining only the feces, livers and carcasses, Hakk et al. (2001a) reported that pentaBDE (BDE-99) and hexaBDE (BDE-153) were the most abundant congeners in the liver and carcass (see Table 2-5). Another pentaBDE (BDE-85) was highest in the feces, representing about 56% of the total administered dose (Tables 2-1 and 2-2). Hakk et al. (2001a) concluded that this study did not indicate any significant differences between the bioavailability and accumulation of lower brominated congeners, which are inferred from the study to refer to pentaBDE congeners and lower brominated congeners.

Huwe et al. (2002a) conducted a comparable study in which male Sprague-Dawley rats were fed 33 ng/day/rat (rats weighed between 250-300 grams) of the commercial product Great Lakes DE-79™ in peanut oil for 21 days. Animals were sacrificed 24 hours after the final feeding and their feces, liver, and carcass were analyzed. Huwe et al. (2002a) reported hexaBDE (BDE-153) to be the dominant congener detected (see also Table 2-5). Feces excretion accounted for 20-40% of the dosed congeners with the exception of another hexaBDE (BDE-154) and one of the octaBDE congeners detected. These concentrations were near background levels and their percent recoveries ranged from 48-80%. Huwe et al. (2002a) believed that these recoveries indicated that

higher BDE congeners were metabolized. However, they did not find any evidence to indicate conversion to lower brominated congeners.

Klasson-Wehler et al. (2001) performed an experiment in which male Sprague-Dawley rats were separated into 3 groups and administered a dose of 30 µmol/kg body weight of a tetraBDE (BDE-47), 15 µmol/kg body weight of a pentaBDE (BDE-99), or 3 µmol/kg body weight of a decaBDE (BDE-209). Both urine and feces were collected every 24 hours. Klasson-Wehler et al. (2001) reported that 9% of the tetraBDE was collected from the feces after 5 days. Additionally, an average of 65% of the oral dose was excreted after 12 days as the pentaBDE (26% after 3 days, and 18% during the first 24 hours). By comparison, after 7 days, 22% of the administered decaBDE was excreted through the feces (Tables 2-1 and 2-2).

The only experimental study addressing dermal bioavailability is an unpublished GLP study reported by GLCC (Inveresk Research, 2001). In this study, human and rat dermis were examined for their ability to absorb tetraBDE using an *in vitro* tissue diffusion chamber system. This system allowed the skin samples to be maintained at a constant physiological temperature of 31.9 to 32.8°C and a dose of 10 mg/cm^2/day was administered. The receptor chamber for collection of absorbed material had a flow rate of 1.5 mL/hr, samples were collected hourly (0-6 hours post dose) and then at 2 hour fractions from 6-24 hours post dose. In the rat dermis model, the mean total unabsorbed material of applied dose was 82.27 %, and the mean dermal delivery of applied dose was 17.94%. The mean absorbed dose was 14.81 %. In the human dermis, the mean total unabsorbed material of applied dose was 97.56 %, the mean dermal delivery of applied dose was 3.13%, and the mean absorbed dose was 1.94%. The discrepancy between the permeability of rat dermis and human dermis was expected because rat dermis is typically 7 to 10 times more permeable than human dermis (Inveresk Research, 2001).

Elimination Half-Life

Limited data are available in the scientific literature on elimination of the various congeners in the commercial pentaBDE product. A study by von Meyerinck et al. (1990) tested the components of Bromkal 70™, a commercial pentaBDE product that is no longer produced in the United States but may be similar to Great Lakes DE-71™. Although the specific congeners were not identified, seven congeners in the mixture were detected by gas chromatography-mass spectrometry (GC-MS) analysis. The mixture was found to contain one tetra-, three penta-, and three hexaBDE congeners. The results demonstrated that the degree of bromination directly influences the rate of elimination. Lower brominated compounds, such as tetraBDE, were eliminated more quickly than higher brominated compounds, such as hexaBDE. In addition, there was no significant variation in elimination between male and female rats for penta- and hexaBDE congeners. However, there was a difference in the elimination rate between male and female rodents for tetraBDE, with females having faster elimination. The elimination rates for the various BDE congeners are presented in Table 2-3.

Table 2-3. Elimination rates for various BDE congeners found in Bromkal 70™ [1]

Component of Bromkal 70™	Half-Life in female rats (days)	Half-Life in male rats (days)
tetraBDE	29.9	19.1
pentaBDE	47.4	36.8
pentaBDE	25.4	24.9
penta- and hexaBDE	44.6	55.1
hexaBDE	90.9	119.1

[1] von Meyerinck et al., 1990.

In addition to bioassays, scientists have used biological models to predict the uptake and half-life of BDEs based on chemical structure. For example, Palm et al. (2002) used Estimation Program Interface for Windows (EPIWIN) software to estimate the half-lives of congeners BDE-47 (a tetraBDE), BDE- 99 (a pentaBDE), and BDE-209 (decaBDE) in various environmental media. EPIWIN predicts physical, chemical, and environmental fate properties of a chemical. Palm et al. (2002) reported the predicted half-life of the tetraBDE (BDE-47) to be around 11 days; a longer half-life, 318 days, was predicted for decaBDE (BDE- 209). The predicted half-life of the tetraBDE and pentaBDE (BDE-47 and BDE-99) in water, soil, and sediment was about the same (150 days); the half-life of decaBDE (BDE-209) was approximately 4-times longer (600 days).

Distribution in Biological Tissues

Only five studies describe the distribution of BDE congeners in different biological tissues in rodents exposed orally to the commercial octaBDE product, commercial pentaBDE product(s) and one of the congeners, a pentaBDE (BDE-99). The results are presented in Tables 2-4 and 2-5.

Örn and Klasson-Wehler (1998) reported that after 5 days, 86% of a tetraBDE (BDE-47) was retained in the adipose tissue of male Sprague-Dawley rats, and 47% of the tetraBDE was retained in the adipose tissue of male C57B1 mice (see Table 2-4). Similar results were reported by Klasson-Wehler et al. (2001) in a study where male Sprague-Dawley rats were fed a combination of 30 µmol/kg bw of a tetraBDE (BDE-47), 15 µmol/kg bw of a pentaBDE (BDE-99), and 3 µmol/kg bw of decaBDE.

Hakk et al. (1999) reported that a pentaBDE (BDE-99) preferentially accumulated in the carcass, gastrointestinal tract, adipose tissue and blood of male Sprague-Dawley rats after 3 days. Hakk et al. (2001a) reported that on average, after 21 days, 0.3-1.2% of dose of commercial pentaBDE product, Great Lakes DE-71^{TM}, remained in the liver and 17.9-42.7% remained in the carcass. Furthermore, Huwe et al. (2002a) reported that of the dose administered of the commercial octaBDE product, Great Lakes DE-79^{TM}, 0.9-1.7% was detected in the liver and 16.2-62.9% was detected in the carcass, also after 21 days.

In a study reported by Hakk et al. (2001b), 672 ng/rat of the commercial product Great Lakes DE-73 (sic.)[2] was fed to male Sprague-Dawley rats (weighing between 258-288 grams) in peanut oil for 21 days, and only the livers and carcasses were analyzed. However, the total concentration fed to the rats was equal to approximately 2.9 ppb, which Hakk et al. (2001b) believed to be representative of ambient environmental levels. Hakk et al. (2001b) suspected that most of the congeners that were consumed, but not detected, were eliminated through the feces. Retention of the most prevalent BDE congeners ranged from 0.15-0.59 % of the administered dose in the liver and 11-56% in the carcass. A tetraBDE (BDE-47) had the greatest bioaccumulation in both tissues; 0.69% in the liver and 55.9% in the carcass. Hakk et al. (2001b) reported that the retention of the BDE congeners in the tissues seemed to decrease with increasing bromination, except for a pentaBDE (BDE-85) (Table 2-5).

Table 2-4. Congener concentration in nmol/g lipid weight in different tissues of male Sprague-Dawley rats [1]

Tissue	BDE-47 nmol/g lw	BDE-99 nmol/g lw	DecaBDE nmol/g lw
Adrenal	ND	157	4.4
Adipose Tissue	693	116	0.17
Kidney	203	19	1.9
Liver	128	224	13.9
Lungs	134	80.5	2

ND = Not Determined

[1] Dose was orally 30 µmol/kg body weight (bw) of BDE-47, 15 µmol/kg bw of BDE-99, and 3 µmol/kg bw of BDE-209 in corn oil for a maximum of 12 days (Klasson-Wehler et al., 2001).

[2] It is necessary to note that according to GLCC (personal communication), the commercial product Great Lakes DE-73 has never been produced and does not exist; it is not known whether Hakk et al. (2001b) are aware of a printing error in their study.

Table 2-5. The percentage of various BDEs in different tissues of male Sprague-Dawley rats[1]

Tissue	Hakk et al., 1999	Hakk et al., 2001a						Hakk et al., 2001b					
		Great Lakes DE-71™						Great Lakes DE-73					
	% of BDE-99	% of BDE-47	% of BDE-85	% of BDE-99	% of BDE-100	% of BDE-153	% of BDE-154	% of BDE-47	% of BDE-85	% of BDE-99	% of BDE-100	% of BDE-153	% of BDE-154
Adrenals	0.1	--	--	--	--	--	--	--	--	--	--	--	--
Adipose Tissue	3.8	--	--	--	--	--	--	--	--	--	--	--	--
Blood	1.4	--	--	--	--	--	--	--	--	--	--	--	--
Carcass	38.8	32.6	32	45.2	33.9	44.1	27.4	55.9	10.62	30.6	56.09	49.78	21.07
G.I. tract	6.1	--	--	--	--	--	--	--	--	--	--	--	--
Heart	0.03	--	--	--	--	--	--	--	--	--	--	--	--
Kidney	0.1	--	--	--	--	--	--	--	--	--	--	--	--
Liver	0.9	0.7	0.6	1	0.7	1.2	0.4	0.69	0.15	0.46	0.4	0.25	0.16
Lungs	0.1	--	--	--	--	--	--	--	--	--	--	--	--
Testes	0.06	--	--	--	--	--	--	--	--	--	--	--	--
Thymus	0.06	--	--	--	--	--	--	--	--	--	--	--	--
Total % retained	***51.45***	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

-- = the tissue was not analyzed

N/A = Not Applicable. Because the carcass and the liver were the only tissues analyzed, it is not appropriate to calculate the total percent retained in the male Sprague-Dawley rats.

[1] Dose was orally 2.2 mg/rat of BDE-99 in peanut oil for three days; 672 ng/rat of commercial pentaBDE product DE-71 for 21 days; and 672 ng/rat of commercial pentaBDE product DE-73 for 21 days.

2.2 Toxicology Profile

2.2.1 Systemic Toxicity

The acute, subacute, subchronic, and chronic effects of the commercial pentaBDE product via oral and inhalation pathways have been evaluated and summarized in the following sections.

Acute Studies

The acute effects of pentaBDE have been well characterized in several studies. The acute effects via inhalation exposure of pentaBDE have been evaluated at only two concentrations. The acute oral and inhalation effects of pentaBDE are summarized below with details provided in Tables 2-6 and 2-7.

Oral

Studies evaluating the potential acute oral effects of pentaBDE have been conducted in both male and female rats (British Industrial Research Association, 1977; IRDC, 1975; Pharmakon Research International, Inc., 1984) and in female mice (Fowles et al., 1994). The 44-day LD_{50} in male and female rats was determined to be 7,400 and 5,800 mg/kg, respectively (British Industrial Research Association, 1977). Pharmakon Research International Inc. (1984, as reported in ATSDR, 2002) reported a 14-day LD_{50} of 5,000 mg/kg. IRDC (1975) concluded that pentaBDE should be considered "toxic, but not highly toxic" based on the death of 4 of 5 rats at a dose of 5,000 mg/kg. In mice, no effects on serum T_4 or liver cytochrome P450 content were reported following single oral doses in mice ranging from 0.8 to 500 mg/kg (Fowles et al.,1994).

Inhalation

Only one study was located that evaluated the potential acute effects of pentaBDE following inhalation exposure (IRDC, 1975). The study exposed male and female rats to 2,000 or 200,000 mg/m^3 pentaBDE mixed with corn oil in the form of a mist for 1 hour in a closed glass chamber. The rats were examined immediately following exposure and at 14 days after exposure. All animals in both dose groups survived the 14 day observation period. At the end of the 14-day study period, all rats in both dose groups appeared normal and exhibited normal body weight gains.

Table 2-6. Summary of Acute Oral Studies with the commercial pentaBDE product.

Species/ Sex/ Strain	Duration/ Dosing method	Doses (mg/kg)	Endpoints Evaluated	NOAEL (mg/kg)	LOAEL (mg/kg)	Reference
Rats/male/ Wistar; 5 animals per dose group	1day/ intra-esophageal intubation	2400, 4800, 6043, 7021, 9600	Weight gain; mortality	--	7400 (44-day LD_{50})	British Industrial Biological Research Association (1977)
Rats/female/ Wistar; 5 animals per dose group	1day/ intra-esophageal intubation	2400, 4800, 6043, 7021, 9600	Weight gain; mortality	--	5800 (44-day LD_{50})	British Industrial Biological Research Association (1977)
Rats/male/ Charles River CD (Sprague Dawley); 5 animals per dose group	1day/gavage	50, 500, 5000	Body weight; mortality	--	5000 (4 of 5 died)	IRDC (1975)
Rat/Sprague Dawley; Number of animals unknown	1day/gavage	--	Mortality	--	5000 (14-day LD_{50})	Pharmakon Research International Inc. (1984) (as reported in ATSDR 2002)
Mice/female C57BL/6; 6 animals per dose group	1day/gavage	0, 0.8, 4, 20, 100, 500	Serum total T_4; Liver (P450) enzyme content	500	--	Fowles et al. (1994)

Table 2-7. Summary of Acute Inhalation Studies with the commercial pentaBDE product

Species/ Sex/ Strain	Duration/ Dosing method	Doses (mg/m^3)	Endpoints Evaluated	NOAEL (mg/m^3)	LOAEL (mg/m^3)	Reference
Rats/male and female/ Charles River CD (Sprague-Dawley); 10 animals per sex per dose group	1 hour	2,000, 200,000	Weight gain; clinical signs; mortality	200,000	--	IRDC (1975)

Subacute and Subchronic Studies

Oral

The toxicity of the commercial pentaBDE product has been evaluated in longer term repeated dose studies (e.g., 14 to 90 days). These studies are summarized below with details provided in Table 2-8.

In a study designed to examine thyroid hormone concentrations and hepatic enzyme activity, Zhou et al. (2001) administered Great Lakes DE-71™ (a mixture of pentaBDE, octaBDE, hexaBDE, and tetraBDE) orally to female rats at doses of 0, 0.3, 1, 3, 10, 30, 100, or 300 mg/kg/day for 4 days. A dose-dependent decrease in serum T_4 levels was seen with significant decreases reported at doses of 10 mg/kg/day and higher. An increase in hepatic microsomal ethoxy-resorufin-O-deethylase (EROD), pentoxyoxy-resorufin-O-deethylase (PROD), and uridinediphospahte-glucuronosyltransferase (UDPGT) activity was also reported, which was statistically significant at 10 mg/kg/day for PROD activity, and 30 mg/kg/day for EROD and UDPGT activity. The NOAEL for this study was 3 mg/kg/day, based on the decreased T_4 levels.

Two 14-day oral studies were available that evaluated the potential subacute toxic effects of pentaBDE in rats (Carlson, 1980a) and mice (Fowles et al., 1994). In rats, pentaBDE was administered at a concentration of 0.1 mmol/kg/day (approximately 56 mg/kg/day) in corn oil for 14 days (Carlson, 1980a). PentaBDE exposure produced liver enlargement and increases in NADPH cytochrome C reductase, cytochrome P-450, EPN detoxification, UDPGT, and benzo(a)pyrene hydroxylase. Sorbitol dehydrogenase levels, an indicator of liver damage, were not increased. Exposure to pentaBDE induced xenobiotic metabolism and the large increase in EPN detoxification, p-nitroanisole demethylation, and NADPH cytochrome c reductase activity accompanied by small increases in UDPGT and benzo(a)pyrene hydroxylase resembled phenobarbital-like induction. In the absence of microscopic changes, induced xenobiotic metabolism was not considered an adverse affect. Thus, the NOAEL for this study was 56 mg/kg/day.

Fowles et al. (1994) treated mice with pentaBDE at doses ranging from 18 to 72 mg/kg/day for 14 days. Total serum T_4 and liver cytochrome P450 (EROD and PROD) content were measured at the end of the exposure period. PentaBDE produced significant decreases in total and free serum T_4 in all dose groups. Liver to body weight ratios were significantly increased at both doses, when compared with controls, and thymus weight was significantly decreased in the high-dose group. Total liver cytochrome P450 and EROD and PROD metabolism were significantly increased in both dose groups; however, in the absence of microscopic changes in the liver, these results were not considered adverse. The LOAEL for this study was 18 mg/kg/day, based on the changes in T_4 levels.

Enzyme induction was not evaluated in a 28-day subacute dietary toxicity study (IRDC, 1976); however, other observations consistent with enzyme inducted were noted. There were increases in absolute and relative liver weights and increased liver cell size reported in male and female rats that received doses of approximately 78 or 83 mg/kg/day, respectively. At lower doses (approximately 8 mg/kg/day), a significant increase in relative liver weight in female rats and an increased incidence of increased liver cell size in male rats was reported. In the high-dose group, relative liver weights were significantly increased in both males and females. Thyroid hyperplasia was observed in 3 out of 5 of the high-dose male rats, but was not observed in any of the high-dose female rats. No other treatment related histopathological findings were reported. NOAELs for this study were 8 and 83 mg/kg/day for males and females, respectively.

In a 28-day study conducted by WIL (1984), significant decreases in body weights, accompanied by significant decreases in food consumption, were reported in male and female rats that received 100 mg/kg/day of pentaBDE in the diet. Absolute and relative liver weights were increased in the high-dose (100 mg/kg/day) males and females. Microscopic changes observed in the liver consisted of hepatocytomegaly. There were no increases in clinical chemistry enzymes indicative of hepatocellular damage. T_4 levels were significantly decreased in the mid- (10 mg/kg/day) and high-dose males and in the mid-dose females. Thyroid hyperplasia was observed in 7 out of 10 males and females in the high-dose group, and in 1 out of 10 males and 4 out of 10 females in the mid-dose group. In the low-dose group (2 mg/kg/day), 2 out of 10 males and 2 out of 10 females had thyroid hyperplasia. This effect was not observed in the control male and female rats. NOAELs for this study were 10 mg/kg/day for males and 2 mg/kg/day for females.

In a 30-day dietary study, liver weights in treated rats were comparable to controls and there were no microscopic changes attributable to treatment in the liver or thyroid gland (the only organs examined) at the highest dose tested (1 mg/kg/day) (WIL, 1985). There were no treatment-related effects on liver and urine porphyrins. The NOAEL for this study was 1 mg/kg/day.

Two 90-day studies have been conducted with pentaBDE (Carlson, 1980b; WIL, 1984). The study by Carlson (1980b) evaluated the potential for pentaBDE to induce metabolism. Exposures to pentaBDE induced metabolism, based on increases in EPN detoxification, p-nitroanisole demethylation, cytochrome c reductase, and cytochrome P-450 protein in all dose groups. These effects were persistent and remained at 30 or 60 days after cessation of dosing. However, there were no microscopic changes reported in the livers of rats that received the low-dose series of 0, 0.44, 0.88, or 1.77 mg/kg/day (the livers in the high-dose groups were not examined microscopically); therefore, these effects were not considered adverse. The NOAEL for this study was 1.77 mg/kg/day.

Enzyme induction was not evaluated in a 90-day dietary conducted by WIL (1984); however, other endpoints typically evaluated in a subchronic toxicity study were evaluated. The findings in this study included an increase in absolute and relative liver weights in the mid- (10 mg/kg/day) and high-dose (100 mg/kg/day) males and females, and an increase in the incidence of

hepatocytomegaly in male rats in the low- (2 mg/kg/day), mid- and high-dose groups, and in female rats in the mid- and high-dose groups. Liver effects (very slight or slight hepatocyte degeneration and necrosis) were also reported after a 24-week recovery period in female rats in the 2 and 100 mg/kg/day dose groups. However, statistical significance was not achieved and the severity of this change in the treated groups was comparable to the severity in the control group. Thyroid hyperplasia was observed in 5 out of 10 high-dose male and 4 out of 10 high-dose female rats. This effect was transient and had resolved after 24-weeks of recovery. Significant decreases in T_4 were reported in the mid-dose males at week 13; however, T_4 levels in the high-dose group were not significantly different from the controls. T_3 levels were not affected. A NOAEL for this study was 10 mg/kg/day for females and 100 mg/kg/day for males.

Summary of liver and thyroid effects

Effects have been observed in the liver and the thyroid gland in rats exposed to pentaBDE. In the liver, the only effect noted was enzyme induction along with observations characteristic of adaptive responses to enzyme induction to include increased liver weights and microscopic changes, such as increased size of the hepatocytes (hepatocytomegaly). However, enzymes indicative of hepatocellular damage were not increased following treatment with pentaBDE (Carlson, 1980b; WIL, 1984).

The pattern of enzyme induction in the Carlson (1980a,b) studies in rats is suggestive of a phenobarbital-like metabolic induction pattern. The large increases in three enzyme markers (EPN detoxification, p-nitroanisole demethylation, and NADPH cytochrome c reductase) compared to two other enzymes (UDPGT and benzo[a]pyrene hydroxylase) resemble phenobarbital-like induction (Carlson, 1980a). Further, induction of PROD occurred at lower doses than either EROD or UDPGT (Zhou et al., 2001). However, induction of both PROD and EROD is suggestive of a mixed-type inducer.

Other systemic toxicity studies conducted with pentaBDE did not evaluate enzyme induction. However, liver weights and microscopic changes in the liver were evaluated. Significant increases in liver weights have been reported in rats that received doses of approximately 80 mg/kg/day pentaBDE or greater for 28 days (IRDC, 1976; WIL, 1984) or 10 mg/kg/day for 90 days (WIL, 1984). The primary microscopic change observed was hepatoctomegaly (IRDC, 1976; WIL, 1984), or enlargement of hepatocytes, which is consistent with an adaptive response to enzyme induction (Popp and Cattley, 1991).

Following exposures to pentaBDE, changes in thyroid hormone levels and microscopic changes (thyroid hyperplasia) in the thyroid gland have been reported. T_4 levels were significantly decreased in rats that received 10 mg/kg/day of pentaBDE for 4 days (Zhou et al., 2001). This effect was not observed at lower doses (e.g., 3 mg/kg/day or less). In mice, significant decreases in T_4 levels were reported at doses of 18 mg/kg/day, the lowest dose tested, and greater after 14 days of treatment (Fowles et al., 1994). When DE-71 was administered to pregnant female rats from gestation day (GD) 6 through postnatal day (PND) 21, T_4 was significantly decreased in the

dams in the high-dose group (30 mg/kg/day) at GD 20 and PND 22 and in the offspring of dams that received 10 or 30 mg/kg/day at PND 4 and 14 (Zhou et al., 2002). Decreased T_4 levels were reported in male rats that received 10 or 100 mg/k/day pentaBDE for 28 days, but not in rats that received 2 mg/kg/day (WIL, 1984). After 90 days of dosing, T_4 levels were significantly decreased in male rats that received 10 mg/kg/day or pentaBDE, but not in male rats that received 100 mg/kg/day or in female rats that received doses of ≤ 100 mg/kg/day (WIL, 1984). T_3 levels were not affected at doses up to 300 mg/kg/day for 4 days (Zhou et al., 2001) or 100 mg/kg/day for 90 days (WIL, 1984).

The other thyroid effect, thyroid hyperplasia, was observed in 3/5 male rats that received approximately 80 mg/kg/day pentaBDE for 28 days, but not in male rats that received approximately 8 mg/kg/day or in female rats that received up to approximately 80 mg/kg/day (IRDC, 1976). Thyroid hyperplasia was also seen only in male rats that received 100 mg/kg/day males for 28 days and in female rats that received 10 or 100 mg/kg/day (WIL, 1984). After 90 days of dosing, thyroid hyperplasia was reported in male and female rats that received 100 mg/kg/day (WIL, 1984). In lower, dose groups (2 or 10 mg/kg/day) the incidence of thyroid hyperplasia was not significantly increased after 90 days of dosing (WIL, 1984).

Collectively, the results of the studies that evaluated the potential effects of pentaBDE suggest there is an initial effect on T_4 levels followed by compensation and recovery. For example, in the 4-day (Zhou et al., 2001) and 28-day (WIL, 1984) studies, T_4 levels were decreased at doses of approximately 19 mg/kg/day and greater. However, after 90 days of dosing, T_4 levels were not significantly decreased in male or female rat that received 200 mg/kg/day. These data suggest that over longer periods of exposure the rat is able to compensate and the effects on T_4 levels do not worsen and are not observed at lower doses with increasing exposure duration. Thyroid hyperplasia followed a similar pattern. Thyroid hyperplasia was observed at higher doses (e.g., 80 mg/kg/day or greater) in 28- and 90-day studies, but not at lower doses (e.g., 10 mg/kg/day) after 90 days of exposure. Thus, as with the changes in T_4 levels, thyroid hyperplasia did not increase at lower doses following longer durations of exposure.

Table 2-8. Summary of Subacute and Subchronic Oral Studies with the commercial pentaBDE product

Species/ Sex/ Strain	Protocol	Endpoints Evaluated	Dose (mg/kg/day)	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Reference
Rats/ female/ Long-Evans; 8 animals per dose group except in low dose group which had 4 animals.	Great Lakes DE-71™ was administered orally once a day for 4 days	T_3, T_4; EROD; PROD; UDPGT	0, 0.3, 1, 3, 10, 30, 100, 300	3	10 (decreased T_4)	Zhou et al. (2001)
Rats/male/ Sprague-Dawley; 4 animals	A commercial mixture of pentaBDE was administered orally once a day for 14 days	*O*-ethyl *O*-*p*-nitrophenyl phenylphosphonothioate (EPN) detoxification; p-nitroanisol demethylation; NADPH-cytochrome c reductase; cytochrome P-450; liver weight; UDP-glucuronyltransferase; benzo(a)pyrene hydroxylase; serum sorbitol dehydrogenase	0.1 mmol/kg/day (approximately 56 mg/kg/day based on a molecular weight of 564 g/mol)	56	--	Carlson (1980a)
Mice/female/ C57BL/6J; 6-8 mice per dose group	Great Lakes DE-71™ was administered orally once a day for 14 days	Total serum T_4; liver Cytochrome P450 (EROD and PROD)	0, 18, 36, 72	--	18 (decreased T_4)	Fowles et al. (1994)
Rat/male and female/Charles River CD (Sprague Dawley; 10 per sex per dose group	PentaBDE was administered in the diet for 28 days	Food consumption, body weight, organ weights, liver bromine analysis, macroscopic pathology and microscopic pathology (liver, kidneys and thyroid)	0, 100 or 1000 ppm (approximately 0, 8 or 78 mg/kg/day in males and 0, 8 or 83 mg/kg/day in females	8 (males) 83 (females)	78 (males) Increased liver cell size; increased incidence of thyroid hyperplasia	IRDC (1976)

Species/ Sex/ Strain	Protocol	Endpoints Evaluated	Dose (mg/kg/day)	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Reference
Rat/male and female/ Sprague Dawley; 10 animals per sex per group	Great Lakes DE-71™ administered via the diet for 28 days	Food consumption, body weights, hematology, clinical chemistry, T_3 and T_4 levels, tissue bromine levels and gross and microscopic pathology	0, 2, 10 or 100	2	10 Decreased T_4 and Increased incidence of thyroid hyperplasia	WIL (1984)
Rat/male and female/ Sprague Dawley; 20 per sex per group	PentaBDE was administered in the diet for 30 days with recovery periods of 6, 12 or 24 weeks.	Food consumption, body weight, GGT, cholesterol, serum bromide levels, tissue bromine levels, liver and urine porphyrins, organ weights, macroscopic pathology and microscopic pathology (liver and thyroid)	0, 0.01, 0.05, 0.1, 0.5 or 1	1	--	WIL (1985)
Rat/male/ Sprague Dawley; number of animals per group not specified	Commercial mixture of pentaBDE was administered via corn oil gavage for 13 weeks with sacrifices at 1, 30 or 60 days after the last dose.	EPN detoxification, p-nitroanisole demethylation, cytochrome c reductase, and cytochrome P-450 protein	Low dose series : 0, 0.44, 0.88, or 1.77 High dose series 0, 5.01, 10.02, or 20.04	1.77[1]	--[1]	Carlson (1980b)
Rat/male and female/ Sprague Dawley; 30 animals per sex per group	Great Lakes DE-71™ administered via the diet for 90 days with sacrifices at weeks 4, 13, 19 and 37 weeks	Food consumption, body weights, hematology, clinical chemistry, T_3 and T_4 levels, tissue bromine levels and gross and microscopic pathology	0, 2, 10 or 100	10 (females) 100 (males)	100 (females – liver degeneration and necrosis after 24 weeks of recovery)	WIL (1984)

[1]– In this study groups of Sprague Dawley rats were administered a high dose series and a low dose series. Enzyme induction was increased in all dose groups in both series; however, in the derivation of the RfD for pentaBDE, the EPA did not consider enzyme induction an adverse effect in the absence of microscopic changes. There were no microscopic changes in the livers of the rats in the low dose series (the high dose series was not examined microscopically). Consequently, the NOAEL for this study was 1.77 mg/kg/day (the highest dose tested in the low dose series) and a LOAEL was not determined.

2.2.2 Genotoxicity Studies

PentaBDE was not mutagenic when tested in reverse mutation assays by the overlay, agar incorporation, and preincubation methods (Litton Bionetics, 1976, 1980; Zeiger et al., 1987) (Table 2-9). Several strains of *Salmonella typhimurium* and the D4 stain of the *Saccharomyces cerevisiae* yeast were tested. These microorganisms are sensitive to compounds that induce mutagenic effects, such as frame shifts or base-pair substitutions. In each of these assays pentaBDE was tested up to concentrations that induced cytotoxicity. Further, in order to evaluate the potential for mutagenic metabolites, pentaBDE was tested with and without metabolic activation.

The potential for pentaBDE to induce chromosomal aberrations was evaluated in human peripheral blood lymphocytes collected from healthy individuals with no recent history of radiotherapy, viral infections or drug therapy (Microbiological Associates, 1996). In an initial assay and in an independent repeat assay, pentaBDE did not increase the frequency of chromosomal aberrations in human peripheral blood lymphocytes *in vitro*, when compared with the negative controls (Table 2-9). In these chromosomal aberration assays, pentaBDE was tested at concentrations that induced cytotoxicity in the presence or absence of Aroclor 1254-induced rat liver metabolic activation.

Table 2-9. Summary of Genotoxicity Studies with the commercial pentaBDE product

Assay Type	Strains Tested	Protocol	Results	Reference
Reverse Mutation Assay (overlay method)	*Salmonella typhimurium* strains TA-1535, TA-1537, TA-1538, TA-98, TA-100 and *Saccharomyces cerevisiae* strain D4	0, 0.005, 0.01, 0.1, or 1 µL/plate with and without Aroclor 1254-induced rat liver S9 metabolic activation	Negative	Litton Bionetics (1976)
Reverse Mutation Assay (preincubation method)	*Salmonella typhimurium* strains TA-1535, TA-1537, TA-98, TA-100	0, 100, 333, 1000, 3333 or 10000 µg/plate with and without Aroclor 1254-induced rat and hamster liver S9 metabolic activation	Negative	Zeiger et al. (1987)
Reverse Mutation Assay (agar incorporation method)	*Salmonella typhimurium* strains TA-1535, TA-1537, TA-1538, TA-98, TA-100	0, 0.5, 1, 10, 100, 500, 1000, 2500 or 5000 µg/plate with and without Aroclor 1254-induced rat liver S9 metabolic activation	Negative	Litton Bionetics (1980)
Chromosomal Aberration Assay	Human peripheral blood lymphocytes	0, 0.75, 2.5, 7.5, 25, 75, 250, 750 or 2500 µg/ml without metabolic activation for 20 hours[1]	Negative[2]	Microbiological Associates (1996)
		0, 0.75, 2.5, 7.5, 25, 75, 250, 750 or 2500 µg/ml for 4 hours with a 16 hour recovery with Aroclor 1254-induced rat liver S9 metabolic activation	Negative[2,3]	
		0, 63, 125, 250, or 500 µg/ml without metabolic activation for 20 or 44 hours without metabolic activation[4]	Negative[2]	
		0, 313, 625, 1250, 2500, 3759 µg/ml for 4 hours with a 16 hour recovery with Aroclor 1254-induced rat liver S9 metabolic activation[4,5]	Negative[2]	
		0, 313, 625, 1250, 2500, 3759 µg/ml for 4 hours with a 40-hour recovery with Aroclor 1254-induced rat liver S9 metabolic activation metabolic activation[4]	Negative[2]	

[1] In the initial assay without metabolic activation, concentrations of 750 and 2500 µg/ml were not scored due to absence of scorable cells.

[2] The percentage of cells with structural or numerical aberrations was evaluated.

[3] In the initial assay, a significant increase in the percentage of cells with aberrations was reported at 2500 µg/ml. However, the authors concluded that because the response (4%) was only slightly outside the range for historical controls (1-3%) and was not observed in the repeat assay, this response had no biological significance.

[4] Independent repeat assay

[5] Concentrations of 2500 and 3750 µg/ml were not scored due to absence of scorable cells

2.2.3 Reproductive/Developmental Studies

Developmental toxicity of a commercial pentaBDE mixture has been evaluated in rats (DE-71: Zhou et al., 2002; Taylor et al., 2002; Gilbert and Crofton, 2002; and Saytex 115: Ethyl Corporation, 1985c)[3] and 2,2',4,4',5-pentaBDE has been tested in mice (Branchi et al., 2002) (Table 2-10). In rats, in addition to standard developmental parameters, effects on enzyme induction and thyroid hormone levels were assessed in dams sacrificed on GD20 or PND 21 or in fetuses (GD20) or off-spring on PNDs 4, 14, 36 or 90. In rats, there were no effects on maternal body weight gain, live litter size, gestation length or sex ratio in dams given DE-71 at doses up to 30 mg/kg/day from GD 6 through PND 21 (Zhou et al., 2002; Taylor et al., 2002) or in rats given Saytex 115 at a dose of 10 mg/kg/day on GDs 6 to 15 (Ethyl Corporation, 1985b). No effects on pup body weight or any measure of growth or viability were seen in the offspring of rats treated at maternal doses up to 30 mg/kg/day (Zhou et al., 2002; Taylor et al., 2002) or 200 mg/kg/day (Ethyl Corporation, 1985b). Similarly no effects on pregnancy were seen in mice (body weight gain, duration of pregnancy, pup sex ratio) given pentaBDE at doses up to 30 mg/kg/day from GD 6 through PND 21. In mice, live litter size was reduced in the mid-dose group (6 mg/kg/day) only; however, the number of dams tested was small (n = 3) and litter size was evaluated on PND 1.

Enzyme induction in the fetus and neonate has been evaluated in rats (Zhou et al., 2002), but not mice. There was an increase in liver to body weight ratios in both pregnant (GD 20) and lactating dams (PND 21) in the 30 mg/kg/day dose group consistent with increases in enzyme induction. In dams, circulating levels of EROD and PROD were significantly increased on GD 20 and PND 21, in the mid- `and high-dose groups (10 and 30 mg/kg/day), while UDPGT was significantly increased in the high-dose group only. Liver weights were not significantly increased in the dams or in fetuses, but pup liver weights were significantly increased in the mid- and high-dose groups on PND 4, in the high-dose group on PND 14, but comparable to controls on PNDs 36 and 90. The pattern of enzyme induction in offspring was similar to that seen in the dams. EROD activity was significantly increased in off-spring on GD20. EROD and PROD activity were significantly increased in the mid- and high-dose groups and UDPGT in the high-dose group PNDs 4 and 14 but returned to control levels by PND 36 (UDPGT and PROD) and PND 90 (all). Induction of PROD and UDPGT were of similar magnitude in both dams and off-spring but induction of EROD was greater in the off-spring.

The effect of DE-71 on thyroid hormone levels in the dams, fetuses and neonates has been evaluated in rats (Zhou et al., 2002; Taylor et al., 2002). Significant reductions in T_4 levels were seen in dams on GD 20 and PND 21 in the 30 mg/kg/day dose group (Zhou et al., 2002). T_4 levels were significantly reduced in the mid- and high doses in the fetus (GD 20) and in offspring (PND 4 and 14), but were comparable to controls by PND 36. Levels of T_3 were unaffected in dams and offspring at any dose tested.

Neurobehavioral activity has been evaluated in off-spring of rats (Taylor et al., 2002; Gilbert and Crofton, 2002). No age-dependent changes in motor or sensory behavior (Taylor et al., 2002) or in either long-term potentiation or synaptic transmission in the dentate gyrus of the hippocampus (Gilbert and Crofton, 2002) were noted in off-spring of dams given up to 30 mg/kg/day of DE-71 on GD 6 through PND 21. The hippocampus was evaluated because thyroid hormones are thought to regulate the development of cholinergic and dopaminergic systems in the hippocampus (McDonald, 2002). Gilbert and Crofton (2002) concluded that the modest reductions in T_4 in off-spring of dams given DE-71 up to 30 mg/kg/day were not sufficient to alter these indices of hippocampal function.

Neurobehavioral effects were evaluated in off-spring of female mice administered pentaBDE up to 30 mg/kg/day on GDs 6 to PND 21 (Branchi et al., 2002) or to neonates administered up to 12 mg/kg/day on PNDs 3, 10 or 19 (Eriksson et al., 1998, 1999, 2001, 2002; Viberg et al., 2002). Branchi et al. (2002) reported that pentaBDE had no effect on hair growth, incisor eruption, eye opening, ultrasonic vocalization, homing performance or on all but one test (of seven) in a modified Fox battery. According to Branchi et al. (2002), the main behavioral alteration found was a transient, age-dependent alteration in activity level in the open field test that ended by 4 months of age. However, an inverse dose-response was noted in that significant increases in activity on PNDs 34 and 60 occurring only in the off-spring of rats in the low dose group (0.6 mg/kg/day) and mid-dose group (6 mg/kg/day), but not the high dose group (30 mg/kg/day). No plausible explanation for the inverted dose-response was given. However, only one male and one female per litter were tested per dose group (6 to 8 animals total) and statistical analyses that considered litter-based effects and repeated measures within subjects were not applied.

Eriksson and colleagues conducted several experiments in which a single dose of pentaBDE was administered to male mice (Eriksson et al., 1998, 1999, 2001, 2002; Viberg et al., 2002). In this group of studies administration of pentaBDE resulted in significant changes in the normal habituation response in adult mice (2 or 4 months of age) treated on PND 3 (8 mg/kg/day) or PND 10 (0.8, 8, or 12 mg/kg/day), but not when treated on PND 19 (8 mg/kg/day). According to the authors, habitation capability, i.e., locomotion, rearing, and total activity that normally decreases as the novelty of the test chamber decreases, was significantly altered. In contrast to normal spontaneous behavior patterns, locomotion, rearing, and total activity were below (hypoactive) controls in the early part of the test (first 20 minutes), but above controls (hyperactive) during the third 20 minutes. The authors noted that the effect was more pronounced at 4 months than at 2 months. The results of a Morris swim maze test indicated that neonatal exposure (12 mg/kg/day on PND10) affected performance on retest, but not the initial test in 5 month old animals. The protocol and analyses were the same in all of the Eriksson studies in that 8 to 12 mice were randomly picked from 3 to 4 litters for each dose group. However, litter-based

[3] Abstracts only were available for Taylor et al. (2002) and Gilbert and Crofton (2002); only a summary in ATSDR (2002) was available of the Ethyl Corporation (1985c) study.

statistical analyses were not conducted nor repeated measures tests used (animals tested at 2 months were not re-tested at 4 months).

Table 2-10. Summary of Reproductive and Developmental Studies

Species/ Strain	Exposure/duration/ Frequency	Dose (mg/kg/day)	Endpoints	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Referen-ces
Mouse NMRI Males 8 per dose group	Single gavage dose at 10 days of age; Tested at 2, 4 and 5 (swim maze only) months of age	0, 0.8 or 12 of 2, 2',4,4',5-PentaBDE in a 20% fat emulsion vehicle	Spontaneous behavior (locomotion, rearing, total activity)	-	0.8	Eriksson et al. (1998, 2001)
			Habituation capability	-	0.8	
			Swim maze (only high dose tested)	-	12	
Mouse NMRI Males 10 per dose group	Single gavage dose at 3, 10, or 19 days of age; Tested at 4 months of age	0 or 8 of 2,2',4,4',5- PentaBDE in a 20% fat emulsion vehicle	Spontaneous behavior (locomotion, rearing, total activity)	- 8 (day19)	8 (days 3 and 10)	Eriksson et al. (1999, 2002)
Mouse NMRI Males 12 per dose group	Single gavage dose at 10 days of age needs control; Tested at 2 months of age	0 or 8 of 2,2',4,4',5- PentaBDE in a 20% fat emulsion vehicle	Cholinergic nicotinic response	-	8	Viberg et al. (2002)
			Spontaneous behavior (locomotion, rearing, total activity)	-	8	
Mouse Swiss (CD-1) Female 4 dams per dose group for low and high dose; 3 dams per dose for control and mid-dose	Daily gavage dose from gestation day 6 (GD 6) through Post-natal day 21 (PND 21); Off-spring tested	0, 0.6, 6, or 30 of 2,2',4,4',5-PentaBDE in a 20% fat emulsion vehicle	Effects on pregnancy	30	-	Branchi et al. (2002)
			Somatic and neurobehavioral development (8 males and 8 females per dose group tested every 2 days from PND 2 to 20. A modified Fox battery was used.	Screen Climbing Test 6	30	
				All other tests 30	-	
			Ultrasonic vocalizations (4 males and 4 females tested on PNDs 4, 8, and 12)	30	-	
			Homing Test (4 males and 4 females tested on PND 11)	30	-	
			Open-field test (PNDs 22, 34, 60 and 120)			

Species/ Strain	Exposure/duration/ Frequency	Dose (mg/kg/day)	Endpoints	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Referen-ces
			PND 22	30	-	
			PNDs 34, 60,	30	6	
				(inverse dose response)		
			PND 120	30	-	
Rats Long Evans Females 39 to 48 dams per dose group	Daily gavage doses from GD 6 through 21 except for PND 0; Groups of dams (GD 20 and PNDs 4, 14, 36, and 90) were evaluated; Off-spring evaluated at PNDs 4, 14, and 36.	0, 1, 10 or 30 mg/kg/day of DE-71 (a mixture of tetra- and penta- BDE)	Reproductive/ developmental parameters (gestation length, litter size at birth, sex ratio at birth, viability index)	30	-	Zhou et al. (2002)
			Dam: Body and organ weights (increase in liver weight)	10	30	
			Eye opening	30	-	
			Off-spring: Body and organ weights liver to body weight ratio on PND 4 PND 14 PND 36	 1 10 30	 10 30 -	
			Dam: Thyroid hormones (T_4)- (GD 20 and PND 22)	10	30	
			Off -spring: Thyroid hormones (T_4) Fetal	1	10	
			PND 4 and 14	1	10	
			PND 36 and 90	30	-	
			Dam and Off-spring: Thyroid: Hormones (T_3)	30	-	
Rats Long Evans	Daily, gavage doses to females on GD 6 to PND 21	0, 1, 10 or 30 mg/kg/day in corn oil	Body weight (dams and pups)	30	-	Taylor et al. (2002)
			Survival (dams and pups)	30	-	
			Eye Opening	30	-	

Species/ Strain	Exposure/duration/ Frequency	Dose (mg/kg/day)	Endpoints	NOAEL (mg/kg/day)	LOAEL (mg/kg/day)	Referen-ces
		of commercial mixture, DE-71 (a mixture of tetra-and penta- BDE)	Motor activity development	30	-	
			Auditory startle response	30	-	
			Thyroid hormones (T_4) PND 14 PND 36	 10 30	 30 -	
			Hepatic enzymes (hepatic glucuronidation) PND 14 PND 36	 10 30	 30 -	
Rats Strain not specified Females Number not given	Daily gavage doses to females from GD 0 to PND 21; Male off-spring tested at PND 14 and PND 36 for thyroid hormones; Tested male off-spring at 4 to 8 months for hippocampal effects	0 or 30 mg/kg/day in corn oil of a commercial mixture, DE-71 (a mixture of mostly tetra- and penta- BDE)	Thyroid hormones (T_4) Early post- natal PND 36	 - 30	30 -	Gilbert and Crofton (2002)
			Thyroid Hormones (T_3)	30	-	
			Long term potentiation	30	-	
			Synaptic transmission in hippocampus	30	-	
Rats Strain not specified Females 25 per dose group	Daily gavage doses on GDs 6 to 15; Dams sacrificed on GD 20 and fetuses examined	0, 10, 100 or 200 mg/kg/day of Saytex 115 (a commercial mixture containing penta-BDE)	Maternal body weight	10	100	Ethyl Corporation (1985b)
			Reproductive/developmental parameters (resorptions, litter size, fetal mortality, skeletal or soft tissue malfunctions)	200	-	
			Fetal body weight	200	-	

2.2.4 Immunotoxicity Studies

The potential effects of the commercial pentaBDE product on the immune system have been evaluated by in human lymphocytes *in vitro* (Fernlof et al., 1997) and in rats and mice (Fowles et al., 1994; Darnerud and Thuvander, 1998). In the study by Fernlof et al. (1997), there were no effects on mitogen-induced proliferation or on immunoglobulin (IgG) production in human lymphocytes that were collected from healthy women and were exposed to 1 x 10^{-9} M to 1 x 10^{-5} M of the commercial pentaBDE product *in vitro*. When the potential immunotoxicity of pentaBDE was assessed by measuring the plaque-forming cell (PFC) response to sheep erythrocytes in female mice, there were no effects on PFC response following a single gavage dose of 500 mg/kg (Fowles et al., 1994). However, when 72 mg/kg was administered for 14 days the PFC response was significantly decreased, when compared with the untreated control. Doses of 18 mg/kg/day or 36 mg/kg/day for 14 days, had no effect on the PFC response (Fowles et al.,1994). Natural killer cell activity was not altered following the administration of 0, 18, 36 or 72 mg/kg/day for 14 days (Fowles et al.,1994). Darnerud and Thuvander (1998) reported that splenocyte numbers were decreased in mice exposed to 18 mg pentaBDE/kg-day for 14 days. These effects were not observed in mice that received 18 or 36 mg/kg/day of Bromkal (a commercial pentaBDE product) or in rats that received 18 or 36 mg/kg/day for 14 days. However, IgG levels were decreased in mice that received 36 mg/kg Bromkal for 14 days (Darnerud and Thuvander, 1998).

In addition to the immunotoxicity studies discussed above, immune system organs (e.g., spleen, lymph nodes, thymus, bone marrow, etc.) have been evaluated microscopically as part of a 90-day dietary study in rats (WIL, 1984). In this study groups (30/sex/group) of male and female Sprague Dawley rats were fed 0, 2, 10 or 100 mg/kg/day for 90 days. There were no treatment-related microscopic changes in any of the organs listed above.

2.2.5 Carcinogenicity Studies

The commercial pentaBDE product has not been evaluated in a two-year animal bioassay and the potential for the commercial product to induce cancer in animals is not known. The genotoxicity studies conducted with the commercial pentaBDE product have been negative, and based on the results of these studies; the commercial product would not be expected to induce cancer by a genotoxic mode of action. Potential nongenotoxic modes of action warrant further investigation.

2.3 Robust Toxicological Summaries

An IUCLID data set, prepared by GLCC, containing hazard summaries describing each of the available relevant toxicology studies is included in Appendix III. In accordance with USEPA VCCEPP guidance, the IUCLID format is consistent with USEPA guidance for preparing Robust Summaries for the HPV Chemicals Challenge Program (USEPA, 1999b).

2.4 Tier 1 Assessment Absorption Factors and Toxicity Values

There is limited data in the scientific literature describing the absorption of the commercial pentaBDE product or its constituents. For the purpose of this Tier I assessment, oral, dermal, and inhalation absorption factors were determined based on the results of five recent studies (Hakk et al., 2001a; Hakk et al., 1999; Örn and Klasson-Wehler, 1998; Klasson-Wehler et al., 2001; and Inveresk Research, 2001). These studies assumed that difference between the administered dose and the amount of the parent compound excreted in both the feces and urine represented the bioavailable fraction. In the absence of data describing the absorption of the commercial pentaBDE product, bioavailability was based on the lowest brominated BDE isomer retained in exposed animals. Consequently, a tetraBDE (BDE-47) was assumed to be representative of the absorption rate for the commercial pentaBDE product.

Oral, dermal, and inhalation absorption factors used in the Tier 1 exposure assessment are summarized in Table 2-11. For oral routes of exposure, 86.3% of the predicted exposure concentration of the commercial pentaBDE product was bioavailable based on results reported by Örn and Klasson-Wehler (1998). The dermal absorption value was assumed to 3.13 % based on the results of a human skin absorption study sponsored by GLCC (Inveresk Research, 2001). In the absence of any data describing inhalation absorption, the default value of 75% used by ECB (2000) was adopted to represent the bioavailable fraction of commercial pentaBDE product exposure by inhalation routes.

Table 2-11 Oral, dermal, and inhalation bioavailability values used in the Tier 1 exposure assessment

Substance	Exposure Route	Absorption	Reference
Commercial PentaBDE Product	Oral	86.3%	Örn and Klasson-Wehler (1998)
	Dermal	3.13%	Inveresk Research (2001)
	Inhalation	75%	ECB (2000)

For the purposes of this Tier 1 assessment, which represents a screening-level evaluation based on the currently available scientific information, the relevant toxicity endpoints for evaluating exposures to children and prospective parents based on the available data are (1) thyroid hormone disruption, (2) thyroid hyperplasia, and (3) liver enzyme induction. In the absence of any evidence, cancer is not included as a human health endpoint in the Tier 1 assessment. Toxicity values used in the Tier 1 exposure assessment are summarized in Table 2-12.

Table 2-12. Human health endpoints and toxicity values used in the Tier 1 exposure assessment

Human Health Endpoint	Toxicity Value	Relevant Study
Liver enzyme induction	0.002 mg/kg/day	USEPA Reference Dose (RfD) for the commercial pentaBDE product based on Carlson (1980b)
Developmental Effects: Change in T_4 homeostasis	0.07 mg/kg/day	Zhou et al. (2002)
Systemic Effects: Thyroid Hyperplasia	0.04 mg/kg/day	IRDC (1976)

Thyroid Effects

It is known that pregnant women and the developing fetus are sensitive to thyroid hormone disruption (Morreales de Escobar, 2001). Key stages of development, which may involve thyroid hormone homeostasis, e.g., neurobehavioral development, in humans may begin in the first trimester, during the last trimester (brain development begins in the first trimester), and post-natally for the first 1 to 2 years (McDonald, 2002). Consequently, *in utero* exposure should be considered. Evidence of decreased fetal T_4 levels in rats exposed to the commercial pentaBDE product at doses at which maternal T_4 levels were also reduced have been reported (Zhou et al., 2002). It is not known if the decreases in the fetus were the direct action of one or more BDE congeners in the fetus or the decrease in maternal transfer of T_4. Maternal transfer is the major source of T_4 during the first two trimesters in humans (McDonald, 2002).

The potential for fetal thyroid effects is likely dependent on the mode of action for decreases in T_4 levels. If these fetal changes are due to enzyme induction, in particular UDPGT, which would increase conjugation of T_4, or by inducing its own metabolism to the metabolite that competes with T_4, the differences in the presence and activity of enzymes *in utero* and in the neonate and young child in the rat and human should be considered. If the key enzyme is UDPGT, this particular enzyme is not thought to be active in the human *in utero (*Clewell et al., 2002) and was not detected in the fetal rat on GD 20 (Crofton, 2003). Consequently, exposures that do not result in decreases in maternal transfer of T_4 should be protective of the fetus. However, UDPGT becomes active soon after birth and, therefore, the potential for thyroid effects should be considered for children of all ages. If the mode of action for disruption of thyroid homeostasis in the fetus is associated with binding to the transport protein, transthyretin (TTR), then, pentaBDE could be transported from the mother to the fetus (McDonald, 2002). Several pentaBDE isomers, in particular, 2,2',4,4'5-pentaBDE, require metabolic activation to competitively bind with TTR and then only when metabolized by phenobarbital-type microsomes (CYP2B) and not CYP1A2

or CYP4C3 (Meerts et al., 2000). In humans, the key transport protein for T_4 is thyroid-binding globulin (TBG) (McDonald, 2002). The affinity of BDEs for TBG is not known but the affinity of hydroxyl-chlorinated biphenyls is low (McDonald, 2002). Therefore, placental transport of pentaBDE in humans may be quite low or non-existent. It is unlikely that BDEs act directly on the hypothalamic-pituitary axis as there were no changes in TSH levels in response to BDE exposure (Zhou et al., 2001). Decreases in T_4 levels (55%) in the early postnatal period were not sufficient to alter synaptic transmission in the dentate gyrus of the hippocampus of rat pups born to dams that received 30 mg/kg/day of DE-71 (commercial pentaBDE product) from gestation day 6 through postnatal day 21 (Gilbert and Crofton, 2002). While not known with certainty, effects on the fetus from maternal exposure to pentaBDE may only occur at doses that produce sufficient hypothyroidism in the mother, such that maternal transfer of T_4 is sufficiently impaired.

Children older than 1 to 2 years of age may be more sensitive to these changes in thyroid hormone levels than as adults. Therefore, children are evaluated in this assessment. As with enzyme induction, effects on thyroid levels may be proportional to body burden in children for a given exposure; the body burden would be expected to be greater than in an adult. The exposure assessment accounts for this in that children would have a higher amount of intake for the same level of exposure as adults.

The biological significance of subtle changes in T_4 levels in the rat should be considered. Changes in T_4 levels occurred at approximately the same estimated doses in both the 4-day (Zhou et al., 2001) and the 90-day (WIL, 1984) studies. Significant changes did not occur at 3 mg/kg/day or less (next highest dose was 10 mg/kg/day) in the 4-day study (Zhou et al., 2001) or at 2 mg/kg/day in the 90-day study (WIL, 1984). Unlike the potential for enzyme induction, which may occur at the same blood concentration in rats and humans, there is considerable evidence that humans are more resistant to changes in circulating thyroid hormone levels, because of the significant buffering capacity of binding proteins, and are lower sensitivity to alterations in those changes (USEPA, 1998a). The relevance of changes in T_4 and TTR binding in rats to humans is questionable. The rat thyroid is known to have greater sensitivity to physiological perturbations than humans due to the shorter half-life of T_4 in rats than in humans, as well as differences in transport proteins and the more easily induced UDPGT in rats compared to humans (Capen, 1996, 1997). Therefore, rats are more susceptible than humans to decreases in circulating thyroid hormone levels, especially when brought about by hepatic enzyme inducers (Capen, 1997). Even if it is assumed that these findings in rats may be qualitatively relevant for human health assessment, differences in sensitivity to small changes in thyroid hormone levels in the rat should be considered quantitatively.

In this screening level Tier 1 assessment, changes in thyroid hormone levels in the rat were used to develop toxicity values to compare to levels of exposure to children of all ages and prospective parents. However, the biological significance of changes in T_4 in the rat relative to human development is uncertain.

Benchmark doses ($BMDL_{10}$s) were derived using the data from Zhou et al. (2002), WIL (1984), IRDC (1976) and Fowles et al. (1994) and were evaluated using BMDWin software. The T_4 datasets reported by Zhou et al. (2001) (female rats after 4 days of exposure), Zhou et al. (2002) (dams on GD 20 or PND 22), WIL (1984) (male and female rats after 28 or 90 days of exposure) and Fowles et al. (1994) (male and female mice after 14 days of exposure) were modeled. The incidence of thyroid hyperplasia reported in male and female rats after 28 (IRDC, 1976; WIL, 1984) or 90 days (WIL, 1984) of exposure were also modeled. $BMDL_{10}$s for the modeled T_4 datasets ranged from 9.04 mg/kg/day in the Zhou et al. (2002) study (decreased T_4 levels at GD 20 in the dams) to 78 mg/kg/day for the WIL (1984) 28-day study (male rats) (Table 2-13). The $BMDL_{10}$s for thyroid hyperplasia ranged from 4 mg/kg/day for male rats in the IRDC (1976) study to 9.6 mg/k/day for female rats in the WIL (1984) study.

For the development of a toxicity benchmark for the thyroid effects, the lowest $BMDL_{10}$ (4 mg/kg/day for thyroid hyperplasia (Table 2-13) was selected for the point of departure. This value was used as the basis for the toxicity benchmark, with a composite uncertainty factor of 100 (3 for intraspecies extrapolation, 10 for human variability, and 3 for duration of dosing). Each of these factors takes into account both the differences in kinetics and sensitivity (USEPA, 1994). With intraspecies extrapolation, a factor of 3 was used in the absence of data on the kinetic differences in the rat and human, but a factor of 1 was used for sensitivity because it is expected that humans would be less sensitive to thyroid effects than the rodent (USEPA, 1998a). A factor of 3 was used for duration of dosing because the doses where thyroid hyperplasia was observed were similar in the 28-day and 90-day studies and thyroid hyperplasia was not observed at lower doses following longer exposure durations. Therefore, a factor of 3 was considered sufficient for study duration. Also, at the initial exposures, decreases in thyroid hormone levels may be manifested in both rats and humans, which both have significant capacity to compensate for these small changes and maintain homeostasis. Humans, in particular, have the capability to compensate for changes in thyroid hormone levels due to the presence of thyroid binding globulin. Further, thyroid hyperplasia, which likely occurs in the rat in response to increased demand for thyroid hormone from an organ that is functioning near maximum capacity to maintain homeostasis, would not be expected in humans due to the presence of binding proteins (e.g., thyroid binding globulin) that allow humans to compensate for small decreases in thyroid hormone levels. For this screening level Tier 1 assessment, the resulting toxicity value for thyroid hyperplasia is 0.04 mg/kg/day.

Table 2-13. Results of Benchmark Dose-Response Modeling for Thyroid Endpoints

Reference	Endpoint	Model	BMD (mg/kg/day)	$BMDL_{10}$ (mg/kg/day)
Zhou et al. (2001)	Serum T_4 Female Rat - Day 4	K-power	76.79	45.54
WIL (1984)	T_4 Female Rat - week 4	K-power	164.56	65.17
WIL (1984)	T_4 Male Rat - week 4	K-power	160.84	78.304
WIL (1984)	T_4 Female Rat - week 13	K-power	621.39	121.67
WIL (1984)	T_4 Male Rat - week 13	K-power	295.97	54.35
Fowles et al. (1994)	Free T_4	K-power	21.11	13.56
Zhou et al. (2002)	Serum T_4 (Dam GD20)	K-power	13.89	9.04
Zhou et al. (2002)	Serum T_4 (Dam PND22)	K-power	23.59	15.03
IRDC (1976)	Thyroid Hyperplasia Male - 28 Days	Polynomial	7.62	3.55
IRDC (1976)	Thyroid Hyperplasia Male - 28 Days	Weibull	7.62	3.55
WIL (1984)	Thyroid Hyperplasia Female - 4 weeks	Polynomial	8.13	4.17
WIL (1984)	Thyroid Hyperplasia Female - 4 weeks	Weibull	8.13	4.17
WIL (1984)	Thyroid Hyperplasia Female - 13 weeks	Polynomial	18.85	9.58
WIL (1984)	Thyroid Hyperplasia Female - 13 weeks	Weibull	18.85	9.58
WIL (1984)	Thyroid Hyperplasia Male - 4 weeks	Polynomial	16.42	5.05
WIL (1984)	Thyroid Hyperplasia Male - 4 weeks	Weibull	9.46	5.02
WIL (1984)	Thyroid Hyperplasia Male - 13 weeks	Polynomial	13.73	6.49
WIL (1984)	Thyroid Hyperplasia Male - 13 weeks	Weibull	13.73	6.49

Reproductive/Developmental Effects

Reproductive and developmental outcomes are of importance for prospective parents and *in utero* exposure and exposure to children below the age of 2. Developmental toxicity of commercial pentaBDE mixtures have been evaluated in rats (DE-71: Zhou et al., 2002; Taylor et al., 2002; Gilbert and Crofton, 2002; and Saytex 115: Ethyl Corporation, 1985b) and 2,2',4,4',5-pentaBDE has been tested in mice (Branchi et al., 2002) (Table 2-10). No treatment-related effects on pregnancy (maternal body weight gain, live litter size, gestation length or sex ratio) were seen in rats given DE-71 at doses up to 30 mg/kg/day from GD 6 through PND 21 (Zhou et al., 2002; Taylor et al., 2002), in rats given Saytex 115 at a dose of 10 mg/kg/day on GDs 6 to 15 (Ethyl Corporation, 1985c), or in mice given 2,2',4,4',5-pentaBDE at doses up to 30 mg/kg/day from GD 6 through PND 21. No effects on pup body weight or any measure of growth or viability were seen in the offspring of rats treated at maternal doses up to 30 mg/kg/day (Zhou et al., 2002; Taylor et al., 2002) or 200 mg/kg/day (Ethyl Corporation, 1985b).

The only treatment-related effected considered to be a potentially adverse effect (note: see discussion above about differences in sensitivity of rat and human thyroid and the relevance of changes in thyroid hormone levels in rats to humans) was the significant reductions in T_4 levels seen in dams on GD 20 and PND 21 in the 30 mg/kg/day dose and in the mid- and high doses in the fetus (GD 20) and in off-spring (PND 4 and 14) group. Levels of T_3 were unaffected in dams and offspring at any dose tested. As stated above, thyroid hormone balance is important in developing fetus; however, the stage at which the central nervous system is sensitive to decreases in thyroid hormone levels has not been clearly defined (Morreales de Escobar, 2001). Thyroid hormones regulate the development of the cholinergic and dopaminergic systems in the cerebral cortex and hippocampus (McDonald, 2002). Administration of a single dose of 8 mg/kg on PND 10 altered cholinergic response to nicotine (Viberg et al., 2002). However, decreases in T_4 levels (55%) in the early postnatal period were not sufficient to alter synaptic transmission in the dentate gyrus of the hippocampus of rat pups born to dams that received 30 mg/kg/day of DE-71 (commercial pentaBDE product) from gestation day 6 through postnatal day 21 (Gilbert and Crofton, 2002).

Toxicity values were based on changes in T_4 levels in the fetus or neonate exposed *in utero* (Zhou et al., 2002). The data used were for PNDs 4 and 14. While significant changes were noted on GD20 in the fetus, the magnitude of these changes were uncertain because control levels were already very close to the detection limit and, at higher doses, a number of the animals had values that were below the detection limit. The data were modeled using Benchmark model software. The $BMDL_{10}$ was estimated to be 10.3 mg/kg/day for data from PND 4 and 2.2 mg/kg/day for data from PND 14. By applying an uncertainty factor of 30 (3 each for inter- and 10 intraspecies extrapolation), the resulting toxicity values calculated for changes in T_4 homeostasis were 0.3 and 0.07 mg/kg/day. The lower value was used as the screening toxicity benchmark in this assessment.

Neurobehavioral Changes

In addition to effects on development, neurobehavioral changes have been evaluated in mice (Branchi et al., 2002; Eriksson et al., 1998, 1999, 2001, 2002) and rats (Taylor et al., 2002). No effects on any measure of sensory-motor development were noted in the off-spring of rats treated with up to 30 mg/kg/day of the commercial DE-71 mixture (Taylor et al., 2002). The studies in mice suggest that administration of 2,2',4,4'5-pentaBDE, rather than the commercial mixture, can affect sensory-motor development; however, these data are difficult to interpret. In mice exposed *in utero* and through lactation, no effects on several indices of neurobehavior were noted in off-spring of dams treated with up to 30 mg/kg/day (Branchi et al., 2002). Transient changes were noted in the open-field test on locomotion; however, there was an inverse dose-response. No effects were noted by PND 22 in any dose group. By PNDs 34 and 60, hyperactivity was noted in the off-spring in the lowest dose group (0.6 mg/kg/day) to a greater extent and at an earlier time point than the off-spring of the mid-dose group (6 mg/kg/day), while no changes were noted in the off-spring of the high dose group females (30 mg/kg/day). No hyperactivity in any dose group was noted at 120 days. No plausible explanation for this inverse dose-response was given by the authors and use of these data in a quantitative risk assessment is not advised

Locomotion and habituation capability was evaluated in mice administered 2,2',4,4',5-pentaBDE at either 0.8 or 12 mg/kg/day on PND 10 and evaluated at 2 and 4 months of age (Eriksson et al., 1998, 2001) or administered 8 mg/kg/day on PNDs 3, 10 or 19 and evaluated at 2 months (Viberg et al., 2002) or 4 months (Eriksson et al., 1999, 2002). When neonates were dosed on PND 10, a significant reduction (hypoactive) in total activity (open field test) during the first 20 minutes, comparable to controls in the next 20 minutes, but a significant increase (hyperactive) in total activity during the last 20 minutes, was reported. Habituation (adjustment to a new environment with time) occurred in treated mice but not to the same extent as control mice (as indicated by the fact that total activity in the last 20 minutes for treated mice was less than their activity in the first and second 20-minute periods). This habituation capability was measured as the ratio of the activity in the last 20 minutes to that in the first 20 minutes. These ratios were significantly increased in treated mice compared to control; however, the treated mice were hypoactive (unexplained by the authors) in the first 20 minutes, thereby increasing this ratio. Similar results to those seen in adults dosed (8 mg/kg/day) on PND 10, but of a lesser magnitude, were noted in neonates dosed on PND 3 and no changes in spontaneous behavior were seen in mice dosed on PND 19. The authors noted that pentaBDE remained in the neonate brain when evaluated 7 days after administration and some compound administered on PND 3 would remain on PND 10, the day defined by the authors as the critical period of neonatal brain development in mice. While these data suggest that administration of pentaBDE has the potential to cause neurobehavioral changes, the use of these data quantitatively is not advised. First, litter-based statistical analyses were not conducted, nor repeated measures tests used (animals tested at 2 months may not have

been re-tested at 4 months).[4] The authors refer to PND 10 as the period of "brain growth spurt." This period varies with mammalian species and begins prenatally in humans and certain forms of neurodevelopment continue for up to 2 years of age. It is difficult to extrapolate from a single day in the mouse to an equivalent *in utero* dose in humans that could be received from exposure to the commercial pentaBDE over the same sustained period. Studies of neurobehavioral effects with dosing over a greater portion of development of the nervous system did not show these effects in the rat (Taylor et al., 2002). Altered neurobehavioral function in mice appears to return to control values by 4 months of age (Branchi et al., 2002).

Enzyme Induction

The USEPA has derived an RfD for the commercial pentaBDE product based on increased hepatic enzyme induction in rats reported by Carlson (1980b). In this study, groups of Sprague Dawley rats were administered a high dose series (series 0, 5, 10, or 20 mg/kg/day) and a low dose series (0, 0.44, 0.88, or 1.77 mg/kg/day). Enzyme induction was increased in all dose groups in both series. There were no microscopic changes in the livers of the rats in the low dose series; however, the high dose series was not examined microscopically. Consequently, the USEPA considered the highest dose tested in the low-dose series (1.77 mg/kg/day) a NOAEL. To derive the RfD, a composite uncertainty factor of 1000 (10 for interspecies extrapolation, 10 for intraspecies variation and 10 for study duration) was applied, resulting in an RfD of 0.002 mg/kg/day. This value is still reported on IRIS (USEPA, 2003).

A review of the basis for the RfD value suggests that the current value in IRIS (USEPA, 2003) is not appropriate and does not reflect the current available data. Carlson (1980b) did not examine the livers of rats in the high-dose series microscopically. However, in the WIL (1984) 90-day study, where rats were exposed to much higher doses (100 mg/kg/day), the microscopic changes observed in livers of these rats were considered adaptive changes, consistent with enzyme induction (Popp and Cattley, 1991). It is likely that only these adaptive changes would have been observed in the livers of the high-dose series in the Carlson (1980b) study, if those tissues had been examined microscopically. As a result, a higher NOAEL may have been identified for the Carlson (1980b) study, and the RfD would also be higher.

It is important to note that the enzymes induced in the Carlson (1980b) study were those potentially involved in endogenous and xenobiotic metabolism, not enzymes indicative of cellular damage in the liver (e.g., SGOT and SGPT were unchanged even at high doses) and other tissues. If enzyme induction by the commercial pentaBDE product in the rat were also to occur in humans, the pattern of xenobiotic metabolism may be altered, although the potential consequences of enzyme induction, particularly for children, are unclear. Depending on the

[4] This may be of importance in identifying a NOAEL for this study because the increase in the 0.8 mg/kg/day dose group was not significantly and increased only at 2 months and increased only marginally at 4 months.

enzyme induced and the endogenous or exogenous substrate metabolized, it can not be said *a priori* that enzyme induction is an adverse effect.

There are data that suggest that *in utero* exposures to pentaBDE resulted in significant increases in EROD induction in the rat fetus at the highest dose tested (30 mg/kg/day) (Zhou et al., 2002). The enzyme, UDPGT, which is involved in T_4 conjugation, was not detected in the fetus (Crofton, 2003). Many of the types of enzymes present in the fetus and neonate in the rat and human are heterogeneous and some are homogeneous; the timing of their activation (and likely inducibility) differ (Clewell et al., 2002). Nevertheless, it is possible that for children and adults of child-bearing age, key metabolic enzymes could be induced resulting in changes in metabolic patterns in children. If induction of key enzymes is linked to other biological consequences not limited to liver toxicity, then while enzyme induction in itself is not an adverse effect, it would be health protective to limit exposure to levels that would not result in enzyme induction.

$BMDL_{10}$s were estimated based on continuous data for markers of induction for 4 different enzymes following 90 days of exposure of up to 20 mg/kg/day in Carlson (1980b) study using the K-power hybrid model. These BMDLs ranged from 0.35 to 21.04 mg/kg/day. BMDLs reported by Zhou et al. (2001) for enzyme induction of 3 separate enzymes ranged from 0.54 to 5.83 mg/kg/day following 4 days of exposure of up to 100 mg/kg/day. Zhou et al. (2002) reported BMDLs that ranged from 1.66 to 11.22 and 0.42 to 3.41 in dams and neonates exposed to up to 30 mg/kg/day from gestation day 6 to postnatal day 21. Although different enzymes were evaluated by Carlson (1980b) and Zhou et al. (2001, 2002), the magnitude of the increases, based on the BMDLs was comparable for the studies suggesting that increasing exposure duration did not result in an increased magnitude of response.

For this screening level assessment and to provide an upper-bound estimate of hazard, rather than focus on a specific enzyme evaluated in the rat that may not be present or be active in humans, the BMDLs developed from the Carlson (1980b) study and the BMDLs reported by Zhou et al. (2001, 2002) for enzyme induction were averaged to develop a mean BMDL for enzyme induction. The resulting mean BMDL was 4 mg/kg/day. This value would represent a baseline for enzyme induction in the rat. The application of uncertainty factors to this mean BMDL is not recommended because enzyme induction does not represent an adverse effect and relevance of the resulting number would be questionable and highly uncertain.

However, it is possible that enzymes present in humans could be induced by exposure to the commercial pentaBDE product potentially resulting in some adverse effect due to altered metabolic patterns. Further, in general, induction of liver enzymes in the rat may be quantitatively similar in humans for the same body burden, although different isozymes may be induced in rats and humans. Toxicity benchmarks for the other endpoints observed in studies of the commercial pentaBDE product (e.g., thyroid and developmental effects) were 0.07 and 0.04 mg/kg/day, respectively, as discussed above. If enzyme induction in the rat and human is

comparable at comparable body burdens, significant enzyme induction would not be expected in the human at these toxicity benchmarks for other endpoints (e.g., thyroid effects).

3.0 Exposure Assessment

This section of the Tier 1 assessment includes four components: (1) a summary of general substance information, (2) primary manufacturing methods and chain-of-commerce uses for the commercial pentaBDE product, (3) a summary of available data describing environmental levels in the United States and North America, and (4) the conceptual model and exposure calculations describing the scenarios and exposure pathways used to predict child and prospective parent exposures to the commercial pentaBDE product. The first component summarizes general substance information on the commercial pentaBDE product including physical/chemical properties, the environmental fate, and exposure limits. The second component summarizes the primary manufacturing methods and chain-of-commerce uses for the commercial pentaBDE product. This component also includes a description of primary manufacturing activities, highlighting those activities associated with the manufacture of the commercial pentaBDE product that have the potential for significant worker exposures. Similarly, the predominant chain-of-commerce activities involving the use of the commercial pentaBDE product as part of the manufacture of consumer products also addresses the potential for significant worker exposures. A description of downstream uses of consumer products containing the commercial pentaBDE product, along with an assessment of the potential for general population exposure, is described. The third component is a compilation of U.S. and North American environmental data that are used to identify exposure point concentrations in the screening-level exposure models. The fourth component is the conceptual exposure model describing the possible routes of exposure to children and prospective parents. The conceptual exposure model for the commercial pentaBDE product describes three different scenarios: exposures in the workplace, exposures in the indoor home/school/office environments, and exposures associated with ambient environmental levels (including food).

3.1 General Substance Information

The current commercial pentaBDE product is composed of a mixture of primarily tetraBDE (approximately 34%), pentaBDE (approximately 55%), and hexaBDE (approximately 12%) congeners (ECB, 2000). The predominate congeners in the commercial pentaBDE product are 2,2',4,4',5- pentaBDE (BDE-99) and 2,2',4,4'-tetraBDE (BDE-47). The next most prominent congeners in the commercial mixture are 2,2',3,4,4'-pentaBDE (BDE-85) followed by lesser amounts of 2,2',4,4',6-pentaBDE (BDE-100), 2,2',4,4',5,6-hexaBDE (BDE-154) and 2,2',4,4',5,5'-hexaBDE (BDE-153). Several chemical additives can be included in the commercial pentaBDE product and comprise between 18 and 32% by weight of the product formulation currently sold in the United States (i.e., Great Lakes DE-60F Special™). The relative proportions of the different BDE constituents in commercial pentaBDE products sold in the United States and elsewhere are presented in Tables 1-1 and 1-2.

At present, GLCC is the sole producer of commercial pentaBDE and offers 4 commercial products. The four products are Great Lakes DE-60F Special™, DE-61™, DE-62™, and Great

Lakes DE-71™, where the number indicates the percent total bromine by weight. Great Lakes DE-71™ is the tradename used for commercial pentaBDE product that contains no additives. The other tradename products contain aromatic phosphate additives (i.e., isopropylated triaryl phosphates and triphenyl phosphate). Material Safety Data Sheets for each of the four commercial pentaBDE products are included in Appendix II.

Physical and Chemical Properties

The physical and chemical properties of the commercial pentaBDE product are presented in Table 3-1. The information in Table 3-1 is pertinent to the commercial pentaBDE product, as indicated, unless stated otherwise.

Table 3-1. Physical and chemical properties of the commercial pentaBDE product

Physical / Chemical Property	Characteristic
Chemical formula	$C_{12}H_5Br_5O$
Molecular weight	564.66 g/mole (70.8% bromine by weight)
Melting point	-7 to -3ºC (commercial product)
Boiling point	Decomposes at >200ºC (commercial product)
Relative density	2.25 to 2.28 (commercial product)
Vapor pressure	4.69×10^{-5} Pa (commercial product)
Water solubility	13.3 µg/L (commercial product) pentabromodiphenyl ether component = 2.4 µg/L tetrabromodiphenyl ether component = 10.9 µg/L
Log octanol-water partition coefficient (Log K_{ow})	6.57 (measured; commercial product) 7.88 (calculated)
Flammability	Not applicable – flame retardant
Autoflammability	Decomposes above 200ºC (commercial product)
Explosive properties	None
Oxidizing properties	None
Viscosity	Highly viscous at room temperature (greater than 2×10^5 cps at 25ºC)
Conversion factor	1 ppm = 23.48 mg/m^3 at 20ºC

With the application of heat, the commercial pentaBDE product decomposes more quickly than the host polymer matrix, hindering the formation of flammable gases. The release of bromine molecules upon heating interferes with the radical chain mechanism that typically takes place in the gas phase during combustion. High-energy hydroxyl and hydrogen radicals formed during combustion are removed by bromine (Rahman et al., 2001).

Physical Appearance

The commercial pentaBDE product is a thick, sap-like, semi-solid material that typically is heated or mixed with aromatic phosphate ester diluents to reduce the viscosity of the material when transferring from primary formulation vessels to product containers. Technical-grade pentaBDE is an amber viscous liquid or semi-solid at 20°C and 101.325 kPa. In contrast, pure pentaBDE is characterized as a white crystalline solid (ECB, 2000).

Melting Point

A melting point is not available for the commercial pentaBDE product. The melting range of technical-grade pentaBDE has been reported as -7 to -3°C (WHO, 1994).

Boiling Point

A boiling point is not available for the commercial pentaBDE product. Technical-grade pentaBDE decomposes in the temperature range 200-300°C. Because the commercial product is a mixture, it is expected to exhibit a wide temperature range for decomposition (WHO, 1994).

Density

The relative density of the commercial pentaBDE product is reported as 2.25 (Dead Sea Bromine Group), 2.28 (WHO, 1994), and 2.3 at 25ºC (GLCC MSDS; see Appendix II).

Vapor Pressure

The vapor pressure for the substance analyzed by the ECB (2000) has been measured as 4.69 x 10^{-5} Pa at 21°C using a spinning rotor gauge in a Good Laboratory Practice (GLP) study (Stenzel and Nixon, 1997). According to Watanabe and Tatsukawa (1990), vapor pressure increases as the degree of bromination decreases. According to ECB (2000), the material tested was a composite sample from three manufacturers and had the following composition: 33.7% tetraBDE, 54.6% pentaBDE and 11.7% hexaBDE. The method used was not able to separate the contributions of the individual components to the total vapor pressure; the results likely represent only the vapor pressures of the more volatile constituents of the commercial mixture and, therefore, the value likely represents the upper limit of the vapor pressure of the commercial mixture. GLCC has reported values of <0.01 mmHg at 95°F for the Great Lakes DE-60F Special™ commercial product and 4.69 x 10^{-5} Pa at 21°C for the Great Lakes DE-71™ commercial product.

Solubility

According to ECB (2000), the water solubility of the commercial pentaBDE product is <0.1 g/100g at 25°C (Great Lakes DE-60F Special™ , Great Lakes DE-61™ , and Great Lakes DE-62™) and 13.3 µg/L at 25°C (Great Lakes DE-71™).

Octanol-Water Partition Coefficient

According to Watanabe and Tatsukawa (1990), the logarithm of the octanol-water partition coefficient (log K_{ow}) for the commercial pentaBDE product is 6.46-6.97 using a high-

performance liquid chromatography method. MacGregor and Nixon (1997) have reported a similar value, 6.57 at 25°C.

Flash Point

The commercial pentaBDE product does not have a flash point and is used as a flame retardant, making this parameter irrelevant.

Autoignition

The commercial pentaBDE product does not undergo autoignition and, instead, decomposes at elevated temperatures (>200-300ºC). The decomposition properties are consistent with the use of this material as a flame retardant.

Explosivity

Explosive properties are not expected on the basis of chemical structure and physical properties. The commercial pentaBDE product is not known to exhibit explosive properties with other materials.

Oxidation Potential

Testing for this property is not applicable due to the physical nature of the substance (semi-solid or highly viscous liquid). The commercial pentaBDE product does not contain any substance with structural alerts for oxidizing effects and, therefore, is not considered to be an oxidizer.

Granulometry

Not applicable. The commercial pentaBDE product is a thick, semi-solid liquid.

Surface Tension

No value could be found for surface tension of an aqueous solution.

Other Physical-Chemical Properties

The viscosity is quoted as >200,000 cps at 25°C and 220 cps at 70°C (GLCC MSDS; see Appendix II). WHO (1994) cites a value of 150,000 cps at 25°C, although no further details of the origin of this value are known.

Hazardous Chemical Reactions

According to ECB (2000), pyrolysis of the commercial pentaBDE product at up to 900°C can result in the formation of brominated dibenzofurans and brominated dibenzo-p-dioxins.

3.2 Environmental Fate

Abiotic Degradation

No information is currently available on the abiotic degradation of the commercial pentaBDE product in aqueous solutions. A rate constant for the reaction of 2,2',4,4',5-pentaBDE (BDE-99) with atmospheric hydroxyl radicals has been estimated as 1.27×10^{-12} cm^3 $molecule^{-1}$ s^{-1} using the Syracuse Research Corporation Atmospheric Oxidation Program (AOP) estimation software.

Assuming an atmospheric concentration of hydroxyl radicals of 5 x 10^5 molecules/cm^3, an atmospheric half-life of around 12.6 days has been estimated for this reaction (ECB, 2000).

Biodegradation

The commercial pentaBDE product is not readily biodegradable (ECB, 2000). Information is not available regarding anaerobic biodegradation.

Volatilization

Volatilization of the commercial pentaBDE product is likely minimal because low vapor pressures have been estimated or measured for many BDEs. However, once in the atmosphere, BDEs are likely adsorbed strongly onto atmospheric particles and subsequently removed by wet or dry deposition.

Adsorption

For sediment, water partition coefficients ($K_{p(sed)}$) have been measured for several constituents of the commercial pentaBDE product (Watanabe, 1988). The values obtained are shown in Table 3-2. In general, higher brominated BDE compounds tend to adsorb more tightly to soil and sediment than lower brominated compounds. ECB (2000) calculated organic carbon partition coefficient (K_{oc}) values of between 21,508-55,680 L/kg for the commercial pentaBDE product based on fractional organic carbon contents of soil, sediment and suspended sediment of 0.02, 0.05 and 0. 1, respectively.

Table 3-2. Measured sediment-water partition coefficients ($K_{p(sed)}$) for relevant BDEs

Component	Concentration in Sediment (µg/kg)	Concentration in Water (µg/L)	$K_{p(sed)}$ (L/kg)
Tetrabromodiphenyl ether	116	0.0041	28,293
Pentabromodiphenyl ether	118	0.0024	49,167
Hexabromodiphenyl ether	138	0.0022	62,727

From the above information, the main BDE constituents of the commercial pentaBDE product can be characterized as relatively immobile in soil and unlikely to leach into groundwater. Using the Surface Area Modeling System (SAMS) model for both the tetraBDE and pentaBDE congeners, ECB (2000) concluded that releases of tetra- and pentaBDEs to the soil surface would result in contamination of only the top few centimeters of soil, with an insignificant amount, if any, leaching into groundwater.

Bioaccumulation Potential

The tetraBDE and pentaBDE congeners have been reported to be more bioaccumulative than other BDE congeners (WHO, 1994; Burreau et. al., 1997).

3.3 Occupational Exposure Limits

Occupational exposure limits for the commercial pentaBDE product have not been established in the United States.

3.4 Environmental Limits

Environmental limits for the commercial pentaBDE product have not been established in the United States.

3.5 Production Volumes

Historically, there may have been as many as nine different manufacturers of commercial BDE products worldwide (WHO, 1994). Production of commercial pentaBDE ceased in the European Union in 1997 (ECB, 2000). At present, three commercial BDE formulations are produced and sold in the United States and elsewhere. Of these, the commercial decaBDE product constitutes 82% of the reported global BDE demand, the commercial octaBDE product contributes an additional 5-7% of global demand, and the commercial pentaBDE products contributes the remaining 11-13% (Hale et al., 2002). The annual worldwide production of all three commercial BDE products was estimated in 1994 as 40,000 metric tons/year (KEMI, 1994). The majority of worldwide production is associated with the commercial decaBDE product (30,000 metric tons/year), followed by the commercial octaBDE product (6,000 metric tons/year) and the commercial pentaBDE product (4,000 metric tons/year; KEMI, 1994).

In 1999, overall annual global demand for these products increased to 67,125 tons/year with North America accounting for approximately 50% of the increased demand (Hale et al., 2002). Other major markets for the commercial BDE products in 1999 included Asia (37%) and Europe (12%) (Hale et al., 2001). At present, North America accounts for nearly 98% of the total global demand for the commercial pentaBDE product (Hale et al., 2002; Renner, 2000). GLCC is currently the sole producer in the United States for the commercial pentaBDE product with a total production level of several million pounds per year (GLCC, personal communication).

3.6 Production Methods

The commercial pentaBDE product is produced by the direct bromination of diphenyl ether using a Friedel-Crafts catalyst. The resulting mixture is a viscous liquid or semi-solid at ambient temperature and is supplied drummed as either the pure product or blended with a synergist (ECB, 2000). In the United States, only one chemical manufacturing plant, owned and operated by GLCC and located in the state of Arkansas, manufactures the commercial pentaBDE product.

3.7 Chain-of-Commerce Product Uses

Primary Use

The commercial pentaBDE product is used only for flame retardant purposes as an additive in consumer products manufactured by the furniture industry. It is used almost exclusively to flame retard flexible polyurethane foam (FPUF) used in bed mattresses and cushioning in upholstered products (IPCS, 1994). According to the limited data provided by companies that purchase the commercial pentaBDE product from GLCC for use as a flame retardant additive in FPUF (GLCC, personal communication), mattress FPUF contains approximately 2-3% flame retardant and cushion FPUF contains 3-5% flame retardant. Scrap material from both industries have been used as padding beneath carpets, and, as a result, carpet padding likely contains 3-5% flame retardant similar to that for cushion FPUF. In both FPUF products and in carpet padding, only 75% of the flame retardant additive is the BDE portion of the commercial pentaBDE product. The remaining 25% is the aromatic phosphates.

According to the Alliance for Flexible Polyurethane Foam (AFPF, 2002), not all of the FPUF found in cushion, mattress, and carpet padding products are treated with the commercial pentaBDE product. Approximately 7.5% of the more than 2.1 billion pounds of FPUF produced annually in the United States uses the commercial pentaBDE product as a flame retardant additive. The majority of FPUF products treated with the commercial pentaBDE product are sold in California, the only state requiring by law that upholstered products achieve a prescribed level of ignition resistance.

The commercial pentaBDE product is typically used in FPUF as an additive mixture with aromatic phosphate esters (Larsen and Ecker, 1988; Rose and Hughes, 1982). Polyurethanes are step addition polymers made by reacting isocyanate compounds with compounds containing

active hydrogen groups, usually hydroxyl groups, on the ends of long polyether or polyester chains. The isocyanate groups can also react with water to form carbon dioxide. This reaction is used as the principle source of gas for blowing the foam, as well as a source of heat for the expansion and curing of the foam. Other blowing agents may also be added to the foam.

FPUF is manufactured using both continuous and batch processes. In a batch manufacturing process, approximately 20 tons of the initial ingredients (mainly water, isocyanate, polyether polyols and other additives, including flame retardants) are mixed and extruded into pre-formed moulds. The extruded material is cured for 48 hours and the hardened FPUF product cut into 100-foot sections (Woods, 1982). In the continuous process, the reactants and additives are combined at the end of a continuously moving three-sided conveyor inside a long tunnel. As the reactants combine, a continuous "bun" of material exits the tunnel and is cut into long sections.

Other Possible Uses

At present, the commercial pentaBDE product is sold for use predominately (95-98%) in the manufacture of FPUF used by the furniture industry. A small percentage is used in commercial adhesive products.

Historical uses of the commercial pentaBDE product included coatings for specialty textiles, printed circuit board components, hydraulic and oilfield completion fluids, and rubber products. All of these uses have been discontinued in recent years. In the past, automotive and airplane seating cushions contained FPUF with commercial pentaBDE product. However, this use was discontinued in the early 1990s. Prior to approximately 1990, the commercial pentaBDE product may have been used in small quantities as a flame retardant in specialty fire-resistant clothing using polyurethane treatment and in polyurethane coatings in carpets.

According to the ECB (2000), with the exception of small quantities used in rigid polyurethane elastomers for instrument casings, the commercial pentaBDE product is not used in electronic equipment manufactured in the United States. However, it is possible that electronic equipment manufacturing in other countries involves the use of the commercial pentaBDE product (ECB, 2000). For example, the results of a German study examining the presence of flame-retardants in electrical and electronic equipment revealed that nearly every fraction of waste equipment prepared for recycling contained between 10 and 80 mg/kg of the tetra- through pentaBDEs (Doedens and Cuhls, 1997). The source of BDEs was not known.

According to Sjödin et al. (2001a), measurable levels of BDEs occur in air in computer rooms and electronics dismantling/recycling sites in Sweden, including BDEs typically associated with the commercial pentaBDE product. However, these results should be treated with caution. The overall data set describing air levels is not extensive and is generally lacking in control data regarding background levels and other possible sources. The commercial pentaBDE product is not used in acrylonitrile-butadiene styrene plastics and electronics equipment.

There is evidence that the commercial pentaBDE product was used as an additive in hydraulic fluids in underground mines as a replacement for PCBs (de Boer, 1990). Similarly, there has been some evidence from patents that the commercial pentaBDE product was used as an additive in completion fluids used in oil well drilling in the North Sea and the Yellow Sea. At present, investigations conducted by KEMI (1999) to confirm these uses have been unsuccessful.

3.8 Summary of Available Environmental Data

This section summarizes the available data describing environmental levels in the United States, North America, and elsewhere. Additional supporting information, including the environmental sampling data from various studies used to develop media-specific data summaries, is provided in Appendix V.

Environmental Data Compilation Strategy

During data compilation, preference was given to data relevant to the ambient environment, home environment, and workplace in the United States. In the absence of U.S. data, secondary preference was given to data from Canada; third preference was given to data from the European Union and elsewhere. The environmental data presented in this Tier 1 assessment were collected from both primary and secondary literature sources and may not represent an exhaustive survey of the published scientific literature. In most cases, the values published in different documents were not verified, nor was the quality of the study rigorously considered during the selection and compilation of data.

Among the challenges in the Tier 1 assessment of the commercial pentaBDE product is discerning the differences in environmental and hazard information pertaining to a specific BDE congener, a homologue group, and the product itself. Although the commercial pentaBDE product clearly represents a mixture of many chemicals, it is not uncommon to find both environmental and hazard studies imprecisely describing their purpose, methods, and results as pertinent to the commercial product when, in fact, the study is pertinent to only one or more constituents of the commercial product. For example, environmental monitoring data reported in the scientific literature typically describe the occurrence of BDEs in one of several ways: the concentrations of one or more homologue groups (e.g., tetra- through nonaBDEs), total BDE concentrations, or the concentrations of certain BDE congeners. It is not uncommon for studies to inadequately specify or inconsistently use chemical terms to describe homologue groups, congeners, and the commercial product.

In general, 10-12 individual BDE congeners are commonly reported (although not consistently) by different researchers involved in environmental investigations or human exposure studies (noted in bold in Appendix I). Data reported in this manner greatly hinder efforts to develop reliable exposure estimates describing plausible human exposure. Interpretation of the available environmental data also is complicated by the fact that much more extensive analysis has been performed only in recent years and for only a few constituents of the commercial products. In

addition, the chemical composition of the commercial pentaBDE product has been shown to vary between commercial products and over the past 10-20 years (e.g., see Table 1-1).

To address these limitations in the scientific literature, a chemical profile of the commercial pentaBDE product is used to identify the dominant individual BDEs that can reasonably be associated with the commercial product (yet, recognizing that a few BDE isomers may occur in both the commercial pentaBDE and the commercial octaBDE products). In studies that do not report specifically on the occurrence of the commercial pentaBDE product and, instead, report on the occurrence of specific BDE congeners, the concentrations of one or more of the individual BDE congeners will be assumed to be representative of the occurrence of the entire commercial pentaBDE product. If more than one BDE congener associated with the commercial pentaBDE product is reported in a study, the results are assumed to be additive and the occurrence of the commercial pentaBDE product is assumed to be the sum of the concentrations of each of the individual BDE congeners. In cases where the same BDE congener may occur in both the commercial pentaBDE and the commercial octaBDE products, the same study used to support the commercial pentaBDE product Tier 1 assessment also may be used to support the commercial octaBDE product Tier 1 assessment. In general, however, the technical limitations associated with changes in the chemical composition of the commercial product over time and the occurrence of BDE constituents in the environment that originate from the commercial product cannot be resolved in Tier 1; instead, both of these challenges are identified as sources of uncertainty in the Tier 1 assessment.

Levels in Ambient Air

In general, few environmental data describing levels of BDEs in U.S. ambient air are available in the scientific literature. Table 3-3 summarizes the available U.S. data, as well as the results of ambient air monitoring conducted elsewhere in the world. None of the studies found in the scientific literature attribute the occurrence of individual BDEs to the commercial pentaBDE product. The predominant BDEs reported in ambient air, BDE-47, BDE-99, BDE-100, and BDE-153, are found in the commercial pentaBDE product.

Table 3-3. Levels of BDEs reported in ambient air in the U.S. and other countries.

Location	BDE-47	BDE-49	BDE-85	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	BDE-209	Total PBDE	Reference and Comments
Levels Reported in the United States											
Chicago, USA (urban)	33.00	--	--	16.00	2.00	0.53	0.41	--	0.30	52.00	Strandberg et al. (2001); values averaged over a three year period (1997, 1998, and 1999)
Eagle Harbor, USA (rural)	2.90	--	--	2.10	0.29	0.13	0.09	--	<0.10	5.50	
Sturgeon Point, USA (rural)	3.80	--	--	2.80	0.39	0.19	0.11	--	<0.10	7.20	
Sleeping Bear Dunes, USA (rural)	8.40	--	--	5.30	0.80	0.25	0.15	--	<0.10	15.00	
Virginia, USA (rural) (% of total PBDE)	53%	ND	--	38%	4%	3%	2%	--	--	--	Hale et al. (2001); values were extrapolated from a graph, rural air
Levels Reported in Other Countries											
Sweden	<100	--	<6	<60	<9	<4	<2	<0.7	<40	--	Sjodin et al. (2001a); total air concentration of particle-associated and semivolatile BFR, two samples collected
Sweden	<100	--	--	<50	--	<3	--	<0.4	<20	--	Bergman et al. (1999); ambient outdoor air, two samples, limit of quantification = 10 times the blank sample amount
Ammarnos, Sweden	6.3	--	--	1.6	0.4	--	--	--	ND	--	Bergander et al. (1995) as cited in Palm et al. (2002)
Hoburgen, Sweden	0.7	--	--	0.35	0.07	--	--	--	ND	--	
Japan & Taiwan	--	--	--	--	--	--	--	--	--	7.1-53 (tri-hexa)	Watanabe et al. (1992) as cited in Palm et al. (2002)
Osaka, Japan	--	--	--	--	--	--	--	--	83-3060	--	
Dunai, Russia	--	--	--	--	--	--	--	--	--	5	Alaee et al. (1999) as cited in Palm et al. (2002)
Tagish, Yukon, Canada	<10-210	--	--	<10-490	--	--	--	--	--	2000	
Alert, Canada	--	--	--	--	--	--	--	--	--	1.0-28.0	
Bobcaygeon, Ontario, Canada	47-650	--	--	<10-330	--	--	--	--	--	88-1300	Gouin et al. (in review) as cited in Palm et al. (2002)
Stoke Ferry, UK	4.0-50.0	--	--	5.5-13	1.1-3.9	--	--	--	--	--	Peters et al. (1999) as cited in Palm et al. (2002)
Hazelrigg, UK	3.2-61.0	--	--	3.1-22.0	0.62-5.4	--	--	--	--	--	

ND = Not Detected

< = did not meet the limit of quantification

Few ambient air studies have been conducted in the United States. Air samples collected in the United States from urban, rural, and remote shorelines of the Great Lakes all contained measurable levels of BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154 (Dodder et al., 2000). Strandberg et al. (2001) reported a total BDE concentration range in air between 5.5 pg/m^3 in rural environments and 52 pg/m^3 in urban air in Chicago, Illinois. The concentrations of total BDEs represent the sum of six congeners: BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, and BDE-209. Air measurements were averaged over a three-year period between 1997 and 1999. The results of ambient air data collected by Hale et al. (2001) report the levels of different BDEs as a percentage of total BDE levels. The occurrence and relative distributions of BDEs reported by Hale et al. (2001) are consistent with the data reported by both Dodder et al. (2000) and Strandberg et al. (2001), as well as by scientists in other countries (see Table 3-3).

With the exception of Canada, the occurrence and relative distributions of BDEs reported in the United States are consistent with data reported by scientists in other countries (see Table 3-3). The highest ambient air levels have been reported in Canada. By comparison, the levels of BDE-47, BDE-99, BDE-100, and BDE-153 in ambient air in Sweden, where considerable environmental monitoring has been conducted in recent years, range from 0.7-100 pg/m^3, 0.35-60.0 pg/m^3, 0.07-9.0 pg/m^3, and 2.94-4.0 pg/m^3, respectively (Bergman et al., 1999; Sjödin, 2001a; Palm et al., 2002).

Levels in Sediment

In general, few environmental data describing levels of BDEs in sediments in U.S. waterways are available in the scientific literature. Table 3-4 summarizes the available U.S. data, as well as the results of environmental sampling conducted elsewhere in the world describing the concentrations of BDEs found in sediment from rivers, estuaries, and lakes near industrial plants, sewage plants, and landfills. Only one study found in the scientific literature (conducted in the United Kingdom) attributes the occurrence of individual BDEs to the commercial pentaBDE product (Allchin et al., 1999).

In two U.S studies conducted by Hale et al. (2002) and Dodder et al. (2002), total BDE concentrations in sediment ranged from non-detect to 132 ng/g dry weight (dw). The highest value was reported in sediment immediately downstream from a FPUF manufacturing plant. The predominant BDEs reported in sediment (BDE-47, BDE-99, and BDE-100) are found in the commercial pentaBDE product.

In general, sediment levels reported in Sweden and the United Kingdom appear to be higher than in the United States, ranging from 0.5 to 2250 ng/g dw. BDE-47, BDE-99, and BDE-100 are the predominant analytes in the studies reviewed. The study conducted by Nyland et al. (1992) has been cited extensively as evidence of the increased global use of the commercial products during the past decade. In the United Kingdom, Allchin et al. (1999) reported sediment concentrations of the commercial products Great Lakes DE-71™ (a commercial pentaBDE product), DE-79™

(an octaBDE commercial product), and DE-83™ (an octaBDE commercial product). The concentrations of the commercial products ranged from 0.38 to 366 ng/g dw, 0.44 to 1405 ng/g dw, and 0.6 to 3190 ng/g dw, respectively.

Table 3-4. Levels of BDEs reported in sediments (ng/g dw) in the United States and other countries

Location	BDE-47	BDE-49	BDE-99	BDE-100	BDE-153	BDE-154	BDE-190	BDE-209	Total PBDE	DE-71	DE-79	DE-83	Reference and Comments
Levels Reported in the United States													
Virginia and North Carolina, Sediment from stream leaving foam plant, Sample 1	6.37	--	9.47	1.34	--	--	--	--	17.2	--	--	--	Hale et al. (2002)
Virginia and North Carolina, Sediment from stream leaving foam plant, Sample 2	<0.1	--	<0.1	<0.1	--	--	--	--	ND	--	--	--	
Virginia and North Carolina, Sediment from stream leaving foam plant, Sample 3	36.2	--	86.3	9.01	--	--	--	--	132	--	--	--	
Virginia and North Carolina, Pond Sediment, Sample 1	0.5	--	<0.1	<0.1	--	--	--	--	0.5	--	--	--	
Virginia and North Carolina, Pond Sediment, Sample 2	<0.1	--	<0.1	<0.1	--	--	--	--	ND	--	--	--	
Hadley Lake, IN - West	4.7	--	22	2.5	4.2	4.2	<0.02	33	71	--	--	--	Dodder et al. (2002)
Hadley Lake, IN -Middle	0.85	--	1.6	0.32	1.4	1	<0.03	36	41	--	--	--	
Hadley Lake, IN - Middle	1.1	--	2.8	0.53	1.5	1.2	<0.03	27	34	--	--	--	
Hadley Lak, IN - East	0.8	--	1.9	0.36	1.1	1.3	<0.03	19	24	--	--	--	
Virginia, USA (% of total PBDE)	55	0	35	7	2	1	--	--	--	--	--	--	Hale et al. (2001)
Levels Reported in Other Countries													
Sweden - Core depth mm/(date):		--											Nylund, K. et al. (1992); Sediment core from the Baltic Sea
Sweden - Core depth mm/(date): 0-5 (1987)	1.6	--	0.98	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 5-10 (1986)	0.76	--	0.2	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 10-15 (1984)	0.68	--	0.36	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 15-20 (1982)	0.5	--	0.13	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 20-25 (1980)	0.35	--	0.09	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 25-30 (1978)	0.28	--	0.1	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 30-35 (1976)	0.24	--	0.07	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 35-40 (1974)	ND	--	0.05	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 40-45 (1971	0.13	--	0.06	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 45-50 (1967)	ND	--	0.03	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 50-60 (1961)	0.21	--	ND	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 60-70 (1953)	0.15	--	0.07	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 70-80 (1946)	0.12	--	ND	--	--	--	--	--	--	--	--	--	
Sweden - Core depth mm/(date): 80-90 (1939	0.06	--	ND	--	--	--	--	--	--	--	--	--	
Baltic Sea, surface sediment	--	--	--	--	--	--	--	--	0.5	--	--	--	Nylund et al. (1992) as cited in Manchester-Neesvig et al. (2001)
Osaka, Japan, surface sediment	--	--	--	--	--	--	--	--	57.2	--	--	--	Watanabe et al., 1987 as cited in Manchester-Neesvig et al. (2001)

Location	BDE-47	BDE-49	BDE-99	BDE-100	BDE-153	BDE-154	BDE-190	BDE-209	Total PBDE	DE-71	DE-79	DE-83	Reference and Comments
UK, River Tweed	<0.3-0.4	--	<0.6	--	--	--	--	--	--	<0.38	<0.44	<0.6	Allchin et al. (1999) as cited in de Wit (2002)
UK, River Tees, upstream a suspected source/manufacturing plant	<0.3	--	<0.6	--	--	--	--	--	--	<0.38	<0.44	<0.6	
UK, River Tees/Skerne, downstream of confluence	8.0-51.0	--	11.0-85.0	--	--	--	--	--	--	34.0-35.0	25.0-129.0	<0.6-7	
UK, River Skerne, Newton Ayecliffe	239	--	319	--	--	--	--	--	--	130	397	64	
UK, River Skerne, downstream Newton Ayecliffe	68-112	--	111-159	--	--	--	--	--	--	45-68	264-1405	23-294	
UK, Tees estuary	8.9-368	--	16-898	--	--	--	--	--	--	19-366	29-1348	<0.6-9	
UK, River Calder, upstream of sewage plant	2.3-7.6	--	0.6-16	--	--	--	--	--	--	<0.38-6.1	3.0-9.0	<0.6-399	
UK, River Calder, downstream of sewage plant	24	--	46	--	--	--	--	--	--	18	17	3190	
UK, River Ribble	1.2	--	1.7	--	--	--	--	--	--	<0.38	4.4	111	
UK, River Nith	<0.3-1.7	--	<0.6-3.5	--	--	--	--	--	--	<0.38-0.6	<0.44-2	<0.6	
UK, Avonmouth	2.4-3.6	--	2.9-4.7	--	--	--	--	--	--	0.6-1.0	<0.44	<0.6-7	
UK, Great Ouse and Elstow Landfill	0.4-4.2	--	<0.6-5.7	--	--	--	--	--	--	<0.38-1.5	<0.44-13	<0.6	
UK, River Humber	21	--	36	--	--	--	--	--	--	6.6	29	17	
Sweden, Upstream a plastics industry	3.7	--	8.8	1.6	--	--	--	--	14.1	--	--	--	Sellstrom (1996, 1999), as cited in de Wit (2002); Ignition loss % = 67
Sweden, Downstream a plastics industry	780	--	1200	270	--	--	--	--	2250	--	--	--	Sellstrom (1996, 1999), as cited in de Wit (2002); Ignition loss % = 62.5

< = did not meet the limit of quantification

Levels in Water

It is highly unlikely that BDEs will be detected in water because of the hydrophobic nature of this class of compounds. The environmental data reported in the literature are from studies conducted in Japan. A summary of the results, indicating the inability to detect BDEs in water, is presented in Table 3-5. Additional studies (not included in Table 3-5) in Japan between 1977 and 1989 reported non-detect levels of BDEs in 75-200 water samples (IPCS, 1994). All of the analyses reported hexaBDE, octaBDE, and decaBDE as not detected in water (IPCS, 1994; Darnerud et al., 2001).

Table 3-5. Levels of BDEs reported in water (µg/L) in Japan

Location	BDE-209 [1]	Reference and Comments
Japan, 1977	<0.2-<2.5	EU (2000) as cited in Palm et al. (2002)
Japan, 1987	<0.1	Watanabe and Tatsukawa (1989) as cited in Palm et al. (2002)
Japan, Kino River	<0.1	EU (2000) as cited in Palm et al. (2002)
Japan, 1988	<0.06	EU (2000) as cited in Palm et al. (2002)

< = did not meet the limit of quantification
[1] Data were reported only for BDE-209 and not for other BDE congeners.

Levels in Soil

Few environmental data describing levels of BDEs in soil in the United States are available in the scientific literature. Table 3-6 summarizes the available U.S. data. Environmental data on levels in soil from other countries were not found in the scientific literature. Soil samples collected by Hale et al. (2002) adjacent to a FPUF manufacturing plant likely represent higher levels than are reasonably expected in rural and, possibly, urban areas in the United States and elsewhere. Total BDE levels ranged from non-detect to 76 µg/kg dw. BDE-99 was the predominant isomer detected in soil, followed by BDE-47 and BDE-100. The Hale et al. (2002) study does not attribute the occurrence of individual BDEs to the commercial pentaBDE product.

Table 3-6. Levels of BDEs reported in soil (µg/kg dw) in the United States

Location	BDE-47	BDE-99	BDE-100	Total BDE	Reference and Comments
Mid-Atlantic region, USA					Hale et al. (2002)
Soil near FPUF production building	31.6	41.2	3.15	76	
Soil downwind of FPUF plant, Sample 1	8.11	4.75	0.77	13.6	
Soil downwind of FPUF plant, Sample 2	<0.10	<0.10	<0.10	ND	

< = did not meet the limit of quantification
ND = not detected

Levels in Food

Two studies conducted in Japan and Sweden report levels of BDEs in different food products as part of market surveys of contaminant levels in consumer food products. The only study of food in the United States was conducted by Huwe et al. (2002b), who reported total BDE levels in

farm chickens raised in two different regions of the United States. A summary of the available data describing the levels of BDEs found in food in the United States, Japan, and Sweden is presented in Table 3-7. None of the studies found in the literature attribute the occurrence of BDE congeners to the commercial pentaBDE product. The predominant BDEs reported in these studies included BDE-28, BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154.

Table 3-7. Levels of BDEs reported in different foods in the United States and other countries

Location and Food	BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154	Total BDE	Reference and Comments
USA - (ng/g whole weight)								Huwe et al. (2002b); data represent 13 discrete samples of chickens raised in Arkansas and one composite sample of chickens raised in North Dakota.
North Dakota, chicken	--	--	--	--	--	--	1.7	
Arkansas, chicken	--	--	--	--	--	--	39.4	
USA - (ng/g lipid weight)								Huwe et al. (2000)
Chickens fed ball clay	--	--	--	--	--	--	3.6 - 35	
Store-bought chicken	--	--	--	--	--	--	0.5	
Japan - (ng/g fresh weight)								Ohta et al. (2002); data represent approximations derived from graphs.
Spinach	0.023	0.04	0.012	0.003	0.01	0.002	0.134	
Potato	0.004	0.006	0.004	0.001	0.03	0.003	0.0476	
Carrot	0.006	0.012	0.005	0.002	0.012	0.002	0.0384	
Pork	0.002	0.025	0.025	0.006	0.003	0.002	0.0636	
Beef	0.002	0.006	0.002	0.002	ND	0.001	0.0162	
Chicken	ND	0.002	0.002	0.002	ND	0.0005	0.00625	
Sweden (ng/g whole weight)								Darnerud et al. (2000; unpublished); congeners included BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154.
Fish	--	--	--	--	--	--	0.634	
Meat	--	--	--	--	--	--	0.0458	
Dairy	--	--	--	--	--	--	0.0182	
Egg	--	--	--	--	--	--	0.0425	
Fat/Oil	--	--	--	--	--	--	0.158	
Pastry	--	--	--	--	--	--	0.0925	

ND = Not Detected

In the study by Huwe et al. (2002b), the concentration of total BDEs in one composite sample of chickens raised in North Dakota was reported as 1.7 ng/g on a whole weight basis. The total BDE concentration in 13 chickens raised in Arkansas and tested individually was 39.4 ng/g on a whole weight basis. The range of results reported by Huwe et al. (2002b), though limited, are 1-2 orders of magnitude higher than the level reported in chickens in Japan (0.00625 ng/g fresh weight) by Ohta et al. (2002). In an earlier study by Huwe et al. (2000), chickens fed ball clay and chicken bought at a grocery store were analyzed for total BDEs (i.e., BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, and BDE-183). BDE-99 was the dominant congener in all

samples; total BDE concentrations ranged between 4 and 35 ng/g lipid weight in chickens fed ball clay, and 0.5 ng/g lipid weight in store-bought chicken.

The only studies describing the levels of BDEs in other meats and food products available in the scientific literature were conducted in Japan by Ohta et al. (2002) and in Sweden by Darnerud et al. (2000; unpublished). In addition to chicken, Ohta et al. (2002) reported measurable levels of BDEs in spinach (0.134 ng/g fresh weight, fw), potato (0.048 ng/g fw), carrot (0.038 ng/g fw), pork (0.064 ng/g fw), and beef (0.016 ng/g fw). In general, BDE-47 and BDE-99 were the predominant isomers, followed by BDE-28, BDE-153, BDE-154, and BDE-100. Darnerud et al. (2000; unpublished) reported comparable levels of total BDEs (on a whole weight basis) in Swedish foods. BDEs included in the data reported by Darnerud et al. (2000; unpublished) included BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154, which are found in the commercial pentaBDE product.

Levels in Fish

Relative to other environmental matrices, the levels of BDEs in fish have been studied extensively in the United States and in other countries. Table 3-8 summarizes the available U.S. and Canadian data. The levels of BDEs reported in fish from elsewhere in the world are summarized in Table 3-9. With the exception of one study conducted in the United Kingdom, none of the studies found in the scientific literature measure the commercial pentaBDE product or attribute the occurrence of individual BDEs to the commercial pentaBDE product. The predominant BDEs reported in fish – BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154 – are found in the commercial pentaBDE product.

Based on seven U.S studies reported in the scientific literature, total BDEs in either whole or fillet fish tissues range from non-detect to 1250 ng/g wet weight (ww) in the United States. Table 3-10, shows the available U.S. fish data reporting BDEs in edible fish tissues and the calculated 95th percent upper confidence limit of the mean (UCL_{95} = 72 ng/g ww). By comparison, total BDEs in either whole or fillet fish tissues ranged between 6 and 9,680 ng/g ww in Sweden, including fish from the Baltic Sea, and from 0.04 to 4 ng/g ww in Canada. Fish data reported in other measurement units are not included in this data range.

In general, fish levels in the United States are comparable to those reported in Europe and higher than levels reported in Canada. However, data in the United States and Canada are limited, and results reported in the published literature describe different BDEs and use different units of measure (e.g., whole weight, wet weight, lipid weight) that make direct comparisons difficult. A Swedish market basket study reported the total concentration of five congeners as 0.0634 ng/g whole weight in fish sold in stores (Darnerud et al., 2000). A similar market basket survey of BDEs in fish has not been conducted in the United States. For the purposes of the Tier 1 assessment, the exposure model addressing fish consumption uses the calculated 95th percent upper confidence limit of the mean (UCL_{95} = 72 ng/g ww) for fish fillets found in the United States. This assumption is substantially higher than the levels of BDEs reported in most fish

studies conducted in the United States. The only study to report on levels of the commercial pentaBDE product was conducted by Allchin et al. (1999) in the United Kingdom. The concentrations of Great Lakes DE-71™ ranged between 47 and 1,200 ng/g lipid weight.

Table 3-8. Levels of BDEs reported in fish (ng/g ww) in the United States and Canada

Location	BDE-28/33	BDE-47	BDE-49	BDE-66	BDE-99	BDE-100	BDE-153	BDE-154	BDE-181	BDE-183	BDE-155	BDE-190	BDE-209	Total PBDE	Reference and Comments
Levels Reported in the U.S.															
Laurentian Great Lakes, USA - (ng/g wet)															Luross et al. (2002); values are mean concentrations in freshwater fish and represent whole fish tissues
Lake Ontario lake trout	--	58 ± 15	--	1.3 ± 0.38	14 ± 3.5	5.7 ± 1.1	4.9 ± 1.6	--	--	--	--	--	--	--	
Lake Erie lake trout	--	16 ± 4.2	--	0.18 ± 0.12	2.0 ± 0.48	2.5 ± 0.89	0.89 ± 0.14	--	--	--	--	--	--	--	
Lake Huron lake trout	--	27 ± 8.6	--	0.82 ± 0.38	7.7 ± 3.8	3.8 ± 1.8	2.3 ± 0.98	--	--	--	--	--	--	--	
Lake Superior lake trout	--	29 ± 9.8	--	0.89 ± 0.31	12 ± 5.9	4.1 ± 1.3	1.5 ± 0.54	--	--	--	--	--	--	--	
Lake Michigan, USA															Manchester-Neesvig et al. (2001); n = 21 coho and chinook salmon; the average % lipid was 3.89 with a range b/w 1.83-7.19; a 100g "steak" was tested which included skin, muscle, bone, and organ tissues and then blended - the steak was extracted from in front of the dorsal fin.
Salmon - Range	--	26.0-95.1	--	1.2-2.5	5.9-18.9	5.2-18.8	1.8-4.8	2.8-8.5	--	--	--	--	--	44.6-148.0	
Mean	--	52.1	--	1.7	9.3	9.7	2.7	4.5	--	--	--	--	--	80.1	
Washington State, USA															Johnson and Olson (2001); Samples were collected at different times between 9/94 - 7/99; Total PBDEs can be broken down into TBDEs, PeBDEs, and HxBDEs
Rock Island Creek - Rainbow trout (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	ND	
RI - Rainbow trout (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	ND	
SR - Rainbow trout (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	297.0	
Douglas Creek - Rainbow trout (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	1.4	
Douglas Creek - Rainbow trout (split)	--	--	--	--	--	--	--	--	--	--	--	--	--	1.5	
Douglas Creek - Rainbow trout (split)	--	--	--	--	--	--	--	--	--	--	--	--	--	1.4	
Douglas Creek - Rainbow trout (split)	--	--	--	--	--	--	--	--	--	--	--	--	--	1.4	
Spokane River - Raibow trout (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	20.0	
Spokane River - Raibow trout (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	119.0	
Spokane River - Raibow trout (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	166.0	
Spokane River - Raibow trout (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	174.0	
Sole Duck River - Moutain whitefish (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	ND	
Spokane River - Mountain whitefish (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	1250.0	
Spokane River - Largescale sucker (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	120.0	
Spokane River - Largescale sucker (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	105.0	
Yakima River - Largescale sucker (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	64.0	
Yakima River - Carp (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	22.0	
Snake River - Channel catfish (fillet)	--	--	--	--	--	--	--	--	--	--	--	--	--	8.0	
Columbia River - Starry flounder (whole)	--	--	--	--	--	--	--	--	--	--	--	--	--	30.0	
USA															Dodder et al. (2002); This lake is in close proximity(1.3 km) to a suspected PBDE manufacturing facility
Hadley Lake - white crappie and bluegill	--	13 ± 2	--	--	16 ± 2	7.4 ± 1	15 ± 2	13 ± 1	--	--	--	<0.007	<1.4	65 ± 8	
Hadley Lake - Carp	--	3.20	--	--	0.07	0.89	0.10	2.10	--	--	--	<0.007	<1.4	6.2	
Hadley Lake - Carp	--	9.80	--	--	0.12	3.60	0.04	6.90	--	--	--	<0.006	<1.3	20.0	
Lake of the Ozarks - white crappie and bluegill	--	3.5 ± 0.7	--	--	1.9 ± 0.4	1.1 ± 0.2	0.27 ± 0.11	0.2 ± 0.05	--	--	--	<0.007	<1.4	6.9 ± 1.4	
Lake Superior - Smelt	--	5.7 ± 0.3	--	--	1.8 ± 0.2	0.98 ± 0.09	0.2 ± 0.02	0.45 ± 0.03	--	--	--	<0.007	<1.5	9.1 ± 0.6	
Lake Ontario - Smelt	--	10 ± 1	--	--	5.3 ± 0.7	1.6 ± 0.1	0.49 ± 0.02	0.9 ± 0.05	--	--	--	<0.007	<1.6	18 ± 1	
Michigan, USA															Rice et al. (2002); n=12, composite subsamples of whole fish
Detroit River - Large mouth bass - Mean	--	2.80	--	--	0.48	0.45	0.44	0.43	0.26	0.26	--	--	--	5.25	
Detroit River - Carp - Mean	--	3.00	--	--	0.50	0.48	0.47	0.45	0.24	0.25	--	NA	--	5.39	Rice et al. (2002); n=10, composite subsamples of whole fish

Table 3-8. Levels of BDEs reported in fish (ng/g ww) in the United States and Canada

Location	BDE-28/33	BDE-47	BDE-49	BDE-66	BDE-99	BDE-100	BDE-153	BDE-154	BDE-181	BDE-183	BDE-155	BDE-190	BDE-209	Total PBDE	Reference and Comments
Illinois, USA															Rice et al. (2002); results may be attributable to industrial discharges
Des Plaines River (lower) Carp - Mean	--	2.54	--	--	0.50	0.44	1.01	1.89	3.28	2.99	--	1.75	--	14.40	Rice et al. (2002); n=10, composite subsamples of whole fish
Des Plaines River (upper) Carp - Mean	--	1.34	--	--	0.50	0.49	0.66	0.98	1.44	1.31	--	0.97	--	7.68	Rice et al. (2002); n=4, composite subsamples of whole fish
Buffalo, New York															Loganathan et al. (1995) as cited in Manchester-Neesvig et al. (2001)
Carp Muscle	--	--	--	--	--	--	--	--	--	--	--	--	--	18.7	
Virginia, USA															Hale et al. (2001); values were extracted and approximated from a graph, Number of samples: Catfish n=15; flathead n=20; carp n=5; stiped n=24; white n=6
Range (% of total PBDEs)	--	45-74%	2-12.0%	--	0-27%	16-24%	0-7%	3.0-5.0%	--	--	--	--	--	--	
Levels Reported in Canada.															
Canada															Easton et al. (2002); NDR = Peak detected, but did not meet quantification criteria, ND = Not detected
Salmon feed	0.061; 0.067	0.84; 1.1	0.19; 0.17	--	0.13; 0.18	0.17; 0.23	0.065; 0.038	0.17; 0.48	--	--	0.14; 0.019	--	--	1.875; 1.902	Easton et al. (2002); NDR = Peak detected, but did not meet quantification criteria, ND = Not detected; 2 samples were taken at different locations. Both are listed.
Farmed Salmon	0.036; 0.11	0.69; 2.6	0.11; 0.21	--	0.14; 0.39	0.13; 0.47	NDR; 0.08	0.041; 0.13	--	--	0.016; 0.067	--	--	1.187; 4.147	
Wild Salmon	NDR-0.019	0.029-0.28	NDR-0.029	--	NDR-0.097	0.004-0.043	ND-0.003	ND-0.005	--	--	NDR-0.003	--	--	0.039-0.485	Easton et al. (2002); NDR = Peak detected, but did not meet quantification criteria, ND = Not detected; 4 samples were taken at different locations. Both are listed.

ND = Not Detected
NDR = Peak detected, but did not meet quantification criteria (Easton et al., 2002)
NA = Not Applicable
< = did not meet the limit of quantification

Table 3-9. Levels of BDEs reported in fish in other countries.

Location	BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154	DE-71	DE-79	Total PBDE	Reference and Comments
Levels Reported in Other Countries										
(ng/g ww)										
Klosterfjorden, Sweden										Andersson et al. (1981) as cited in Manchester-Neesvig et al. (2001)
Sea Trout	--	--	--	--	--	--	--	--	15.0	
Viskan River, Sweden										
pike muscle	--	--	--	--	--	--	--	--	124.0	
pike liver	--	--	--	--	--	--	--	--	9680.0	
Baltic Sea										
Salmon muscle	--	--	--	--	--	--	--	--	14.0	Asplund et al. (1999) as cited in Manchester-Neesvig et al. (2001)
Salmon egg	--	--	--	--	--	--	--	--	9.0	
Salmon blood	--	--	--	--	--	--	--	--	6.0	
Wakayama, Japan										
Sardine	--	--	--	--	--	--	--	--	0.8	Watanabe et al. (1987) as cited in Manchester Neesvig et al. (2001)
North Sea										
North - cod liver	--	--	--	--	--	--	--	--	26.0	de Boer. (1989) as cited in Manchester-Neesvig et al. (2001)
Central - cod liver	--	--	--	--	--	--	--	--	54.0	
South - cod liver		--	--	--	--	--	--	--	170.0	
(ng/g lipid weight)										
North Sea										Boon et al. (2002)
Herring filet - Mean	1.9	37	12	9.2	0.9	1.5	--	--	--	
Range	1.2-2.4	23-47	9.9-17	6.3-12	0.6-1.3	1.3-1.9	--	--	--	
Cod filet - Mean	2.7	43	6.3	13	<LOD	3.9	--	--	--	
Range	1.5-4.5	26-74	3.1-16	5.9-21	<LOD	3.9-3.9	--	--	--	
Whiting filet - Mean	1.8	26	9	8.6	<LOD	3.3	--	--	--	
Range	1.3-2.4	7.1-40	5.3-14	4.2-12	<LOD	2.2-4.4	--	--	--	
Herring liver - Mean	2.1	30	13	9.1	2.1	2.6	--	--	--	
Range	1.6-2.5	19-52	8.0-21	5.6-17	1.1-3.9	1.5-4.4	--	--	--	
Cod liver - Mean	6.7	133	15	40	0.7	6.4	--	--	--	
Range	2.0-12	63-307	1.4-53	18-93	0.5-1.3	4.3-12	--	--	--	
Whiting liver - Mean	3.6	70	15	16	1.4	4.5	--	--	--	
Range	0.7-6.3	7.6-132	1.9-34	1.7-31	0.3-3.1	0.6-11	--	--	--	
Baltic Sea										Bergman et al., Cambridge Isotope Labs
Salmon muscle	--	200.0	54.0	47.0	--	--	--	--	--	
Sweden - Baltic Sea										Bergman et al. (1999)
Salmon - Mean	--	201	55	--	--	--	--	--	--	
Range	--	103-411	26-74	--	--	--	--	--	--	
Herring - Homogenate	--	83	27	--	--	--	--	--	--	

Table 3-9. Levels of BDEs reported in fish in other countries.

Location	BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154	DE-71	DE-79	Total PBDE	Reference and Comments
Sweden										Sellstrom et al. (1993)
Herring Muscle-Bothnian Sea Nov. 1986	--	82	27	--	--	--	--	--	--	
Baltic Proper June 1987	--	450	46	--	--	--	--	--	--	
Skagerrak April 1987	--	59	9.8	--	--	--	--	--	--	
Fladen Nov. 1987	--	12	3.4	--	--	--	--	--	--	
Utlangan Sept. 1987	--	38	17	--	--	--	--	--	--	
Landsort Oct. 1987	--	35	9.2	--	--	--	--	--	--	
Angskarsklubb Sept.-Oct 1987	--	27	17	--	--	--	--	--	--	
Harufjarden Oct 1987	--	19	7.8	--	--	--	--	--	--	
Bream muscle - River Viskan Spring 1987	--	250	2.3	--	--	--	--	--	--	
Bream muscle - River Viskan Spring 1987	--	750	2.4	--	--	--	--	--	--	
Pike muscle - River Haggen Spring 1987	--	6500	1100	--	--	--	--	--	--	
Pike muscle - River Viskan Spring 1987	--	2000	78	--	--	--	--	--	--	
Perch muscle - River Viskan Spring 1987	--	24000	9400	--	--	--	--	--	--	
Perch muscle - River Viskan Spring 1987	--	2200	380	--	--	--	--	--	--	
Trout muscle - Kesnacksalven Fall 1988	--	460	590	--	--	--	--	--	--	
Trout muscle - Bengtsbroholjen Fall 1988	--	120	130	--	--	--	--	--	--	
Trout muscle - Kesnacksalven Fall 1988	--	140	130	--	--	--	--	--	--	
Trout muscle - Bengtsbroholjen Fall 1988	--	250	220	--	--	--	--	--	--	
Trout muscle - Skifors Fall 1988	--	190	64	--	--	--	--	--	--	
Pike muscle - Kesnacksalven Fall 1988	--	98	79	--	--	--	--	--	--	
Pike muscle - Bengtsfors Fall 1988	--	94	60	--	--	--	--	--	--	
Arctic char muscle - Lake Vattern May 1987	--	400	64	--	--	--	--	--	--	
Whitefish muscle - Lake Storvindeln Nov. 1986	--	15	7.2	--	--	--	--	--	--	
Great Britain										Allchin et al. (1999) as cited in de Wit (2002)
Tees Bay - plaice, flounder and dab	--	520-9500	83-370	--	--	--	920-1200	500-1200	--	
Lune/Wyre (off River Calder) - flounder	--	400	54	--	--	--	100	120	--	
Nith estuary - flounder	--	73-120	ND-19	--	--	--	47-120	ND-83	--	
Bay (off Avonmouth) - paice, flounder, and dab	--	ND-370	ND-100	--	--	--	94-120	ND-970	--	
The Wash (off Great Ouse) - dab	--	380	74	--	--	--	110	58	--	
Off River Humber	--	1600	160	--	--	--	110	900	--	

ND = Not Detected
< LOD = did not meet the limit of quantification

Table 3-10. Levels of BDEs in the edible tissues of fish in the United States

Fish	Sample Size	Total BDE (ng/g ww)	Reference and Comments
Spokane River - Rainbow trout (fillet)	n=1	20	Johnson and Olson (2001)
Spokane River - Rainbow trout (fillet)	n=1	119	
Spokane River - Rainbow trout (fillet)	n=1	166	
Spokane River - Rainbow trout (fillet)	n=1	174	
Sole Duck River - Mountain whitefish (fillet)	n=5	1.05*	
Spokane River - Large-scale sucker (fillet)	n=1	120	
Yakima River - Carp (fillet)	n=5	22	
Snake River - Channel catfish (fillet)	n=5	8.0	
Columbia River - Starry flounder (whole)	n=5	30	
Hadley Lake –Carp	n=1	6.2	Dodder et al. (2002)
Hadley Lake -Carp	n=1	20	
Buffalo, NY - Carp Muscle	n=45, mean	18.7	Loganathan et al. (1995) as cited in Manchester-Neesvig et al. (2001)
UCL_{95} on the Mean	**--**	**72**	Calculated by ENVIRON

*Value represents half the detection limit

Levels in Indoor Air

Data on indoor air levels of BDEs in U.S. homes, schools, or the workplace are not reported in the scientific literature. The few available data describing levels in the workplace are summarized in Table 3-11 from studies conducted in Sweden and Norway by Sjödin et al. (2001a), Bergman et al. (1999) and Thomsen et al. (2001a). Indoor air concentrations of BDE-47, BDE-85, BDE-99, BDE-100, BDE-153, BDE-154, BDE-183, and BDE-209 were tested in an electronics dismantling hall, computer repair facilities, laboratory, around electronics shredders, and during the assembly of circuit boards. The levels of BDE-183 and BDE-209 were more prevalent than other BDEs, with the exception of laboratory air where BDEs were not analyzed. The mean concentrations of seven BDEs in 12 samples from a dismantling hall ranged from 1.2 to 19 ng/m^3; at the shredder, BDE concentrations in two samples ranged between 0.4 and 87 ng/m^3 (Sjödin et al., 2001a). Air concentrations were highest in samples collected by Sjödin et al. (2001a) adjacent to the electronic equipment shredder.

Indoor air levels are also reported in teaching halls and offices in Sweden (Sjödin et al., 2001a). Only two samples were taken in the teaching hall that contained 20 computers. Both results are listed on Table 3-11. Congeners found in the commercial pentaBDE product such as BDE-47, BDE-85, BDE-99, BDE-100, BDE-153, and BDE-154 were detected at levels of 0.8 ng/m^3, 0.05 ng/m^3, 0.4 ng/m^3, 0.01 ng/m^3, 0.01 ng/m^3, and 0.02 ng/m^3 respectively. However, the levels of these congeners did not meet the level of quantification in the office samples. These are the only data reported in the literature.

Table 3-11. Levels of BDEs in indoor air (ng/m^3)

Location	BDE-47	BDE-85	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	BDE-209	Reference and Comments
Sweden									
In a dismantling hall	1.2	0.17	2.6	0.25	3.9	0.57	19	36	Sjodin (2001a); total air concentration of particle-associated and semivolatile BFR, Only two samples (n=2) were collected around shredder - both concentrations are listed
Range	0.35-2.1	0.10-0.24	0.54-5.5	0.063-0.52	0.88-11	0.13-1.0	6.3-44	12.0-70	
Around a shredder	2.0;2.1	0.42;0.52	4.0;4.5	0.56;0.61	14;15	2.4;2.4	84;87	150;200	
During the assembly of circuit boards	0.35	--	0.15	0.041	0.019	0.0069	0.053	0.22	
Range	<0.1-0.39	<0.006	<0.06-0.15	<0.009-0.058	<0.004-0.033	<0.002-0.013	0.014-0.11	<0.04-0.32	
In a computer repair facility	<0.10	<0.006	<0.06	<0.009	<0.004;0.014	<0.002;0.0043	0.048;0.063	<0.04;0.093	
Sweden - Dismantling of Electronics									Bergman et al. (1999); (n=12); limit of quantification was determined as 10 times the blank sample amount
Mean	1.225	--	2.621	--	3.963	--	18.967	36.83	
Range	0.358-2.106	--	0.57-5.583	--	0.909-11.043	--	6.347-43.77	11.63-70.752	
Sweden									Sjodin (2001a); total air concentration of particle-associated and semivolatile BFR, Only two samples (n=2) were collected in the teaching hall - both concentrations are listed
- In an office with computers									
Mean	--	--	--	--	--	--	0.0082	0.083	
Range	<0.1	<0.009	<0.06	<0.006	<0.002	<0.004	0.0046-0.012	<0.04-0.087	
- In a taching hall									
sample 1	0.72	0.053	0.35	0.0085	0.012	0.022	0.011	<0.04	
sample 2	0.8	0.059	0.41	0.011	0.013	0.023	0.012	0.17	
Norway									Thomsen et al. (2001a);samples were taken by adsorption to glass at ordinary laboratory circulation and by pumping air through a SPE column.
Laboratory Air - Range	0.012-0.059	--	0.007-0.02	<0.0045	<0.0045	<0.0045	--	--	

Levels in Indoor Dust

Two studies report levels of BDEs in indoor dust. Knoth et al. (2002) collected 25 dust samples from household vacuum cleaners in Germany. Santillo et al. (2001) collected 13 samples from vacuum cleaners used in European parliament buildings and 3 similar samples from the offices of an Internet provider in the Netherlands. None of the studies attribute the occurrence of BDE congeners to the commercial pentaBDE product. Congeners BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, and BDE-209 predominated in all of the indoor dust samples. The data are presented in Tables 3-12 and 3-13.

Table 3-12. Levels of BDEs in indoor dust from homes in Germany

Dust Sample Number	Total BDE Concentration (ng/g)
S040	1326.7
S041	1328.9
S042	4302.1
S043	204.6
S045	490.2
S046	943.8
S047	309.9
S048	666.8
S049	145
S050	208.1
S051	837.2
S052	451.2
S053	286.6
S054	271.2
S055	282.2
S056	694
S057	451.1
S058	19124.2
S059	624.7
S060	16.3
S061	302.5
S062	6774.2
S063	2707.8
S064	233.9
S065	309
UCL_{95} of the Mean (Calculated by ENVIRON)	**410**

Knoth et al. (2002)

Table 3-13. Levels of BDEs in indoor dust from European Parliament buildings (ug/g)

Location	BDE-28	BDE-47	BDE-66	BDE-71	BDE-75	BDE-77	BDE-85	BDE-99	BDE-100	BDE-119	BDE-138	BDE-153	BDE-154	BDE-190	BDE-209	Reference and Comments
Netherlands	<0.16	97	<0.16	<0.16	0.49	<0.17	7.4	130	30	<0.16	3.5	48	11	<0.16	800	Santillo et al. (2001); samples were taken from vacuum cleaner bags after cleaning offices or parliament buildings
Finland	1.6	180	2.4	<0.60	<0.60	<0.62	7.5	160	36	<0.59	1.9	22	9.4	<0.60	1100	
Sweden	0.95	78	1.6	<0.36	<0.36	<0.38	3.1	68	19	<0.36	<0.36	9.8	5	<0.36	700	
Italy 1	2.8	89	2.3	<0.26	<0.26	<0.27	3.3	59	15	<0.26	2.3	21	5.4	<0.26	6900	
Italy 2	1.5	110	2.7	<0.20	<0.20	<0.21	11.2	170	23	<0.20	4.7	59	9.2	<.20	4600	
Denmark 1	0.47	21	0.48	<0.29	<0.29	<0.30	1.6	27	5	<0.28	<0.28	6.1	5.2	<0.29	470	
Denmark 2	0.91	39	0.88	<0.26	<0.26	<0.27	2.2	40	8.3	<0.26	0.76	8.5	3	<0.26	330	
Netherlands (Office) 1	<0.16	15	0.58	<0.16	<0.16	<0.17	0.9	15	3.9	<0.16	0.89	17	2.3	<0.16	490	
Netherlands (Office) 2	<0.17	10	0.5	<0.17	<0.17	<0.18	<0.06	10	2.5	<0.17	<0.17	6.3	<0.43	<0.17	330	
Netherlands (Office) 3	<0.12	17	0.69	<0.12	<0.12	<0.12	0.99	14	3.4	<0.12	0.49	13	1	<0.12	260	
Austria 1	2.8	66	2.3	<0.03	<0.04	<0.04	5.5	68	26	<0.03	1.6	26	11	<0.08	340	
Austria 2	3	64	<0.04	<0.04	<0.04	<0.04	5.5	72	23	<0.04	<0.04	18	9.4	<0.09	510	
Germany 1	6.9	80	17	<0.04	<0.04	<0.04	2.9	50	14	<0.04	<0.04	17	6.3	<0.09	1500	
Germany 2	<0.03	8.6	<0.03	<0.03	<0.03	<0.03	0.87	12	3.4	<0.03	<0.03	4.8	<0.07	<0.06	290	
UK 1	16	320	36	<0.09	<0.09	<0.10	6.6	92	22	<0.09	<0.09	31	8.9	<0.2	4500	
UK 2	<0.08	19	<0.08	<0.08	<0.08	<0.08	2.2	29	8.2	<0.08	<0.08	11	<0.19	<0.17	550	

< = did not meet the limit of quantification

Natural Sources of BDEs in the Environment

This section summarizes the available information regarding natural sources of BDEs in the environment, which is described in further detail in ECB (2000). Several brominated compounds that are structurally similar to the BDEs have been found in some marine species, especially marine sponges (Faulkner, 1988; Gribble, 2000). The compounds identified by Faulkner (1988) and Gribble (2000) are characterized by the diphenyl ether ring structure and contain an additional hydroxyl and methoxy group on one or both of the aromatic rings. Many of the compounds have been shown to possess anti-microbial properties (Sharma et al, 1969). Carte and Faulkner (1981) isolated several substituted BDE compounds from marine sponges *(Dysidea herbacea, Dysidea chlorea* and *Phyllospongia foliascens).* The compounds identified were:

from *Dysidea herbacea:*

- 2-(2',4'-dibromophenoxy)-3,4,5-tribromophenol,
- 2-(2',4'-dibromophenoxy)-4,5,6-tribromophenol, and
- 2-(2',4'-dibromophenoxy)-3,5-dibromophenol.

from *Dysidea chlorea:*

- 2-(2',4'-dibromophenoxy)-4,6-dibromophenol.

from *Phyllospongia foliascen:*

- 2-(3',5'-dibromo-2'-methoxy-phenoxy)-3,5-dibromoanisole,
- 2-(3',5'-dibromo-2'-hydroxyphenoxy)-3,5,6-tribromophenol, and
- 2-(3',5'-dibromo-2'-hydroxyphenoxy)-3,4,5,6-tetrabromophenol.

Similar compounds have been isolated from *Dysidea* species by Salva and Faulkner (1990), Norton and Wells (1980), Norton et al. (1981), Fu et al. (1995), Llin et al. (1996) and Anjaneyulu et al. (1996). Generally, compounds with between 4 and 6 bromine atoms per molecule have been detected. Salva and Faulkner (1990) found that the brominated compounds appeared to be found only in the tropical species of *Dysidea* that also contained large populations of cyanophytes in their tissues. Unson et al. (1994) demonstrated that the presence of 2-(2',4'-dibromophenyl)-4,6-dibromophenol in *Dysidea herbacea* was associated with the symbiotic filamentous cyanobacterium (similar to *Oscillatoria spongeliae)* present within the organism, rather than the sponge cells, and concluded that the brominated compounds are biosynthesized by the cyanobacterium.

Similar compounds have also been found to be produced by the acorn worm *Ptychodera flava laysanica* from Hawaii (Higa and Scheuer, 1977) and the green alga *Cladophora fascicularis* (Kuniyoshi et al., 1985) in marine waters around Japan. Species of the green algal genus

Cladophora are known to occur in a variety of marine and freshwaters, including the Baltic Sea (Dodds and Gudder, 1992).

It is possible that some of these naturally occurring brominated compounds may cause interferences in analytical methods used to detect BDEs in the environment. Since the natural brominated compounds generally have between 4 and 6 bromine atoms per molecule, this interference is likely to be a consideration only in the determination of the levels of pentaBDEs. For example, in a recent paper by Haglund et al. (1997), both pentaBDEs and methoxyBDEs were detected in biotic samples using gas chromatography-mass spectrometry. The study confirmed the presence of 2,2',4,4'-tetraBDE, 2,2',4,4',5- and 2,2',3,4,4'-pentaBDE, and 2,2',3,4,4',5'-hexaBDE in samples of ringed and grey seals and herring from along the Swedish coast, and also salmon, fish oil and human adipose tissue by comparison of the mass spectra with that from reference material. The source of the methoxy-derivatives found in the environment was not identified, but the possibility of a natural source of these compounds could not be ruled out.

3.9 Summary of Available Human Data

Levels in Human Breast Milk

The available data reported in the scientific literature describing the levels of various BDEs in breast milk from women in the United States and other countries are summarized in Table 3-14. The current understanding of BDE levels in human breast milk is based primarily on the results of monitoring studies reported during the past decade in Sweden. The most frequently reported study is that by Meironyté et al. (1999), who reported that BDE levels in breast milk from primaparous women in Sweden doubled approximately every five years between 1972-1997. More recently, however, Meironyté-Guvenius and Norén (2001) reported that BDE levels appear to be declining in monitoring data collected during the period 1998-2000. Similar results were reported by Strandman et al. (2000) in a study of Finnish women. These and other studies reported in the literature do not attribute the occurrence of BDE congeners to the commercial pentaBDE product. Congeners BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154 are typically the primary congeners reported in the various studies. The data clearly indicate that BDE-47 is the predominant congener in breast milk, followed by BDE-99, BDE-100, and BDE-153.

At present, very limited data on the levels of BDEs in breast milk from women in North America are available, and only two data points describing BDE levels in U.S. women are reported in the scientific literature. As shown in Figure 3-1, these data indicate higher BDE levels in breast milk from women in both Canada and United States than in women from Sweden and Finland. One composite sample of breast milk collected in the United States from women in Denver, Colorado, and Austin, Texas, suggests that BDE concentrations may be higher than reported by Meironyté

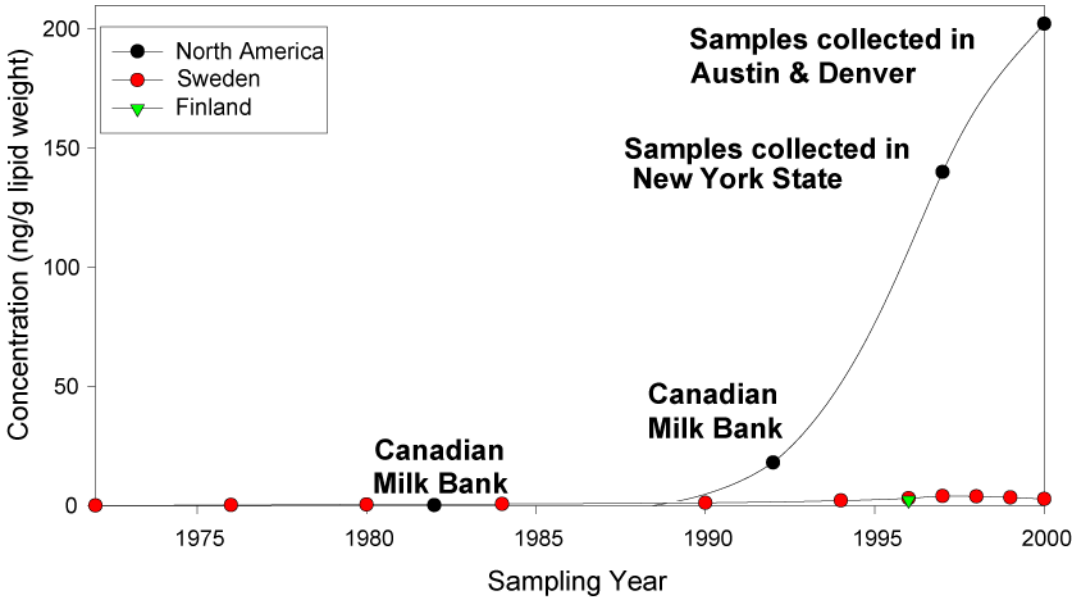
et al. (1999). According to Päpke et al. (2001), levels in the United States are 40 times higher than the Swedish concentrations.

Table 3-14. Levels of BDEs in breast milk of women from the United States and other countries (ng/g lipid weight)

Location	BDE-17	BDE-28	BDE-47	BDE-66	BDE-85	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	Total PBDE	Reference and Comments
United States												Päpke et al. (2001); composite sample - Samples were collected in Austing and Denver, USA, and sent to a laboratory in Hamburg, Münster, and Stockholm.; other congeners detected, but not individually reported are BDE-17 and BDE-138
Hamburg Laboratory results												
using LRMS, NCI	--	2.92	112	0.92	3.18	29.9	23.6	14.2	3.4	0.17	190.6	
using HRMS, EI	--	3.14	124	0.53	----	35.4	25.1	14.1	1.6	0.16	204.1	
Münster Laboratory results - using HRMS, EI	--	5.4	122	1.2	2.6	21.7	24.7	17.2	1.4	0.2	196.4	
Stockholm Laboratory results - using HRMS,EI	--	5	150	0.5	3	23	21	14	1	0.1	217.6	
Vancouver, Canada												Ryan et al. (2002); Congeners included BDE-28, BDE-47, BDE-99, BDE-100, BDE-153, BDE-154, and BDE-183; n=20 samples collected between 2001-2002
Mean	--	--	--	--	--	--	--	--	--	--	42.8	
Range	--	--	--	--	--	--	--	--	--	--	0.9-281.9	
Canada												Ryan and Patry (2001); n=10, samples collected in 1992 from Ontario and Quebec, other congeners were present but in lesser
Mean	--	0.22	3.39	--	--	1.19	0.44	0.41	--	0.15	5.79	
Stockholm, Sweden												Meironyté-Guvenius et al. (2003); samples were collected in 2000-2001 from 15 mothers ranging in age from 28-38 years old.
Median	<0.01	0.06	1.15	0.02	0.04	0.21	0.14	0.32	0.02	0.01	2.14	
Range	<0.01	0.02-0.18	0.26-4.01	<0.01-0.07	<0.01-0.17	0.07-2.2	<0.01-0.69	0.03-1.16	<0.01-0.14	<0.01-0.14	0.56-7.72	
Sweden												Meironyté-Guvenius and Norén (2001)
2000	--	0.08	1.7	0.02	0.03	0.23	0.22	0.45	0.02	0.05	2.79	
1999	--	0.1	1.97	0.03	0.05	0.43	0.24	0.54	0.04	0.07	3.46	
1998	--	0.1	2.29	0.03	0.06	0.6	0.31	0.47	0.02	0.02	3.88	
Sweden												Meironyté et al. (1999); % lipids ranged between 2.5 and 3.6%.
1997	--	0.19	2.28	0.07	0.07	0.48	0.42	0.46	0.05	--	4.02	
1996	--	0.12	2.08	0.05	0.05	0.41	0.15	0.24	0.01	--	3.11	
1994	--	0.13	1.48	0.04	ND	0.26	0.09	0.15	0.02	--	2.17	
1990	--	0.03	0.81	0.02	ND	0.15	0.06	0.1	0.04	--	1.21	
1984/1985	--	0.03	0.49	ND	ND	0.08	0.06	0.05	0.02	--	0.73	
1980	--	0.03	0.28	ND	ND	0.09	0.04	0.03	0.01	--	0.48	
1976	--	0.04	0.18	0.01	ND	0.04	0.05	0.02	0.01	--	0.35	
1972	--	ND	0.06	ND	ND	ND	ND	0.01	ND	--	0.07	
Sweden												Darnerud et al. (1998); n = 39
Mean	--	--	2.516	--	--	0.717	0.475	0.648	0.07	--	4.452	
Range	--	--	0.331-16.1	--	--	0.181-4.47	0.06-5.140	0.255-4.320	0.03-0.27	--	1.139-28.17	
Mean (pg/g fresh weight)	--	--	77	--	--	24	14	19	2.1	--	137	
Range (pg/g fresh weight)	--	--	8-358	--	--	4-222	1.5-114	8.0-96	1.5-6	--	26-626	
Sweden - Mean (Range)												Hagmar and Bergman (2001)
Pooled samples from 40 primiparas	--	--	1.7	--	--	0.23	--	--	--	--	--	
Primiparous women	--	--	1.8 (0.3-16)	--	--	0.44 (0.18-4.5	--	--	--	--	--	
Finland												Strandman et al. (2000); Human milk and placenta samples collected between 1994-1998 were analyzed from eleven (n=11) donors between 25-42 years old (median age = 34); 6 samples were from donors of their first childbirth, 1 from a donor of her second, 1 from a donor of her third, 2 from donors of their fourth chilbirth, and 1 was unknown.
Mean	--	0.16	1.31	--	--	0.39	--	0.39	--	--	--	
Range	--	0.04-0.59	0.3-4.25	--	--	0.14-0.94	--	0.19-0.72	--	--	--	

ND = Not Detected

Figure 3-1. Comparison between the concentrations of BDEs reported in human milk from North America And Europe[1]



[1] Results from the Canadian Milk Bank and New York State are from Ryan and Patry (2001); results from Denver and Austin are from Päpke et al. (2001); Swedish results are from Meironyté-Guvernius and Norén (2001), and Finnish data are from Strandman et al. (2000).

As shown in Figure 3-1 and summarized in Table 3-14, levels of BDEs in breast milk from Swedish women range from 0.07 to 28 ng/g lipid weight; in the United States, levels range from 191 to 218 ng/g lipid weight (Päpke et al., 2001). The results of breast milk monitoring conducted by Ryan and Patry (2001) in Canada are lower than the concentrations reported by Päpke et al. (2001) in the United States, ranging from 0.9-282 ng/g lipid weight for total BDEs.

Data are not reported in the literature describing BDE levels in the human fetus. Using data on the levels of total BDE (including BDE-28, BDE-47, BDE-99, and BDE-153) in human breast milk, Strandman et al. (2000) calculated possible levels in the human placenta based on an average lipid content of 0.5%. According to Strandman et al. (2000), levels in the placenta would be similar to that in breast milk; concentrations ranged from 0.88-5.89 ng/g lipid weight in breast milk and 1-4.4 ng/g lipid weight in placenta samples. Strandman et al. (2000) observed that lower brominated BDEs were more likely to be transferred from the mother to the fetus, based on data showing BDE-47 as the most predominant BDE in human milk.

In a more recent study, Mieronyté-Guvenius et al. (2003) colleted cord blood plasma at partus, maternal blood, and breast milk samples from 15 mothers living in Stockholm, Sweden. Similar levels were measured to those reported by Strandman et al. (2000). Total PBDE concentrations ranged from 0.46-4.28 ng/g lipid weight with a median of 1.69 ng/g lipid weight for cord blood plasma, 0.71-8.39 ng/g lipid weight with a median of 2.07 ng/g lipid weight for maternal blood plasma, and 0.56-7.72 ng/g lipid weight with a median of 2.14 ng/g lipid weight for breast milk. Levels of individual congeners in cord blood plasma were the same or lower than the concentrations reported in maternal blood plasma or breast milk.

Levels in Human Blood Serum

In contrast to studies conducted in Sweden, Norway, and Japan, few data are available describing levels of BDEs in blood serum in the U.S. population. A summary of the data available in the scientific literature describing the concentrations of BDEs in human blood serum is presented in Table 3-15. Similar to studies describing levels in human breast milk, the available studies do not attribute the occurrence of BDE congeners to the commercial pentaBDE product. Congeners BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154 are the primary congeners reported in these studies; other congeners, such as BDE-183 and BDE-209, have been detected, but not at appreciable levels.

The results of one U.S study of blood serum samples collected from a blood donor facility located in Illinois (Sjödin et al., 2001b) suggest that BDE levels in human blood serum (based on the occurrence of the 5 primary BDE congeners) ranges between 0.82 and 54 ng/g lipid weight. According to Sjödin et al. (2001b), BDE concentrations ranged from 0.39-24 ng/g lipid weight for BDE-47, 0.23-3.8 ng/g lipid weight for BDE-99, 0.11-24 ng/g lipid weight for BDE-100, and 0.08-2 ng /g lipid weight for BDE-153. Similar levels are reported in Sweden, Norway, and Japan (Bergman et al., 1999; Hagmar and Bergman, 2001; Jakobsson et al., 2002; Thomsen et al., 2001b; and Nagayama et al, 2001).

Levels in Human Adipose Tissues

A summary of the data available in the scientific literature describing the concentrations of BDEs in human adipose tissue is presented in Table 3-16. Similar to studies describing levels in human breast milk and blood serum, the available studies do not attribute the occurrence of BDE congeners to the commercial pentaBDE product. Congeners BDE-47, BDE-99, BDE-100, BDE-153, and BDE-154 are the primary congeners reported in various studies; other congeners, such as BDE-183 and BDE-209, have been detected, but not at appreciable levels. She et al. (2002) reported BDE levels in breast adipose tissue ranging from 1 to as high as approximately 196 ng/g lipid weight. For the individual BDEs measured, She et al. (2002) reported in the San Francisco Bay Area, USA, levels of 7.01-196.0 ng/g lipid weight for BDE-47, 2.17-72.2 ng/g lipid weight for BDE-99, 0.77-60.6 ng/g lipid weight for BDE-100, 1.52-124.0 ng/g lipid weight for BDE-153, and 2.86-70.5 ng/g lipid weight for BDE-154. According to She et al. (2002), a strong correlation could be established between levels in breast adipose tissue and abdominal adipose tissue.

3.10 Conceptual Exposure Model

This section is the fourth component of the exposure assessment. It includes a conceptual exposure model describing the possible routes of exposure to children and prospective parents. The conceptual exposure model for the commercial pentaBDE product illustrates three different scenarios: exposures in the workplace, exposures in the indoor home/school/office environments (which includes child daycare centers), and exposures associated with ambient environmental levels (e.g., via the diet, both direct and indirect contact with soil, and particulates in air). For

direct and indirect exposure routes illustrated in the conceptual model where exposures are plausible and quantifiable, screening-level exposure models developed in Microsoft® Excel are presented for children of different ages and adult men and women (prospective parents).

Table 3-15. Levels of BDEs in human blood serum in the U.S. general population and in other countries (ng/g lipid weight)

Location and Population	BDE-17	BDE-28	BDE-37	BDE-47	BDE-66	BDE-77	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	BDE-203	BDE-209	OctaBDE	OctaBDE	OctaBDE	NonaBDE	NonaBDE	NonaBDE	Total PBDEs	Reference and Comments
Levels in the U.S Population																					
Illinois, USA																					Sjodin et al. (2001b); samples were collected in 1988 from blood donors; octaBDE and nonaBDE concentrations were estimated from the instrument
Median	--	--	--	0.64	--	--	0.32	0.2	0.35	--	0.18	<0.1	<1	0.13	0.78	0.1	<0.1	0.39	<0.1	--	
Range	--	--	--	0.4-24	--	--	0.2-3.8	0.1-24	0.08-2.0	--	0.09-1.3	<0.1-0.2	<1-34	<0.1-0.3	0.17-1.7	<0.1-0.15	<0.1-0.37	0.11-2.1	<0.1-0.29	--	
San Francisco, USA																					Petreas et al. (2003); samples were collected from 1997-1999 from 50 Laotian immigrant women between the ages of 19 and 40 years old from a study on organochlorine exposures and menstrual cycle function. Samples between 1959-1967 were collected from 420 pregnant women and used for historic comparison
1997-1999	--				--																
Mean	--	--	--	50.6 ± 94.6	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
Median (Range)	--	--	--	10 (<10-511)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
1959-1967	--				--																
Mean	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
Median (Range)	--	--	--	<10	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
Levels in Populations from Other Countries																					
Japan																					Nagayama et al. (2001); subjects included 54 Japanese adults aging from 37-49 years old with a mean of 43.9 yrs.
Mean (Range)	--	--	0.032 (ND-0.1)	0.52 (0.1-2.0)	--	ND	1.2 (0.39-2.5)	--	2.3 (ND-6.4)	--	--	--	--	--	--	--	--	--	--	4.5 (1.2-18.0)	
Stockholm, Sweden																					Meironyté-Guvenius et al. (2003); blood plasma samples were collected in 2000-2001 from 15 mothers ranging in age from 28-38 years old.
Median	<0.01	0.07	--	0.83	0.02	--	0.19	0.17	0.56	0.04	0.06	--	--	--	--	--	--	--	--	2.07	
Range	<0.01-0.03	<0.01-0.2	--	0.3-5.1	<0.01-0.14	--	<0.01-1.43	<0.01-0.52	0.27-1.03	<0.01-0.16	0.01-0.44	--	--	--	--	--	--	--	--	0.71-8.39	
Sweden																					Bergman et al. (1999); for BDE-47, BDE-153, and BDE-183, the limit quantification was defined as twice the blank sample amount; for BDE-209, the limit of quantification was defined as <0.7 pmol/g lipid and the limit of detection was <0.3 pmol/g lipid and also defined as the signal to noise ratio of
No fish Intake - 1991	--	--	--	0.407	--	--	--	--	NA	--	NA	--	NA	--	--	--	--	--	--	--	
High fish Intake - 1991	--	--	--	2.155	--	--	--	--	NA	--	NA	--	NA	--	--	--	--	--	--	--	
Cleaners - 1997	--	--	--	1.567	--	--	--	--	0.578	--	0.117	--	<0.678	--	--	--	--	--	--	--	
Clerks - 1997	--	--	--	1.469	--	--	--	--	0.844	--	0.175	--	<0.678	--	--	--	--	--	--	--	
Dismantling Electronics - 1997	--	--	--	2.89	--	--	--	--	4.547	--	8.025	--	4.846	--	--	--	--	--	--	--	
Sweden - Median																					Jakobsson et al. (2002); Subjects included 19 hospital computer technicians, 20 hospital cleaners and 20 computer clerks
Computer Technicians	--	--	--	1.3	--	--	--	--	2.7	0.6	0.95	--	1.6[d]	--	--	--	--	--	--	--	
Hospital Cleaners	--	--	--	1.6	--	--	--	--	0.58	0.38	0.12	--	<0.7[a]	--	--	--	--	--	--	--	
Computer Clerks	--	--	--	1.5	--	--	--	--	0.84	0.51	0.18	--	<0.7[a]	--	--	--	--	--	--	--	
Range	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
Computer Technicians	--	--	--	<1[a]-14	--	--	--	--	<1[a]-5.8	0.23-1.2	0.18-4.7	--	<[c]-6.9	--	--	--	--	--	--	--	
Hospital Cleaners	--	--	--	<0.5-17	--	--	--	--	0.42-4.9	0.16-0.91	0.018-0.29	--	<0.3[c]-3.8	--	--	--	--	--	--	--	
Computer Clerks	--	--	--	<0.5[a]-4.9	--	--	--	--	0.5-3.3	0.28-0.97	<0.02[a]-1.0	--	<0.3[c]-7.8	--	--	--	--	--	--	--	
Norway																					Thomsen et al. (2002); Between 1975 to 2003, serum had been sampled from patients at five different county hospitals, regardless of reason for hospitalization or disease.
1977	--	ND	--	0.25	--	--	0.087	ND	0.1	ND	--	--	--	--	--	--	--	--	--	--	
1981	--	0.096	--	0.32	--	--	0.13	0.079	0.18	0.22	--	--	--	--	--	--	--	--	--	--	
1986	--	ND	--	0.41	--	--	0.13	0.12	0.14	0.26	--	--	--	--	--	--	--	--	--	--	
1990	--	0.066	--	0.89	--	--	0.24	0.13	0.27	0.23	--	--	--	--	--	--	--	--	--	--	
1995	--	0.14	--	1.4	--	--	0.33	0.32	0.52	0.5	--	--	--	--	--	--	--	--	--	--	
1999	--	0.24	--	1.5	--	--	0.31	0.35	0.59	0.35	--	--	--	--	--	--	--	--	--	--	
0-4 years old	--	0.26	--	6.2	--	--	1.6	1.7	1.5	0.45	--	--	--	--	--	--	--	--	--	--	
4-14 years old	--	0.2	--	2	--	--	0.37	0.66	0.86	0.39	--	--	--	--	--	--	--	--	--	--	
15-24 years old, female	--	0.088	--	2.5	--	--	0.71	0.49	0.56	0.27	--	--	--	--	--	--	--	--	--	--	
15-24 years old, male	--	0.094	--	2.3	--	--	0.68	0.61	0.66	0.19	--	--	--	--	--	--	--	--	--	--	
25-59 years old, female	--	0.093	--	1.3	--	--	0.32	0.28	0.34	0.23	--	--	--	--	--	--	--	--	--	--	
25-59 years old, male	--	0.093	--	2.3	--	--	0.4	0.52	0.71	0.24	--	--	--	--	--	--	--	--	--	--	
> 60 years old, female	--	0.36	--	1.2	--	--	0.25	0.37	0.36	0.26	--	--	--	--	--	--	--	--	--	--	
> 60 years old, male	--	0.096	--	3.4	--	--	0.36	0.45	0.59	0.38	--	--	--	--	--	--	--	--	--	--	
Norway - Mean																					Hagmar and Bergman (2001); subjecst tested included laboratory personnel, circuit board producers, and electronic dismantlers are the only worker categories listed for BDE-99.
Laboratory Personnel	--	--	--	1.5	--	--	0.4	--	0.54	--	--	--	--	--	--	--	--	--	--	--	
Circuit Board Producers	--	--	--	1.6	--	--	0.32	--	0.95	--	--	--	--	--	--	--	--	--	--	--	
Electronics dismantlers	--	--	--	4	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	2.9	--	--	0.97	--	1.7	--	--	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	4.5	--	0.33	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	--	--	3.2	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	--	--	7.8	--	4.8	--	--	--	--	--	--	--	
- Range																					
Laboratory Personnel	--	--	--	1.0-3.0	--	--	0.17-0.73	--	0.43-0.63	--	----	--	----	--	--	--	--	--	--	--	
Circuit Board Producers	--	--	--	0.4-3.4	--	--	ND-0.77	--	0.5-1.8	--	----	--	----	--	--	--	--	--	--	--	
Electronics dismantlers	--	--	--	0.9-15	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	<0.5-23	--	--	0.18-3.6	--	1.2-2.3	--	--	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	2.1-12	--	0.09-1.1	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	--	--	2.5-12	--	--	--	--	--	--	--	--	--	
(Sweden)	--	--	--	--	--	--	--	--	--	--	2.3-20	--	<0.3-9.5	--	--	--	--	--	--	--	
Norway - Mean																					Thomsen et al. (2001b); total PBDEs consist of BDE-28,47,99,100,153,154,183,TriBP, and TBBPA.
electronics dismantlers	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	8.8	
Circuit board producers	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	3.9	
Laboratory Personnel	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	3	
- Range																					
electronics dismantlers	--	--	--	0.43-14.6	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	----	
Circuit board producers	--	--	--	0.43-14.6	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	----	
Laboratory Personnel	--	--	--	0.43-14.6	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	----	

ND = Not Detected
NA = Not Analyzed
< = did not meet the limit of quantification

Table 3-16. Levels of BDEs in human adipose tissue in the U.S. general population and in other countries (ng/g lipid weight)

Location and Population	BDE-28	BDE-47	BDE-85	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	Total PBDEs	Reference and Comments
San Francisco, USA										Petreas et al. (2003); samples were collected from 1996-1998 from 32 women between the ages of 25 and 65 years old from a case-control study on breast cancer and organochlorine
Mean	--	28.9 ± 39.8	--	--	--	--	--	--	--	
Median (Range)	--	16.5 (5.2-196)	--	--	--	--	--	--	--	
San Francisco, USA										She et al. (2002); Breast adipose tissue was sampled from 23 women ages 28-62 (mean of 47); percent fat in sample ranged from 9.9-93.6 with a mean of 72.2
Range	--	7.01-196.0	--	2.17-72.2	0.77-60.6	1.52-124.0	2.86-70.5	--	17.2-462.0	
Mean	--	33.3	--	10.7	9.1	16.2	16.5	--	85.7	
Spain										Meneses et al. (1999); tissues from 13 people were sampled (3 women and 10 men) ages 28 to 83 years (mean age of 57)
Mean (Range)	--	1.36 (0.2-5.8)	--	0.42 (<0.07-2.1)	--	1.83 (0.67-4.2	--	--	--	
Sweden	--	8.8	1.8	1.1	--	1.7	--	--	--	Haglund et al. (1997); based on 90% lipids
Japan										Choi et al. (2003); tissues from 10 women in their 40's and 50's, living in the Tokyo area were sampled Lipid contents of the samples ranged from 72% to 95 %. According to Choi et al. the sample ratio for the 1970 samples ranged from 0.6-1.5 while the samples from 2000 varied from 2.0-9.2.
Median	0.0023	0.017	--	0.0039	0.0021	<0.0063	<0.0063	<0.0063	0.0292	
Range	<0.001-0.0076	0.0044-0.0604	--	<0.0025-0.0139	<0.0025-0.006	<0.0063	<0.0063	<0.0063	0.0068-0.078	
Median	0.076	0.459	--	0.118	0.25	0.382	0.06	0.047	1.288	
Range	0.047-0.487	0.109-0.979	--	0.042-0.362	0.041-0.527)	0.122-0.631	0.014-0.104	0.020-0.177	0.466-2.753	

< = did not meet the limit of quantification

Consistent with the Tier 1 screening approach, exposure models rely on exposure point concentrations for different environmental compartments (e.g., air, soil, indoor dust, food stuffs, and in the workplace) representing the 95th upper confidence limit of the mean for robust data sets or the high end of the range of the available data, if the available data are limited. Exposure factors, based primarily on USEPA's *Exposure Factors Handbook* (USEPA, 1997) and *Child-Specific Exposure Factors Handbook* (USEPA, 2000a), are identified for each of the exposure scenarios and associated exposure pathways. All assumptions and parameter values used in the screening-level exposure calculations are specified and referenced to the relevant supporting technical information.

The conceptual exposure model describing how potentially exposed populations might be exposed to the commercial pentaBDE product is presented in Figure 3-2. Figure 3-2 provides a general overview of how BDEs associated with the commercial pentaBDE product might migrate through the environment, culminating in human exposure. In Figure 3-3, the conceptual exposure model is further refined to indicate the specific receptors and exposure pathways that are considered in the Tier 1 exposure assessment.

Potentially Exposed Populations

The populations that are potentially exposed to the commercial pentaBDE product are as follows: primary product manufacturers, FPUF manufacturers, end product manufacturers, and children and adults in both the indoor home/school/office environments and the ambient environment. Twelve different workplace job functions in primary production and chain-of-commerce manufacturing activities were considered in the Tier 1 assessment, as shown in Table 3-17. The entire population of workers engaged in primary manufacturing of the commercial pentaBDE product is fewer than 100 persons employed by GLCC at one chemical plant located in Arkansas and is typically male workers ranging in age from 20-45 year old. Nonetheless, female workers are included in this Tier 1 screening-level assessment. The population of workers engaged in secondary, or chain-of-commerce, activities involving the use of the commercial pentaBDE product as an additive in the manufacture of FPUF or products containing FPUF is large and cannot be estimated with any degree of accuracy. Similarly, the population of adult men and women and children potentially exposed to ambient levels of BDEs in the environment and through contact with consumer products containing the commercial pentaBDE product (primarily upholstered furniture, mattresses, and cushions) in the indoor home/school/office environments also is large, and cannot be estimated with any degree of accuracy. Child exposures are evaluated for seven different age groups: <1 year, 1-2 years, 3-5 years, 6-8 years, 9-11 years, 12-14 years, and 15-18 years. Girls and boys are evaluated separately in the 9-11, 12-14, 15-18 year age groups.

Figure 3-2. General conceptual exposure model depicting how BDEs associated with the commercial pentaBDE product might migrate through the environment, culminating in human exposure

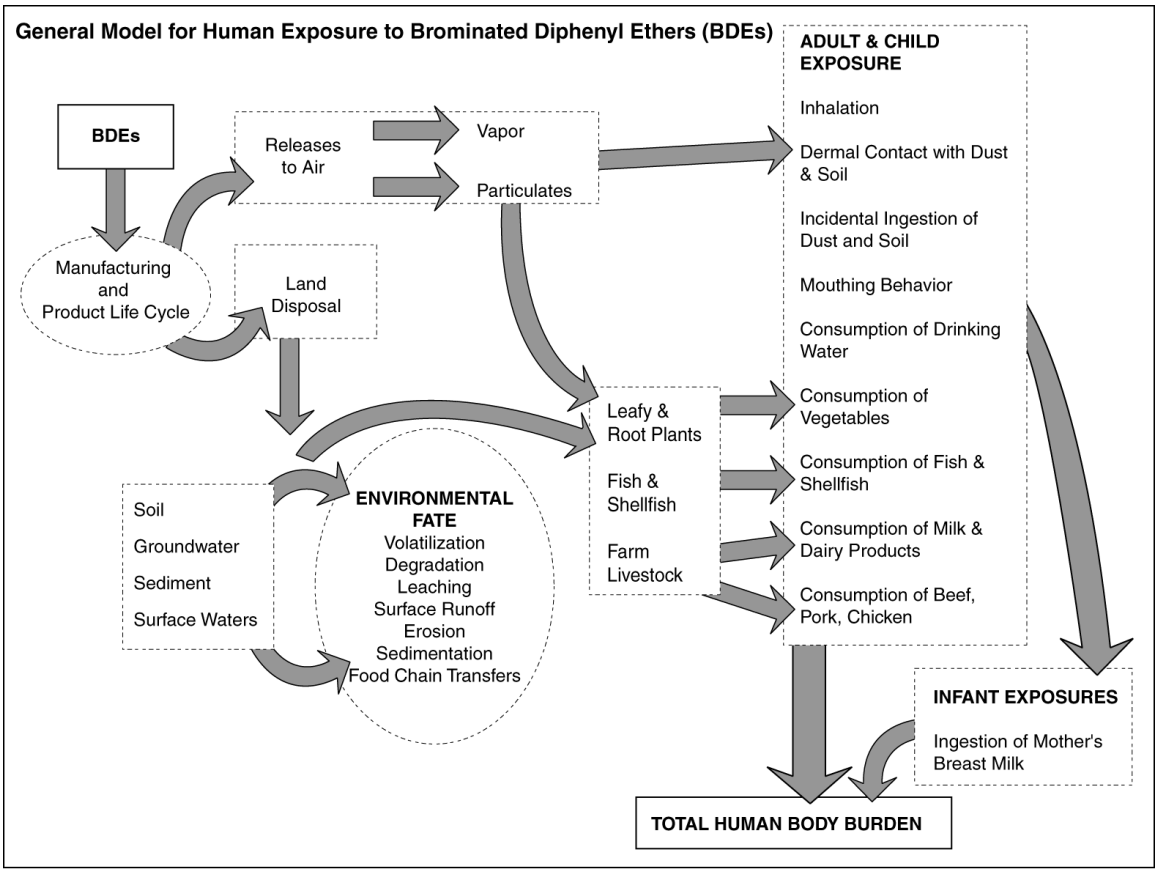
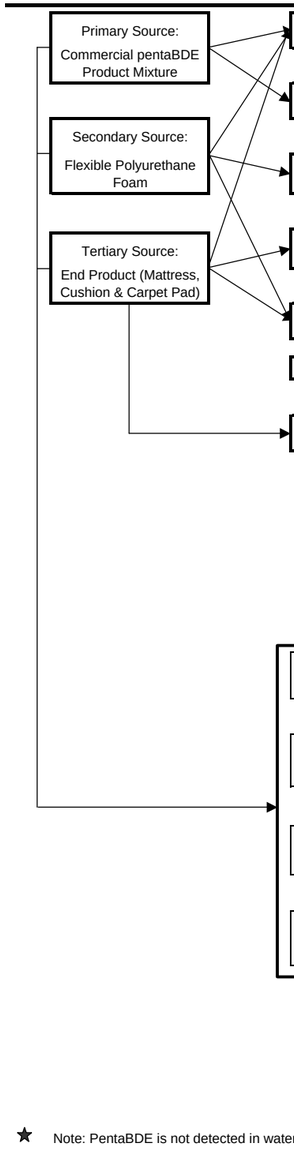


Figure 3-3. Specific conceptual exposure model indicating the specific receptor and exposure pathways considered in the Tier 1 assessment.



Sources	Primary Release Mechanism	Secondary Release Mechanism	Exposure Media	Exposure Route	RECEPTORS: Primary & Chain-of-Commerce Manufacturing Exposure: Primary Exposure (Drummer & Cleaner)	Foam Manufacturer (Mixer & Cutter)	End Product Manufacturer (Mattress, Cushion, Furniture & Carpet Pad)	Carpet Installer	Distributor (Covered & Uncovered)	Recycler	Worker's Child	Exposures in the Indoor Home / Office / School Environment: Child	Adult	Exposure to Ambient Environmental Levels: Child	Adult	Adult Angler (Marine & Freshwater)
Primary Source: Commercial pentaBDE Product Mixture	Volatilization			Inhalation of Vapors	X	X	X	X	X	X	--	--	--	--	--	--
Secondary Source: Flexible Polyurethane Foam	Drumming & Cleaning Activities			Dermal Contact	X	X	X	X	X	X	--	--	--	--	--	--
				Incidental Ingestion	X	X	X	X	X	X	--	--	--	--	--	--
	Foam Mixing & Cutting Activities	Adherence to Clothing	Dirty Laundry	Dermal Contact	X	X	X	X	X	X	X	--	--	--	--	--
				Incidental Ingestion	X	X	X	X	X	X	X	--	--	--	--	--
Tertiary Source: End Product (Mattress, Cushion & Carpet Pad)	End Product Manufacture	Adherence to Shoes	Dirty Floors	Dermal Contact	X	X	X	X	X	X	X	--	--	--	--	--
				Incidental Ingestion	X	X	X	X	X	X	X	--	--	--	--	--
	Distribution (Foam & End Product)															
	Recycling															
	Purchase & Use of End Product	Mechanical Poofing	Indoor Air	Inhalation of Particulates	--	--	--	--	--	--	--	X	X	--	--	--
				Ingestion of Swallowed Particulates	--	--	--	--	--	--	--	X	X	--	--	--
			Indoor Dust	Dermal Contact	--	--	--	--	--	--	--	X	X	--	--	--
				Incidental Ingestion	--	--	--	--	--	--	--	X	X	--	--	--
		Mouthing Behavior	Saliva	Ingestion	--	--	--	--	--	--	--	X	--	--	--	--
	Effluent Discharge															
	Waste Incineration															
	Landfill Disposal															
	Sewage Sludge															
		Volatilization	Ambient Air	Inhalation of Particulates	--	--	--	--	--	--	--	--	--	X	X	--
				Ingestion of Swallowed Particulates	--	--	--	--	--	--	--	--	--	X	X	--
		Leaching														
		Surface Runoff	Soil	Dermal Contact	--	--	--	--	--	--	--	--	--	X	X	--
				Incidental Ingestion	--	--	--	--	--	--	--	--	--	X	X	--
		Erosion														
		Sedimentary	Fish	Consumption	--	--	--	--	--	--	--	--	--	X	X	--
		Food Chain Transfer		Additional Consumption by Anglers	--	--	--	--	--	--	--	--	--	--	--	X
			Other Food (Meat, Dairy, Fats and Oils, Eggs & Vegetables	Consumption	--	--	--	--	--	--	--	--	--	X	X	--
			Breast Milk	Consumption	--	--	--	--	--	--	--	--	--	X	--	--
			Drinking Water	Consumption	--	--	--	--	--	--	--	--	--	--	--	--

★ Note: PentaBDE is not detected in water

Table 3-17. Human populations evaluated in the commercial pentaBDE product Tier 1 exposure assessment

Receptor	Description
Worker: Occupational Exposure Scenarios	
Primary Production Drummer (Male/Female)	Places product (either mixed or unmixed) into drums.
Primary Production Cleaner (Male/Female)	Cleans drums that have been returned empty yet with some product residue.
Foam Manufacture Mixer (Male/Female)	Mixes product with polymer in order to produce flame retarded flexible polyurethane foam. Includes molding activities.
Foam Manufacture Cutter (Male/Female)	Cuts the foam into blocks or according to specifications.
Mattress Manufacture (Male/Female)	Includes the foam in mattresses and covers the assembly.
Cushion Manufacture (Male/Female)	Covers foam with fabric to make cushions.
Furniture Manufacture (Male/Female)	Attaches fabric-covered cushions to furniture frame (chair, sofa).
Carpet Pad. Manufacture (Male/Female)	Makes carpet padding from scraps of foam.
Carpet Installer (Male/Female)	Installs carpet padding.
Distributor Uncovered (Male/Female)	Handles and transports the uncovered foam.
Distributor Covered (Male/Female)	Handles and transports the fabric-covered foam.
Recycler (Male/Female)	Handles foam products destined for recycling.
Child & Adult: Indoor Home, School, and Office Exposure Scenarios	
<1 year Child	Exposed to consumer goods indoors that contain foam at home (sofa, chair, carpet pad, mattress) and school (sofa, chair, carpet pad). Worker's children also are exposed to residues brought home on clothing and shoes.
1-2 yrs Child	
3-5 yrs Child	
6-8 yrs Child	
9-11 yrs Male/Female	
12-14 yrs Male/Female	
15-18 yrs Male/Female	
Adult Male (Prospective Parent)	In addition to the above, also exposed at the office (sofa, chair, carpet pad).
Adult Female (Prospective Parent)	
Child & Adult: Ambient Environmental Exposure Scenarios	
<1 year Child	Exposed to residues that have been measured in different environmental compartments (e.g., air, soil, food products, and fish). Exposure to water (drinking, surface, and ground water) is not considered due to the hydrophobic nature of the compounds of interest. Additional fish consumption by anglers is considered.
1-2 yrs Child	
3-5 yrs Child	
6-8 yrs Child	
9-11 yrs Male/Female	
12-14 yrs Male/Female	
15-18 yrs Male/Female	
Adult Male (Prospective Parent)	
Adult Female (Prospective Parent)	

Scenario #1a - Exposures to Commercial PentaBDE Product Manufacturers

The potential exposure routes for workers engaged in primary manufacturing activities at the single U.S. manufacturing facility located in the state of Arkansas where the commercial pentaBDE product is manufactured include the following:

Primary Exposure Pathways

- Inhalation of vapors originating from the commercial pentaBDE product in the workplace;
- Dermal contact with the commercial pentaBDE product in the workplace;
- Incidental ingestion via hand-to-mouth contact in the workplace;

Secondary Exposure Pathways

- Dermal contact with the commercial pentaBDE product on dirty laundry brought home from the workplace;
- Incidental ingestion via hand-to-mouth contact after handling dirty laundry in the home;
- Dermal contact with the commercial pentaBDE product brought indoors in the home on shoes from the workplace; and,
- Incidental ingestion via hand-to-mouth contact after touching dirty floors in the home.

In addition to direct skin contact, two indirect exposure pathways involving dirty laundry and indoor floors at home also were considered. For contact with dirty laundry and shoes, clothing (i.e., socks, shirts, and pants) and shoes were assumed to contact the commercial pentaBDE product either during the manufacturing process, during the transfer of the product from the primary reaction vessels to storage containers, and/or during cleaning of residues on product reaction vessels and other equipment. When clothing was laundered in the home, only the bare hands were assumed to contact dirty clothing. Dermal contact and incidental ingestion with dirty floors theoretically occurs when workers redistribute the chemicals of concern adhered to shoes across the entry level of the home, with subsequent exposure from contacting the dirty floor. Workers' children are also present in the home and were evaluated for these last two pathways.

Scenario #1b - Exposures to FPUF Manufacturing Workers

The major sources of worker exposures during the manufacture of FPUF are likely to be associated with:

- Handling of the commercial pentaBDE product prior to mixing with other ingredients;
- Volatilization of BDEs from FPUF during the mixing and molding processes; and,

- Handling of residues, including FPUF scrap, during the cleaning of molding equipment.

According to ECB (2000), the main source of exposure to the commercial pentaBDE product is associated with the handling of the material prior to the foaming process, where releases to waste water are estimated to be around the order of 0.01% (i.e. 0.1 kg/ton). Although the default emission factors suggest a zero release to air of the commercial pentaBDE product during FPUF processing, there is a potential release during the curing phase, since the foam material is at elevated temperatures, e.g. up to 160ºC for several hours (depending on the size of the block). The Use Category Document model (Jolly, 1998) estimated releases of 0.1% (i.e., 1 kg/ton) to air during the manufacture of FPUF in open mould systems.

Large blocks of FPUF typically are cut into smaller blocks or specialty sizes at the manufacturing facility. Some portion of the commercial pentaBDE product will be lost during the cutting process and also contained within the scrap FPUF. FPUF scrap is often recycled into carpet underlayment (rebond), particularly in the United States. At present, the EU is an exporter of scrap foam (around 40,000 tons/year) to the United States for use in carpet underlayment (ENDS, 1998). In the process, the scrap FPUF from various sources is shredded into small pieces and mixed with an adhesive under pressure to form large blocks of material. Handling this material also represents a possible route of exposure to the commercial pentaBDE product.

The potential exposure routes for workers engaged in FPUF manufacturing activities in the United States includes the following:

Primary Exposure Pathways

- Inhalation of vapors originating from the commercial pentaBDE product or FPUF;
- Dermal contact with the commercial pentaBDE product or FPUF;
- Incidental ingestion via hand-to-mouth contact in workplace;

Secondary Exposure Pathways

- Dermal contact with the commercial pentaBDE product or FPUF on dirty laundry brought to the home from the workplace;
- Incidental ingestion via hand-to-mouth contact after handling dirty laundry in the home;
- Dermal contact with the commercial pentaBDE product or FPUF brought indoors on floors in the home from shoes in the workplace; and,
- Incidental ingestion via hand-to-mouth contact after touching dirty floors in the home.

In addition to direct skin contact, two indirect exposure pathways involving dirty laundry and indoor floors in a home also were considered. For contact with dirty laundry and shoes, clothing

(i.e., socks, shirts, and pants) and shoes were assumed to contact the commercial pentaBDE product or FPUF during the FPUF manufacturing process. When clothing was laundered in the home, only the bare hands were assumed to contact dirty clothing. Dermal contact and incidental ingestion with dirty floors theoretically occurs when workers redistribute the chemicals of concern adhered to shoes across the entry level of the home, with subsequent exposure from contacting the dirty floor. Workers' children are also present in the home and were evaluated for these last two pathways.

Scenario #1c - Exposures to Cushion and Mattress Manufacturers, Distributors, and Recyclers using FPUF Containing the Commercial PentaBDE Product

FPUF is supplied to a countless number of end product manufacturers primarily involved in the manufacture of cushions, mattresses, and carpet padding. In some cases, end product manufacturers may shape FPUF blocks using a hot wire cutter, which may result in either trace volatilization or release of small particles containing the commercial pentaBDE product additive. The same exposure pathways identified for workers engaged in the manufacture of FPUF are likely applicable to workers engaged in shaping FPUF for use in the manufacture of cushions, distribution, and recycling mattresses and carpet padding.

Scenario #2 – Exposures to Adults and Children in the Indoor Home/School/Office Environments

Three different locations were evaluated in this scenario: the home, school and office environments. All three indoor environments typically contain different consumer upholstered FPUF products (e.g., the home generally contains bed mattresses, while schools and offices do not).

The potential exposure routes for adults and children in the indoor home/school/office environments in the United States includes the following:

Primary Exposure Pathways

- Inhalation of respirable particulates originating from covered FPUF products;
- Ingestion of particulates inhaled and swallowed;
- Dermal contact with indoor dust that contains FPUF;
- Incidental ingestion by young children (less than 5 years old) mouthing cushions containing FPUF treated with the commercial pentaBDE product; and,
- Incidental ingestion of indoor dust via hand-to-mouth contact.

It is possible, although extensive environmental measurements are lacking, that the commercial pentaBDE product may either volatilize, leach during washing, or be released via particulates from cushions, mattresses, and carpet padding during the lifetime of the consumer product in the home, school, or office environment. Anecdotal evidence from Hale et al. (2002) suggests that

FPUF cushions exposed to sunlight will degrade over time, releasing small particles of dried FPUF with minimal abrasion.

The commercial pentaBDE product has a very low vapor pressure; losses from FPUF due to volatilization are expected to be minimal and may not be measurable in the indoor home/school/office environments. An equation for estimating this possible loss for an additive in plastics is given by UCD (Jolly, 1994) as follows:

$$\text{Percentage loss due to volatilization} = 1.1 \times 10^{6} \times P \times N$$

Where:

P = vapor pressure of flame retardant (mmHg at 200°C); and,
N = service life of product (estimated to be 10 years for furniture foam).

This equation was derived for the loss of plasticizer additives in various plastics, and was used by ECB (2000) to estimate possible releases of the commercial pentaBDE product from FPUF as a worst case. Assuming a vapor pressure of 3.5×10^{-7} mmHg (4.69×10^{-5} Pa), the predicted loss of the commercial pentaBDE product during the lifetime of cushions and mattresses containing FPUF would be less than 0.39% per year over a 10-year product life cycle. At this trace level, human exposures from volatilization of FPUF are expected to be insignificant.

Similar to volatilization, leaching of the commercial pentaBDE product from FPUF used in furniture seating and in automobile and airplane seat cushions is expected to be an insignificant exposure pathway. Sales of FPUF treated with the commercial pentaBDE product ceased after approximately 1990; in general, FPUF seat cushions in automobiles no longer contain the commercial pentaBDE product. And, although it is possible that upholstered seat covers may be washed during the lifetime of furniture, it is highly unlikely that the actual foam cushion will be washed.

Children and adults are rarely exposed directly to FPUF in cushions and mattresses because these consumer products typically are covered with fabric. It is possible that the most likely source of exposure to the commercial pentaBDE product in FPUF is via direct or indirect contact with particles released into indoor air at the end of a product life cycle or by accidental destruction of the consumer product indoors. For young children (less than 5 years old), mouthing activity on cushions with or without exposed FPUF (e.g., through tears or holes in the fabric covering) also may be a source of exposure to the commercial pentaBDE product. This exposure pathway was evaluated only in the home environment, and not in the school or office environments in this exposure scenario.

Scenario #3 – Ambient Environmental Exposures to Children and Adults

In addition to the indoor home/school/office environments, children and adults may be exposed to the commercial pentaBDE product via releases to the ambient environment. Potential exposures are primarily through contact with ambient air, soil, and the consumption of food products, including human breast milk for infants. In addition, fish consumption through recreational fishing is another source of ambient environmental exposure. It is not well understood at this time to what extent (or how) the commercial pentaBDE product contributes to levels reported in various environmental compartments.

The potential exposure routes for adults and children associated with ambient levels in the U.S. environment includes the following:

Primary Exposure Pathways

- Inhalation of respirable particulates in ambient air;
- Ingestion of particulates inhaled and swallowed;
- Dermal contact with outdoor soil;
- Incidental ingestion of outdoor soil via hand-to-mouth contact;
- Ingestion of meat, dairy, and vegetable food products;
- Ingestion of recreationally caught fish; and,
- Ingestion of human breast milk by infants.

Because the commercial pentaBDE product is used mainly in FPUF, the potential for release of particulate waste from weathering, wear, etc., during the service life of consumer products containing FPUF treated with the commercial pentaBDE product is low. FPUF is used mainly in indoor applications (e.g. home and office furniture) and typically includes a protective covering. Releases to the environment could occur during or after disposal at the end of the product's life cycle, when FPUF particles containing the commercial pentaBDE product could be generated (ECB, 2000). The actual volume of FPUF treated with the commercial pentaBDE product disposed in landfills or incinerated in the United States is not known. In addition, releases to the environment are possible during accidental fires in the home or work place, but data are lacking. As a result, only the primary exposure pathways indicated above are included in the Tier 1 screening-level exposure assessment.

Tier 1 Human Exposure Concentration Values

A summary of the environmental and human exposure data used to support the Tier 1 exposure assessment is presented in Table 3-18. Rather than rely on environmental fate models to predict exposure point concentrations, the values used to calculate exposure were based on the most representative (yet protective) levels reported in available environmental studies. The values selected for use in the exposure assessment calculations were chosen to evaluate potential

workplace, indoor home/school/office, and ambient environmental exposures to the commercial pentaBDE product. Where data were reported for individual BDE congeners or groups, the congeners associated with the commercial pentaBDE product (hexaBDE and lower brominated BDEs) were summed and used in this Tier 1 assessment.

Table 3-18. Environmental and human exposure values used in the Tier 1 exposure assessment

Exposure Scenario	Exposure Media	Concentration	Units	Reference and Comments
Workplace	Indoor Air	18	$\mu g/m^3$	ECB (2000); based on the saturated vapor pressure of the commercial pentaBDE product
Home School Office	Indoor Air	1.3	ng/m^3	Sjödin et al. (2001a)
	Indoor Dust	410	ng/g	Knoth et al. (2002); represents the 95th upper confidence limit of the mean of the data set.
	Saliva	14	µg/L	ECB (2000) water solubility; based on 13.3 µg/L at 25°C; used to estimate mouthing behavior exposure
Ambient Environment	Ambient Air	52	pg/m^3	Strandberg et al. (2001)
	Soil	76	µg/kg	Hale et al. (2002)
	Leafy and Root Vegetables	0.134	ng/g ww	Ohta et al. (2002)
	Meat	0.0458	ng/g ww	Darnerud et al. (2000) (unpublished)
	Dairy	0.0182	ng/g ww	
	Other Fats and Oils	0.158	ng/g ww	
	Eggs	0.0425	ng/g ww	
	Fish Fillet	72	ng/g ww	Johnson and Olson (2001); Dodder et al. (2002); and Loganathan et al. (1995) as cited in Manchester-Neesvig et al. (2001); represents the 95th percent upper confidence limit of the mean of 72 fish fillet samples.
	Breast Milk	42.8	ng/g lipid weight	Ryan et al. (2002)

To evaluate the potential for exposure to the commercial pentaBDE product in the workplace, indoor air levels in GLCC workplaces where the commercial pentaBDE product is manufactured were conservatively based on the saturated vapor concentration of the commercial product.

To evaluate exposures to the commercial pentaBDE product in the indoor home/school/office environments, levels in indoor dust were based on the 95th percent upper confidence limit of the mean of the data set reported by Knoth et al. (2001). Indoor air levels were based on the data from Sjödin et al. (2001a). For mouthing behavior, the commercial pentaBDE product was

assumed to transfer to children at its solubility limit in water, which is assumed to be representative for saliva.

Environmental exposures to the commercial pentaBDE product through direct and indirect contact with soil was based on levels reported by Hale et al. (2002) in soil adjacent to a FPUF cushion manufacturing plant. Exposures to the commercial pentaBDE product in ambient air were based on levels reported in Chicago ambient air by Stranberg et al. (2001). In the absence of U.S. data, exposures through the food pathway were based on the results of market basket surveys reported by Darnerud et al. (2000; unpublished) for meat, dairy, eggs, and other fats and oils, and by Ohta et al. (2002) for vegetables. The 95th percent upper confidence limit of the mean of the data describing total BDE levels in edible fish tissue (see Table 3-8) was used to evaluate fish consumption by the general population and the additional consumption by recreational anglers in the United States. Infant exposures to the commercial pentaBDE product through consumption of human breast milk were based on the levels reported by Ryan et al. (2002). The data reported by Päpke et al. (2001) was not used because it is based on a single pooled sample of as many as 30 women and does not adequately represent the U.S. population.

3.11 Theoretical Exposure Pathways and Calculations

The Tier 1 exposure assessment presented in this section represents a screening-level analysis. The USEPA (2001) characterizes a Tier 1 exposure assessment as containing screening-level information on exposure from manufacturing activities, downstream processing and use activities, and specific information on children's (and relevant adult) exposure scenarios. The Agency defines the typical screening level exposure assessment as one involving readily available measured data, existing release and exposure estimates and other exposure-related information. Where actual measures of exposure are not available, the use of models is considered appropriate. USEPA has specified that the populations of concern to the VCCEPP are children and, in certain situations, prospective parents. Exposures that can affect children are those that occur prior to conception (to either parent), during prenatal development, and post-natal to the age of sexual maturation, which USEPA indicates is completed around 18-21 years of age. Although adult exposures are not intended to be a major focus, the Agency believes that prospective parent exposures are relevant to the evaluation of risks due to fertility and reproductive effects, as well as developmental effects from *in utero* exposures.

Exposure modeling was performed in accordance with USEPA exposure assessment guidelines (USEPA, 1992). In addition, other available exposure assessment procedures and guidance such as the Agency's draft exposure factors handbook for children (USEPA, 2000a), which consolidates all child exposure factors and related data contained in USEPA's *Exposure Factors Handbook* (USEPA, 1997), also was used to develop exposure models for the different exposure scenarios. The Agency's recent guidance on risk assessment of chemical mixtures (USEPA, 2001) also was consulted during the development of this Tier 1 assessment.

General Approach to Calculating Theoretical Exposures

Inhalation, incidental ingestion, dermal contact, and food consumption were evaluated in one or more of the three exposure scenarios (i.e., manufacturing workplace, adult and child exposure in the indoor home/school/office environments, and adult and child exposure to ambient environmental levels, including food, breast milk, and fish consumption) included in the Tier 1 exposure assessment of the commercial pentaBDE product. Both incidental ingestion and dermal contact were evaluated in all three scenarios. Airborne vapor inhalation was limited to workers, while inhalation of airborne particulates (and swallowing the exhaled fraction) was evaluated for indoor and ambient environmental exposures to adults and children. Food consumption was also evaluated. Exposures through the consumption of surface water or groundwater were not included in the Tier 1 assessment (see Section 3.5).

In this Tier 1 assessment, exposures to the commercial pentaBDE product by workers and by adults and children in the indoor home/school/office environments and through ambient environmental exposures were evaluated by modeling the physical characteristics and behaviors of the reasonably highest exposed individual, similar to the description of the RME profile described in USEPA (1989) risk assessment guidance. The RME was defined as an individual who is of typical behavior and physical characteristics and receives the high end of the reasonable range of exposure (in this assessment, either the 95th percentile upper confidence limit of the mean exposure or the upper end of the range of available data). The theoretical chronic daily intakes (CDIs) for non-cancer effects were calculated for dermal, oral and inhalation uptake. The relevant CDIs were summed to provide the cumulative, or total, daily intake by each receptor group evaluated in the three exposure scenarios. Cancer risks were not evaluated in this Tier 1 assessment, so lifetime averaging of intake was not required.

The exposure assumptions and values used to calculate theoretical exposures are presented in Table 3-19 for workers and workers' children; Table 3-20 for adults and children in the home, school, and office environments; and Table 3-21 for adults and children exposed to ambient environmental levels and through consumption of food, fish, and breast milk (infants only). Oral, dermal, and inhalation bioavailability values are summarized in Table 2-7.

Table 3-19. Exposure assumptions and parameters used to calculate theoretical exposures to the commercial pentaBDE product by primary and chain-of-commerce manufacturing workers and the children of workers.

Exposure Parameter	Receptors																			
	Variable	Units	Primary Production Drummer, Male	Primary Production Drummer, Female	Primary Production Cleaner, Male	Primary Production Cleaner, Female	Foam Manufacture Mixer, Male	Foam Manufacture Mixer, Female	Foam Manufacture Cutter, Male	Foam Manufacture Cutter, Female	Mattress Manufacturer, Male	Mattress Manufacturer, Female	Cushion Manufacturer, Male	Cushion Manufacturer, Female	Furniture Manufacturer, Male	Furniture Manufacturer, Female	Carpet Pad Manufacturer, Male	Carpet Pad Manufacturer, Female	Carpet Installer, Male	Carpet Installer, Female	Distributor Covered, Male
Physiological and General Assumptions																					
Body Weight	BW	kg	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1	65.4	78.1
Inhalation Rate	InR	m3/day	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8	12.8
Exposed Skin Surface Area, Total Body	SAt	cm2	19400	16900	19400	16900	19400	16900	19400	16900	19400	16900	19400	16900	19400	16900	19400	16900	19400	16900	19400
Exposed Skin Surface Area, Palms of Hands	SAh	unitless	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026	0.024	0.026
Fraction of Skin Surface Area Exposed while Doing Laundry	FSAl	unitless	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Fraction of Skin Surface Area Exposed to Floor	FSAf	unitless	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391	0.305	0.391
Fraction of Skin Surface Area that Contacts Mouth	FSAftip	unitless	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002
Fraction of Laundry that Contains the Commercial Product	FCLdy	unitless	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25
Floor Surface Area	FSA	cm2	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644	756644
Fraction of Skin Surface Area that is Shoes	SAs	cm2	613	613	613	613	613	613	613	613	613	613	613	613	613	613	613	613	613	613	613
Exposure Frequency Assumptions																					
Fraction of Work Day Spent Around Commercial Product	FWD	unitless	0.33	0.33	0.33	0.33	0.33	0.33	1	1	1	1	1	1	1	1	1	1	1	1	1
Exposure Frequency, Doing Laundry	EFL	hr/day	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22	0.22
Exposure Frequency, Contact with Floors	EFF	hr/day	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
Environmental Assumptions																					
Absorption Factor, Oral Route	AFo	unitless	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Absorption Factor, Inhalation Route	AFi	unitless	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
Absorption Factor, Dermal Route	AFd	unitless	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313
Chemical Potency Assumptions																					
Reference Dose, oral and inhalation	RfD	mg/kg-day	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002
Thyroid Effects Benchmark, oral and inhalation	TEB	mg/kg-day	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Developmental Effects Benchmark, oral and Inhalation	DEB	mg/kg-day	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07
Chemical Assumptions																					
Saturated Vapor Concentration	SVC	µg/m3	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18
Fraction of Saturated Vapor Concentration in Workplace Air	Fsvc	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Total Particulate Concentration	AirTP	mg/m3	4.6	4.6	4.6	4.6	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67	1.67
Fraction of Particulate Phase that is Respirable	Fresp	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Fraction of Particulate that is Commercial Product	Fpp	unitless	0.75	0.75	0.75	0.75	0.75	0.75	0.0375	0.0375	0.0225	0.0225	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375
Fraction of Handled Item that is Commercial Product	Fip	unitless	0.75	0.75	0.75	0.75	0.75	0.75	0.0375	0.0375	0.0225	0.0225	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375	0.0375
Adherence Rate of Commercial Product on Skin	ARskin	mg/cm2-day	0.1	0.1	0.1	0.1	0.1	0.1	1	1	1	1	1	1	1	1	1	1	1	1	1
Adherence Rate of Commercial Product on Shoes	ARshoe	mg/cm2-day	0.1	0.1	0.1	0.1	0.1	0.1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table 3-19. Exposure assumptions and parameters used to calculate theoretical exposures to the commercial pentaBDE product by primary and chain-of-commerce manufacturing workers and the children of workers.

Exposure Parameter	Receptors							Worker's Children Receptors									Reference and Notes
	Variable	Units	Distributor Covered, Female	Distributor Uncovered, Male	Distributor Uncovered, Female	Recycler, Male	Recycler, Female	<1 yr Child	1-2 yr Child	3-5 yr Child	6-8 yr Child	9-11 yr Male Child	9-11 yr Female Child	12-14 yr Male Child	12-14 yr Female Child	15-18 yr Male Child	15-18 yr Female Child	
Physiological and General Assumptions																	
Body Weight	BW	kg	65.4	78.1	65.4	78.1	65.4	9.1	11.3	16.35	23.75	33.75	34	47.05	48.65	64.93	57.6	USEPA (1997) EFH, page 7-4, table 7-3 for children; table 7-2 for adults; mean values
Inhalation Rate	InR	m^3/day	12.8	12.8	12.8	12.8	12.8	--	--	--	--	--	--	--	--	--	--	USEPA (1997) EFH, page 5-24, table 5-23; value of 1.6 m3/hr for adult moderate work activity multiplied by 8-hr workday
Exposed Skin Surface Area, Total Body	SAt	cm2	16900	19400	16900	19400	16900	5,910	5,910	7,200	9,250	11,600	11,600	14,900	14,800	17,500	16,000	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents total body skin surface area
Exposed Skin Surface Area, Palms of Hands	SAh	unitless	0.024	0.026	0.024	0.026	0.024	--	--	--	--	--	--	--	--	--	--	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value for the palms of both hands represents 1/2 the estimate for the skin surface area of both hands
Fraction of Skin Surface Area Exposed while Doing Laundry	FSAl	unitless	1	1	1	1	1	--	--	--	--	--	--	--	--	--	--	Assumes entire hand skin surface contacts dirty laundry
Fraction of Skin Surface Area Exposed to Floor	FSAf	unitless	0.305	0.391	0.305	0.391	0.305	0.337	0.342	0.379	0.361	0.374	0.374	0.383	0.383	0.403	0.403	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents the estimate for the skin surface area of the hands, arms, legs, and feet
Fraction of Skin Surface Area that Contacts Mouth	FSAftip	unitless	0.002	0.002	0.002	0.002	0.002	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.003	0.002	0.002	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value for the fingertips of all ten digits of both hands represents 1/2 the estimate for the skin surface area of both hands and assumes 25% of each hand is fingertips that could be placed in the mouth
Fraction of Laundry that Contains the Commercial Product	FCLdy	unitless	0.25	0.25	0.25	0.25	0.25	--	--	--	--	--	--	--	--	--	--	Assumes that 1/4 of the laundry may contain the product on clothing used in the workplace
Floor Surface Area	FSA	cm2	756644	756644	756644	756644	756644	--	--	--	--	--	--	--	--	--	--	USEPA (1997) EFH; table 17-31 central estimate; assume two-story home
Fraction of Skin Surface Area that is Shoes	SAs	cm2	613	613	613	613	613	--	--	--	--	--	--	--	--	--	--	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile; value for the soles of both feet represents 1/2 the estimate for the skin surface area of the feet
Exposure Frequency Assumptions																		
Fraction of Work Day Spent Around Commercial Product	FWD	unitless	1	1	1	1	1	--	--	--	--	--	--	--	--	--	--	For primary production and mixing of product with FPUF ingredients, one-third of the day is assumed spent handling the primary product directly; for other work categories, the entire work day is assumed to involved handling materials containing the product
Exposure Frequency, Doing Laundry	EFL	hr/day	0.22	0.22	0.22	0.22	0.22	--	--	--	--	--	--	--	--	--	--	USEPA (1997) EFH, page 15A-22, table 15A-5; mean value of 13.35 min/weekday
Exposure Frequency, Contact with Floors	EFF	hr/day	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	Assume workday evenings are spent on floor by adults and older children, youngest children are often in cribs or play pens, and pre-schoolers spend the most time on the floor
Environmental Assumptions																		
Absorption Factor, Oral Route	AFo	unitless	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	ECB (2000) RA, page 132; high end of 70-90% range for oral absorption of PBBs and PCBs
Absorption Factor, Inhalation Route	AFi	unitless	0.75	0.75	0.75	0.75	0.75	--	--	--	--	--	--	--	--	--	--	ECB (2000); assumes 75% absorption by the inhalation route
Absorption Factor, Dermal Route	AFd	unitless	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	Inveresk Research (2001); mean dermal delivery of applied dose of radio-labeled tetrabromodiphenyl ether, 12 samples of human skin
Chemical Potency Assumptions																		
Reference Dose, oral and inhalation	RfD	mg/kg-day	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	USEPA (2003) IRIS; based on Carlson (1980b) rat study that administered commercial product mixture (24.6% tetra, 58.1% penta, 13.3% hexa, 2.6% hepta, 0.3% octa, 0.2% nona, 0.8% deca); critical effect: induction of hepatic enzymes; uncertainty factor = 1000, modifying factor = none; low level of confidence in RfD
Thyroid Effects Benchmark, oral and inhalation	TEB	mg/kg-day	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	IRDC (1976); based on the incidence of thyroidf hyperplasia in rodents
Developmental Effects Benchmark, oral and Inhalation	DEB	mg/kg-day	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	Zhou et al. (2002); based on changes in T4 levels in the rodent fetus or neonate exposed in utero
Chemical Assumptions																		
Saturated Vapor Concentration	SVC	µg/m3	18	18	18	18	18	--	--	--	--	--	--	--	--	--	--	ECB (2000) RA, page 125; saturated vapor concentration calculated to be 0.0007 ppm at 25°C.
Fraction of Saturated Vapor Concentration in Workplace Air	Fsvc	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	Assumes that one-half of the SVC value cited in ECB (2000) reaches ambient air in the workplace
Total Particulate Concentration	AirTP	mg/m3	1.67	1.67	1.67	1.67	1.67	--	--	--	--	--	--	--	--	--	--	GLCC (2002), for primary product workers took the high value of three samples taken in the packaging area of a primary product processing plant, data are for DE-79, an octa mixture (no data available for penta mixture); Breysse (2000),for other workers took average dust value from nine facilities that use deca mixture (no data available for penta mixture), personal sample used rather than area sample
Fraction of Particulate Phase that is Respirable	Fresp	unitless	0.5	0.5	0.5	0.5	0.5	--	--	--	--	--	--	--	--	--	--	Standard value used by regulatory agencies
Fraction of Particulate that is Commercial Product	Fpp	unitless	0.0375	0.0375	0.0375	0.0375	0.0375	--	--	--	--	--	--	--	--	--	--	GLCC (personal communication); 75% of commercial product is penta mix, 2-3% of mattress foam is commercial product, and 3-5% of cushion is commercial product (used upper end of ranges)
Fraction of Handled Item that is Commercial Product	Fip	unitless	0.0375	0.0375	0.0375	0.0375	0.0375	--	--	--	--	--	--	--	--	--	--	GLCC (personal communication); 75% of commercial product is penta mix, 2-3% of mattress foam is commercial product, and 3-5% of cushion is commercial product (used upper end of ranges)
Adherence Rate of Commercial Product on Skin	ARskin	mg/cm2-day	1	1	1	1	1	--	--	--	--	--	--	--	--	--	--	ECB (2000) RA, page 128; lower end of ranges predicted by EASE program for primary product (0.1 to 1.0 mg/cm2-day) and foam (1 to 5 mg/cm2-day) handling
Adherence Rate of Commercial Product on Shoes	ARshoe	mg/cm2-day	1	1	1	1	1	--	--	--	--	--	--	--	--	--	--	Assumes that the bottom of shoes are equivalent to bare skin

Table 3-20. Exposure assumptions and parameters used to calculate theoretical exposures to the commercial pentaBDE product by adults and children in the home, school, and office environment

Exposure Parameter	Receptors														Reference and Notes
	Variable	Units	<1 yr Child	1-2 yr Child	3-5 yr Child	6-8 yr Child	9-11 yr Male Child	9-11 yr Female Child	12-14 yr Male Child	12-14 yr Female Child	15-18 yr Male Child	15-18 yr Female Child	Adult Male (Prospective Parent)	Adult Female (Prospective Parent)	
Physiological and General Assumptions															
Body Weight	BW	kg	9.1	11.3	16.35	23.75	33.75	34	47.05	48.65	64.93	57.6	78.1	65.4	USEPA (1997) EFH, page 7-4, table 7-3 for children; table 7-2 for adults; mean values
Inhalation Rate	InR	m3/day	4.5	6.8	8.3	10	14	13	15	12	17	12	15.2	11.3	USEPA (1997) EFH, page 5-24, table 5-23; value of 1.6 m3/hr for adult moderate work activity multiplied by 8-hr workday
Indoor Dust Ingestion Rate	DIR	mg/day	10	10	10	3	3	3	3	3	3	3	0.56	0.56	USEPA (1997) EFH, page 4-12, table 4-11 for children (winter indoor rate); page 4-17, table 4-16 for adults
Exposed Skin Surface Area, Total Body	SAt	cm2	5,910	5,910	7,200	9,250	11,600	11,600	14,900	14,800	17,500	16,000	19,400	16,900	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents total body skin surface area
Fraction of Skin Surface Area Exposed to Floor	FSAf	unitless	0.337	0.342	0.379	0.361	0.374	0.374	0.383	0.383	0.403	0.403	0.319	0.305	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents the estimate for the skin surface area of the hands, arms, legs, and feet
Fraction of Skin Surface Area Exposed at School	FSAs	unitless	0.272	0.279	0.306	0.289	0.301	0.301	0.308	0.308	0.332	0.332	0.251	0.238	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents the estimate for the skin surface area of the hands, arms, and legs
Fraction of Skin Surface Area Exposed at Office	FSAo	unitless	--	--	--	--	--	--	--	--	--	--	0.119	0.109	USEPA (2001) Dermal Exposure Guidance, Exhibit C-1 page C-3 and C-4; 50th percentile, though in some cases the value represents the midpoint of the age bin; value represents the estimate for the skin surface area of the hands and arms
Salivary Flow-rate in the Child's Mouth	Vs	ml/minute	0.22	0.22	0.22	--	--	--	--	--	--	--	--	--	Watanabe et al. (1990); based on unstimulated salivary flow-rate in 5-year old children
Fractional Rate of Extraction by Saliva	FR	unitless	0.038	0.038	0.038	--	--	--	--	--	--	--	--	--	NRC (2000) upper end of range for laundering tests of fabric with flame retardant backing
Exposure Duration/Frequency Assumptions															
Exposure Frequency at Home	EFh	hr/day	24	24	20	17	17	17	17	17	17	17	24	24	For children, time at school subtracted from 24-hour day; adults assumed to be stay-at-home parents
Exposure Frequency at School	EFs	hr/day	0	0	4	7	7	7	7	7	7	7	8	8	Assume 4-hour school day for pre-school and kindergarten, 7-hour school day for older children, and adult teachers work 8 hours/day
Exposure Frequency at Office	EFo	hr/day	0	0	0	0	0	0	0	0	0	0	8	8	Assume 8-hour work day at office
Exposure Frequency Mouthing All Items	EFmouth	minutes/day	9	5.5	2	0	0	0	0	0	0	0	0	0	USEPA (2000a) CEFH, page 6-9; weighted average for mouthing behavior from two studies for children 3 to 60 months; adjusted the reported average of 46 min/day to 1 hr/day
Environmental Assumptions															
Absorption Factor, Oral Route	AFo	unitless	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	ECB (2000) RA, page 132; high end of 70-90% range for oral absorption of PBBs and PCBs
Absorption Factor, Inhalation Route	AFi	unitless	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	ECB (2000); assumes 75% absorption by the inhalation route
Absorption Factor, Dermal Route	AFd	unitless	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	Inveresk Research (2001); mean dermal delivery of applied dose of radio-labeled tetrabromodiphenyl ether, 12 samples of human skin

Table 3-20. Exposure assumptions and parameters used to calculate theoretical exposures to the commercial pentaBDE product by adults and children in the home, school, and office environment

Exposure Parameter	Receptors														Reference and Notes
	Variable	Units	<1 yr Child	1-2 yr Child	3-5 yr Child	6-8 yr Child	9-11 yr Male Child	9-11 yr Female Child	12-14 yr Male Child	12-14 yr Female Child	15-18 yr Male Child	15-18 yr Female Child	Adult Male (Prospective Parent)	Adult Female (Prospective Parent)	
Toxicological Assumptions															
Reference Dose, Oral and Inhalation	RfD	mg/kg-day	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	USEPA (2003) IRIS; based on Carlson (1980b) rat study that administered commercial product mixture (24.6% tetra, 58.1% penta, 13.3% hexa, 2.6% hepta, 0.3% octa, 0.2% nona, 0.8% deca); critical effect: induction of hepatic enzymes; uncertainty factor = 1000, modifying factor = none; low level of confidence in RfD
Thyroid Effects Benchmark, Oral and Inhalation	TEB	mg/kg-day	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	IRDC (1976); based on the incidence of thyroidf hyperplasia in rodents
Developmental Effects Benchmark, Oral and Inhalation	DEB	mg/kg-day	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	Zhou et al. (2002); based on changes in T4 levels in the rodent fetus or neonate exposed in utero
Chemical Assumptions															
Product Concentration in Dust at Home, School and Office	Cdust	ng/g	784	784	784	784	784	784	784	784	784	784	784	784	Knoth (2002); 95th upper confidence limit of the mean of 23-25 samples collected from vacuum cleaner bags from German households; sum of BDE-47,49,85,99,100,153,154 (hepta and deca excluded)
Product Concentration in Particulate Phase at Home, School and Office	AirP	ng/m3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	Sjodin (2001a); air concentration of particle-associated and semivolatile compounds associated with penta commercial product; sum of BDE-47,85,99,100,153 and 154 (hepta and deca excluded), high of two air samples collected in a teaching hall with computers in Sweden
Fraction of Particulate Phase that is Respirable	Fresp	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	Standard value used by regulatory agencies
Adherence Rate of Dust to Skin	ARdust	mg/cm2-day	0.050	0.050	0.050	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	USEPA (2001) RAGS Part E, page 3-17, assumed same as soil adherence rate; 50th percentile of high-end activity (gardening) for adults; For children under 6 yr, USEPA (2001) Exhibit C-3, p.C-6 representing the 95th percentile for indoor daycare children.
Commercial Product Concentration in FPUF Cushions	Ccush	mg/cm3	0.75	0.75	0.75	--	--	--	--	--	--	--	--	--	Concentration calculated based on average content (7.5%) of commercial product in FPUF, typical volume (10 cm3) of a furniture cushion.
Water Solubility of Commercial PentBDE Product	WS	µg/L	14	14	14	14	14	14	14	14	14	14	14	14	Reported as 13.3 ug/L in GLCC (2002 MSDS; see Appendix II) Materials Safety Data Sheet for the commercial pentaBDE product; the value was rounded to 14 for this assessment.

Table 3-21. Exposure assumptions and parameters used to calculate theoretical exposures to the commercial pentaBDE product by adults and children exposed to ambient environmental levels and through consumption of foods, fish, and breast milk (Infants Only)

Exposure Parameter	Receptor Scenarios																		Reference and Notes
	Variable	Units	<1 yr Child	1-2 yr Child	3-5 yr Child	6-8 yr Child	9-11 yr Male Child	9-11 yr Female Child	12-14 yr Male Child	12-14 yr Female Child	15-18 yr Male Child	15-18 yr Female Child	Adult Male (Prospective Parent)	Adult Female (Prospective Parent)	Recreational Marine Angler Adult Male	Recreational Marine Angler Adult Female	Recreational Freshwater Angler Adult Male	Recreational Freshwater Angler Adult Female	
Physiological and General																			
Body Weight	BW	kg	9.1	11.3	16.35	23.75	33.75	34	47.05	48.65	64.93	57.6	78.1	65.4	78.1	65.4	78.1	65.4	USEPA (1997) EFH, page 7-4, table 7-3 for children; table 7-2 for adults; mean values
Inhalation Rate	InR	m3/day	4.5	6.8	8.3	10	14	13	15	12	17	12	15.2	11.3	--	--	--	--	USEPA (1997) EFH, page 5-24, table 5-23
Soil Ingestion Rate	SIR	mg/day	200	200	200	200	200	200	200	200	200	200	100	100	100	100	100	100	USEPA (1997) EFH, page 4-21, recommended value for adults (agricultural/residential setting); page 4-20, recommended conservative estimate for children
Fish Consumption Rate	CRfish	g/day	1.001	4.181	5.232	6.175	8.775	8.84	9.41	9.73	12.986	11.52	20.1	20.1	7.2	7.2	17	17	USEPA (2000a) CEFH, page 3-78, table 3-35, Mean values for children; USEPA (1997) EFH, page 10-79, table 10-81, Mean Intake for total fish; page 10-79, tables 10-83 and 84, Highest Mean intake for anglers
Meat (Beef, Poultry and Pork) Consumption Rate	CRmeat	g/day	53.69	115.26	153.69	161.5	229.5	231.2	230.545	238.385	318.157	282.24	398.31	333.54	--	--	--	--	USEPA (2000a) CEFH, page 3-78, table 3-35, 95th percentile values for children; USEPA (1997) EFH, page 11-31, table 11-30, 95th percentile values for adults
Dairy Consumption Rate	CRdairy	g/day	2138.5	1019.26	797.88	795.625	1130.625	1139	837.49	865.97	1155.754	1025.28	2319.57	1942.38	--	--	--	--	USEPA (2000a) CEFH, page 3-78, table 3-35, 95th percentile values for children; USEPA (1997) EFH, page 11-31, table 11-30, 95th percentile values for adults
Other Fats and Oils Consumption Rate	CRfat	g/day	3.3	16	22	30	30	30	49	49	49	49	118.28	76.52	--	--	--	--	USEPA (2000a) CEFH, page 3-45, table 3-19, 95th percentile values for children (no overlap with other fatty foods); USEPA (1997) EFH, page 11-28, table 11-27, mean values for adult total fat intake (overlaps other fatty foods)
Egg Consumption Rate	CRegg	g/day	38	70	63	66	66	66	88	88	88	88	83.33	69.78	--	--	--	--	USEPA (2000a) CEFH, page 3-45, table 3-35, 95th percentile values for children; USEPA (1997) EFH, page 9-44, table 9-29, 95th percentile values for adults
Vegetable Consumption Rate	CRveg	g/day	220.22	263.29	299.205	320.625	455.625	459	437.565	452.445	603.849	535.68	781.00	654.00					USEPA (2000a) CEFH, page 3-78, table 3-19, 95th percentile values for children; USEPA (1997) EFH, page 11-15, table 11-7, 95th percentile values for adults
Breast Milk Consumption Rate	CRbm	mL/day	980	0	0	0	0	0	0	0	0	0	0	0	--	--	--	--	USEPA (2000a) CEFH, page 2-19, table 2-12 ,12-month average
Fraction of Breast Milk that is Lipid	Fbm	unitless	0.04	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	--	--	--	--	USEPA (2000a) CEFH, page 2-19, table 2-12 , footnote
Exposed Skin Surface Area, Total Body	SAt	cm2	5,910	5,910	7,200	9,250	11,600	11,600	14,900	14,800	17,500	16,000	19,400	16,900	--	--	--	--	USEPA (1997) EFH , page 6-14, table 6-4 for adults; page 6-15, tables 6-6 and 6-7 for children; mean values
Fraction of Skin Surface Area Exposed Outdoors	FSAout	unitless	0.337	0.342	0.379	0.361	0.374	0.374	0.383	0.383	0.403	0.403	0.319	0.305	--	--	--	--	USEPA (1997) EFH, page 6-14, table 6-5 for adults; page 6-16, table 6-8 for children; mean value for head + hands + arms + legs + feet
Environmental Assumptions																			
Absorption Factor, Oral Route	AFo	unitless	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	ECB (2000) RA, page 132; high end of 70-90% range for oral absorption of PBBs and PCBs
Absorption Factor, Inhalation Route	AFi	unitless	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	ECB (2000); assumes 75% absorption by the inhalation route
Absorption Factor, Dermal Route	AFd	unitless	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	0.0313	Inveresk Research (2001); mean dermal delivery of applied dose of radio-labeled tetrabromodiphenyl ether, 12 samples of human skin
Chemical Potency Assumptions																			
Reference Dose, Oral	RfD	mg/kg-day	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	USEPA (2003) IRIS; based on Carlson (1980b) rat study that administered commercial product mixture (24.6% tetra, 58.1% penta, 13.3% hexa, 2.6% hepta, 0.3% octa, 0.2% nona, 0.8% deca); critical effect: induction of hepatic enzymes; uncertainty factor = 1000, modifying factor = none; low level of confidence in RfD
Thyroid Effects Benchmark, Oral and Inhalation	TEB	mg/kg-day	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	IRDC (1976); based on the incidence of thyroidf hyperplasia in rodents
Developmental Effects Benchmark, Oral and Inhalation	DEB	mg/kg-day	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	0.07	Zhou et al. (2002); based on changes in T4 levels in the rodent fetus or neonate exposed in utero
Chemical Assumptions																			
Soil Adherence to Skin Rate	SAR	mg/cm2-day	0.200	0.200	0.200	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	0.070	USEPA (2001) RAGS Part E, page 3-17; 50th percentile of high-end activity (gardening) for adults, 95th percentile value for children under 6 yr
Air Concentration of Product in Particulate Phase	AirP	pg/m3	52	52	52	52	52	52	52	52	52	52	52	52	52	52	52	52	Strandberg et al.(2001); urban location (Chicago) over three-year period (1997-99), total of BDE-47,99,100,153,154 (excludes deca)
Fraction of Particulate Phase that is	Fresp	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	Standard value used by regulatory agencies
Concentration in Soil	Csoil	µg/kg	76	76	76	76	76	76	76	76	76	76	76	76	76	76	76	76	Hale et al. (2002); Soil near foam production plant in mid-Atlantic region, USA
Concentration in Fish	Cfish	ng/g wet wt	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	71.67	The 95th percent upper confidence limit of the mean value of 72 fish fillet samples from the USA, total PBDE
Concentration in Meat (Beef, Poultry	Cmeat	ng/g wet wt	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	0.0458	Darnerud et al. (2000; unpublished); Swedish market basket study, sum
Concentration in Dairy	Cdairy	ng/g wet wt	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	0.0182	Darnerud et al. (2000; unpublished); Swedish market basket study, sum of BDE-47,99,100,153,154
Concentration in Other Fats and Oils	Cfat	ng/g wet wt	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158	Darnerud et al. (2000; unpublished); Swedish market basket study, sum
Concentration in Eggs	Cegg	ng/g wet wt	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	0.0425	Darnerud et al. (2000; unpublished); Swedish market basket study, sum
Concentration in Vegetables	Cveg	ng/g wet wt	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	0.134	Ohta et al. (2002); Concentration detected in spinach, sum of BDE-
Concentration in Breast Milk	Cbm	ng/g lipid	42.8	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	Ryan et al. (2002); 95 UCL of the sum of BDE-28,47,99,100,153,154,183 (minor contribution according to graph); 20 samples from Vancouver, Canada, collected in 2001-2002

The exposure models created in Microsoft Excel® used to calculate exposure to the commercial pentaBDE product are presented in Appendix VI. A summary of the exposure variables used in the different exposure model calculations is presented in Table 3-22.

Table 3-22. Exposure variables and their definitions used in the Tier 1 exposure model calculations

Variable	Definition
A	area of cushion fabric mouthed (cm^2/hr)
AFd	dermal absorption factor (unitless)
AFi	inhalation absorption factor (unitless)
AFo	oral absorption factor (unitless)
AirP	product concentration in particulate phase at home, school and office (ng/m^3)
AirTP	total particulate concentration (mg/m^3)
ARdust	adherence rate of dust to skin (mg/cm^2-day)
ARshoe	adherence rate of item to shoe (mg/cm^2-day)
ARskin	adherence rate of handled item to skin (mg/cm^2-day)
BW	body weight (kg)
Cbm	concentration in breast milk (ng/g lipid weight)
Ccush	product concentration on surface of cushion (mg/cm^2)
CDI	chronic daily intake (mg/kg-day)
Cdairy	concentration in dairy (ng/g ww)
Cdust	concentration of the commercial pentaBDE product in indoor dust (ng/g)
Cegg	concentration in egg (ng/g ww)
CF	a conversion factor
Cfat	concentration in other fats and oils (ng/g ww)
Cfish	concentration in fish (ng/g ww)
Cfloor	concentration of the commercial pentaBDE product on the floor (mg/cm^2) for the maximally exposed worker and is calculated using the variable definitions provided in the previous section
Cmeat	concentration in meat, beef, poultry, and pork (ng/g ww)
Cveg	concentration in vegetables (ng/g ww)
CRbm	breast milk consumption rate (mL/day)
CRdairy	dairy consumption rate (g/day)
CRegg	egg consumption rate (g/day)
CRfat	other fats and oil consumption rate (g/day)
CRfish	fish consumption rate (g/day)
CRmeat	meat (beef, poultry, and pork) consumption rate (g/day)
CRveg	vegetable consumption rate (g/day)
CSF	cancer slope factor (mg/kg-day) $^{-1}$
Csoil	concentration of the commercial pentaBDE product in outdoor soil (µg/kg)
DEB	developmental effects benchmark, oral and inhalation (mg/kg-day)
DIR	indoor dust ingestion rate (mg/day)
EFF	exposure frequency for contact with floors (hr/day)
EFh/s/o	exposure frequency at home/school/office (hr/day)
EFL	average amount of time spent daily doing laundry (hr/day)
EFmouth	exposure frequency mouthing all items (minutes/day)
Fbm	fraction of breast milk that is lipid (unitless)
FCLdy	fraction of laundry that is contaminated (unitless)
Fcush	fraction of items mouthed that are cushions (unitless)
Fip	fraction of handled item that is commercial product (unitless)
Fpp	fraction of particulate that is commercial product (unitless)
FQ	frequency of hand-to-mouth activity in the workplace (events/day)

Variable	Definition
FR	fractional rate of extraction by saliva (unitless)
Fresp	fraction of particulate phase that is respirable (unitless)
FSA	floor surface area (cm^2)
FSAf	fraction of skin surface area exposed to floor (unitless)
FSAftip	fraction of skin surface area that contacts mouth (unitless)
FSAw	fraction of skin surface area exposed at work (unitless)
FSAh/s/o	fraction of skin surface area exposed at home/school/office (unitless)
FSAl	fraction of skin surface area exposed while doing laundry (unitless)
FSAout	fraction of skin surface area exposed outdoors (unitless)
FSAs	fraction of skin surface area that is shoes (unitless)
Fsvc	fraction of saturated vapor concentration in workplace air (unitless)
FWD	fraction of work day spent around commercial product (unitless)
InR	inhalation rate (m^3/day)
RfD	reference dose, oral and inhalation (mg/kg-day)
SAR	adherence rate of soil to skin (mg/cm^2-day)
SAt	exposed skin surface area, total body (cm^2)
SIR	soil ingestion rate (mg/day)
SVC	saturated vapor concentration (µg/m^3)
TEB	thyroid effects benchmark, oral and inhalation (mg/kg-day)
Vs	salivary flow rate in children under 5 years of age (mL/minute)
WS	water solubility limit for the commercial pentaBDE product

Dermal Uptake Pathway

Dermal uptake is a function primarily of the degree of skin contact. The exposed skin surface area of primary and chain-of-commerce manufacturing workers was assumed to include only the palms of the hand. Restrictions on direct contact with primary product or FPUF by workers (e.g., by wearing gloves) were not considered, even though the sap-like and highly viscous nature of the commercial pentaBDE product typically necessitates the use of protective gloves to avoid direct skin contact. Laundry contact was limited to the hands only. Floor contact included hands, arms, legs and bare feet. For adults and children, exposed skin contact with indoor dust varied according to the three settings: home (hands, arms, legs and feet), school (hands, arms and legs) and office (hands and arms). Values for exposed skin surface areas were adopted from USEPA (2001, 2000a, 1997).

Assumptions regarding soil and dust adherence to skin were adopted from USEPA guidance for dermal risk assessment (USEPA, 2001, 2000a, 1997). The fraction of material containing the commercial pentaBDE product was based on information from GLCC. Laundry was assumed to be 25% soiled for all job categories. Soil adherence to shoes was assumed to be the same as that of bare skin.

Exposure Pathway – Direct Dermal Contact

Workers. The potential for dermal contact with the commercial pentaBDE product was calculated using the following equation:

CDI = SAt * SAh * ARskin * Fip * AFd * FWD * 1/BW

Exposure Pathway – Dermal Contact with Dirty Laundry in the Home

Workers. The potential for dermal contact with the commercial pentaBDE product from handling dirty laundry was calculated using the following equation:

CDI = SAt * SAh * FSAl * ARskin * FCLdy * AFd * EFL * CF2 * 1/BW

Exposure Pathway – Dermal Contact with Floor Surfaces Contaminated by Dirty Shoes in the Home

Workers. The standard home floor surface area (756,644 cm^2) that potentially retains dirt from outdoor sources was adopted from USEPA (1997). The potential for dermal contact with the commercial pentaBDE product from touching floor surfaces that have been contaminated by dirty shoes brought home from the workplace was calculated using the following equation:

Cfloor = (SAs * ARshoe) / FSA
CDI = Cfloor * SAt * FSAf * AFd * EFF * CF2 * 1/BW

Workers' Children. The potential for dermal contact with the commercial pentaBDE product by children touching floor surfaces that have been contaminated by a parent's dirty shoes brought home from the workplace was calculated using the following equation:

Cfloor = (SAs * ARshoe) / FSA
CDI = Cfloor * SAt * FSAf * AFd * EFF * CF2 * 1/BW

Exposure Pathway – Dermal Contact with Indoor Dust

Adults and Children in the Home, School and Office. The potential for dermal contact with the commercial pentaBDE product in indoor dust in the home, school and office was calculated using the following equation:

CDI = Cdust * CF1 * ARdust * SAt * FSAh/s/o * AFd * EFh/s/o * CF3 * 1/BW

Exposure Pathway – Dermal Contact with Outdoor Soil

Adult and Child Ambient Environmental Exposures. The potential for dermal contact with the commercial pentaBDE product in outdoor soil was calculated using the following equation:

CDI = Csoil * CF1 * SAR * SAt * FSAout * AFd * 1/BW

Incidental Ingestion

Incidental ingestion via hand-to-mouth contact was modeled for workers and workers' children. The person's fingertips were assumed to contact the mouth and transfer any adhered commercial pentaBDE product to the mouth. Swallowing and absorption were assumed to follow. The fraction of both hands contacting the mouth was assumed to include the fingertips of all ten fingers. For adults and children, dust or soil ingestion rates were used to calculate incidental ingestion exposures. Exposure parameter values were adopted primarily from the USEPA Exposure Factors Handbook (1997) and USEPA risk assessment guidelines (1989).

Estimates of soil ingestion by the average adult and children of different ages are the focus of continued scientific study (Paustenbach, 2000). The dust ingestion values used in this Tier 1 assessment are from USEPA *Exposure Factors Handbook* (1997) as follows: 100 mg/day for the youngest children, 3 mg/day for older children and 0.56 mg/day for adults. Soil ingestion values, also from the same source, were 200 mg/day for children and 100 mg/day for adults. The results of nearly all studies used in risk assessment for incidental ingestion are based on the measurement of nonmetabolizable soil tracers in feces and urine (Paustenbach, 2000; Davis et al., 1990; van Wijnen et al., 1990; Calabrese et al., 1989, 1990; Clausing et al., 1987; Binder et al., 1986).

Exposure Pathway – Incidental Ingestion from Hand-to-Mouth Contact

Workers. The potential intake of the commercial pentaBDE product through incidental ingestion was calculated using the following equation:

CDI = SAt * FSAftip * ARskin * Fip * AFo * FWD * 1/BW

Exposure Pathway – Incidental Ingestion from Dirty Laundry

Worker. The potential intake of the commercial pentaBDE product through incidental ingestion while handling dirty laundry was calculated using the following equation:

CDI = SAt * FSAftip * ARskin * FCLdy * AFo * EFL * CF2 * 1/BW

Exposure Pathway – Incidental Ingestion from Hand-to-Mouth Contact with Floor Surfaces Contaminated by Dirty Shoes in the Home

Workers. Standard home floor surface area (756,644 cm^2) that potentially retains dirt from outdoor sources was adopted from USEPA (1997). The potential intake of the commercial pentaBDE product through incidental ingestion from touching floor surfaces that have been contaminated by dirty shoes was calculated using the following equation:

Cfloor = (SAs * ARshoe) / FSA

CDI = Cfloor * SAt * FSAftip * AFo * EFF * CF2 * 1/BW

Workers' Children. The potential intake of the commercial pentaBDE product by children touching floor surfaces that have been contaminated by a parent's dirty shoes was calculated using the following equation:

Cfloor = (SAs * ARshoe) / FSA
CDI = Cfloor * SAt * FSAftip * AFo * EFF * CF2 * 1/BW

Exposure Pathway – Incidental Ingestion of Indoor Dust from Hand-to-Mouth Contact

Adults and Children in the Home, School and Office. The potential for incidental ingestion with the commercial pentaBDE product in indoor dust in the home, school and office was calculated using the following equation:

CDI = Cdust * CF1 * DIR * AFo * EFh/s/o * CF2 * 1/BW

Exposure Pathway – Incidental Ingestion from Hand-to-Mouth Contact with Outdoor Soil

Adult and Child Ambient Environmental Exposures. The potential for incidental ingestion from hand-to mouth contact with the commercial pentaBDE product in outdoor soil was calculated using the following equation:

CDI = Csoil * CF * SIR * AFo * 1/BW

Ingestion of Food

Exposure models describing the ingestion of food products, fish, and breast milk (infants only) were developed for children and adults as part of the assessment of ambient environmental exposures to the commercial pentaBDE product. The environmental exposure data used in these models are summarized in Table 3-18. U.S. and Canadian data describing levels in fish and human breast milk were used in the exposure assessment. In the absence of sufficient U.S. data, Swedish and Japanese market basket data were used to evaluate potential exposures through the relevant dietary categories – meat (beef, poultry and pork), dairy, eggs, other fats and oils, and vegetables. Exposure assumptions describing food consumption habits for adults and different child age bins were adopted from the USEPA *Exposure Factors Handbook* (1997). In order to avoid combining multiple high-end values, either the high end estimate of the food concentration or the high-end estimate of the consumption rate was used in the assessment (and not both high-end estimates).

Exposure Pathway – Ingestion of Fish

Adult and Child Ambient Environmental Exposures. The potential intake of the commercial pentaBDE product through ingestion of fish was calculated using the following equation:

CDI = Cfish * CF * CRfish * AFo * 1/BW

Exposure Pathway – Ingestion of Food

Adult and Child Ambient Environmental Exposures. The potential intake of the commercial pentaBDE product through ingestion of different food groups was calculated using an equation similar to that used for fish consumption. The parameters for the concentration term and consumption rate were replaced with appropriate parameters for each of the different food groups evaluated in the assessment: meat (beef, poultry and pork) – Cmeat, CRmeat; vegetables – Cveg, CRveg; dairy – Cdairy, CRdairy; eggs – Cegg, CRegg; and other fats and oils – Cfat, CRfat.

Exposure Pathway – Ingestion of Human Breast Milk by Infants

Infant Exposures. The potential intake of the commercial pentaBDE product through ingestion of human breast milk was calculated only for children <1 year old using the following equation:

CDI = Cbm * CF1 * Fbm * CRbm * CF2 * AFo * 1/BW

Ingestion via Mouthing Behavior

The indoor home/school/office scenario included mouthing of cushions with exposed FPUF treated with the commercial pentaBDE product and subsequent ingestion as a plausible source of exposure to children less than 5 years old. According to the U.S. Consumer Product Safety Commission (USCPSC) Chronic Health Advisory Panel behavioral study of mouthing behavior in U.S. children, mouthing behavior tends to decline dramatically after age 3 (USCPSC, 2002; USEPA, 2000a). Similar observations are reported by Juberg et al. (2001). The behavioral observation data collected as part of the USCPSC evaluation of children exposures to diisononyl phthalate (a chemical plasticizer in plastic toys) and the data reported in the published literature indicate that mouthing behavior declines as children age. According to USCPSC (2002), evaluation of chemical exposures to children over 3 years old via mouthing activity are very likely to experience lower intakes than younger children.

For the purposes of this screening-level Tier 1 exposure assessment, exposure to the commercial pentaBDE product via mouthing behavior was evaluated in children representing the <1 year, 1-2 year, and 3-5 year age groups. Data describing the surface concentration of the commercial pentaBDE product or BDEs on FPUF cushion surfaces are not available in the published literature. In the absence of data, the water solubility limit of the commercial pentaBDE product (14 µg/L; EBC, 2000) was used as a surrogate for the exposure point concentration. The amount of commercial pentaBDE product ingested was calculated by assuming that 100% of the commercial pentaBDE product transferred from the exposed FPUF to saliva at the water solubility limit, and the saliva swallowed by the child. Based on data reported by Juberg et al. (2001), children <1, 1-2, and 3-5 years old mouth surfaces other than toys, pacifiers, and baby teething objects for 9, 5.5, and 2 minutes each day, respectively. For children 1-2 years old, 5.5 minutes/day is the mid-point between results reported by Juberg et al. (2001) for 0-18 months (9

minutes/day) and 19-36 months (2 minutes/day). An unstimulated salivary flow rate of 0.22 mL/minute was assumed for the three age groups based on data reported by Watanabe et al. (1990) for 5-year-old children. The amount of time each day that young children might engage in mouthing behavior of cushions with exposed FPUF treated with the commercial pentaBDE product is consistent with similar models described in NRC (2000) and USEPA (2000a).

Exposure Pathway – Ingestion from Mouthing a Cushion

Children in the Indoor Home, School and Office Environment. Hypothetical exposure via ingestion by mouthing of cushions with exposed FPUF treated with the commercial pentaBDE product was evaluated only for the home environment and only for children <1, 1-2, and 3-5 years old. Exposure was calculated using the following equation:

CDI = WS * Vs * CF1 * CF2 * FR * AFo * EFmouth * 1/BW

Inhalation

The inhalation exposure pathway was considered a plausible source of exposure to primary and chain of commerce manufacturing workers and to adults and children. Workplace exposures were based on the saturated vapor concentration of the commercial pentaBDE product. For adults and children potentially exposed in the indoor home/school/office environments and ambient environment, indoor and outdoor air particulate data reported in the literature were used as exposure point concentrations (see Table 3-18). One-half of the measured particulate level was assumed to be respirable, and the remaining fraction was assumed to be inhaled and swallowed. Inhalation rates for workers were adopted from USEPA (1997) based on the level of activity (light, moderate, or heavy) appropriate for each job category.

Exposure Pathway – Inhalation of Vapor

Workers. The inhalation of vapor was calculated using the following equation:

CDI = SVC * Fsvc * CF * InR * AFi * 1/BW

Exposure Pathway – Inhalation of Respirable Particulates

Adults and Children in the Home, School and Office. The inhalation of respirable indoor particulates that contain the commercial pentaBDE product was calculated using the following equation:

CDI = AirP * CF1 * Fresp * InR * AFi * EFh/s/o * CF2 * 1/BW

Adult and Child Ambient Environmental Exposures. The inhalation of respirable outdoor particulates that contain the commercial pentaBDE product was calculated using the following equation:

CDI = AirP * CF * Fresp * InR * AFi * 1/BW

Exposure Pathway – Swallowing of Inhaled Particulates

Adults and Children in the Home, School and Office. The inhalation and subsequent swallowing of indoor particulates that contain the commercial pentaBDE product was calculated using the following equation:

CDI = AirP * CF1 * (1 - Fresp) * InR * AFo * EFh/s/o * CF2 * 1/BW

Adult and Child Ambient Environmental Exposures. The inhalation and subsequent swallowing of outdoor particulates that contain the commercial pentaBDE product was calculated using the following equation:

CDI = AirP * CF * (1 - Fresp) * InR * AFo * 1/BW

3.12 Results of the Exposure Assessment

Theoretical CDIs of the commercial pentaBDE product by prospective parents associated with the seven different oral, dermal, and inhalation exposure pathways evaluated in scenario #1 (workplace exposures) are presented in Table 3-23. Theoretical CDIs were highest for the vapor inhalation, dermal contact, and incidental ingestion via hand-to-mouth contact pathways.

Theoretical CDIs of the commercial pentaBDE product by children of prospective parents engaged in different primary and chain-of-commerce manufacturing activities associated with two different exposure pathways evaluated as part of scenario #1 are presented in Table 3-24. The highest theoretical exposure was associated with dermal contact with floor surfaces in the worker's home. The pathway assumes that workers bring home clothing and shoes from the workplace that contains the commercial pentaBDE product. The theoretical total CDI was less than approximately 1 µg/kg-day.

Table 3-23. Chronic Daily Intakes (CDI) of the commercial PentaBDE product by different adult receptor groups engaged in primary production of the product and chain-of-commerce flexible polyurethane foam manufacturing or use activities

Receptor	CDI	CDI	CDI	CDI	CDI	CDI	CDI	CDI
	Inhalation of Vapor	Dermal Contact	Incidental Ingestion Hand to Mouth Contact	Dermal Contact with Dirty Laundry	Incidental Ingestion Hand to Mouth Contact with Dirty Laundry	Dermal Contact with Dirty Floors in the Home	Incidental Ingestion Hand to Mouth Contact with Dirty Floors in the Home	Total
	mg/kg-day	mg/kg-day	mg/kg-day	mg/kg-day	mg/kg-day	mg/kg-day	mg/kg-day	mg/kg-day
Primary Production Drummer, Male	1.1E-03	5.0E-03	1.1E-02	4.6E-05	1.0E-04	3.1E-05	4.5E-06	1.74E-02
Primary Production Drummer, Female	1.3E-03	4.8E-03	1.2E-02	4.4E-05	1.1E-04	2.5E-05	4.7E-06	1.78E-02
Primary Production Cleaner, Male	1.1E-03	5.0E-03	1.1E-02	4.6E-05	1.0E-04	3.1E-05	4.5E-06	1.74E-02
Primary Production Cleaner, Female	1.3E-03	4.8E-03	1.2E-02	4.4E-05	1.1E-04	2.5E-05	4.7E-06	1.78E-02
Foam Manufacture Mixer, Male	1.1E-03	5.0E-03	1.1E-02	4.6E-05	1.0E-04	3.1E-05	4.5E-06	1.74E-02
Foam Manufacture Mixer, Female	1.3E-03	4.8E-03	1.2E-02	4.4E-05	1.1E-04	2.5E-05	4.7E-06	1.78E-02
Foam Manufacture Cutter, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Foam Manufacture Cutter, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Mattress Manufacturer, Male	1.1E-03	4.5E-03	1.0E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	1.76E-02
Mattress Manufacturer, Female	1.3E-03	4.4E-03	1.0E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	1.80E-02
Cushion Manufacturer, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Cushion Manufacturer, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Furniture Manufacturer, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Furniture Manufacturer, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Carpet Pad Manufacturer, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Carpet Pad Manufacturer, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Carpet Installer, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Carpet Installer, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Distributor Covered, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Distributor Covered, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Distributor Uncovered, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Distributor Uncovered, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02
Recycler, Male	1.1E-03	7.6E-03	1.7E-02	4.6E-04	1.0E-03	3.1E-04	4.5E-05	2.73E-02
Recycler, Female	1.3E-03	7.3E-03	1.7E-02	4.4E-04	1.1E-03	2.5E-04	4.7E-05	2.79E-02

Table 3-24. Chronic Daily Intake (CDI) of the commercial pentaBDE product by children of adult workers engaged in different primary manufacturing or chain-of-commerce activities associated with FPUF

Receptor	CDI Dermal Contact with Floor Surfaces	CDI Incidental Ingestion Hand-to-Mouth Contact with Floor Surfaces	CDI Total
	mg/kg-day	mg/kg-day	mg/kg-day
<1 yr Child	6.9E-04	1.8E-04	8.7E-04
1-2 yr Child	5.7E-04	1.4E-04	7.1E-04
3-5 yr Child	5.3E-04	1.2E-04	6.5E-04
6-8 yr Child	4.5E-04	1.1E-04	5.5E-04
9-11 yr Male Child	4.1E-04	9.4E-05	5.0E-04
9-11 yr Female Child	4.0E-04	9.3E-05	5.0E-04
12-14 yr Male Child	3.8E-04	8.7E-05	4.7E-04
12-14 yr Female Child	3.7E-04	8.3E-05	4.5E-04
15-18 yr Male Child	3.4E-04	4.9E-05	3.9E-04
15-18 yr Female Child	3.5E-04	5.1E-05	4.1E-04

Theoretical CDIs of the commercial pentaBDE product by prospective parents and children associated with four different oral, dermal, and inhalation exposure pathways evaluated in scenario #2 (exposures in the indoor home/school/office environments) are presented in Table 3-25. A fifth exposure pathway, ingestion of the commercial pentaBDE product via mouthing cushions containing FPUF, was evaluated for three different child age bins (<1 years, 1-2 years, 3-5 years old). Theoretical CDIs were less than approximately 1 µg/kg-day for all pathways and all receptor groups. The highest theoretical exposures were associated with the inhalation and dermal contact pathways.

Theoretical CDIs of the commercial pentaBDE product by prospective parents and children associated with 11 different oral, dermal, and inhalation exposure pathways evaluated in scenario #3 (ambient environmental exposures) are presented in Table 3-26. Theoretical CDIs were generally less than 1 µg/kg-day for all pathways and all receptor groups. The highest theoretical exposures were associated with the food consumption pathways.

3.13 Comparison to Other Exposure Assessment Results

There is limited data available on the manufacturing process and waste disposal of BDEs, the fate and transport of BDEs in the environment, and the uptake and bioavailability of BDEs in humans and biota. As a result, estimating exposure in the U.S. population is difficult. Often, reliance is placed on computer models that use chemical and physical properties to predict how a chemical might behave in the environment. At present, three different exposure assessments of

Table 3-25. Chronic daily intakes (CDI) of the commercial pentaBDE product by different adult and child receptor groups potentially exposed in the home, school, and office environments

Receptor	CDI Inhalation of Respirable Particulates mg/kg-day	CDI Ingestion of Particles Inhaled and Swallowed mg/kg-day	CDI Dermal Contact mg/kg-day	CDI Incidental Ingestion via Hand to Mouth Contact mg/kg-day	CDI Ingestion from Mouthing a Cushion mg/kg-day	CDI Total mg/kg-day
Exposures in the Home Environment						
<1 yr Child	2.41E-07	2.89E-07	2.69E-07	7.75E-07	1.80E-08	1.59E-06
1-2 yr Child	2.93E-07	3.52E-07	2.19E-07	6.24E-07	8.86E-09	1.50E-06
3-5 yr Child	2.06E-07	2.47E-07	1.70E-07	3.60E-07	2.23E-09	9.86E-07
6-8 yr Child	1.45E-07	1.74E-07	1.71E-07	6.31E-08	NA	5.54E-07
9-11 yr Male Child	1.43E-07	1.72E-07	1.56E-07	4.44E-08	NA	5.16E-07
9-11 yr Female Child	1.32E-07	1.58E-07	1.55E-07	4.41E-08	NA	4.90E-07
12-14 yr Male Child	1.10E-07	1.32E-07	1.47E-07	3.19E-08	NA	4.21E-07
12-14 yr Female Child	8.52E-08	1.02E-07	1.42E-07	3.08E-08	NA	3.60E-07
15-18 yr Male Child	9.04E-08	1.08E-07	1.32E-07	2.31E-08	NA	3.54E-07
15-18 yr Female Child	7.19E-08	8.63E-08	1.36E-07	2.60E-08	NA	3.21E-07
Adult Male (Prospective Parent)	9.49E-08	1.14E-07	1.36E-07	5.06E-09	NA	3.50E-07
Adult Female (Prospective Parent)	8.42E-08	1.01E-07	1.35E-07	6.04E-09	NA	3.27E-07
Exposures in the School Environment						
<1 yr Child	NA	NA	NA	NA	NA	NA
1-2 yr Child	NA	NA	NA	NA	NA	NA
3-5 yr Child	4.12E-08	4.95E-08	2.76E-08	7.19E-08	NA	1.90E-07
6-8 yr Child	5.99E-08	7.18E-08	5.64E-08	2.60E-08	NA	2.14E-07
9-11 yr Male Child	5.90E-08	7.08E-08	5.18E-08	1.83E-08	NA	2.00E-07
9-11 yr Female Child	5.44E-08	6.52E-08	5.15E-08	1.82E-08	NA	1.89E-07
12-14 yr Male Child	4.53E-08	5.44E-08	4.88E-08	1.31E-08	NA	1.62E-07
12-14 yr Female Child	3.51E-08	4.21E-08	4.69E-08	1.27E-08	NA	1.37E-07
15-18 yr Male Child	3.72E-08	4.47E-08	4.48E-08	9.51E-09	NA	1.36E-07
15-18 yr Female Child	2.96E-08	3.55E-08	4.62E-08	1.07E-08	NA	1.22E-07
Adult Male (Prospective Parent)	3.16E-08	3.80E-08	3.57E-08	1.69E-09	NA	1.07E-07
Adult Female (Prospective Parent)	2.81E-08	3.37E-08	3.52E-08	2.01E-09	NA	9.90E-08
Exposures in the Office Environment						
<1 yr Child	NA	NA	NA	NA	NA	NA
1-2 yr Child	NA	NA	NA	NA	NA	NA
3-5 yr Child	NA	NA	NA	NA	NA	NA
6-8 yr Child	NA	NA	NA	NA	NA	NA
9-11 yr Male Child	NA	NA	NA	NA	NA	NA
9-11 yr Female Child	NA	NA	NA	NA	NA	NA
12-14 yr Male Child	NA	NA	NA	NA	NA	NA
12-14 yr Female Child	NA	NA	NA	NA	NA	NA
15-18 yr Male Child	NA	NA	NA	NA	NA	NA
15-18 yr Female Child	NA	NA	NA	NA	NA	NA
Adult Male (Prospective Parent)	3.16E-08	3.80E-08	1.69E-08	1.69E-09	NA	8.82E-08
Adult Female (Prospective Parent)	2.81E-08	3.37E-08	1.61E-08	2.01E-09	NA	7.99E-08

NA = not applicable

Table 3-26. Chronic daily intakes (CDI) of the commercial pentaBDE product by different adult and child receptor groups associated with ambient environmental exposures

Receptor	CDI Inhalation of Respirable Particulates mg/kg-day	CDI Ingestion of Particles Inhaled and Swallowed mg/kg-day	CDI Dermal Contact mg/kg-day	CDI Incidental Ingestion via Hand to Mouth Contact mg/kg-day	CDI Fish Consumption mg/kg-day	CDI Meat Consumption mg/kg-day	CDI Dairy Consumption mg/kg-day	CDI Fats & Oil Consumption mg/kg-day	CDI Egg Consumption mg/kg-day	CDI Vegetable Consumption mg/kg-day	CDI Breast Milk Consumption mg/kg-day	CDI Total mg/kg-day
<1 yr Child	9.64E-09	1.16E-08	1.04E-07	1.50E-06	7.10E-06	2.43E-07	3.85E-06	5.16E-08	1.60E-07	2.92E-06	1.66E-07	1.61E-05
1-2 yr Child	1.17E-08	1.41E-08	8.50E-08	1.21E-06	2.39E-05	4.20E-07	1.48E-06	2.01E-07	2.37E-07	2.81E-06	NA	3.03E-05
3-5 yr Child	9.90E-09	1.19E-08	7.93E-08	8.37E-07	2.06E-05	3.87E-07	7.99E-07	1.91E-07	1.47E-07	2.21E-06	NA	2.53E-05
6-8 yr Child	8.21E-09	9.85E-09	2.34E-08	5.76E-07	1.68E-05	2.80E-07	5.49E-07	1.80E-07	1.06E-07	1.63E-06	NA	2.01E-05
9-11 yr Male Child	8.09E-09	9.71E-09	2.14E-08	4.05E-07	1.68E-05	2.80E-07	5.49E-07	1.26E-07	7.48E-08	1.63E-06	NA	1.99E-05
9-11 yr Female Child	7.46E-09	8.95E-09	2.12E-08	4.02E-07	1.68E-05	2.80E-07	5.49E-07	1.25E-07	7.43E-08	1.63E-06	NA	1.99E-05
12-14 yr Male Child	6.22E-09	7.46E-09	2.02E-08	2.91E-07	1.29E-05	2.02E-07	2.92E-07	1.48E-07	7.15E-08	1.12E-06	NA	1.51E-05
12-14 yr Female Child	4.81E-09	5.77E-09	1.94E-08	2.81E-07	1.29E-05	2.02E-07	2.92E-07	1.43E-07	6.92E-08	1.12E-06	NA	1.50E-05
15-18 yr Male Child	5.11E-09	6.13E-09	1.81E-08	2.11E-07	1.29E-05	2.02E-07	2.92E-07	1.07E-07	5.18E-08	1.12E-06	NA	1.49E-05
15-18 yr Female Child	4.06E-09	4.88E-09	1.86E-08	2.38E-07	1.29E-05	2.02E-07	2.92E-07	1.21E-07	5.84E-08	1.12E-06	NA	1.50E-05
Adult Male (Prospective Parent)	3.80E-09	4.55E-09	1.32E-08	8.76E-08	1.66E-05	2.10E-07	4.86E-07	2.15E-07	4.08E-08	1.21E-06	NA	1.89E-05
Adult Female (Prospective Parent)	3.37E-09	4.04E-09	1.31E-08	1.05E-07	1.98E-05	2.10E-07	4.86E-07	1.66E-07	4.08E-08	1.21E-06	NA	2.21E-05
Recreational Marine Angler Adult Male	NA	NA	NA	NA	5.95E-06	NA	NA	NA	NA	NA	NA	5.95E-06
Recreational Marine Angler Adult Female	NA	NA	NA	NA	7.10E-06	NA	NA	NA	NA	NA	NA	7.10E-06
Recreational Freshwater Angler Adult Male	NA	NA	NA	NA	1.40E-05	NA	NA	NA	NA	NA	NA	1.40E-05
Recreational Freshwater Angler Adult Female	NA	NA	NA	NA	1.68E-05	NA	NA	NA	NA	NA	NA	1.68E-05

NA = not applicable

Table 3-27. Comparison of exposure point concentrations developed in the Tier 1 assessment to estimates developed by ECB (2000), Palm et al. (2002), and Wenning et al. (2002).

Environmental Media	Units	VCCEP Assessment[a]	ECB Risk Assessment[b]		Palm et al., 2002[c]			Wenning (2002)
		Commercial PentaBDE Mixture			BDE-49	Total PentaBDEs	BDE-209	Commercial Product mixture
			FPUF Production	Regional Sources				
Fish	ng/g wet weight	170	4,380	22	--	--	--	22
			8,360	41	--	--	--	41
Root Crop Vegetables	ng/g wet weight	0.13	6,800	340	--	--	--	335
Leaf Crop Vegetables			31	0.29	--	--	--	0.29
Drinking Water	mg/L	--	2.7E-04	1.4E-05	--	--	--	--
Meat	ng/g wet weight	0.05	208	6.5	--	--	--	6.9
Milk/Dairy	ng/g wet weight	0.02	66	2.1	--	--	--	2.1
Eggs	ng/g wet weight	0.04	--	--	--	--	--	--
Breast Milk	ng/g lipid	67	--	10[d]	--	--	--	1.2
Air	pg/m^3	52	28,000[e]	270[e]	1.9	2.5	29	75
Soil	µg/kg dry weight	76	2,700[f]	130[f]	0.003	0.01	0.10	--
Sediment	ng/g dry weight	--	4,500[g]	32[g]	0.03	0.05	0.60	--
Water	µg/L	--	0.37	0.0015	1.2E-06	1.8E-06	2.3E-05	--

[a] In the VCCEP Assessment, the commercial pentaBDE product mixture was assumed to be represented by the sum of congeners typically associated with the commercial product. The number of isomers representing the commercial product varied in different media. In all cases, exposure models relied on measured environmental data.

[b] Risk assessments conducted by the ECB (2000) mainly relied on predicted concentrations of the commercial product using the EUSES Environmental Fate model. Limited validation to measured environmental levels was performed.

[c] Exposure models by Palm et al. (2002) relied mainly on predicted concentrations of either BDE-49, the pentaBDE congeners, and BDE-209 using EPIWIN software and the EQC and the SimpleBox 1.0 models. The results were not validated by the authors.

[d] Results in the ECB Risk Assessment are reported for polyBDE.

[e] Results in the ECB Risk Assessment are reported for the human intake of air and do not include the contribution from waste in the environment. Concentrations in air which include this contribution are equal to 34.5 ng/m^3 for FPUF production sources and 0.27 ng/m^3 for regional sources.

[f] Results are reported as µg/kg wet weight.

[g] Results are reported as ng/g wet weight.

the commercial pentaBDE product have been reported in the literature (ECB, 2000; Palm et al., 2002; Wenning, 2002). A comparison of the total theoretical exposure estimates calculated in each study is summarized in Table 3-27.

The exposure models developed by ECB (2000) were based on assumptions that the commercial pentaBDE product contained 50-62% ww pentaBDE, 24-38% ww tetraBDE, 4-12% ww hexaBDE, and trace levels of triBDE and heptaBDE. Exposure estimates were calculated using the EUSES Environmental Fate model. Emissions and discharges from the manufacturing process and the life-cycle of finished consumer products in the European Union (EU) containing the commercial pentaBDE product were quantified and used to calculate potential exposures and health risks. ECB (2000) concluded that leaching and volatilization were insignificant pathways based on the physical/chemical properties of the commercial product and current environmental safeguards at landfills in the EU.

Palm et al. (2002) used a 6-stage system to classify chemical and physical properties, obtain data on emissions/discharges and the concentrations found in the environment, predict environmental fate, evaluate and validate environmental fate on a regional or continental level, evaluate and validate environmental fate on a local level, and compare predicted and measured levels including the toxicity and risk of exposure. Palm et al. (2002) used EPIWIN to predict the chemical and physical properties, the Equilibrium Concentration (EQC) model to estimate the transport between environmental compartments, and Simple Box 2.0 to perform a mass balance analysis. Palm et al. (2002) evaluated BDE-47, BDE-99, BDE-209, and a non-brominated diphenyl ether. Based on the results of the different models, it was concluded that exposure to the commercial product via disposal emissions was negligible since the commercial product would most likely be destroyed during incineration or leaching. Volatization from landfills was not likely to occur due to the current stringent landfill disposal regulations. The results of the exposure assessment developed by Palm et al. (2002) were not validated. Predictions for soil and sediment were found to be lower than data reported in the literature. As a result, Palm et al. (2002) concluded that the emissions were underestimated, degradation rates were overestimated, and monitoring data were not representative of regional concentrations.

Using both deterministic and probabilistic methods to calculate environmental exposures to the three commercial products, Wenning (2002) developed theoretical exposure estimates that were generally lower than those calculated by ECB (1996; 2000; 2002). The differences between the two risk assessments were attributable to various exposure assumptions and the inherent limitations in the EUSES model used by ECB (1996; 2000; 2002) to predict local and regional-scale exposures. Wenning (2002) results indicated that exposures to the commercial products through fish consumption and the consumption of dairy products are the most important and variable sources of exposure in adults and different child age groups. The theoretical levels of exposure associated with the occurrence of the commercial products in air, drinking water, consumption of meat products, and fruits and vegetables represent relatively less significant exposure pathways. Theoretical lifetime CDIs at the 95th percentile for each of the five different

age groups and different exposure pathways were highest in children representing the 0-2 and 6-12 year age groups. The results suggested that children up to the age of 12 may potentially be exposed to the commercial products or their constituents at higher levels than in older children and adults. In general, however, the results of the deterministic and probabilistic risk assessment indicated larger margins of safety than those calculated in the ECB risk assessments when comparing theoretical CDIs to the human toxicity benchmark values derived by ECB (1996; 2000; 2002).

4.0 Risk Assessment

The risk assessment information presented in this section is consistent with USEPA's *Risk Characterization Handbook* (USEPA, 2000e). The characterization of health risks associated with exposure to the commercial pentaBDE product also is in accordance with Agency's guidance for developmental (USEPA, 1991a), cancer (USEPA, 1996a), reproductive (USEPA, 1996b), and neurotoxicity (USEPA, 1998c) risk assessments. This section includes theoretical estimates of the potential health risks to different receptor groups associated with different exposure pathways, based on the findings presented in the hazard and exposure assessments. The risk results include both qualitative and quantitative conclusions about the likelihood that exposure to the commercial pentaBDE product may pose a hazard to children or, where relevant, prospective parents, the nature of the observed effects, and under what conditions (route, dose levels, time, and duration) of exposure these effects may occur. A discussion of uncertainties and their potential impact on the assessment, including the strengths and weaknesses of the assessment, also is presented in this section.

4.1 Method for Calculating Health Hazards

The results of the Tier 1 exposure assessment provide estimates of the theoretical CDI of the commercial pentaBDE product by children and prospective parents (see section 3.12). As a final step in the Tier 1 assessment, these results were integrated with information describing the non-cancer potency of the commercial pentaBDE product to characterize the likelihood for adverse health effects. For the purposes of this Tier 1 assessment, which represents a screening-level evaluation based on the currently available scientific information, the relevant toxicity endpoints for evaluating exposures to children and prospective parents are: (1) change in T_4 homeostasis, (2) thyroid hyperplasia, and (3) liver enzyme induction. In the absence of any evidence, cancer is not included as a human health endpoint in the Tier 1 assessment. Toxicity values used in the Tier 1 risk assessment are summarized in Table 4-1. The basis for each of these toxicity values is discussed in Section 2 of this document.

Table 4-1. Human health endpoints and toxicity values used in the Tier 1 risk assessment

Human Health Endpoint	Toxicity Value	Relevant Study
Liver enzyme induction	0.002 mg/kg/day	USEPA Reference Dose (RfD) for the commercial pentaBDE product based on Carlson (1980b)
Developmental effects: change in T_4 homeostasis	0.07 mg/kg/day	Zhou et al. (2002)
Systemic effects: thyroid hyperplasia	0.04 mg/kg/day	IRDC (1976)

In accordance with USEPA (1989) risk assessment guidance, the probability of adverse non-cancer effects associated with each of these health endpoints was evaluated using a "Hazard Quotient" (HQ) approach. If the intake predicted by the different exposure models is below the toxicity value specified for the health endpoint, then the predicted intake would not be expected to pose a significant health hazard under the conditions evaluated in this screening-level Tier 1 assessment (USEPA, 1989). The hazard quotient is defined as the ratio of the pathway-specific chronic daily intake (CDI) calculated in the exposure assessment and the toxicity value (TV) derived from the hazard assessment:

$$Hazard\ Quotient = \frac{CDI}{TV}$$

USEPA (1989) risk assessment guidance also specifies a second step in the evaluation of potential non-cancer hazards involving evaluation of cumulative exposure associated with all possible pathways of exposure. Consistent with USEPA (1989) guidance, a "Hazard Index" (HI) was calculated by summing the pathway-specific hazard quotients for each receptor group using the following equation:

$$Hazard\ Index = \sum_{i}^{n} \text{Hazard Quotient}_{i\text{-}n}$$

An HI less than or equal to a value of one indicates that exposure to commercial pentaBDE product is unlikely to result in adverse health effects to the receptor of interest.

4.2 Health Hazards to Children

Theoretical non-cancer health hazards to children were calculated for each of the three exposure scenarios considered in the exposure assessment; i.e., exposures to children in the home of

workers engaged in primary and chain-of-commerce manufacturing activities involving the manufacture or use of the commercial pentaBDE product, exposures to children in the indoor home/school/office environment, and exposures to children associated with ambient environmental conditions.

The theoretical health hazards to children associated with exposure in the home of workers engaged in primary and chain-of-commerce manufacturing activities are presented in Table 4-2. Hazard indices associated with each of the three health endpoints were well below a value of one for each age group and for each of the exposure pathways evaluated as part of this exposure scenario. A hazard index less than one suggests that the theoretical level of exposure calculated in the screening-level Tier 1 model for the exposure pathways used to evaluate children of primary and chain-of-commerce workers in the home did not exceed the toxicity values for the non-cancer human health endpoints indicated in Table 4-1.

Table 4-2. Theoretical health hazard indices for children whose parents work in primary and chain-of-commerce manufacturing activities involving the manufacture or use of the commercial pentaBDE product

Receptor	Cumulative Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
Worker's Children Exposure				
<1 yr Child	8.72E-04	0.02	0.01	0.4
1-2 yr Child	7.09E-04	0.02	0.01	0.4
3-5 yr Child	6.49E-04	0.02	0.009	0.3
6-8 yr Child	5.52E-04	0.01	0.008	0.3
9-11 yr Male Child	5.01E-04	0.01	0.007	0.3
9-11 yr Female Child	4.98E-04	0.01	0.007	0.2
12-14 yr Male Child	4.71E-04	0.01	0.007	0.2
12-14 yr Female Child	4.52E-04	0.01	0.006	0.2
15-18 yr Male Child	3.93E-04	0.01	0.006	0.2
15-18 yr Female Child	4.05E-04	0.01	0.006	0.2

[1] The cumulative CDI was calculated from the sum of CDIs for dermal contact with floor surfaces and incidental ingestion of floor dirt in worker homes.

[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor.

[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed *in utero* in the Zhou et al. (2002) study.

[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

The theoretical health risks to children associated with exposures to the commercial pentaBDE product in the indoor home/school/office environments are presented in Table 4-3. Hazard indices associated with exposure to the commercial pentaBDE product in the indoor home and

school environments were well below a value of one. Hazard indices for children in the indoor office environment were not calculated since children are not thought to spend any significant time in an office setting. Similarly, hazard indices for children in the age groups of <1 year and 1-2 year were not calculated because they are not thought to spend a significant amount of time in a school setting. A hazard index less than a value of one suggests that the theoretical level of exposure calculated in the screening-level Tier 1 model for the exposure pathways used to evaluate children at home and school did not exceed the toxicity values for the non-cancer human health endpoints indicated in Table 4-1.

Table 4-3. Theoretical health hazard indices for children and prospective parents associated with exposures to the commercial pentaBDE product in the indoor home/school/office environments

Receptor	Cumulative Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
Adult and Child Exposure at Home				
<1 yr Child	1.59E-06	0.00004	0.00002	0.0008
1-2 yr Child	1.50E-06	0.00004	0.00002	0.0007
3-5 yr Child	9.86E-07	0.00002	0.00001	0.0005
6-8 yr Child	5.54E-07	0.00001	0.00001	0.0003
9-11 yr Male Child	5.16E-07	0.00001	0.00001	0.0003
9-11 yr Female Child	4.90E-07	0.00001	0.00001	0.0002
12-14 yr Male Child	4.21E-07	0.00001	0.00001	0.0002
12-14 yr Female Child	3.60E-07	0.00001	0.00001	0.0002
15-18 yr Male Child	3.54E-07	0.00001	0.00001	0.0002
15-18 yr Female Child	3.21E-07	0.00001	0.000005	0.0002
Adult Male (Prospective Parent)	3.50E-07	0.00001	0.000005	0.0002
Adult Female (Prospective Parent)	3.27E-07	0.00001	0.000005	0.0002
Adult and Child Exposure at School				
<1 yr Child	NC	NC	NC	NC
1-2 yr Child	NC	NC	NC	NC
3-5 yr Child	1.90E-07	0.000005	0.000003	0.0001
6-8 yr Child	2.14E-07	0.000005	0.000003	0.0001
9-11 yr Male Child	2.00E-07	0.000005	0.000003	0.0001
9-11 yr Female Child	1.89E-07	0.000005	0.000003	0.0001
12-14 yr Male Child	1.62E-07	0.000004	0.000002	0.0001
12-14 yr Female Child	1.37E-07	0.000003	0.000002	0.0001
15-18 yr Male Child	1.36E-07	0.000003	0.000002	0.0001
15-18 yr Female Child	1.22E-07	0.000003	0.000002	0.0001
Adult Male (Prospective Parent)	1.07E-07	0.000003	0.000002	0.0001
Adult Female (Prospective Parent)	9.90E-08	0.000002	0.000001	0.00005
Adult and Child Exposure in an Office Environment				
<1 yr Child	NC	NC	NC	NC
1-2 yr Child	NC	NC	NC	NC
3-5 yr Child	NC	NC	NC	NC
6-8 yr Child	NC	NC	NC	NC
9-11 yr Male Child	NC	NC	NC	NC

Receptor	Cumulative Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
9-11 yr Female Child	NC	NC	NC	NC
12-14 yr Male Child	NC	NC	NC	NC
12-14 yr Female Child	NC	NC	NC	NC
15-18 yr Male Child	NC	NC	NC	NC
15-18 yr Female Child	NC	NC	NC	NC
Adult Male (Prospective Parent)	8.82E-08	0.000002	0.000001	0.00004
Adult Female (Prospective Parent)	7.99E-08	0.000002	0.000001	0.00004

NC = Exposures were not calculated for children in this exposure scenario.

[1] The cumulative CDI was calculated from the sum of CDIs for the exposure pathways in Table 3-25.

[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor.

[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

The theoretical health hazards to children associated with ambient environmental (including food) exposures to the commercial pentaBDE product are presented in Table 4-4. Hazard indices associated with each of the three health endpoints evaluated in the Tier 1 assessment were well below a value of one for each age group and for each of the exposure pathways evaluated as part of this exposure scenario. Hazard indices less than a value of one suggest that the theoretical levels of exposure calculated in the screening-level Tier 1 model for the exposure pathways used to evaluate children hypothetically exposed to the commercial pentaBDE product in the ambient environment did not exceed the toxicity benchmark values for the non-cancer human health endpoints indicated in Table 4-1.

Table 4-4. Theoretical health hazard indices for children and prospective parents associated with exposures to ambient environmental levels of the commercial pentaBDE product

Receptor	Cumulative Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
Ambient and Food Exposure				
<1 yr Child	1.61E-05	0.0004	0.0002	0.01
1-2 yr Child	3.03E-05	0.0008	0.0004	0.02
3-5 yr Child	2.53E-05	0.0006	0.0004	0.01
6-8 yr Child	2.01E-05	0.0005	0.0003	0.01
9-11 yr Male Child	1.99E-05	0.0005	0.0003	0.01
9-11 yr Female Child	1.99E-05	0.0005	0.0003	0.01
12-14 yr Male Child	1.51E-05	0.0004	0.0002	0.01
12-14 yr Female Child	1.50E-05	0.0004	0.0002	0.01
15-18 yr Male Child	1.49E-05	0.0004	0.0002	0.01
15-18 yr Female Child	1.50E-05	0.0004	0.0002	0.01
Adult Male (Prospective Parent)	1.89E-05	0.0005	0.0003	0.01
Adult Female (Prospective Parent)	2.21E-05	0.0006	0.0003	0.01
Additional Fish Consumption Exposure				
Recreational Marine Angler Adult Male	5.95E-06	0.0001	0.00008	0.003
Recreational Marine Angler Adult Female	7.10E-06	0.0002	0.0001	0.004
Recreational Freshwater Angler Adult Male	1.40E-05	0.0004	0.0002	0.007
Recreational Freshwater Angler Adult Female	1.68E-05	0.0004	0.0002	0.008

[1] The cumulative CDI was calculated from the sum of CDIs for the exposure pathways in Table 3-26.
[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor.
[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.
[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

4.3 Health Hazards to Prospective Parents

Theoretical non-cancer health hazards to prospective parents were calculated for each of the three exposure scenarios considered in the exposure assessment; i.e., exposure to adults in the workplace associated with primary and chain-of-commerce manufacturing activities involving the manufacture or use of the commercial pentaBDE product, exposures to adults in the indoor home/school/office environments, and exposures to adults associated with ambient environmental conditions.

The theoretical health hazards to prospective parents associated with exposures to the commercial pentaBDE product during primary and chain-of-commerce manufacturing activities are presented in Table 4-5. Hazard indices associated with the potential for developmental and thyroid effects in all workers were below a value of one. Although the toxicity benchmark for liver enzyme induction was exceeded by all worker categories, this endpoint is not considered relevant and is provided to represent an upper-bound estimate of hazard to adults. This toxicity benchmark value has not been revised since the early 1980s. Review of the available toxicology data strongly suggests that the RfD value is inappropriate and does not reflect the current information (see Section 2). Workers handling FPUF had the highest HIs, which were attributable to the inhalation, dermal contact, and hand-to-mouth contact pathways.

Table 4-5. Theoretical health hazard indices for prospective parents associated with exposures to the commercial pentaBDE product during primary and chain-of-commerce manufacturing activities

Receptor	Cumulative Chronic Daily Intake [1]	Hazard Index		
		Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
Worker Exposures				
Primary Production Drummer, Male	0.017	0.4	0.2	9
Primary Production Drummer, Female	0.018	0.4	0.3	9
Primary Production Cleaner, Male	0.017	0.4	0.2	9
Primary Production Cleaner, Female	0.018	0.4	0.3	9
Foam Manufacture Mixer, Male	0.017	0.4	0.2	9
Foam Manufacture Mixer, Female	0.018	0.4	0.3	9
Foam Manufacture Cutter, Male	0.027	0.7	0.4	14
Foam Manufacture Cutter, Female	0.028	0.7	0.4	14
Mattress Manufacturer, Male	0.018	0.4	0.3	9
Mattress Manufacturer, Female	0.018	0.4	0.3	9
Cushion Manufacturer, Male	0.027	0.7	0.4	14
Cushion Manufacturer, Female	0.028	0.7	0.4	14
Furniture Manufacturer, Male	0.027	0.7	0.4	14
Furniture Manufacturer, Female	0.028	0.7	0.4	14
Carpet Pad Manufacturer, Male	0.027	0.7	0.4	14
Carpet Pad Manufacturer, Female	0.028	0.7	0.4	14
Carpet Installer, Male	0.027	0.7	0.4	14
Carpet Installer, Female	0.028	0.7	0.4	14
Distributor Covered, Male	0.027	0.7	0.4	14
Distributor Covered, Female	0.028	0.7	0.4	14
Distributor Uncovered, Male	0.027	0.7	0.4	14
Distributor Uncovered, Female	0.028	0.7	0.4	14
Recycler, Male	0.027	0.7	0.4	14
Recycler, Female	0.028	0.7	0.4	14

[1] The cumulative CDI was calculated from the sum of CDIs for the exposure pathways in Table 3-23.

[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL^{10}$ was extrapolated to humans using a 100-fold safety factor.

[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

The theoretical health hazards to prospective parents associated with exposures to the commercial pentaBDE product in the indoor home/school/office environments are presented in Table 4-3. Hazard indices associated with each health endpoint evaluated for prospective parents were well below a value of one.

The theoretical health hazards to prospective parents associated with ambient environmental exposures to the commercial pentaBDE product are presented in Table 4-4. The predicted cumulative CDIs associated with all pathways of exposure to the commercial pentaBDE product in the ambient environment result in hazard indices associated with each of the three health endpoints well below a value of one.

4.4 Aggregate Theoretical Exposures and Hazards to Children and Prospective Parents

A summary of exposure pathways in each of the three exposure scenarios for which theoretical exposures to children and prospective parents resulted in non-cancer hazard indices greater than or less than a value of one is presented in Table 4-6. A hazard index greater than a value of one suggests that the theoretical level of exposure calculated in the screening-level Tier 1 model for a particular exposure pathway and receptor group exceeds one or more of the toxicity benchmark values for the different non-cancer human health endpoints indicated in Table 4-1.

Theoretical exposures to the commercial pentaBDE product by children did not result in levels that exceeded the any of the screening toxicity benchmark values. Theoretical exposure of prospective parents to the commercial pentaBDE product in the indoor home/school/office and ambient environments also did not result in levels above screening toxicity benchmark values. The theoretical health hazards to prospective parents engaged in primary product production, FPUF production, and chain-of commerce manufacturing activities did not exceed screening toxicity benchmark values for thyroid hyperplasia or disruption of T_4 homeostasis. The screening toxicity benchmark for liver enzyme induction was exceeded for all workplace categories. Given the uncertainties regarding the relevance of this endpoint and the toxicity value, the calculated HIs (9-14) do not support conclusions that workplace exposures pose a hazard to workers.

Table 4-6. Summary of exposure pathways considered in the Tier 1 assessment associated with a theoretical non-cancer hazard index greater than one for one or more of the three non-cancer health endpoints included in the screening-level exposure assessment

	Hazard Index Above 1?	
Exposure Pathway	**Receptor Group**	**Toxicity Endpoint**
Scenario #1 - Workplace Exposures for Adults		
Inhalation of Vapor	▪ Primary production workers ▪ Chain of commerce workers	Yes; liver enzyme induction, only
Dermal Contact	-none-	-none-
Incidental Ingestion Hand to Mouth Contact	-none-	-none-
Dermal Contact with Dirty Laundry	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact with Dirty Laundry	-none-	-none-
Dermal Contact with Dirty Floors	-none-	-none-
Incidental Ingestion of Dust from Dirty Floors	-none-	-none-
Scenario #1 – Exposures for Worker's Children		
Dermal Contact with Dirty Floors at Home	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact with Floor Surfaces	-none-	-none-
Scenario #2 – Home Exposure		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Ingestion from Mouthing a Cushion	-none-	-none-
Scenario #2 - School and Office Exposures		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Ingestion from Mouthing a Cushion	-none-	-none-
Scenario #3 – Ambient Environment Exposures		
Inhalation of Respirable Particulate	-none-	-none-
Ingestion of Particles Inhaled and Swallowed	-none-	-none-
Dermal Contact	-none-	-none-
Incidental Ingestion via Hand to Mouth Contact	-none-	-none-
Fish Consumption	-none-	-none-
Meat Consumption	-none-	-none-
Dairy Consumption	-none-	-none-
Fats & Oil Consumption	-none-	-none-
Egg Consumption	-none-	-none-
Vegetable Consumption	-none-	-none-
Breast Milk Consumption	-none-	-none-

Hazard indices for children based on aggregate exposures to the commercial pentaBDE product from all three exposure scenarios (i.e., workplace, home/school/office environments, and ambient environment) are summarized in Table 4-7. Aggregate exposures to children of different age

groups were calculated by summing the theoretical total average daily intakes associated with exposure to the commercial pentaBDE product in the indoor home (including workers' homes), school, and ambient environment. The aggregate theoretical daily intakes indicate that children in the age groups <1 and 1-2 years receive the highest potential exposures to the commercial pentaBDE product.

Table 4-7. Aggregate total theoretical exposure to the commercial pentaBDE product by children associated with hypothetical exposures in workers' homes, indoor home/school/office environments, and ambient environment and associated hazard indices for three screening toxicity benchmarks

Receptor	Aggregate Total Theoretical Chronic Daily Intake [1]	Comparison to Thyroid Effects Benchmark [2]	Comparison to Developmental Effects Benchmark [3]	Comparison to Liver Enzyme Induction Benchmark [4]
	mg/kg-day	unitless	unitless	unitless
<1 yr Child	8.90E-04	0.02	0.01	0.4
1-2 yr Child	7.41E-04	0.02	0.01	0.4
3-5 yr Child	6.75E-04	0.02	0.01	0.3
6-8 yr Child	5.73E-04	0.01	0.008	0.3
9-11 yr Male Child	5.22E-04	0.01	0.007	0.3
9-11 yr Female Child	5.18E-04	0.01	0.007	0.3
12-14 yr Male Child	4.86E-04	0.01	0.007	0.2
12-14 yr Female Child	4.68E-04	0.01	0.007	0.2
15-18 yr Male Child	4.09E-04	0.01	0.006	0.2
15-18 yr Female Child	4.21E-04	0.01	0.006	0.2

[1] Aggregate exposures to children were calculated by summing the theoretical total CDIs calculated for children hypothetically exposed to the commercial product in a worker home, in the indoor home/school/office environments, and ambient environment. This represents the highest level of total exposure developed in the Tier 1 screening-level assessment.

[2] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL^{10}$ was extrapolated to humans using a 100-fold safety factor.

[3] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[4] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

A similar approach was used to evaluate aggregate exposures to the commercial pentaBDE product in prospective parents. Hazard indices for prospective parents, based on aggregate exposures to the commercial pentaBDE product from all three exposure scenarios (i.e., workplace, indoor home/school/office environments, and ambient environment) are summarized in Table 4-8. Aggregate exposures to prospective adult male and female parents were calculated by summing the average daily intakes associated with each exposure pathway evaluated in each of the three hypothetical exposure scenarios. The hazard index associated with the aggregate

total theoretical chronic daily intake was calculated by summing the hazard indices associated with each of the three exposure scenarios included in the Tier 1 assessment.

The results shown in Table 4-8 demonstrate that, with the exception of primary production and chain-of-commerce workers, potential exposure to the commercial pentaBDE product via all plausible pathways and exposure scenarios does not result in a level of exposure that exceeds relevant toxicity benchmark values. In the workplace, hypothetical exposures exceeded the toxicity benchmark established by USEPA (2003) based on liver enzyme induction. This endpoint (liver enzyme induction) is not considered applicable or relevant to children or prospective parents. The aggregate hazard index was below a value of one for both thyroid and developmental effects. The aggregate hazard index was above a value of one for liver enzyme induction and was driven by the worker scenario. Given the uncertainties regarding the relevance of this endpoint and the toxicity value, the calculated aggregate HI (a value of 14) does not support conclusions that workplace exposures pose a hazard to workers.

Table 4-8. Aggregate total theoretical exposure to the commercial pentaBDE product by adults associated with hypothetical exposures in the workplace, indoor home/school/office environments, and ambient environment and associated hazard indices for three screening toxicity benchmarks

Hypothetical Exposure Scenario	Aggregate Total Theoretical Chronic Daily Intake	Hazard Index		
		Comparison to Thyroid Effects Benchmark [1]	Comparison to Developmental Effects Benchmark [2]	Comparison to Liver Enzyme Induction Benchmark [3]
	mg/kg-day	*unitless*	*unitless*	*unitless*
Prospective Parent, Female				
Workplace Scenario [4]	0.03	0.7	0.4	14
Home, School, and Office Scenario [5]	5.06E-07	0.00001	0.00001	0.0003
Ambient Environmental Scenario [6]	2.21E-05	0.0006	0.003	0.01
Recreational Freshwater Fishing Scenario [7]	1.68E-05	0.0004	0.0002	0.01
SUM	**0.03**	**0.7**	**0.4**	**14**
Prospective Parent, Male				
Workplace Scenario [4]	0.03	0.7	0.4	14
Home, School, and Office Scenario [5]	5.45E-07	0.00001	0.00001	0.0003
Ambient Environmental Scenario [6]	1.89E-05	0.0005	0.0003	0.009
Recreational Freshwater Fishing Scenario [7]	1.40E-05	0.0004	0.0002	0.007
SUM	**0.03**	**0.7**	**0.4**	**14**

[1] The benchmark used to evaluate thyroid effects is based on the incidence of thyroid hyperplasia observed in the IRDC (1976) study. The $BMDL_{10}$ was extrapolated to humans using a 100-fold safety factor

[2] The benchmark used to evaluate developmental effects is based on changes in T_4 levels in the fetus or neonate exposed in utero in the Zhou et al. (2002) study.

[3] The benchmark used to evaluate the potential for liver enzyme induction was based on the Carlson (1980b) animal bioassay used by USEPA to derive the current RfD listed in IRIS (USEPA, 2003). This endpoint is not considered relevant, but is provided to represent an upper-bound estimate of hazard to children and adults.

[4] The theoretical highest exposed adult in the workplace scenario is associated with cutting FPUF treated with the commercial pentaBDE product, as shown in Table 4-5.

[5] The theoretical exposures in the home, office, and school environments were calculated by summing the total CDIs for the prospective female parent in Table 4-3.

[6] The theoretical exposures in the ambient environment represent the prospective parent in Table 4-4.

[7] The theoretical exposures in the ambient environment represent the prospective fishing person in Table 4-4. [2]

4.5 Risk-Benefit Analysis

The commercial pentaBDE product is recognized to be efficient at reducing the potential for loss of life or property as a result of fire (NFPA, 2000). Commercial products containing BDEs decrease fire hazards by (1) reducing the likelihood of ignition, (2) slowing the burning rate and the quantity of smoke produced, if ignition does occur, and (3) reducing the burn rate and as a result increasing time before flashover occurs, which increases egress time (NFPA, 2000). The commercial pentaBDE product is incorporated into highly flammable, petroleum-based materials that compose consumer goods such as upholstery and mattresses. If these consumer products are

not flame-retarded, then, upon accidental ignition, upholstery and mattresses can burn with such intensity and at such high temperatures that resulting fires often "flashover." This phenomenon occurs when the intensity of the fire is such that everything in the general vicinity of this fire suddenly ignites. An estimated 50-60% of residential fires reach flashover and spread from the room of origin (NFPA, 2000). Fires involving goods composed of FPUF (a petroleum-based product) that is not flame-retarded are particularly susceptible to flashover (NFPA, 2000).

According to a study by Long et al. (2002), the available fire statistics data strongly suggest that the benefits of the commercial pentaBDE product, including fire prevention and minimization or elimination of injury or loss of life or property, outweigh the potential health risks associated with human exposure. A tragic example occurred in January 2000 at Seton Hall University in New Jersey where 3 college students were killed and 58 others suffered injuries as a result of a fire in a dormitory. The fire, which started on a sofa and flashed over, engulfed the dormitory in flames. Fire investigators later determined that the sofa had not been flame retarded.

The objective of the Long et al. (2002) analysis was threefold: (1) determine the risk of preventable fire deaths and injuries in the United States and the corresponding potential to reduce such risk with the use of the commercial pentaBDE product, (2) conduct a quantitative analysis of the health risks from exposure to environmental concentrations of the commercial pentaBDE product, and (3) compare the potential health risks from environmental exposures to the risk of injuries from residential fires.

Risk of Death and Injury from Residential Fires

The number of residential fires in the United States annually ranged from 467,000 in 1990 to 383,000 in 1999 (NFPA, 2000). Over 50 percent of these fires spread due to flashover from the point of origin. In 1970, when BFRs were initially incorporated into the materials used to make TV cabinets, there was a significant decrease in flashover from fires that originated within TV cabinets, and the number of deaths resulting from these fires dropped 73%. Clarke (1999) estimated that 250-280 lives are saved annually due to the use of BDEs in TV cabinets, electrical insulation, and drapery materials. An additional 140-220 fire deaths may have been prevented by the increased use of BDEs in these materials, as well as in other highly flammable home furnishings. Since there were 2,920 fire deaths in the home during 1999, the use of BDEs may have reduced the number of fire-related deaths by 13-15% (NFPA, 2000).

To evaluate whether the increased use of commercial flame retardant mixtures in furniture and home furnishings could significantly reduce the potential risk of death by fire in the home, Long et al. (2002) used probabilistic models based on population projections published by the U.S. Census Bureau and fire statistics generated by the National Fire Protection Association (NFPA). Population and residential data were based on 1990 U.S. census data. Using NFPA fire statistics for 1999 and U.S. Census Bureau data, the risk of fire death to an individual in the population was 1×10^{-5} for 1999. In order to address the variability and uncertainty associated with this data set, probabilistic techniques were employed. The results of probabilistic modeling indicated a

10% certainty that the risk of death in a residential fire is between 4.2 x 10^{-6} and 4.6 x 10^{-6} and a 90% certainty that the risk is between 2.6 x 10^{-6} and 9.8 x 10^{-6}. If the commercial pentaBDE product is used to flame-retard consumer goods, then probabilistic risk estimates establish a 10% certainty that the risk of death in residential fire drops to between 3.3 x 10^{-6} and 3.5 x 10^{-6} and a 90% certainty that the risk drops to between 2.9 x 10^{-6} and 4.2 x 10^{-6}.

Using residential fire statistics for all of the years covering the period between 1990 and 1999 (NFPA, 2000), Long et al. (2002) concluded that the risk of fire death to an individual in the U.S. population is 1.3E-5. Evaluated probabilistically, there is a 10% certainty that the risk of death in a residential fire is between 5.2 x 10^{-5} and 5.6 x 10^{-5} and a 90% certainty that that the risk is between 3.1 x 10^{-5} and 1.1 x 10^{-4}. If the commercial pentaBDE product is used, then there is a 10% certainty that the risk of death in residential fire is between 4.2 x 10^{-5} and 4.4 x 10^{-5} and a 90% certainty that the risk is between 3.5 x 10^{-5} and 5.3 x 10^{-5}.

The same approach was used by Long et al. (2002) to evaluate the risk of injury to the general population due to residential fires. Using NFPA fire statistics for 1999 and U. S. Census Bureau data, the risk of injury to an individual in the population was 6.0 x 10^{-5} for 1999. Again, to address the variability and uncertainty associated with the underlying data set, probabilistic techniques were employed. Evaluated probabilistically, there is a 10% certainty that the risk of injury in a residential fire is between 2.4 x 10^{-5} and 2.6 x 10^{-5} and a 90% certainty that the risk is between 1.5 x 10^{-5} and 5.0 x 10^{-5}. If the commercial pentaBDE product is utilized to flame-retard consumer goods, then it can be established that there is a 10% certainty that the risk of injury in residential fire drops to between 1.8 x 10^{-5} and 1.9 x 10^{-5} and a 90% certainty that the risk is between 1.6 x 10^{-5} and 2.4 x 10^{-5}.

Using residential fire statistics for all years between 1990 through 1999 (NFPA, 2000), the risk of injury to an individual in the population is 7.1 x 10^{-5}. Evaluated probabilistically, there is a 10% certainty that the risk of injury in a residential fire is between 2.7 x 10^{-5} and 3.1 x 10^{-5} and a 90% certainty that the risk is between 2.2 x 10^{-5} and 6.5 x 10^{-5}. When considering the additional reduction in fire deaths due to increased usage of the commercial pentaBDE product, there is a 10% certainty that the risk of injury in residential fire is between 2.2 x 10^{-5} and 2.4 x 10^{-5} and a 90% certainty that the risk is between 2.0 x 10^{-5} and 3.0 x 10^{-5}.

Comparison of Fire Safety and Potential Health Risks

For the purpose of comparing fire safety estimates to the potential health risks posed by commercial flame retardant products, Long et al. (2002) developed screening-level exposure models for ingestion and dermal exposure scenarios for adults (worker and residential scenarios) and children. The exposure assessment methods follow USEPA's risk assessment guidelines (USEPA, 1989). Exposure concentrations for the commercial pentaBDE product were based on monitored concentrations in environmental media, including soil, water, sediment, air, as well as animal and human tissue. Based on these data, the following concentrations were used to estimate potential health risks: soil 0.033 mg/kg (dry wt), water 0.001 mg/L, food 0.2 mg per day

(including consumption of fish fillets) and air (0.16 mg/m^3). Daily intakes were estimated for industrial worker and residential scenarios, which included both adults and children. Daily intakes were calculated using the Risk Assessment Information System (RAIS, 2003) and the media concentrations listed above.

The results from Long et al. (2002) are presented in Table 4-9. The trend in residential fire deaths and injuries in the United States has decreased each year for the previous 10 years primarily due to the increased use of smoke detectors and the use of commercial flame retardant products in electronic components, carpet, draperies and upholstery materials (NFPA, 2000). An estimated 45% reduction in the risk of injury or death due to residential fires can be achieved with the continued and increased usage of BDEs in household materials. The Long et al. (2002) results indicate: (1) the benefits of the commercial pentaBDE product use in home furnishings outweigh the potential risks to human health with a large margin of safety, and (2) the use of the commercial pentaBDE product yields a potential 45% decrease in the risk of injury and death in residential fires.

Table 4-9. Comparison of theoretical health risks associated with exposure to ambient environmental levels of BDEs and fire hazards associated with the presence or absence of BDEs in home consumer products [1]

Theoretical Health Risk Predictions	Fire Hazard Predictions
▪ For the residential scenario, the daily intake for the ingestion pathway was 2.7E-8 mg/kg-day and the Hazard Quotient was 0.014. ▪ The dermal pathway intake was 1.0E-8 mg/kg-day and the Hazard Quotient was 0.000024. ▪ The total Hazard Quotient for a resident is 0.014. ▪ These Hazard Quotients are less than 1 indicating no adverse health effects are predicted for a resident based on the exposure concentrations listed above and the most extreme exposure parameters. ▪ For the industrial scenario, the daily intake for the ingestion pathway was 9.8E-6 mg/kg-day and the Hazard Quotient was 0.0049. ▪ The dermal pathway intake was 1.0E-8 mg/kg-day and the Hazard Quotient was 0.00001. ▪ The total Hazard Quotient for the industrial worker was 0.0049. ▪ These Hazard Quotients are less than 1 indicating no adverse health effects are predicted for a worker based on the exposure concentrations listed above and the most extreme exposure parameters.	▪ Based on NFPA fire statistics, the risk of death in a residential fire ranges from 3.1 x 10^{-5} to 1.1 x 10^{-4} (90% certainty level). ▪ When considering the additional benefits from the use of BDEs in electronic components, draperies, carpets, and upholstery materials, the risk of death in residential fire is reduced by approximately 45% to a range of 3.5 x 10^{-5} to 5.3 x 10^{-5} (90% certainty level). ▪ Likewise, the risk of injury in a residential fire is reduced from 2.2 to 6.5 x 10^{-5} to 2.0 to 3.0 x 10^{-5} (90% certainty level). This is a reduction in the risk of injury by approximately 45% due to the use of BDEs.

[1] Analysis from Long et al. (2002).

4.6 Uncertainty Analysis

Several key factors contribute to uncertainty in the Tier 1 assessment. Inherently, the process of estimating hazard, exposure, and health risks is associated with various uncertainties in the assumptions used to predict environmental exposure point concentrations, human activity patterns, chemical potency, and the probabilities of adverse cancer or non-cancer effects. These

and other sources of uncertainty are common in risk assessments and are described in detail elsewhere (e.g., Paustenbach, 2000; USEPA, 2000b;). A discussion of key uncertainties and their effect on the estimation of hazard presented in this Tier 1 assessment are summarized below.

Hazard Assessment

- Few scientific data have been published on the potential adverse health effects of the commercial pentaBDE product on children. Factors such as absorption, metabolism, and elimination of chemicals, routes of exposure, and physical location are known to be different from children to adults (Paustenbach, 2002). Future environmental and hazard studies should be more sensitive to differences in physiological factors at different lifestages. In this Tier 1 evaluation, exposure assumptions and various environmental and chemical factors were purposely selected to provide upper-bound estimates of exposure and hazard. Future studies should provide a foundation to derive more accurate information on the potential hazards to children's health.
- USEPA has characterized the available data for the commercial pentaBDE product as insufficient to make a clear determination of the potential for adverse health effects in humans. The available animal data are limited and insufficient for extrapolation to humans with high confidence. There have been few occupational exposures studies reported in the literature, and none have been able to correlate between exposure and adverse health effects.
- Animal studies currently provide the most reliable information on which to base an estimate of adverse human health effects. Although reliance on experimental animal data has been widely used in general risk assessment practices, chemical absorption, metabolism, excretion, and toxic responses may differ between humans and the species for which the experimental toxicity data are available (Paustenbach, 2002). Uncertainties in using animal data to predict the potential human effects are introduced when routes of exposure in animals studied differ from that in humans, when exposure in animals are acute or sub-chronic, and when relatively high exposure levels used in animals studies are used to predict effects that may be caused by much lower environmental levels (USEPA, 2000e).
- The current RfD reported by USEPA in the IRIS (USEPA, 2003) database is based on a 90-day rodent bioassay that evaluated the commercial pentaBDE product in 1980 (Carlson, 1980b). The data and RfD were given a low confidence rating due to study limitations (e.g., only one species and sex exposed). Since the Carlson bioassay in 1980, the relative proportions of the different BDE congeners in the commercial pentaBDE product have changed. The limited available data describing the distribution of BDE congeners in current formulations of the commercial pentaBDE product indicate a shift in the congener profile to lower brominated compounds. Review of the results of the Carlson (1980b) study and more recent information indicates the current RfD value (0.002 mg/kg/day) may be as much as 1,000-fold lower than current estimates and does not adequately represent the potency of the commercial pentaBDE product.

- Route-to-route extrapolation (from oral to inhalation pathways) also may introduce uncertainties in the hazard assessment. The available information concerning the bioavailability of the commercial pentaBDE product suggests that there are significant differences in human exposure and uptake of the product and/or its constituents, which are not fully addressed in the Tier 1 assessment.

Exposure Assessment

- Consistent with the purposes of the Tier 1 assessment, conservative assumptions and values for the key environmental data and exposure factors were used and likely exaggerate actual exposure conditions.
- A significant source of uncertainty concerns product uses and environmental releases of the commercial pentaBDE product throughout the chain-of-commerce after the initial manufacture of the product. Limited data are available describing the levels of commercial pentaBDE product used in consumer and industrial products. Because most product specifications are focused on flame retardant factors, the concentrations of the commercial pentaBDE product either on surfaces or embedded in different products (which are important to quantify for the purpose of exposure modeling) are not known precisely in the United States and other countries.
- Currently available data are insufficient to provide a thorough understanding of environmental levels of the commercial pentaBDE product in soil, surface waters, air, foods, and biological tissues. The reliance in this assessment on data that may or may not be related to releases of the commercial pentaBDE product to soil, water, or air and levels reported in humans and wildlife introduces significant uncertainties regarding the likely true levels of exposure encountered by children and prospective parents.
- Several scientists have described concerns regarding the prevalence of lower brominated BDEs in the environment and the possibility that their occurrence is due to debromination or photolytic degradation of higher brominated congeners (deWit, 2002). At present, the mechanism for debromination is not known (Strandberg et al., 2001). The assumption used in this Tier 1 assessment was that all hexa and lower BDEs found in environmental samples are attributable to commercial pentaBDE product. Should degradation not occur, this assumption would overestimate the potential for exposure to the commercial pentaBDE product.
- Most of the data reported in the scientific literature is congener specific. Other BDE congeners in the environment that could be attributed to the commercial pentaBDE product have not been analyzed, and the resulting levels might underestimate levels of BDEs in the U.S. environment.
- Overlap exists between the constituents of the commercial pentaBDE product and constituents found in the commercial octaBDE product. Attributing the BDE congeners found in environmental samples to either the commercial pentaBDE or octaBDE product is virtually impossible for constituents that are known or suspected to be present in both

products. Therefore, as part of this Tier 1 assessment BDEs with six or fewer bromine atoms were attributed to the commercial pentaBDE product. This approach likely overestimates levels in the environment.

- The characterization of human activity patterns in the Tier 1 exposure models likely has a substantial influence on the exposure estimates and theoretical health risks. A variety of assumptions ranging from inhalation rates to dermal contact rates were developed to estimate human exposure. Many of these assumptions were adopted from USEPA (1997) guidance and represented the upper range of possible values. The combination of several upper-bound estimates used as exposure parameters may substantially overestimate chemical intake. Thus, calculated approximations of exposure may be higher than what actually occurs.
- Some of the simplistic assumptions used in the exposure assessment are likely to over estimate actual exposure. For example, incidental ingestion of indoor dust is included in the indoor home/school/office scenario and incidental ingestion of soil is included in the ambient environmental scenario. Both of these two exposure pathways are aggregated; however, the soil ingestion rate actually includes ingestion of indoor dust. In the final aggregation, dust ingestion is double counted. As another example, time spent breathing outdoor air is assumed to be 24 hours per day, as is the breathing period for indoor air. Aggregation of these two exposures overestimates actual exposure.
- Factors such as species differentiation, dose, and exposure duration can create uncertainty in the bioavailability of the commercial pentaBDE product. There has been very little data published on the subject of bioavailability with regard to BDEs. The studies that have been conducted are usually performed on male Sprague-Dawley rats and the doses are usually administered orally. Exposure to humans will more likely occur through direct dermal contact or through inhalation pathways. Furthermore, Örn and Klasson-Wehler (1998) observed a species differentiation between the Sprague-Dawley rats and C57B1 mice in regard to absorption. Therefore, agreement as to which organism is more similar to humans must be established.
- In many instances, U.S. data were not available or inadequate for characterization of environmental levels and the derivation of exposure point concentrations. In the absence of U.S. data, environmental data from studies conducted in Canada, Europe, and other countries were assumed to be representative of environmental levels that may occur in the United States. This assumption introduces considerable uncertainties to the Tier 1 exposure assessment because the available information suggests that manufacturing methods and product uses may differ between Europe and the United States, which may result in releases of different congeners or congener concentrations that are different from levels reported in Europe. The available data describing BDE levels in fish in the U.S. and European countries, for example, may be attributable, in part, to different manufacturing methods and product uses in commercial and consumer goods.

Risk Assessment

- The USEPA (1989) notes that the conservative assumptions used in risk assessment are intended to assure that the estimated risks do not underestimate the actual risks posed and that the estimated risks do not necessarily represent actual risks experienced by the affected population.
- The exposure and hazard information and assumptions used in this Tier 1 assessment were intended to provide conservative, upper-bound estimates of exposure and hazard to children and prospective parents. The use of probabilistic methods such as those used by Wenning (2002) could provide further insight on the factors that most influence the results described in this Tier 1 assessment. The application of probabilistic methods would be useful to identify the key exposure and hazard assumptions that could be evaluated further in Tier 2 or Tier 3 VCCEPP activities.

5.0 Data Needs Assessment

The Data Needs Assessment presented in this section addresses the USEPA's request for chemical sponsors to specify the additional hazard and/or exposure information needed to further evaluate the potential risks to children and, where relevant, prospective parents. The assessment presented herein conforms to USEPA's recommendations contained in its HPV guidance for determining the adequacy of existing data (USEPA, 1999c).

The data needs identified in this Tier 1 assessment were developed based on consideration of the available toxicology and exposure information using a weight-of-evidence evaluation process similar to that used by ECB (2000) in their risk assessment for the commercial pentaBDE product. USEPA risk assessment guidance (1997, 1999a, 2000b) supports the use of a weight-of-evidence evaluation of exposure and risk information when considering data gaps and recommendations for additional studies. This approach considers the degree to which available hazard and exposure information sufficiently addresses the data needed to perform both an exposure and risk assessment. In situations where adequate data may be lacking for a particular hazard endpoint or exposure parameter, a weight-of-evidence approach considers the impact of the limitation on the ability to adequately evaluate the potential hazard or exposure to children from environmental releases. The weight-of-evidence approach is the appropriate method for evaluating whether data gaps significantly (or does not significantly) impact the ability to evaluate the risks to children.

While data is available and adequate for many of the Tier 1, 2, and 3 endpoints, there are data gaps. Table 5-1 summarizes the Tier 1, Tier 2, and Tier 3 toxicology studies specified by USEPA in the Federal Register notice for the VCCEPP program. For each type of toxicology study specified by USEPA, Table 5-1 indicates whether one or more studies are either available and adequate for the VCCEPP or have not been conducted at the present time for the commercial pentaBDE product. The use of the term "adequate" to describe the different types of toxicology information listed in Table 5-1 is defined in accordance with the HPV Challenge Program and GLPS (40 CFR Part 792 Subpart A-J), which outlines the provisions that USEPA has set forth to determine if the data are sufficient to assess the health and environmental effects of a chemical substance.

In addition to the toxicology studies listed in Table 5-1, several additional hazard bioassays and exposure assessment studies are currently underway or planned by government regulatory and private research organizations in the United States and Canada. For example, the U.S. National Toxicology Program (NTP) has initiated several toxicokinetic studies, including 90-day and two-year chronic bioassays in mice, B6C3F1 rats, and Fischer 344 rats and a 26-week bioassay in TG/RASH2/CB6F1 (transgenic) mice to evaluate neurobehavioral developmental endpoints in rodents. The U.S. Centers for Disease Control and Prevention (CDC) has included the measurement of the commercial pentaBDE product and several BDE congeners in the next national biomonitoring study scheduled to begin in 2003. The National Health and Nutrition

Examination Survey (NHANES) is a periodic survey conducted by the CDC's National Center for Health Statistics (NCHS) to collect information about the health and diet of people in the United States. The next round of NHANES data collection will involve both dietary and personal habit questionnaires, medical evaluations, and laboratory testing of blood and other tissues. Similar research is underway in Canada with exposure and environmental testing work jointly conducted by Health Canada and Environment Canada. Several environmental studies – primarily to determine levels in ambient air, municipal wastewater, and fish and other foods – are planned by California EPA in 2003 as part of ongoing research conducted by the Office of Environmental Health and Hazard Assessment (OEHHA).

It is anticipated that the results of current and planned hazard and exposure studies in the United States and Canada should improve the understanding of BDE levels in the environment and the potential risks posed to children and prospective parents. The most notable data gap in the available data is the absence of a two-year chronic bioassay. ***A two-year chronic bioassay of the commercial pentaBDE product is identified as a possible data gap.*** At present, chronic studies have been completed only for the commercial decaBDE product. The available hazard data do not provide sufficient information concerning the potential for carcinogencity as an endpoint of concern for pentaBDE.

In addition to a chronic bioassay, mechanistic studies also are identified as possible data gaps in the VCCEPP to improve the understanding of the effects of BDE constituents of the commercial pentaBDE product on children. Further, single or multiple generation reproductive toxicity studies have not been conducted. For example, studies may be appropriate to evaluate the potential for changes in thyroid function in children and adults. According to USEPA (1998a), rodents are generally believed to be more sensitive to thyroid hormone disruption than humans; consequently, additional research may be appropriate to determine whether the changes in thyroid function reported in animal bioassays are relevant to humans. At present, the significance of changes in T_4 levels reported in rodents exposed to certain BDEs is not well understood. Neurobehavioral developmental studies, in addition to those underway or planned by regulatory researchers in the United States and elsewhere, may be appropriate to evaluate the potential for effects in laboratory animals and whether the effects observed in rodents should be extrapolated to humans. However, given that conservative exposure estimates are used in estimating children's exposures, the hazard indices indicate a large margin of safety. Therefore it is difficult to characterize these additional studies as needs, although additional studies would enhance our understanding. The studies planned by U.S. and/or Canadian agencies along with existing data should provide enough information to fully characterize the potential hazards potentially posed by exposure to the commercial pentaBDE product.

Environmental monitoring data for pentaBDE product-related congeners, particularly in food products and human breast milk, represents another possible data gap. At present, limited data are available describing environmental levels in ambient air, soil, water, and different foods in the United States. The California Air Resources Board has listed BDEs as analytes for future

environmental monitoring. Recently, the San Francisco Estuaries Institute has initiated a monitoring study of fish and sludge in the San Francisco Bay Area. In the city of Palo Alto, California, scientists have initiated a study of waste effluents, sludges, and incinerator gas emissions to evaluate the presence of BDEs. In addition, the California Department of Toxic Substance Control has begun testing leachate from various electronic materials; similar studies, including more extensive environmental monitoring, are planned by the Canadian government within the next few years.

Table 5-1. Tier 1, Tier 2, and Tier 3 toxicology studies specified by USEPA as part of the VCCEPP

VCCEPP Tier	Toxicology Studies Specified by USEPA [1]	Status
Tier 1 Assessment Studies	▪ Acute oral toxicity OR acute inhalation toxicity;	✓ Available and adequate for VCCEPP
	▪ *In vitro* gene mutation: bacterial reverse mutation assay;	✓ Available and adequate for VCCEPP
	▪ Combined repeated dose toxicity with reproductive and developmental toxicity screens OR repeated dose oral toxicity AND reproductive toxicity (1-generation);	✗ Not adequate for VCCEPP
	▪ *In vitro* chromosomal aberrations OR i*n vivo* chromosomal aberrations OR *in vivo* mammalian erythrocyte micronucleus;	✓ Available and adequate for VCCEPP
Tier 2 Assessment Studies	▪ 90-day subchronic toxicity in rodents;	✓ Available and adequate for VCCEPP
	▪ Reproduction and fertility effects;	✗ Not available
	▪ Prenatal developmental toxicity (two species);	✓ Available and adequate for VCCEPP
	▪ *In vivo* mammalian bone marrow chromosomal aberrations OR *in vivo* mammalian erythrocyte micronucleus (triggered off results from *in vitro* mammalian chromosomal aberration test if conducted in Tier 1);	✗ Not conducted
	▪ Immunotoxicity	✓ Available and adequate for VCCEPP
	▪ Metabolism and pharmacokinetics	✓ Metabolism studies not available
Tier 3 Assessment Studies	▪ Carcinogenicity OR chronic toxicity/carcinogenicity	✗ Not conducted
	▪ Neurotoxicity screening battery	✗ Not conducted
	▪ Developmental neurotoxicity	✗ Not available

[1] From Federal Register FRP-6758-5.

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