DEPARTMENT OF ECOLOGY
State of Washington

Children's Safe Products Reporting Rule

Chemicals of High Concern to Children Added or Delisted during the 2017 Rule Update

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Contact Information

Hazardous Waste and Toxics Reduction Program
P.O. Box 47600
Olympia, WA 98504-7600
Phone: 360-407-6700
Ecology website: www.ecology.wa.gov

Children's Safe Products Reporting Rule

Added and Delisted
Chemicals of High Concern to Children

Department of Ecology
Olympia, Washington

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Introduction

The Children's Safe Products Reporting Rule (CSPA Reporting Rule, Chapter 173-334 Washington Administrative Code (WAC)[1]) is authorized by the Children's Safe Products Act (CSPA, Chapter 70.240 Revised Code of Washington (RCW)[2]).

The CSPA Reporting Rule requires manufacturers to annually report to Ecology the presence of Chemicals of High Concern to Children (CHCCs) in children's products offered for sale in Washington. The rule identifies the CHCCs and details the process for manufacturers to report to Ecology. In 2016, the law was amended identifying six flame retardants to be considered for inclusion on the CHCCs list in the CSPA Reporting Rule.

Ecology and the Washington Department of Health (Health) evaluated those six flame retardants and other chemicals against the CHCC criteria, using the same basic process followed during original rule development in 2011.[3] Ecology solicited and considered stakeholder comments for the chemicals identified for CHCC addition or delisting during the preliminary and proposed rulemaking efforts.

The CSPA Reporting Rule update included the following efforts:

- Determine whether the six flame retardants should be proposed as CHCCs.
- Identify for inclusion other chemicals that meet the criteria in the law.
- Identify chemicals that may need to be removed from the CHCC list.
- Streamline the rule to make compliance easier.

This document provides the chemical evaluations of CHCC additions and delistings for the adopted CSPA Reporting Rule.

The adopted CSPA Reporting Rule (published on September 21, 2017 added 20 CHCCs and expanded one listed CHCC to three listings in Section 130 of the rule. The first section of this report includes 22 CHCC evaluations. Three existing CHCCs are delisted from the CHCC list. Evaluations for the delisted CHCCs are located at the end of this document. CHCC evaluations are listed in numerical order by chemical abstract service (CAS) number, first for the additions followed by the delistings. The table of contents offers links to the start of each evaluation.

For more information about the CSPA Reporting Rule and the content of these evaluations, contact the Hazardous Waste & Toxic Reductions Program at the Department of Ecology in Lacey, Washington.

[1] app.leg.wa.gov/wac/default.aspx?cite=173-334&full=true
[2] app.leg.wa.gov/rcw/default.aspx?cite=70.240&full=true
[3] https://fortress.wa.gov/ecy/publications/SummaryPages/1704022.html

Criteria for Chemicals of High Concern to Children

During the 2017 rulemaking, changes to the list of CHCCs followed the same basic process that was used to create the original CHCC list in 2011 and update it in 2013. The 2011 CHCC listing process prioritized three toxicity endpoints: carcinogenicity, reproductive/developmental toxicity, and endocrine disruption. Other toxic endpoints (like liver toxicity, neurotoxicity, or aquatic toxicity) were not considered for listing purposes. The process also prioritized potential for exposure as being in children's products or in people.

CHCCs selected for addition or delisting either did or did not meet the listing criteria. CHCC listing criteria are based on authoritative sources that identify chemical toxicity (RCW 70.240.010) and evidence of potential for exposure (RCW 70.240.030(1)). Source references are provided at the end of this document.

Authoritative sources used to determine toxicity:

- California's Proposition 65 list for cancer, birth defects, or other reproductive harm (OEHHA 2017).
- National Toxicology Program (NTP) Center for the Evaluation of Risks to Human Reproduction monographs and Report on Carcinogens (NTP 2016).
- The International Agency for Research on Cancer (IARC 2017).
- Consumer Product Safety Commission's Chronic Hazard Advisory Panel (CHAP) Report on Phthalates (CPSC 2014).
- U.S. EPA sources:
  - Alternatives assessments on flame retardants (EPA 2015).
  - Integrated Risk Information System (IRIS; EPA 2017).
- European Union sources:
  - Substances restricted or authorized under the EU Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) regulation (ECHA 2017).
  - Candidate list of Substances of Very High Concern (SVHC) under REACH (ECHA 2017).
  - Existing Substances Regulation (ECHA 2017).
  - Priority list of chemicals identified as suspected endocrine disruptors (EC 2017).

Authoritative sources used to determine potential for exposure:

- Scientific studies published in peer-reviewed journals showing presence in children's products, house dust, indoor air, or biomonitoring data.
- Danish environmental agency surveys on chemicals in consumer products (DEPA 2017).
- Centers for Disease Control (CDC) National Health and Nutrition Examination Survey (NHANES) (CDC 2015).
- Washington State list of persistent, bioaccumulative, and toxic (PBT) chemicals (Chapter 173-333 WAC).

CHCC Additions

Twenty-two chemicals are added as CHCCs in the adopted CSPA Reporting Rule. The evaluations completed by Ecology and Health are provided in this document. Each evaluation includes the CAS number, a common chemical name, summary of toxicity, summary of potential for exposure, and a list of references. Some CHCC evaluations cover more than one chemical (see Nonylphenol and Short-chain chlorinated paraffins).

CAS	Name	Acronym
80-09-1	Bisphenol S	BPS
84-61-7	Dicyclohexyl phthalate	DCHP
84-69-5	Diisobutyl phthalate	DIBP
115-86-6	Triphenyl phosphate	TPP
117-82-8	Di-(2-methoxyethyl) phthalate	DMEP
126-72-7	Tris (2,3-dibromopropyl) phosphate	TDBPP
126-73-8	Tri-n-butyl phosphate	TNBP
131-18-0	Dipentyl phthalate	DPP
335-67-1	Perfluorooctanoic acid	PFOA
620-92-8	Bisphenol F	BPF
1241-94-7	Ethylhexyl diphenyl phosphate	EHDPP
1330-78-5	Tricresyl phosphate	TCP
13674-84-5	Tris (1-chloro-2-propyl) phosphate	TCPP
25154-52-3 84852-15-3	Nonylphenol 4-Nonylphenol branched	
26040-51-7	Bis (2-ethylhexyl) tetrabromophthalate	TBPH
38051-10-4	Bis(chloromethyl)propane-1,3-diyl tetrakis-(2-chloroethyl) bis(phosphate)	V6
68937-41-7	Isopropylated triphenyl phosphate	IPTPP
84852-53-9	Decabromodiphenyl ethane	DBDPE
85535-84-8 108171-26-2	Short-chain chlorinated paraffins Chlorinated paraffins	SCCP
183658-27-7	2-ethylhexyl-2,3,4,5-tetrabromobenzoate	TBB

CAS 80-09-1 - Bisphenol S (BPS)

Summary of Toxicity

EPA classified Bisphenol S (BPS) as high hazard for toxicity from repeated exposure based on no-observed-adverse-effect-level (NOAEL) of 10 and 40 mg/kg-day in repeated dose rat studies [1]. A 28-day oral study of BPS in rats showed effects on body weight, increased kidney weight, hyperplasia and necrosis in mucosal epithelium of the cecum, and increased incidence of proteinuria and urobilinogen at 200 mg/kg-day. The NOAEL was 40 mg/kg-day [1].

EPA classified BPS as a moderate hazard for reproductive and developmental toxicity based on prolonged estrus cycle, decreased fertility index, decreased number of live offspring, and liver effects observed at 300 mg/kg-day in a reproductive and developmental toxicity test in orally exposed rats. Although the NOAEL for reproductive effects was 60 mg-kg-day, pathology was noted at this dose in the cecum [1]. A recent 90-day oral study in rats reported atrophy of mammary glands in male rats treated with at 300 mg/kg-day of BPS. This study also observed a dose-dependent increase in focal squamous cell metaplasia of glandular epithelium in the uterus of female rats across all doses (100, 300, and 1000 mg/kg-day) but it was unclear when the increase became statistically significant [2].

BPS has been assessed as part of the NTP's Tox21 High Throughput Screening Program where it was classified as an estrogen agonist with some affinity for the estrogen receptor [3]. *In vitro* assays demonstrate that BPS can bind to estrogen receptors, elicit estrogen-induced gene transcription, induce cell proliferation in MCF7 cancer cells, and inhibit the androgenic activity of dihydrotestosterone [1]. In a systematic review of BPS, BPA, and BPF endocrine studies, BPS had estrogenic activity in whole organism testing (Zebrafish, Daphnia magna) and in a number of *in vitro* tests. On average, BPS was about 1/3 as potent as BPA in estrogenic activity *in vitro* assays [4].

Summary of Potential for Exposure

BPS exposures can occur through oral, dermal, or inhalation routes. However, primary exposure likely occurs through the oral route. Information on distribution in the body, metabolism, and excretion is mostly lacking [3].

Washington State banned BPA for use in baby bottles, infant sippy cups, and sports water bottles starting in 2010 (Washington State Law; Chapter 70.280 RCW). BPS is used as a replacement for BPA in polymer production and thermal papers. BPS is used in polyethersulfone (PES) plastics used to make baby bottles [3,5,6]. BPS has been detected in personal care products [7], and sales receipt paper and other paper products [8,9]. National U.S. production volume was reported to be 1-10 million pounds in 2012 [10].

BPS was found in 81% of the human urine samples analyzed from general populations in the United States and several Asian countries collected in 2010-2011. Urine concentrations in U.S. samples had a median of 0.26 ng/mL and a maximum detection of 21 ng/mL [11]. In another biomonitoring study, archived urine samples from U.S. adults collected from 2000-2014 showed increasing levels of BPS over time 12]. BPS was also measured in the serum and urine of cashiers and a control group of adults in a North Carolina study. Urinary levels of BPS were higher in cashiers following a shift handling receipt paper that contained BPS [9]. BPS was detected in 100% of 38 indoor dust samples collected in New York in 2006 and 2010. Median detected concentration was 630 ng/g dust and the maximum was 25,500 ng/g dust [13]. BPS has also been found in a variety of foods collected from retail grocery stores in Albany, NY, in 2008-2010. It was detected in 43% of meats and meat products and about ¼ of seafood, fruit, and vegetable samples [14].

BPS was considered to have moderate persistence and low potential for bioaccumulation by EPA [1].

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CAS 84-61-7 - Dicyclohexyl phthalate (DCHP)

Summary of Toxicity

Dicyclohexyl phthalate (DCHP) is identified as an endocrine disruptor based on the EU Category 1 designation as an endocrine disruptor [1]. The EU developed the priority list in stages (2000, 2002, and 2007), putting chemicals in three categories. The EU **Category 1** endocrine distruptor designation has been used as an authoritative source for CSPA. Category 1 requires evidence of endocrine disrupting activity in at least one species using intact animals. **Category 2,** which requires at least some in vitro

evidence, is too preliminary. **Category 3** is no evidence of endocrine disrupting activity or no data available.

The Consumer Product Safety Improvement Act of 2008 (CPSIA) directed the U.S. Consumer Product Safety Commission (CPSC) to convene the CHAP on Phthalates and Phthalate Alternatives "to study the effects of all phthalates and phthalate alternatives as used in children's toys and child care articles." The CHAP assessed the risks of fourteen phthalates and six phthalate alternatives, including three phthalates permanently banned by the CPSIA and three phthalates subject to an interim ban. CHAP is an authoritative source for CSPA [2].

DCHP was included in the CHAP report, which found studies in rodents suggest that exposure to DCHP can induce adverse effects in reproductive organs and suggests that DCHP is a developmental toxicant [2]. The CHAP panel found the toxicological profile of DCHP is very similar to other antiandrogenic phthalates and thus, exposure to DCHP contributes to the cumulative risk from other antiandrogenic phthalates. The CHAP report recommends that DCHP be permanently banned from use in children's toys and child care articles at levels greater than 0.1%.

Summary of Potential for Exposure

There is new information on the presence of DCHP in indoor dust (2.9 ug/g, 0.3 ug/g) and air (4-5 ng/m^3, 0.07 ug/m^3) in several studies [3]. DCHP was also found in soap (100 ug/g), modeling clay (4,000 mg/kg), and pajamas (3,400 mg/kg), but they are not noted as being for children [3].

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3. European Chemicals Agency (ECHA), Sweden and Denmark, 2015. Annex XV Report: Proposal for Identification of a Substance of Very High Concern on the Basis of the Criteria Set Out in REACH Article 57 Substance Name(s): Dicyclohexyl phthalate (DCHP) EC Number(s): 201-545-9 CAS Number(s): 84-61-7. http://echa.europa.eu/documents/10162/b2fbb22c-72d7-491d-b417-39105e35b792.

CAS 84-69-5 - Diisobutyl phthalate (DIBP)

Summary of Toxicity

Diisobutyl phthalate's (DIBP) is identified as a SVHC as toxic for reproduction [1]. Substances that may have serious and often irreversible effects on human health and the environment can be identified as SVHCs under the EU Registration, Evaluation, Authorisation, and Restriction of Chemicals (REACH) law. If a substance is identified as an SVHC, it is added to the Candidate List for eventual inclusion in the Authorisation List.

The Consumer Product Safety Improvement Act of 2008 (CPSIA) directed the U.S. Consumer Product Safety Commission (CPSC) to convene a Chronic Hazard Advisory Panel (CHAP) on Phthalates and Phthalate Alternatives "to study the effects of all phthalates and phthalate alternatives as used in children's toys and child care articles." The CHAP assessed the risks of fourteen phthalates and six

phthalate alternatives, including three phthalates permanently banned by the CPSIA and three phthalates subject to an interim ban. CHAP is an authoritative source for CSPA [2].

DIBP was included in the CHAP report [2], which found that animal and human studies suggest that exposure to DIBP can cause reproductive and developmental effects. The CHAP found the toxicological profile of DIBP is very similar to other antiandrogenic phthalates and thus, exposure to DIBP contributes to the cumulative risk from other antiandrogenic phthalates and its use should be permanently banned from use in children's toys and child care articles at levels greater than 0.1%.

Summary of Potential for Exposure

DIBP and it's metabolites have been detected in people in biomonitoring studies [2]; in blood samples up to 541 ng/g [3], and DIBP metabolites in urine samples from 6 to 9 year old girls up to 363 ug/L [4].

DIBP has been reported in indoor air (0.50 ug/m^3 [5] and max 990 ng/m3 [6]) and dust (max 39.1 ug/g [6] and 3.81 mg/g [7]). DIBP has been reported in children's products [8, 9, 10, 11, 12].

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CAS 115-86-6 - Triphenyl phosphate (TPP)

Summary of Toxicity

EPA classified Triphenyl phosphate (TPP) as a high hazard for toxicity from repeated exposures [1]. Decreased body weight gain in adult rats was the most sensitive endpoint reported following repeated oral exposure; the lowest-observed-adverse-effect-level (LOAEL) was 161 mg/kg-day. At higher doses, reproductive and fetal effects were observed [1]. TPP appears to be active in endocrine tissues. In a recently published study, mice exposed to 300 mg/kg-day TPP orally for 35 days had decreased testes weight, histopathological damage, decreased testicular testosterone levels, decreased expression of genes related to testosterone synthesis, and signs of oxidative stress in the liver [2]. *In vitro* testing shows that TPP is a moderate androgen-receptor binder and can inhibit receptor function (testosterone-induced androgen-receptor-dependent activity) [1]. TPP and its hydroxylated metabolites acted as estrogen receptor agonists in other *in vitro* studies [3, 4]. Only limited human evidence of endocrine disruption is available. A study in Boston, Massachusetts, reported that men living in homes with higher TPP in house dust had decreased sperm counts and altered hormone (prolactin) levels [5].

There is also emerging evidence that TPP may cause long-lasting metabolic disruption in rats exposed during fetal and nursing periods [6, 7]. Green et al. 2016 showed that developmental exposure to TPP alone caused accelerated onset of type 2 diabetes in a rat diabetes model and increased body fat later in life [7]. The very low dose used in this study (17 µg/rat-day; <0.5 mg/kg-day) was not associated with overt toxicity or weight change in treated dams or offspring at birth. It was equivalent to the dose of TPP present in a study by Patisaul et al. 2013, that observed metabolic disruption in offspring following developmental exposure to 1 mg/rat-day of Firemaster® 550 [6]. These study results suggest a high hazard for developmental toxicity.

Investigators at the National Toxicology Program used cell-based *in vitro* assays and assays in rapidly developing whole organisms (in this case, the nematode *C. elegans*) to screen for potential developmental toxicity and neurotoxicity of a number of phosphate flame retardants [8, 9]. TPP had a more potent impact on larval development than PBDE[4] flame retardants and was a relatively strong inhibitor of mitochondrial activity in *in vitro* testing [9].

Summary of Potential for Exposure

TPP is a plasticizing flame retardant in polyvinyl chloride (PVC). It is also used as a flame retardant in other polymers, textiles, polyurethane foam, electronic circuit boards, photographic films, and building materials. [10, 11]. It is a component of Firemaster® 550 used in polyurethane foams and has been detected in baby products [11, 12], other children's products, carpet pads, and plastic parts of LCD

[4] Pbde - polybrominated diphenyl ether

monitors [13]. TPP is an additive flame retardant and migrates from computer monitors and television sets [11]. TPP is also used as a plasticizer and may be in clothing, textiles, cosmetics, and personal care products [14]. It is listed as an ingredient in nail polish and a recent biomonitoring study showed short-term spikes in exposure following application of nail polish [15]. U.S. national production volume was reported to be 10,796,422 million pounds per year in 2012 [16].

Because of its physical properties, TPP that escapes from consumer products, either by emission or abrasion, is likely to end up in indoor dust. TPP was detected at high levels in indoor dust in studies of homes in North Carolina, Boston, California, and Canada [17-20]. Maximum detected level was 1,800 µg/g dust. It has also been detected in U.S. office and vehicle dust [21]. TPP has also been measured in the indoor air of homes and public buildings in a number of countries. Maximum level reported was 100 ng/m^3 [11].

Diphenyl phosphate (DPHP), a metabolite of TPP, has been found in urine at high frequency (>90%) in North American biomonitoring studies including Boston adults [22], New Jersey mothers and toddlers [23], California mothers and their children aged 2-70 months [24], and North Carolina babies [25]. Levels measured in children were higher than their mothers [23, 24, 26] and were higher in children with more reported hand-to-mouth behaviors [23, 24]. Mean and median levels of DPHP in urine reported across these studies have been less than 3.2 ng/mL with a maximum reported level of 140 ng/mL. TPP has been measured up to 140 ng/g lipid in human breast milk in Asian and Swedish studies [27, 28]. TPP was detected in 98% of hair samples and 74% of finger and toenail samples in a population of young adults in Indiana [29].

TPP appears to be ubiquitous in the environment and has been detected in drinking water, river water, seawater, rainwater, snow, wastewater effluent, ambient air, and indoor air [1, 11, 18, 30-33].

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CAS 117-82-8 – Di(2-methoxyethyl) phthalate (DMEP)

Summary of Toxicity

In December 2011, the European Union added di(2-methoxyethyl) phthalate (DMEP) to the candidate list of substances of very high concern (SVHC) based on a determination that DMEP is toxic for reproduction (1, 2). This is part of implementing the EU law Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). Substances that may have serious and often irreversible effects on human health and the environment can be identified as SVHCs. If a substance is identified as an SVHC, it will be added to the Candidate List for eventual inclusion in the Authorisation List.

Summary of Potential for Exposure

Australian Department of Health reported use of DMEP as a plasticizer in toys, however no product testing results were reported (3, 4).

A Canadian screening assessment of DMEP points out many studies where DMEP was tested for but not detected and concludes that "available data do not indicate the existence of consumer products containing DMEP in the Canadian marketplace." For example, DMEP was not detected in a Health Canada survey of phthalates in 70 soft vinyl children's products (5).

In contrast, investigations reported DMEP in vacuum cleaner bag dust and house carpets in Germany in studies conducted between 1998 and 2000 (Kersten 2003 and Pfordt 1999 as cited in BAuA Dossier [6]). DMEP was detected in 100 percent of the 153blood samples of Hong Kong residents in 2013 (7). Phthalate testing of a variety of cosmetic products in Shanghai detected DMEP in one baby care product (shampoo) in 2013 (8).

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CAS 126-72-7 - Tris (2,3-dibromopropyl) phosphate (TDBPP)

Summary of Toxicity

Tris (2,3-dibromopropyl) phosphate (TDBPP) is reasonably anticipated to be a human carcinogen by the National Toxicology Program [1], is listed as carcinogen on California's Proposition 65 List, and is classified as possible (2A) carcinogen by the International Agency for Research on Cancer (IARC). According to the European Food Safety Authority (EFSA) (2012), there is convincing evidence that TDBPP is genotoxic and carcinogenic [2].

Summary of Potential for Exposure

TDBPP was used as a flame retardant in children's clothing until banned in 1977 [3]. According to the National Toxicology Program, it has been used as an additive flame retardant in polyurethane foams, polystyrene foam, acrylic carpets and sheets, water flotation devices, polyvinyl and phenolic resins, paints, lacquers, paper coatings, styrene-butadiene rubber, and latex [1]. These types of materials are

used in children's products and the chemical is still available for sale from overseas suppliers. A disclosure requirement could confirm that imported children's products do not contain this flame retardant. No current information on uses or national production volume is available [4].

TDBPP has not been included in many house dust sampling studies. It was identified in one study of house dust in California [5]. No biomonitoring studies were located.

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CAS 126-73-8 – Tri-n-butyl phosphate (TNBP)

Summary of Toxicity

According to the European Chemicals Agency (ECHA) Tri-n-butyl phosphate (TNBP) is suspected to cause cancer and is a category 2 cancer hazard [1]. TNBP caused dose-related increases in the incidence and severity of urinary bladder tumors in male and female rats with dietary exposure for two years [2]. Male mice with chronic dietary exposure developed liver tumors [3]. The American Conference of Governmental Industrial Hygienists (ACGIH) classified TNBP as a confirmed animal carcinogen with unknown relevance to humans [4].

The U.S. Agency for Toxic Substances and Disease Registry (ATSDR) evaluated available toxicity data for TNBP and developed human health screening values [3]. Acute oral exposure guidelines were based on reduced weight gain in rats dosed during pregnancy. The lowest-observed-adverse-effect-level (LOAEL) for this maternal effect was 125 mg/kg-day [3]. No observable birth defects in fetuses were observed at gestation day 20 at this dose. Higher oral doses in subacute rat testing caused neurological signs and symptoms, changes in liver and spleen weights, and degenerative changes in the testes (Laham et al. 1983 and Noda et al. 1994 as cited in ATSDR review [3]). Urinary bladder hyperplasia was the most sensitive effect observed in three oral rat studies of longer duration (Arnold et al. 1997, FMC 1985, Tyl et al. 1997 reviewed in ATSDR [3]). ATSDR selected the study by Arnold et al. 1997, with a LOAEL of 33 mg/kg/d, to derive human screening levels for both intermediate and chronic duration [3]. ATSDR's human health screening value for TNBP is 0.08 mg/kg-day for intermediate and chronic exposures [3].

In vitro tests show that TNBP, but not its metabolite di-n-butyl phosphate (DNBP), may act as an antagonist for androgen and the glucocorticoid nuclear receptors [4, 5]. Neither TNBP nor its metabolite DNBP had an effect on estrogen receptors *in vitro* [5, 6].

Summary of Potential for Exposure

TNBP is mainly used as an additive in fire-resistant aircraft hydraulic fluids and as a plasticizer for cellulose esters, lacquers, plastics, and vinyl resins [4]. It may be present in floor finish, floor wax, paints, and glues. It also has a number of industrial applications [3]. U.S. national volume production was reported to be 8,877,744 pounds/year in 2012 [7].

TNBP has been measured in indoor dust and air in U.S. and European studies [8-12]. The maximum level reported was 7,100 ng/g in house dust [10]. Two European studies included air measurements and found TNBP more commonly in indoor air than in dust at homes and daycare centers [8, 11]. Recent residential sampling in Norway by Xu et al. reported 98% detection in residential indoor air with a median of 14 ng/m^3 and a maximum detection of 119 ng/m^3 [11]. Inhalation exposure was the predominant route of estimated human residential exposure [11].

Biomonitoring studies indicate that TNBP is making its way into people's bodies. Dodson et al. measured metabolites of TNBP in urine from adults in Northern California [13]. Fromme et al. reported slightly higher mean levels of TNBP urinary metabolite in a population of 312 children attending 63 German day care centers [8]. TNBP has been detected in breast milk samples from Sweden and several Asian counties [14, 15]. TNBP was recently measured directly in blood of 237 adults in a Chinese study [16]. The median level reported was 37.8 ng/mL, which was much higher than the other organophosphorus flame retardants measured.

There is some evidence of TNBP in the U.S. diet, drinking water, and ambient air. TNBP has been found at low parts per billion levels in cereal products including baby food in the U.S. [3, 4, 17]. Focazio et al. 2008, detected TNBP in a study of 74 public drinking water systems from 25 states and Puerto Rico. TNBP was detected in 8.1% of the samples with a maximum of 0.74 µg/L as cited in [3]. TNBP was detected in 100% or urban air samples from the Great Lakes area with an average concentration of 150-250 pg/m^3. Lower air concentrations (average of 34 pg/m^3) were detected at remote locations [4].

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CAS 131-18-0 – Dipentyl phthalate (DPP)

Summary of Toxicity

Di pentyl phthalate (DPP) is designated by the EU as a Category 1 endocrine disruptor [1, 2]. The EU developed their priority list of endocrine disruptors in stages (2000, 2002, and 2007), grouping chemicals into three categories. The EU **Category 1** endocrine distruptor designation has been as an authoritative source, because Category 1 requires evidence of endocrine disrupting activity in at least one species using intact animals. **Category 2** requires at least some *in vitro* evidence but is considered insufficient evidence of endocrine activity, while **Category 3** indicates either no evidence of endocrine disrupting activity or no data available.

DPP has been identified as a SVHC based on a toxic for reproduction designation [3,4]. This is part of implementing the EU law Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). Substances that may have serious and often irreversible effects on human health and the

environment can be identified as SVHCs. If a substance is identified as an SVHC, it will be added to the Candidate List for eventual inclusion in the Authorisation List.

The Consumer Product Safety Improvement Act of 2008 (CPSIA) directed the U.S. Consumer Product Safety Commission (CPSC) to convene a Chronic Hazard Advisory Panel (CHAP) on Phthalates and Phthalate alternatives "to study the effects of all phthalates and phthalate alternatives as used in children's toys and child care articles." The CHAP assessed the risks of 14 phthalates and 6 phthalate alternatives, including three phthalates permanently banned by the CPSIA and three phthalates subject to an interim ban. CHAP is an authoritative source for CSPA [2].

In 2014, DPP was included in the CHAP report, which found "DPENP is clearly among the most potent phthalates regarding developmental effects" [5]. The CHAP panel found the toxicological profile of DPP is very similar to other antiandrogenic phthalates and thus, exposure to DPP contributes to the cumulative risk from other antiandrogenic phthalates and its use should be permanently banned from use in children's toys and child care articles at levels greater than 0.1%.

Summary of Potential for Exposure

DPP was detected in house dust in northern California [6]. A metabolite of DPP, MnPeP, was detected in children's urine in Austria [7] and Germany [8].

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CAS 335-67-1 - Perfluorooctanoic acid (PFOA)

Summary of Toxicity

In 2013 PFOA was identified by the European Union to be a substance of very high concern (SVHC) as toxic for reproduction [1]. This is part of implementing the EU law Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). Substances that may have serious and often irreversible effects on human health and the environment can be identified as SVHCs. If a substance is identified as an SVHC, it will be added to the Candidate List for eventual inclusion in the Authorisation List.

In 2016 the International Agency for Research on Cancer (IARC) classified PFOA as possible carcinogenic to humans (category 2B) [2]. IARC is part of the World Health Organization and its mission is to coordinate and conduct research on the causes of human cancer, the mechanisms of carcinogenesis, and to develop scientific strategies for cancer control. IARC publishes monographs which identify carcinogenic chemicals.

Summary of Potential for Exposure

PFOA has been detected in biomonitoring studies [3, 4, 5, 6] and house dust [7, 8].

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CAS 620-92-8 - Bisphenol F (BPF)

Summary of Toxicity

EPA classified bisphenol F (BPF) as high hazard for toxicity from repeated exposures based on reduced body weight and decreased total serum cholesterol, glucose, and albumin at 20 mg/kg-day in a 28-day oral rat study. BPF was classified by EPA as a moderate hazard for reproductive toxicity and a high developmental hazard based primarily on toxicity of its structural analog BPA [1].

In a systematic review of BPS, BPA, and BPF endocrine studies, BPF had estrogenic and anti-androgenic activity in *in vitro* testing [2]. On average, BPF was as potent as BPA in estrogenic activity assays and about half as potent as BPA in anti-estrogenic activity assays [2].

Summary of Potential for Exposure

In rodents, bisphenol F is readily absorbed following oral exposure, metabolized, and excreted primarily in the urine [1].

Washington State banned BPA for use in baby bottles, infant sippy cups and sports water bottles starting in 2010 (Chapter 70.280 RCW). BPF is used as a replacement for BPA in epoxy resins used to line food cans and in polymer plastics [3]. BPF has been detected in personal care products such as lotions and cosmetics [4]. National U.S. production volume was reported to be 355,000 pounds in 2012 [5].

BPF was detected in 68% of indoor dust samples collected between 2006 and 2010 in New York. Median detected concentration was 49 ng/g dust and the maximum detected was 240 ng/g. Of 8 bisphenol analogs measured, it was the third most common bisphenol detected after BPA and BPS [6].

BPF was detected in urine collected between 2000 and 2014 from U.S. adults. Depending on the collection time, BPF was detected in 42-88% of samples and the mean detection was 0.15-0.54 ng/mL [7].

BPF was detected more frequently than other BPA analogs in a variety of foods collected from retail grocery stores in Albany, NY, between 2008 and 2012 [3]. The maximum concentration detected (1130 ng BPF/g sample) was in a salad dressing packaged in a plastic container. BPF was most frequently detected in fats and oils, dairy products, fish and seafood, meat products, and vegetables, and was mostly associated with foods packaged in cans. The authors estimated daily dietary exposure to BPF through U.S. food for different age groups and found toddlers had the highest estimated intakes (mean 22.3 ng/kg bw-day, 95th percentile 70.2 ng/kg bw-day) [3].

BPF may be slower to degrade in the environment than BPA [8], but is not expected to have high persistence or high potential for bioaccumulation [1]. BPF has been reported to occur in surface water, sewage, and sediments [9].

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CAS 1241-94-7 - Ethylhexyl diphenyl phosphate (EHDPP)

Summary of Toxicity

Toxicity data for Ethylhexyl diphenyl phosphate (EHDPP) was reviewed by the United Kingdom Environmental Agency in 2009 [1]. Dose-related changes to the blood, liver, kidney, adrenal glands, testes, and ovaries were observed in laboratory rats exposed to 375-425 mg/kg-day of commercial EHDPP in their food over 90 days [1, 2]. The lowest-observed-adverse-effect-level (LOAEL)[5] from three 90-day feeding experiments was 15 mg/kg/day for increase in liver enzymes in male rats (NOAEL[1] was 6 mg/kg-day). A fertility and reproductive toxicity study in rats reported that mating and reproductive performance were unaffected by treatment (up to 0.8% EHDPP in food). Reduced pup weight and survival were noted at mid- and high-doses, respectively. Relative and absolute liver and adrenal weight were increased in a dose-dependent manner in both sexes and both generations. Liver and adrenal pathology was also reported. The reproductive NOAEL for both parental and pup generations was 0.2 percent EHDPP in the diet: equivalent to 144 mg/kg/day [1].

U.K. assessors judged EHDPP to have a low potential to cause cancer in humans based on negative results in *in vitro* and *in vivo* mutagenicity and genotoxicity assays and an absence of proliferative lesions in repeat-dose studies [1].

Investigators at the National Toxicology Program have used high-throughput assays and rapidly developing whole organisms, such as zebrafish and the nematode *C. elegans*, to screen for potential developmental toxicity and neurotoxicity of a number of organophosphorus flame retardants [3, 4].

[5] LOAEL Low Observed Adverse Effect Level and NOAEL No Observed Adverse Effect Level

Based on results, EHDPP was prioritized for additional neurodevelopmental testing. Briefly, EHDPP reduced firing rate in a neural network assay and inhibited larval development in the nematode *C. elegans* [3, 4]. EHDPP caused significant inhibition of mitochondrial activity which may partly explain the observed developmental arrest in *C. elegans* [4]. In two developmental rat studies, no clearly treatment-related developmental effects were seen at oral doses of up to 3,000 mg/kg- day [1].

Summary of Potential for Exposure

EHDPP is primarily used as a flame retardant and plasticizer in flexible PVC. It is used in food-wrapping films such as those used to wrap meats and skinless sausages [1, 2]. According to a 2009 assessment by the U.K., other current uses are in PVC plastics, rubber, polyurethanes, photofilms, paints, pigment dispersions, adhesives, and PVC coatings on textiles and fabrics [1]. These are materials that could be in children's products. It is also used in inflammable hydraulic fluids like those used in large aircraft [2]. U.S. national volume production was reported to be one million to ten million pounds/year in 2012 [5].

EHDPP has been detected in U.S. house dust with levels ranging from 140 to 3,000 ng/g [6]. EHDPP has been detected in U.S. diet studies, primarily in fats and oily foods [1, 2]. A sample of margarine for example had 20 ppm. Estimates of mean daily dietary intake in the U.S. by Gunderson et al. 1995, were 339 ng/kg bodyweight for infants and 1236 ng/kg body weight for toddlers based on data from 1986-1991 surveys [2].

Biomonitoring studies have measured EHDPP or metabolites in breast milk, urine, and blood. EHDPP was detected in breast milk of Swedish women and women from three Asian countries [7, 8]. It was recently detected in the blood of Chinese adults at a median level three times higher than TPHP [9]. A urinary metabolite of EHDPP called DPHP has also been measured in human urine. It is not specific to EHDPP as it can be generated from at least two other flame retardants, TPHP and RDP[6] [10]. The DPHP metabolite has been detected in urine of California adults, 91% of children in a German day care study, and 93% of the infants in a North Carolina study [11-13]. Urinary levels of DPHP in children were higher than their mothers in two studies [14, 15].

Two studies looked for evidence that household sources of TPHP flame retardant contributed to children's exposure. No correlations with indoor dust or air concentrations of TPHP were detected in the German study [12]. No correlations between DPHP in infant urine and the number of infant products in the home were detected in the North Carolina study [11]. Either another flame retardant is contributing to this metabolite (for example EHDPP) or there are more important sources of exposure.

If EHDPP is released into the environment, biodegradation is expected to occur with conservative estimated half-lives of 50 days in surface water and 300 days in soil and sediment [1]. It has potential to build up in aquatic organisms [2]. A 2009 review for measurements in environmental media located some soil, water, and air studies conducted in the 1980s, but no positive detections, including in samples collected near industrial production sites [1].

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CAS 1330-78-5 - Tricresyl phosphate (TCP)

Summary of Toxicity

Tricresyl phosphate (TCP) is classified by EPA as high hazard for reproductive and repeated dose toxicity, and a moderate hazard for developmental and neurological toxicity [1].
Endocrine organs appear to be sensitive to TCP toxicity. Studies carried out by the National Toxicology Program (NTP) in 1994 showed that long-term oral exposure (13 weeks and 104 weeks) to TCP induced adrenal gland and ovarian lesions in rats and adrenal and liver lesions in mice. The lowest-observed-adverse-effect-level (LOAEL) was 7 mg/kg-day for ovarian lesions in female rats in a 2-year bioassay [2, 3]. TCP was not carcinogenic in NTP oral bioassays in rats and mice [2]. The TCP used in the NTP studies was a mixed isomer preparation of 79% tricresyl phosphate esters consisting of 21% tri-m-cresyl phosphate, 4% tri-p-cresyl phosphate, <1% tri-o-cresyl phosphate, and other unidentified tricresyl phosphate esters [2].

At higher doses, TCP reduced fertility and survival of offspring in rodents [2]. Aside from impacts on female ovaries mentioned above, TCP caused a dose-dependent increase in abnormal sperm morphology, reduced sperm concentration, and caused atrophy of seminiferous tubules in male rodents. TCP reduced the number of litters produced and pups/litter especially when males were treated. It also increased pup mortality postnatally [2, 4, 5]. LOAELs ranged from 63-400 mg/kg-day for these reproductive and developmental effects [1].

NTP studies demonstrated that TCP is neurotoxic to rodents exposed by gavage for 13 weeks to commercial TCP mixtures (with less than 0.1% *ortho* TCP isomer). Briefly, TCP caused neuropathy (axonal degeneration in the spinal cord and sciatic nerve) in rats and mice. The LOAEL was 100 mg/kg-d for neurological lesions in male mice [1, 3]. The *ortho* isomer is reportedly kept to <1% of commercial TCP mixtures [1] because it is a known neurotoxic agent in people. In the early 1930s, an outbreak of delayed neuropathy and paralysis in the United States was traced to tri-o-cresyl phosphate that had been added to Jamaican ginger extract and ingested as an alternative alcoholic drink during the Prohibition era [6].

Summary of Potential for Exposure

Commercial TCP is composed of a mixture of methylated triphenyl phosphate isomers with an unspecified amount of methyl substitution[7] including tri-*meta*-cresylphosphate (CAS 563-04-2), tri-*para*-cresylphosphate (CAS 78–32-0), and tri-*ortho*-cresylphosphate (CAS 78-30-8). TCP is often used as a flame retardant and plasticizer in PVC, cellulosic polymers, thermoplastics, and synthetic rubber. It may be added to polyurethane foam as a flame retardant. It also is a flame retardant additive for industrial lubricants such as hydraulic and brake fluids, and in photographic film [1, 2, 7]. The NTP report indicated it was used in back-coatings for upholstery fabric [3]. U.S. national volume production was reported to be one million to ten million pounds/year in 2012 [8].

TCP has been measured in 100% of dust samples in two North American studies of house dust [9, 10]. The largest study sampled 134 urban Canadian homes and reported mean dust concentrations of 990-2600 ng/g depending on the method. Maximum reported dust concentration was 75,000 ng/g dust [10].

TCP has not been widely measured in biomonitoring studies of the general population or children. All three known isomers of TCP were measured but not detected in urine of German children or indoor dust

[7] Other isomers that might also be present in the TCP mixture include the ortho-ortho-meta (oom), ortho-ortho-para (oop), omm, omp, opp, mmp, and mpp isomers (Van der Veen et al. 2012; reference 14)

in multiple German day care centers [11]. TCP was detected at low levels in breast milk from Swedish women (median was 0.28 ng/g lipid; maximum was 3.7 ng/g lipid) [12]. Median levels in Asian women were similar, but the maximum detected level in breast milk (85 ng/g lipid) was much higher in this population [13].

TCP has a high bioconcentration factor (BCF) of 8.56×10^3 meaning that it is likely to partition to fish and sediments if released into waterways. Potential for TCP bioaccumulation may be low, however. Three fish species cleared this compound after exposure ceased. TCP degraded within five days in river water, and within 7.5 hours in sewage sludge in other studies [5, 14]. Rats also are able to excrete TCP in urine, feces, and expired air.

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CAS 13674-84-5 - Tris (1-chloro-2-propyl) phosphate (TCPP)

Summary of Toxicity

EPA classified Tris (1-chloro-2-propyl) phosphate (TCPP) as high hazard for reproductive and developmental effects based on increased estrus cycle length, decreased uterine weights, and increased number of runts at the 99 mg/kg dose in a 2-generation oral rat study [1, 2]. TCPP has not been tested for cancer, but it is structurally similar to TDCPP and TCEP[8] which are both demonstrated animal carcinogens [2]. The National Toxicology Program has a cancer assay underway to fill this important data gap [3].

Only limited toxicity testing results for TCPP were identified in a review by ATSDR in 2012 [4]. A 1982 study by Kawasaki H. et al. reported that oral dosing in pregnant rats up to 893 mg/kg-day on gestation days 0-20 had no significant effects on the number of implantations or resorptions, fetal weight, external malformations, or pup survival and growth in the first 4 postnatal weeks [4].

Summary of Potential for Exposure

TCPP is an additive flame retardant used in polyurethane furniture foam, textiles, apparel, leather, electronics, and rigid polyurethane foam insulation and roofing laminates used in building construction [3]. Commercial TCPP is a mixture of isomers: primarily CAS 13674-84-5, with lesser amounts of CAS 76025-08-6, and 76649-15-5 [3]. The U.S. national production volume of TCPP was reported to be 54,673,933 pounds in 2012 [3, 5].

TCPP has been detected in U.S. household furniture and in baby products including: polyurethane foam in car seats, changing table pads, sleep positioners, portable mattresses, nursing pillows, and children's furniture [6-8]. Detection rates in foam are reported to be 0.5-2.2% by weight in furniture foam; 1-14% in baby product foam [3, 8].

TCPP has been detected, often with high frequency, in indoor house dust and air by multiple studies in North America [8-12]. Median and mean levels in dust are frequently in the low parts per million (µg/g) with detections up to 140 µg/g dust. Reported air concentration of inhalable TCPP particulate (defined as >4µm) ranged up to 1.36 $\mu g/m^3$ in home indoor air [9]. TCPP has been detected in a variety of foods in the FDA total diet study at low levels (< 7 ppb).

In biomonitoring studies, two metabolites of TCPP have been measured and detected in human urine: bis (1-chloro-2-propyl) phosphate (BCIPP) and 1-hydroxy-2-propyl bis(1-chloro-2-propyl) phosphate (BCIPHIPP). One or both were detected in toddlers and their mothers in New Jersey [13], infants in North Carolina [14], mothers and their children in California [15], and in adults in Northern California [16]. While the frequency of detection and levels detected are generally low for the BCIPP metabolite, a recent study measured the BCIPHIPP metabolite in 100% of mothers and their children. Maximum

[8] TDCPP - Tris(1,3-dichloro-2-propyl)phosphate; TCEP - Tris(2-chloroethyl) phosphate

concentrations in urine for mothers and children were 104 ng/mL and 23.2 ng/mL, respectively [15]. TCPP has also been detected in breast milk in Sweden at concentrations up to 82 ng/g lipid [17]. EPA considers TCPP to have high hazard for persistence and low hazard for bioaccumulation [1]. In rats, TCPP is readily absorbed, is widely distributed to tissues – especially the liver and kidney – and is excreted primarily in urine but also bile and feces. Tissue elimination time was slowest from adipose tissue (adipose $T_{1/2}$ = 103 hours) [4].

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CAS 25154-52-3 - Nonylphenol

Related substance: CAS 84852-15-3 - 4-Nonylphenol (branched)

Summary of Toxicity

Nonylphenol and 4-Nonylphenol have been classified as Category 1 endocrine disruptors by the European Union.[1] The EU developed the priority list in stages (2000, 2002, and 2007), putting chemicals in three categories. The EU **Category 1** endocrine distruptor designation has been used as an authoritative source for CSPA. Category 1 requires evidence of endocrine disrupting activity in at least one species using intact animals. **Category 2,** which requires at least some in vitro evidence, is too preliminary. **Category 3** is no evidence of endocrine disrupting activity or no data available.

Uterotrophic assays indicate that nonylphenol has estrogenic activity, and several other lines of evidence suggest that nonylphenol can adversely affect mammalian reproduction.[2] Uterotrophic assays indicate that 4-nonylphenol has estrogenic activity.[3-5]

Summary of Potential for Exposure

The Danish EPA found nonylphenol in 1 of 3 pencil erasers[6] and 1 of 28 infant sunscreens[7] and 4--nonylphenol in 1 out of 2 nursing pillows.[6] A Dutch study of plastics in children's products found nonylphenol in many samples (mostly polyvinyl chloride).[8]

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CAS 26040-51-7 - Bis (2-ethylhexyl) tetrabromophthalate (TBPH)

Summary of Toxicity

EPA classified bis (2-ethylhexyl) tetrabromophthalate (TBPH) as a moderate hazard for reproductive, developmental, neurological, and repeated dose toxicities based on rodent toxicity of commercial mixtures, structurally similar chemicals, and professional judgement [1]. Significant data gaps were noted. Lowest-observed-adverse-effect-levels (LOAELs) for developmental effects in rats were 100 mg/kg-day in an oral prenatal study of a commercial mixture of TBB and TBPH. A LOAEL of 1 mg/kg-day was reported in a second perinatal oral study with another commercial mixture, Firemaster® 550, which contains TBB[9] and TBPH plus two non-brominated phosphate flame retardants [1]. The latter study, published by Patisaul et al. 2013, found that pregnant rats exposed to the Firemaster® 550 mixture during gestation and lactation had altered thyroid function and produced offspring that were 30–60% heavier by weaning, an effect that persisted into adulthood. Female offspring of treated rats entered puberty sooner and had glucose intolerance and elevated anxiety behaviors in maze testing [2].

TBPH is a brominated analog of phthalate DEHP[12] and may be an endocrine disrupter [3]. A metabolite of TBPH induced proliferative damage in rodent liver and altered serum thyroid hormone (T3) in rats after 2 days exposure to 200 mg/kg per day [3]. A study in Boston, MA, reported house dust concentrations of TBPH were positively associated with higher level of thyroid hormone (T3) in men [4].

Summary of Potential for Exposure

TBPH has been detected in foam baby products [5] and U.S. residential furniture [6]. TBPH is an ingredient in additive flame retardant mixtures used in flexible polyurethane foam. TBPH is also used in construction materials and as a non-flammable plasticizer in PVC electrical equipment, electronics, and appliances. In addition, TBPH is a flame retardant in neoprene and certain rubbers [7].

TBPH has been measured with high frequency in residential indoor dust in the United States [3, 4, 8-10] and Canada [11, 12]. It was found in 100% of indoor dust samples from childcare centers studied in 2010-2011 in Northern California [13]. Across all these studies, mean levels in indoor dust ranged from 144-734 ng/g dust and the maximum level reported was 47,110 ng/g. In a study of pregnant women in

[9] TBB – 2-ethylhexyl-2,3,4,5-tetrabromobenzoate; DEHP – di(2-ethylhexyl) phthalate

North Carolina, levels of TBPH in dust were correlated positively with levels in hand wipes [14]. TBPH was also detected in 100% of office dust and 90% of car dust in Boston study [3].

TBPH was detected in human serum in a 2014 Indiana study of adults aged 19-38 [15] and in maternal serum and breast milk collected in a 2008-2009 study of women living in Québec, Canada [16].

TBPH is classified by EPA as high hazard for persistence and bioaccumulation [1].

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CAS 38051-10-4 - Bis(chloromethyl)propane-1,3-diyl tetrakis-(2-chloroethyl) bis(phosphate) (V6)

Summary of Toxicity

EPA classified V6 a moderate hazard for carcinogenicity based on the toxicity of chemicals with very similar structures [1]. Commercial V6 also contains 4.5-13.5% Tris (2-chloroethyl) phosphate (TCEP) as an impurity [1, 2]. TCEP is classified as a carcinogen by the State of California [3] and a 1b reproductive hazard by the European Union [4].

EPA considered V6 to have high hazard for developmental toxicity and moderate hazard for reproductive toxicity [1]. In a two-generation oral rat study, doses of 86 mg/kg-day caused thyroid effects (follicular hypertrophy and increased organ weight) in the parental generation and caused retarded fetal and pup growth in offspring [5]. The no-observed-adverse-effect-level (NOAEL) was 29 mg/kg-day.

Summary of Potential for Exposure

V6 has been used as an additive flame retardant in polyurethane foam and has been identified in a number of consumer products including foam carpet pads, tent fabric, and baby products [2, 6, 7]. Average concentration in the products that tested positive was 4.6% by weight of the foam [6]. It is reportedly used in interior foam for automotive and furniture foam at typical loadings of ~6% w/w [5]. U.S. national production volume of V6 was between 500,000 and 1 million pounds in 2002, but more current information is withheld as confidential business information [8].

V6 has not been widely studied in house dust or the environment. It was detected in 95% of car dust samples and 75% of house dust samples in a single Boston area study [2]. Concentrations in car dust were significantly higher than the house dust, which is consistent with its reported higher use in automobile foam. Median levels in car dust were 103 ng/g.

We did not identify any biomonitoring studies for V6. The compound is readily absorbed across the gut and less readily across skin. Half-life for elimination from the body was 99-113 hours in orally exposed rats [1].

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CAS 68937-41-7 - Isopropylated triphenyl phosphate (IPTPP)

Summary of Toxicity

Isopropylated triphenyl phosphate (IPTPP) is an isomeric mixture of phosphate esters derived from isopropyl phenols. Commercial mixtures may vary in the number of isopropyl substitutions and may contain some triphenyl phosphate and isopropylated diphenyl phosphates, as well [1, 2]. EPA classified IPTPP a high hazard for reproductive, developmental, and neurological toxicities [1]. Changes in organ weights, reduced fertility, and pup survival were observed in an oral rat study of reproduction and development. The lowest-observed-adverse-effect-level (LOAEL) was 25 mg/kg-d for increased female adrenal weights and relative ovary weights. Relative weights of liver, epididymis, and adrenal glands were also observed in male rats at higher doses. IPTPP caused neurotoxicity (ataxia and degeneration of the spinal cord and peripheral nerves) in hens at and above dose of 90 mg/kg-day in a 91-day test submitted by the industry [3]. Brain cholinesterase inhibition was observed in rodent testing of a commercial mixture which contained 80% IPTPP and 20% TPP [1].

Summary of Potential for Exposure

IPTPP is very likely to be found in children's products. In a European assessment, IPTPP was identified as a flame retardant plasticizer used in a range of PVC products, polyurethanes, textile coatings, adhesives, paints, and pigment dispersions [2]. Uses in the U.S. are largely withheld as confidential business information [4]. However, IPTPP isomers are a listed ingredient of Firemaster®550 which is used as an additive flame retardant in flexible polyurethane foam [5]. U.S. consumer product testing has identified the profile of flame retardants contained in Firemaster®550 in foam baby products and U.S. upholstered furniture [6, 7]. The reported U.S. national production volume of IPTPP was 14,904,236 pounds/year in 2012 [3].

U.S. biomonitoring studies indicate that exposure to adults and children is occurring [8-10]. A urinary metabolite of IPTPP was measured in 100% of 22 mothers and 92% of 26 children in a 2013-14 study of families in Princeton, NJ. Mean and maximum level in the children's urine were 1 ng/mL and 10.1 ng/mL, respectively [9]. This same metabolite was detected at slightly higher mean levels in 100% of mothers and babies in a 2015 California study population [10].

EPA considered IPTPP to have very high aquatic toxicity, moderate persistence in the environment, and high potential for bioaccumulation [1].

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CAS 84852-53-9 - Decabromodiphenyl ethane (DBDPE)

Summary of Toxicity

EPA classified decabromodiphenyl ethane (DBDPE) as a high hazard for developmental toxicity based on its structural similarity to decabromodiphenyl ether (decaBDE) [1]. Available toxicity data has been reviewed by the United Kingdom Environment Agency in 2007; by EPA in 2014; and by Health Canada and Environment Canada in 2016 [1-3]**.** Briefly, DBDPE had low acute toxicity in animals, both orally and dermally, and is predicted to have low acute inhalation toxicity. In a 90-day study in rats, minimal systemic effects were reported at the highest dose tested including increased liver size and hepatic cell hypertrophy at 1,000 mg/kg-day (LOAEL). No effects were reported at 320 mg/kg-day (NOAEL).

These liver changes were reversible after 14 days post-exposure, and the effects were interpreted as an adaptive response to increased demand on the liver to metabolize and excrete DBDPE [1].In another 90-day oral assay in rats, Wang et al. dosed male rats for 90 days with 100 mg/kg-day DBDPE [4]. No alteration in liver, kidney, or body weights was observed indicating no overt toxicity. Authors reported indications of organ impairment in DBDPE-treated rats (decreased serum creatinine, decreased serum liver enzymes alanine transferase and alkaline phosphatase, and increased total bile acids). Liver tissue was not examined for signs of pathology in this study to investigate this observation. DBDPE-treated rats also showed increased serum thyroid hormones T3 and T4 although the difference was not

statistically significant for T4 [4]. Thyroid hormones are central to proper mammalian development, including the brain and reproductive organs, so this observation should be further investigated in assays involving prenatal exposure.

Reproductive toxicity testing has not been conducted. In two developmental toxicity tests in rats and rabbits, neither reported treatment-related malformations at birth or altered pup weight or decrease in survivability. The NOAEL was 1,250 mg/kg-day [1, 2]**.** The developmental tests did not include observations for neurobehavioral effects as the pups matured. DBDPE is structurally similar to decabromodiphenyl ether (decaBDE) and has a similar toxicity profile in acute and short-term toxicity testing [4]. In further investigations of developmental exposures, however, decaBDE has been shown to produce neurodevelopmental toxicity and endocrine disruption in rodents in at much lower doses [5-12]. In fact, EPA used a NOAEL of 2.2 mg/kg-day to establish a reference dose for decaBDE based on neurobehavioral effects of prenatal exposure. Lack of testing for both neurodevelopmental outcomes and endocrine disruption are important data gaps for DBDPE given its very close structural similarity to decaBDE. EPA use of toxicity data from decaBDE to score DBDPE's potential for development toxicity is a reasonable approach to address this important gap in toxicity testing.

No cancer testing was identified. DBDPE was negative in two genotoxicity tests [1].

Summary of Potential for Exposure

DBDPE is a general purpose additive flame retardant for a variety of polymer applications and for textiles. It is a commercially important alternative to decaBDE. It typically comprises 10-15% of the weight of treated plastics (e.g., ABS, HIPS, PVC, polypropylene and polyethylene, etc.). It is used in wire and cable coatings for telecommunications, electrical, and automotive industries. To a lesser extent, it can be used in the latex-based back coating for drapery and upholstery fabrics [2]. DBDPE has been manufactured for more than 20 years and is a High Production Volume (HPV) chemical in the United States today. As of 2012, the National Production volume was 50-100 million pounds per year [13].

DBDPE was detected in one third of baby formula and about one querter of baby cereals collected from the U.S. in 2013 [14]. Median levels of DBDPE detected were 22 and 11 pg/g fresh weight, respectively. The daily median intake for U.S. infants consuming formula and cereal was estimated by authors to be 2.2-3.44 ng DBDPE/day.

DBDPE was detected in a child's tablet and plastics of other consumer products by the Washington Department of Ecology at levels of 1000 ppm or lower [15]. It was also detected at lower levels (<100 ppm) in foam, stuffing, and padding of children's products collected by the Washington Department of Ecology [16]. A study that tested a variety of children's toys for sale in China found DBDPE in 80% of hard plastic toys, 89% of foam toys, 50% of the stuffed toys, and 40% of rubber or soft plastic toys including baby pacifiers. Maximum levels detected was 237 ppm [17]. Potential migration into saliva was tested by volunteers in this study. One out of 5 volunteers had measurable DBDPE in saliva after lightly chewing a segment of a hard plastic toy in the mouth for 15 min [17].

Because DBDPE is not chemically bound to the treated materials, it can escape into the environment. DBDPE has been widely detected in studies of U.S. house dust [18-21]. The dust levels of DPDPE reported ranged <2.6 -11,070 ng/g dust. DBDPE has also been detected in residential indoor air (mean 5 ng/m^3) and at higher levels in a gymnastics facility in Seattle (50 ng/m^3) [22]. In addition to U.S. studies, Harrad et al. 2008, studied DPDPE in dust samples from U.K. homes, offices, and cars. Average (and maximum) concentrations of DPDPE were found to be 270 (3,400), 170 (860) and 900 (2,900) ng/g dust respectively [23].

Only very limited human biomonitoring data are available in the literature for DBDPE. It was measured but not detected in maternal serum in Norway in 2012 [24]. It was detected at low frequency in maternal serum and breast milk collected between 2008 and 2009 in the Sherbrooke region of Canada [25]. Low dermal and oral absorption may explain the low detections in people [1]. DBDPE is listed as a priority for biomonitoring by the California Biomonitoring Program [26].

Two recent government assessments predict that DBDPE has high environmental persistence but came to different conclusions regarding potential for bioaccumulation [2-3]. In a 90-day oral rat study, DBDPE and its metabolic products accumulated in adipose, liver, and kidney tissue [3]. DBDPE has been detected in environmental media from various parts of the world and in wildlife including birds, dolphins, and pandas. There is limited but positive evidence that DPDPE biomagnifies in aquatic food chains [2, 27-28]. More testing is needed to characterize environmental fate, bioavailability, and metabolism of DBDPE in different species. If debromination to nona-, octa-, and hepta-bromodiphenyl ethane occurs following the pathway of debromination established for decaBDE, then degradation products are likely to have high potential for bioaccumulation [3].

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CAS 85535-84-8 - Short-Chain Chlorinated Paraffins (SCCPs)

CAS 108171-26-2 – Chlorinated paraffins

Summary of Toxicity

Short-Chain Chlorinated Paraffins (SCCPs) are classified as carcinogens by authoritative sources [1, 2]. The National Toxicology Program classifies chlorinated paraffins (C12, 60% chlorine) as reasonably anticipated to be human carcinogens based on liver, kidney, and thyroid tumors in rodent testing. California Proposition 65 also lists Chlorinated paraffins (CAS No. 108171-26-2) (average chain length, C_{12}; approximately 60 percent chlorine by weight) as carcinogens.

The European Union lists SCCPs as a substance of very high concern (SVHC), as it meets the criteria for both a persistent bioaccumulative and toxic (PBT) substance and a very persistent, very bioaccumulative substance (vPvB) [3]. This is part of implementing the EU law Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH). Substances that may have serious and often irreversible effects on human health and the environment can be identified as SVHCs. If a substance is identified as an SVHC, it will be added to the Candidate List for eventual inclusion in the Authorisation List.

Summary of Potential for Exposure

SCCPs could be present in children's products as they have been used as plasticizers and a flame retardant in plastics, especially PVC. Other minor domestic SCCP uses are as a plasticizer and a flame-retardant additive to a variety of products including: rubber formulations, paints and other coatings, and adhesives and sealants [8].

SCCPs (CAS No 85535-84-8) are included on Washington State's PBT list (WAC 173-333-320) [4]. SCCPs have been detected in breast milk as well as other human tissues [5,6]. SCCPs are found world-wide in the environment, wildlife, and humans. SCCPs bioaccumulate in wildlife and humans, and are persistent and transported globally in the environment [7].

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CAS 183658-27-7 - 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (TBB)

Summary of Toxicity

EPA classified 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (TBB) as a moderate hazard for reproductive, developmental, neurological, and repeated dose toxicities [1]. This was based on the observed toxicity of a closely related confidential analog, and studies of commercial mixtures which contain TBB as a major component. EPA did not release the name or chemical structure of the confidential analog, but reported that the lowest-observed-adverse-effect-level (LOAEL) for a rodent study of this compound was 25 mg/kg-d for reproductive toxicity. LOAELs for developmental effects of two commercial mixtures were reported at 100 mg/kg-d for Firemaster® BZ-54 and 1 mg/kg-d for Firemaster® 550 [1].

The latter study involved prenatal exposure in rats and was published by Patisaul et al. 2013 [2]. Pregnant rats exposed to the Firemaster® 550 mixture during gestation and lactation had altered thyroid function and produced offspring were 30-60% heavier by weaning, an effect that persisted into adulthood. Female offspring of treated rats entered puberty sooner and had glucose intolerance and elevated anxiety behaviors in maze testing [2].

Summary of Potential for Exposure

TBB is an ingredient in common market replacements for PBDEs[10] in flexible polyurethane foam [3]. Approximately 50% of the Firemaster® 550 mixture is TBB and TBPH[1] at a ratio of 4:1 by mass [1, 4]. Past and current national production volume of TBB is withheld as confidential business information [4, 5]. TBB treated foams may be used in many everyday products such as couches, chairs, other upholstered furniture, children's furniture, baby products, office furniture, foam in gymnastic facilities, and auto cushions. TBB may also be present in products made from recycled foam such as carpet backings and pads [4, 6, 7].

TBB has been measured with high frequency in residential indoor dust in studies in the U.S. [8-11] and Canada [12]. It was found in 100% of indoor dust samples from 39 childcare centers in Northern California [13]. Mean levels from these studies ranged from 310-1,062 ng/g in indoor dust. Maximum level reported was 75,000 ng/g dust. In a study of North Carolina adults, levels of TBB in hand wipes correlated positively with a metabolite of TBB in urine suggesting that dermal contact with dust or treated surfaces contributed to overall exposure [14]. In another investigation, median concentrations of TBB and TBPH in paired hand wipe samples were 2-3 times higher after gymnastics practice compared to before indicating skin exposure was occurring during collegiate gymnast practice [15].

[10] PBDE - polybrominated diphenyl ethers; TBPH – bis (2-ethylhexyl) 2,3,4,5-tetra bromophthalate

Metabolites of TBB were detected in urine of toddlers and their mothers in New Jersey and California studies [16, 17]. Levels measured in children tended to be higher than their mothers in both studies. The maximum concentration reported in children's urine reported across both studies was 225 ng/mL. TBB metabolites were also commonly detected in maternal serum (n=102) and breast milk (n=105) collected in a 2008-2009 study in women living in Québec, Canada [18].

TBB is classified by EPA as high hazard for persistence and bioaccumulation [1].

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CHCC Delistings

Three chemicals are delisted from the CHCC list. An evaluation of each chemical is provided in this document summarizing the reason for the delisting. The evaluations identify the CAS number and chemical name and summarizes the current information about toxicity, potential for exposure, reason for delisting, and provides a list of references.

CAS	Name	Acronym
85-44-9	Phthalic anhydride	None
556-67-2	Octamethylcyclotetrasiloxane	D4
7439-98-7	Molybdenum & molybdenum compounds	Mo

CAS 85-44-9 - Phthalic Anhydride

Summary of Toxicity

In 2011, Ecology based the listing of phthalic anhydride on the Globally Harmonized System (GHS) of Classification and Labelling of Chemicals Cat 2 for reproductive toxicity or germ cell mutagenicity [1]. GHS is a worldwide initiative to promote standard criteria for classifying chemicals according to their health, physical and environmental hazards.

Since 2011, the European Chemicals Agency (ECHA) reviewed phthalic anhydride and did not classify it for either reproductive toxicity or germ cell mutagenicity under the GHS criteria [2].

Summary of Potential for Exposure

Phthalic anhydride is primarily used in the manufacture of phthalate plasticizers and polyester resins. It is also used in small volume in the production of alkyl resins used in dyes, paints, and lacquers [3,4]. It was detected by the Danish EPA in coatings on 4 out of 15 wooden toys tested [5].

Reason for Delisting

The authoritative source used in 2011 to identify phthalic anhydride as toxic was updated. The updated evaluation no longer classifies phthalic anhydride as reproductively toxic. Phthalic anhydride is delisted from the CHCC list.

List of References

1. Ecology, 2011, Children's Safe Products Reporting Rule – Supporting Documents for the 2011 Rulemaking. Rule documents from the 2011 rulemaking were merged into a publication in April of 2017. Publication 17-04-022 https://fortress.wa.gov/ecy/publications/SummaryPages/1704022.html
2. ECHA. REACH Registration Dossier – Phthalic Anhydride. Classification & Labelling & PBT Assessment. https://echa.europa.eu/registration-dossier/-/registered-dossier/15845/1.
3. California EPA, Office of Environmental Health Hazard Assessment (OEHHA). Determination of non-cancer REL for Phthalic anhydride, December 2000. www.oehha.ca.gov/air/chronic_rels/pdf/85449.pdf.
4. ICIS, Chemical Intelligence, Phthalic Anhydride Uses and Market Data, Updated February 2010. www.icis.com/v2/chemicals/9076142/phthalic-anhydride/uses.html.
5. Danish Ministry of the Environment, Environmental Protection Agency. Surveys on Chemicals in Consumer Products. Report 60, 2005 www.mst.dk/English/Chemicals/Consumer_Products/Surveys-on-chemicals-in-consumer-products.htm.

CAS 556-67-2 - Octamethylcyclotetrasiloxane (D4)

Summary of Toxicity

Octamethylcyclotetrasiloxane (D4) is included in a European Commission priority list of chemicals identified for further in depth evaluation of their role in endocrine disruption [1,2]. Although, this 2007 publication focused on low production volume chemicals, D4 was one of the high production volume chemicals included. This European Commission listing was based on effects in a uterotrophic assay [3]. There is more recent evidence for the lack of effect from D4 in a uterotrophic assay [4].

Summary of Potential for Exposure

In 2003, the Danish EPA identified D4 as a listed ingredient in 1 out of 28 sunscreens, 1 of 32 lotions, and 1 out of 208 cosmetics marketed to children [5]. Recently, Ecology has found children's cosmetics that include D4 on the ingredient list [6].

Reason for Delisting

Under our current process for designating chemicals to be reported under CSPA, determination of toxicity is based on listings by selected authoritative sources that provide a robust evaluation of available data in a public process. In 2011, D4 was identified as toxic (for the purposes of CSPA reporting) based only on the European Commission list of potential endocrine disruptors. The EU does not intend to update this list and a 2015 study found no effect in a similar assay. No other CSPA authoritative source classifies D4 as toxic. D4 is delisted from the CHCC list.

List of References

1. DHI Water and Environment for DG Environment (2007). Study on enhancing the Endocrine Disruptor priority list with a focus on low production volume chemicals. http://ec.europa.eu/environment/chemicals/endocrine/pdf/final_report_2007.pdf.
2. European Commission. 2007. Priority List of suspected endocrine disruptors. Available at http://ec.europa.eu/environment/chemicals/endocrine/strategy/substances_en.htm#priority_list.

3. McKim, JM, Wilga, PC, Breslin, WJ, Plotzke, KP, Gallavan, RH, and Meeks, RG. (2001) Potential estrogenic and androgenic activity of the cyclic siloxane octamethylcyclotetrasiloxane (D4) and the linear siloxane hexamethyldisiloxane (HDMS) in immature rats using the uterotrophic assay. *Toxicological Sciences 63*: 37-46.
4. Lee D, Ahn C, An BS, Jeung EB. (2015) Induction of the estrogenic marker calbindn-d_9k by octamethylcyclotetrasiloxane. *Int J Environ Res Public Health* 12:14610-25
5. Danish Ministry of the Environment, Environmental Protection Agency. Survey of Chemical Substances in Consumer Products, Report 88, 2007. www.mst.dk/English/Chemicals/Consumer_Products/Surveys-on-chemicals-in-consumer-products.htm.
6. Ecology, October 2016. An Assessment of Children's Safe Product Act Data. Addendum to Quality Assurance Project Plan: Product Testing Program Version 1. Publication 16-03-021

CAS 7439-98-7 - Molybdenum and molybdenum compounds (Mo)

Summary of Toxicity

In 2011, Ecology identified toxicity for Molybdenum (Mo) from a REPROTEXT grade of B for reproductive toxicity [1,2]. Since 2011 we have reconsidered REPROTEXT and determined it is no longer identified as an authoritative source for CSPA. During this review, we also found the REPROTEXT database and scores have not been updated. The information in REPROTEXT is informative, but not sufficient by itself for the purposes of CSPA CHCC listing. We were not able to identify Mo toxicity from another authoritative source.

REPROTEXT a subscription-based database and the University of Washington no longer subscribes to it, which further limits access for residents of Washington.

Summary of Potential for Exposure

Mo is an essential trace nutrient in humans. Biomonitoring in the general U.S. population by the Centers for Disease Control and Prevention (CDC) show that levels in the general population dropped slightly from 1999 to 2004 [3]. Molybdenum was found in testing of children's school supplies by the Danish EPA [4].

Reason for Delisting

The authoritative source used in 2011 to identify Mo as toxic has been determined to be insufficient for CHCC listing. No other authoritative source classifies Mo as toxic. Mo is delisted from the CHCC list.

List of References

1. Ecology, 2011, Children's Safe Products Reporting Rule – Supporting Documents for the 2011 Rulemaking. Rule documents from the 2011 rulemaking were merged into a publication in April of 2017. Publication 17-04-022 https://fortress.wa.gov/ecy/publications/SummaryPages/1704022.html
2. REPROTEXT, "Molybdenum" in REPROTEXT Database Version 5.1 Greenwood Village, CO: Thomson Reuters (Healthcare) Inc. (accessed 2009).
3. Centers for Disease Control and Prevention (CDC), Fourth National Report on Human Exposure to Environmental Chemicals, December 2009. www.cdc.gov/exposurereport/data_tables/.

4. Danish Ministry of the Environment, Environmental Protection Agency. Surveys on Chemicals in consumer products. Report 84. www.mst.dk/English/Chemicals/Consumer_Products/Surveys-on-chemicals-in-consumer-products.htm.