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ECOLOGY
State of Washington

Priority Consumer Products Report to the Legislature

Safer Products for Washington Implementation Phase 2

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Priority Consumer Products
Report to the Legislature

Safer Products for Washington
Implementation Phase 2

Hazardous Waste and Toxics Reduction Program

Washington State Department of Ecology

Olympia, Washington

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Legislative Report

The Washington Department of Ecology, in consultation with the Washington Department of Health, developed this report during Phase 2 in the implementation process for Chapter 70.365 RCW.[1] This report is required by RCW 70.365.030(1)[2]:

"(1) Every five years, and consistent with the timeline established in RCW 70.365.050, the department, in consultation with the department of health, shall identify priority consumer products that are a significant source of or use of priority chemicals. The department must submit a report to the appropriate committees of the legislature at the time that it identifies a priority consumer product."

What this report includes

This report identifies the eleven products listed in Table 1 as priority consumer products under the Safer Products for Washington program.

Table 1. Priority consumer products.

Priority chemical class	Priority product
Flame retardants	Electric and electronic equipment (plastic device casings)
Flame retardants	Recreational polyurethane foam products
PCBs	Paints and printing inks
PFAS	Carpet and rugs
PFAS	Leather and textile furnishings
PFAS	Aftermarket stain and water resistance treatments
Phenolic compounds (bisphenols)	Food and drink cans (can linings)
Phenolic compounds (bisphenols)	Thermal paper
Phenolic compounds (alkylphenol ethoxylates)	Laundry detergent
Phthalates	Vinyl flooring
Phthalates	Personal care and hygiene products (fragrance)

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[1] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365

[2] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Executive Summary

Legislative requirement

In 2019, the Washington State Legislature directed the Washington Department of Ecology (Ecology), in consultation with Washington State Department of Health (Health), to implement a regulatory program to reduce toxic chemicals in consumer products (Chapter 70.365 RCW).[3]

The law requires Ecology to identify priority consumer products that are significant sources or uses of the five priority chemical classes. The implementation program is called **Safer Products for Washington.**

This report, which identifies our priority products and details how they meet the criteria in the law, does not establish regulations or restrictions on these chemical-product combinations.

Background

Small, steady releases of chemicals coming from millions of consumer products are the largest source of toxics entering Washington's environment. The Safer Products for Washington program includes a regulatory process designed to keep harmful chemicals out of homes, workplaces, schools, and the environment. The law identified the first set of priority toxic chemical classes. In this report, we identify consumer products that are significant sources and uses of priority chemicals. In our identification process, we considered the volume of products and chemicals in Washington, human and ecological exposure, existing regulations, and potential safer alternatives. We found some promising options and will determine whether safer alternatives are feasible and available in Phase 3.

Safer Products for Washington creates a systematic approach to reduce exposure to toxic chemicals found in consumer products. The law directs us to take the following actions:

1. Identify priority chemical classes.
2. Identify priority products that are significant sources or uses of those chemicals.
3. Determine if safer alternatives are available and feasible and decide whether to restrict, require reporting, or take no action on priority chemical-product combinations.
4. Implement restrictions or reporting requirements, if any, through a rulemaking process.

The first set of priority chemical classes identified by the law are:

- Organohalogen flame retardants and flame retardants identified in RCW 70.240.010.[4]

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[3] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365
[4] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240.010

- Per- and polyfluoroalkyl substances (PFAS).
- Polychlorinated biphenyls (PCBs).
- Phenolic compounds.
- Phthalates.

We will not identify any of the exempted products outlined in RCW 70.365.030(5) as priority products under the Safer Products for Washington program. See Appendix 3 for a list of exemptions.

Stakeholder advisory process

RCW 70.365.050(4)[5] requires Ecology to create a stakeholder advisory process. To establish this process, and while this work is ongoing, we developed outreach materials[6] and held a public webinar[7] to explain the law and its requirements.

After the webinar, we shared which product categories we were researching. We also invited interested stakeholders to submit information on chemicals and their concentrations in products, the potential for human or environmental exposure, and possible safer alternatives.

The release of the draft version of this report in January 2020 coincided with a robust outreach effort. We sent more than 200 individual emails sharing the draft report with retailers, manufacturers, environmental advocacy organizations, public health agencies, scientific experts, and more.

In February 2020, we hosted a second webinar[8] to discuss the implementation process and explain our rationale for choosing products for further evaluation.

Through these efforts, our email list grew to more than 230 subscribers; subscribers included representation from every stakeholder group outlined in the law, as well as many members of the public. During the public comment period for our draft report, we received 1,350 comments, most of which came from the public.

Community members and environmental organizations submitted approximately 1,250 requests for Ecology to investigate more products and broader product categories. We addressed these concerns by expanding the scope of three product categories and adding two new products in the current phase.

Manufacturers asked for clearer definitions of product categories. Through our ongoing research and discussions with manufacturers, we expect to refine the scope of the product

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[5] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.050

[6] https://fortress.wa.gov/ecy/publications/documents/2004007.pdf

[7] https://www.ezview.wa.gov/Portals/_1962/Documents/saferproducts/Webinar082919slides.pdf

[8] https://www.ezview.wa.gov/Portals/_1962/Documents/saferproducts/February_2020_Webinar_Presentation.pdf

categories in Phase 3. Any potential regulatory actions will have clear product scopes that we will develop with stakeholder input.

Some manufacturers shared that they have already phased the priority chemical out of a product we are considering. Our regulatory process for some products may require certain actions by manufacturers that have not switched to safer alternatives. Learn more in our frequently asked questions[9] and stakeholder engagement process.[10]

Priority products

Ecology, in partnership with Health, consulted peer-reviewed scientific data, government reports, and publicly-available economic and market information to determine that the products shown below are significant sources or uses of the priority chemicals identified in Chapter 70.365.010(12) RCW.[11]

The sources used to support this report are primarily peer-reviewed studies. Because these studies frequently assess products already in homes or workplaces, they don't necessarily reflect the variety of products currently on the market. Therefore, in some cases, to provide insight into current levels, we included additional non-peer reviewed studies on the concentration or frequency of priority chemicals in products. Our calculations, however, are based only on government and peer-reviewed data.

Our work in Phase 3 will focus on reviewing new information and data we expect to receive from manufacturers. This will help us determine whether priority chemicals are still present in priority consumer products and, if so, whether regulatory actions are needed.

Priority products are significant sources or uses of priority chemicals because they contribute to human exposures or environmental releases. As outlined in the law, at this stage (Phase 2), we included all the chemicals in the class that are reported in a particular priority product. We recognize that not all chemicals in a class serve the same function in products.

We are concerned about the threat to human and environmental health that these chemicals pose when they are used, disposed, and released. Governor Inslee's Southern Resident Orca Task Force[12] listed chemicals within each of these priority chemical classes as chemicals of emerging concern.

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[9] https://ecology.wa.gov/ToxicsInProducts

[10] https://www.ezview.wa.gov/Portals/_1962/Documents/saferproducts/Stakeholder_Engagement%20_Process.pdf

[11] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.010

[12] https://ecology.wa.gov/Water-Shorelines/Puget-Sound/Orca-task-force

Table 2. Priority products and major concerns related to the criteria defined in 70.365.

Priority chemical or chemical class	Priority product	Major concerns
Flame retardants	Electric and electronic equipment (plastic device casings)	Human exposure, volume used, contamination from recycling
Flame retardants	Recreational polyurethane foam products	Human exposure
PCBs	Paints and printing inks	Environmental release
PFAS	Carpet and rugs	Human exposure, volume used, environmental release
PFAS	Leather and textile furnishings	Human exposure, environmental release
PFAS	Aftermarket stain and water resistance treatments for textile and leather products	Human exposure, volume used, environmental release
Phenolic compounds (bisphenols)	Food and drink cans (can linings)	Human exposure, volume used
Phenolic compounds (bisphenols)	Thermal paper	Human exposure, volume used
Phenolic compounds (alkylphenol ethoxylates)	Laundry detergent	Environmental release
Phthalates	Vinyl flooring	Human exposure
Phthalates	Personal care and hygiene products (fragrance)	Volume used

Why we chose these priority products

The following sections summarize the five priority chemical classes and the priority products that are major sources of exposure to each chemical class. Scientific evidence shows that these products contribute significantly to the concentrations of priority chemicals found in homes, workplaces, and the environment.

Flame retardants

Flame retardants from consumer products can accumulate in our homes, schools, and workplaces. Exposure to flame retardants is associated with a number of health concerns, including endocrine and immune system disruption, reproductive toxicity, cancer, neurotoxicity, and adverse impacts on child development.

Flame retardants from consumer products can also be released into the environment, where some accumulate in wildlife and people. Because of these concerns, some flame retardants are considered persistent, bioaccumulative, toxic (Chapter 173-333 WAC[13]), and chemicals of high concern to children (Chapter 70.240 RCW[14]).

Electric and electronic equipment (plastic device casings)

Priority chemical: Organohalogen flame retardants and flame retardants identified under Chapter 70.240 RCW.

Priority product summary: Electric and electronic equipment with plastic device casings are a significant use of flame retardants. This product category is broad in scope because the component we're focusing on, the plastic enclosure, serves a similar function across a wide range of electric and electronic products. Flame retardants are detected in the enclosures of many of these products.

Examples of these types of products include TVs, printers, and game consoles, as well as other household electronics like coffee makers, vacuum cleaners, and adaptors. An average home has 30 items with enclosed electronic circuitry. We are concerned about the potential for exposure to children, workers in occupations involving electronics, and sensitive species.

Recreational polyurethane foam products

Priority chemical: Organohalogen flame retardants and flame retardants identified under Chapter 70.240 RCW.

Priority product summary: Recreational foam products are a significant source of flame retardants. Recreational foam products are things like the mats, pads and foam pit cubes found in recreational gym facilities. We are concerned about exposures to flame retardants from using these products. Children, workers, and people of childbearing age are exposed to flame retardants from recreational foam products.

Gymnasts frequently rely on foam products to cushion falls, and several studies have shown that they have higher exposures to flame retardants than the general population. Replacing the foam in foam pits at gymnastic facilities has successfully reduced gymnasts' and coaches' exposure to flame retardants. While many studies have looked at gymnasts' exposures,

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[13] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-333

[14] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240

recreational foam products with flame retardants can also be found in other gym facilities where they are used by a broader population of Washingtonians, many of whom are children.

Polychlorinated biphenyls (PCBs)

PCBs in consumer products contribute to human and environmental exposure. Exposure is associated with many health problems, including liver toxicity, endocrine disruption, immune toxicity, and impaired reproduction. Based on their toxicity, persistence in the environment, and ability to accumulate in wildlife and people, PCBs are considered persistent, bioaccumulative, and toxic (Chapter 173-333 WAC[15]).

Intentional uses of PCBs were phased out of most consumer products in the late 1970s. However, PCBs can still be inadvertently generated through the pigment manufacturing process. The persistence of PCBs in the environment means reducing current PCB releases from consumer products will help prevent additional long-term environmental consequences.

Paints and printing inks

Priority chemical: Polychlorinated biphenyls (PCBs).

Priority product summary: Paints and printing inks are a significant source of unintentionally generated PCBs. Most bodies of water in Washington are contaminated with PCBs, which have many toxic effects, accumulate in people and animals, and contaminate the food supply. Although banned "legacy" sources of PCBs are still dominant in testing, studies suggest that paints and printing inks are the product group contributing the most new PCBs to the environment.

Per- and polyfluoroalkyl substances (PFAS)

PFAS represent a large class of chemicals that are highly persistent and can be bioaccumulative and toxic. Many PFAS have been associated with health problems, including immune system toxicity, reproductive impairment, developmental toxicity, and liver and kidney toxicity (Ecology, 2019a). As more PFAS studies are conducted, we learn that more are toxic.

PFAS are problematic when released due to their persistence and ability to move through the environment. They can be found in most environmental samples. This is especially concerning when they enter drinking water—removing them from drinking water is extremely challenging. Some chemicals within this class are considered persistent, bioaccumulative, toxic (Chapter 173-333 WAC), and chemicals of high concern to children (Chapter 70.240 RCW[16]).

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[15] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-333

[16] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240

Carpets and rugs

Priority chemical: Per- and polyfluoroalkyl substances (PFAS).

Priority product summary: Pretreated carpets and rugs are significant sources and uses of PFAS. Carpets contribute to human PFAS exposure, particularly for children who spend more time on or near the floor. PFAS are environmentally persistent and decompose very slowly, making cleanup difficult.

Aftermarket stain and water resistance treatments

Priority chemical: Per- and polyfluoroalkyl substances (PFAS).

Priority product summary: Treatments for stain and water resistance are applied to a wide variety of consumer products and are significant sources and uses of PFAS. We estimate that people in Washington use 2,300 metric tons (approximately 5 million pounds) of treatments on carpet alone each year. PFAS in treatments are usually applied as liquids, foams, or aerosols. This makes them more likely to migrate from the products to indoor air and dust versus PFAS in pretreated products, like furniture and carpet.

Since children spend more time on the floor, they are disproportionately exposed to dust that is contaminated with PFAS from treatment products. Further, contact with treated textiles, like sitting on a couch that was treated for stain resistance, for example, could also potentially expose people to PFAS. Spills and improper disposal of these products can also release PFAS directly into the environment.

Phenolic compounds

Phenolic compounds, such as bisphenols and alkylphenol ethoxylates, are found in consumer products. Exposure to both bisphenols and alkylphenol ethoxylates is associated with endocrine disruption, which can lead to obesity. Two bisphenols within this class, bisphenol A and bisphenol S, are listed as chemicals of high concern to children (Chapter 70.240 RCW[17]). Some alkylphenol ethoxylates are persistent in the environment and can accumulate in wildlife and people. These factors, combined with their aquatic toxicity, make environmental releases of alkylphenol ethoxylates particularly concerning.

Food and drink cans with linings

Priority chemical: Phenolic compounds—Bisphenols.

Priority product summary: Can linings are a significant use of phenolic compounds. About 2.5 billion cans are sold each year in Washington, and testing as late as 2017 shows that a large proportion of those may contain bisphenol-based can liners.

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[17] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240

The largest source of exposure to bisphenol A (BPA) is from foods and drinks. Consuming canned food leads to higher concentrations of bisphenols in our bodies. BPA does not persist very long in the body, so its widespread detection in people shows that exposures happen often. This is especially concerning for children, who may be more affected by these chemicals during sensitive stages of physical and cognitive development.

Thermal paper

Priority Chemical: Phenolic compounds—Bisphenols.

Priority product summary: Thermal paper, which is paper coated with a material that changes color when exposed to heat, is a significant source and use of phenolic compounds. About 3,300 tons of thermal paper are used every year in Washington, and testing shows that much of it contains bisphenols.

People absorb the bisphenols from thermal paper through their skin, which raises its concentration in their bodies. Retail workers have higher exposure than the general population due to their frequent contact with thermal paper like receipt tape. Another concern is environmental exposure, especially from recycling of thermal paper—the largest source of BPA to the environment.

Laundry detergent

Priority chemical: Phenolic compounds—Alkylphenol ethoxylates.

Priority product summary: Laundry detergents are a significant source of phenolic compounds. Studies show that institutional cleaners—including laundry detergent—are the largest use of alkylphenol ethoxylates (APEs). Data suggests that Washingtonians use up to two million pounds of laundry detergent per year, which could contain APEs.

Laundry detergents with APEs are washed down the drain and make their way through wastewater treatment plants or domestic septic systems into bodies of surface and ground water. APEs and their alkylphenol components are already restricted in many other countries, which reduced their presence in the environment in those areas.

Phthalates

Consumer products release phthalates into our homes, schools, and workplaces as they degrade. Exposure to phthalates is associated with endocrine disruption, impaired reproduction and prenatal development, neurodevelopmental impacts, and potentially asthma. Some phthalates are listed as chemicals of high concern to children (Chapter 70.240 RCW[18]). While phthalates are not environmentally persistent, their constant release into the environment poses health concerns and has led to recontamination of clean-up sites.

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[18] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240

Vinyl flooring

Priority chemical: Phthalates.

Priority product summary: Vinyl flooring is a significant source of phthalates. As recently as 2016, vinyl flooring was estimated to contain phthalates at concentrations between 9% and 32% by weight. This means flooring can contribute significant amounts of phthalates to homes and the environment.

Multiple studies show the connection between vinyl flooring in the home and higher concentrations of phthalates in both the human body and indoor air and dust. Because phthalates are not chemically bound to the vinyl, they are released into the environment over time. We estimate that vinyl flooring may contribute 0.17 metric tons (374 pounds) of phthalates to Washington's environment each year.

Personal care and beauty products

Priority chemical: Phthalates.

Priority product summary: Fragrances used in personal care and beauty products are a significant source and use of phthalates. Women of childbearing age and pregnant women have higher exposure to the phthalates used in fragrances than men, and this is largely attributed to their use of personal care products. Among women, women of color have higher phthalate levels from exposure attributable to fragrances in beauty products. These findings have important health equity and environmental justice implications.

Further, phthalates are frequently detected in the environment. Studies suggest that fragrances account for one-third of the 34 tons of phthalates released into Puget Sound annually.

In this report: Priority product chapters

The following chapters of this report provide evidence supporting the inclusion of each chemical-product combination listed above based on the criteria in Chapter 70.365 RCW. Each chapter includes a product definition, estimations of the volume used in Washington, potential for human and environmental exposure, and existing regulations. Our next steps include assessing whether safer alternatives are feasible and available, then determining whether a restriction or reporting requirement is appropriate. We will conduct these steps between July 2020 and July 2022.

Chapter 1: Electric and Electronic Equipment (Device Casings)

Overview

Priority product

Residential and office electric and electronic equipment with plastics that encase electric or electronic components. Examples include:

- TVs, computers, and monitors.
- Fax machines, printers, shredders, and copiers.
- Game consoles, audio and video equipment.
- Other household electronics such as vacuum cleaners, corded phones, coffee makers.
- Remote controls, temperature dials, and adaptors.

This priority chemical-product combination only focuses on the device casing. It does not include inaccessible electronic components, which are the parts of an electronic product that are entirely enclosed within the casing and are not capable of coming out of the product or being accessed during any reasonably foreseeable use or abuse of the product (RCW 70.365.010(5)).[19]

Priority chemical

Organohalogen flame retardants and flame retardants identified under Chapter 70.240 RCW[20] were listed as priority chemicals in RCW 70.365.[21]

Organohalogen flame retardants are chemicals intended to slow ignition and progression of fires. They contain one or more halogen elements bonded to carbon.

The non-halogenated flame retardants identified under Chapter 70.240 RCW are triphenyl phosphate (TPP), tri-n-butyl phosphate (TNBP), ethylhexyl diphenyl phosphate (EHDPP), tricresyl phosphate (TCP), and isopropylated triphenyl phosphate (IPTPP).

Exposure to flame retardants is associated with a number of health concerns, including endocrine and immune system disruption, reproductive toxicity, cancer, neurotoxicity, and adverse impacts on child development (NIEHS, 2020).

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[19] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.010
[20] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240
[21] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365

Flame retardants from consumer products can also be released into the environment, where some accumulate in wildlife and people. Because of these concerns, some flame retardants are considered persistent, bioaccumulative, toxic (Chapter 173-333 WAC[22]), and chemicals of high concern to children (Chapter 70.240 RCW[23]).

Priority product summary

Residential and office electric and electronic equipment that contain plastic device casings are a significant use of flame retardants. We considered the criteria in RCW 70.365.030[24] and this product category contributes to the amounts of flame retardants in our homes, workplaces, and the environment. Electric and electronic products have the potential to expose infants, young children, workers in occupations that use or dismantle electronics, and sensitive species, such as orcas. This product category is broad in scope because the plastic enclosure serves a similar function across a wide range of electric and electronic products. Flame retardants are detected in the enclosures of many of these products. Detailed support for our listing of flame retardants in electric and electronic equipment as a priority product is shown below.

Background

Flame retardants are frequently added to plastic components in consumer goods to meet flammability standards. Although these standards don't mandate flame retardants, they do test materials used in products with electrical components to prevent the likelihood of fires starting from electrical failures—such as if a battery fails or wiring overheats. For electric and electronic enclosures, the main flammability standard is UL 94, a standard that provides a method for classifying ignition characteristics of plastic materials (Underwriter's Laboratories Inc., 2007). These enclosures are made from several types of polymer resins such as high impact polystyrene (HIPS), acrylonitrile butadiene styrene polymers (ABS), polycarbonate/ABS blends (PC-ABS) and polyphenylene ether/HIPS blends (Pure Strategies, 2005). Because most plastic resins are highly flammable, many require flame retardants to meet fire safety standards.

Flame retardants can be broadly categorized based on how they are incorporated into the material: additive and reactive. Additive flame retardants are not chemically bound to the polymers or chemical materials used in the product but are mixed in the product materials. The rate that additive flame retardants escape from plastics depends on their size, chemical properties, and their compatibility with the plastic resin. Reactive flame retardants chemically bond to the polymers or chemical materials used in the product and become an integral part of the product structure.

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[22] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-333
[23] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240
[24] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

In addition to these two categories, there are three main classes of flame retardants: halogenated (also known as organohalogens), non-halogenated (such as organophosphates), and inorganic (such as aluminum trihydrate). Organohalogens are the most common class of flame retardants used in electronic enclosures.

Estimated volume of flame retardants in device casings for electric and electronic equipment

Historically, octabromodiphenyl ether (octaBDE) and decabromodiphenyl ether (decaBDE) were flame retardants commonly used in resins for electronic enclosures. But with their phase out, a number of alternatives have replaced their use. Several studies have screened or tested electric and electronic enclosures. Common analytes are decaBDE, DBDPE, TTBP-TAZ, TBBPA and RDP. Table 3 summarizes several flame retardants that have been found in products with electronic enclosures above 0.1% of total mass. The products noted in bold are items in which an individual analyte was found in concentrations above 1% of the total mass.

Table 3. Product testing results of flame retardants in electric and electronic enclosures.

Flame retardants	Concentration (%)	Product types detected above 0.1% (bold above 1%)	References
Decabrominated Diphenyl Ether (decaBDE)	N.D. – 8.8	**Computer monitor, TV, power adaptor,** laminator	Ballesteros-Gómez, de Boer, & Leonards, 2014; Gallen, 2014; Latimer, 2013
Decabromodiphenylethane (DBDPE)	N.D. – 16	**TV, power strip**, power adaptor	Ballesteros-Gómez et al., 2014; Jonker, 2016
2,4,6-tris(2,4,6-tribromophenoxy)-1,3,5-triazine (TTBP-TAZ)	N.D. – 1.9	**TV, power strip,** electrical adaptors, heat sealer	Ballesteros-Gómez et al., 2014; Jonker, 2016
Tetrabromobisphenol A (TBBPA)	N.D. – 16	**TV, power adaptor, CD player, laminator, heated mattress, foot warming pad controller, shredder, battery charger and car charger, power strip, router, heat sealer,** tower fan	Ballesteros-Gómez et al., 2014; Ecology, 2014a; Gallen, 2014; Jonker, 2016; Latimer, 2013

Flame retardants	Concentration (%)	Product types detected above 0.1% (bold above 1%)	References
1,2-Bis(2,4,6-tribromophenoxy)ethane (BTBPE)	Unknown	Office copier, power strip, TV, heat sealer	Ballesteros-Gómez et al., 2014; Latimer, 2013

Other studies found total bromine in casings at concentrations greater than 1% using XRF, but individual analytes were not identified. The products that were not mentioned in Table 3 but that likely contained a brominated flame retardant were projectors, deep fryers, a clothing iron, a sandwich press, a microwave oven, a coffee maker, and a space heater (Ecology, 2014a; Gallen et al., 2014; Latimer, 2013).

In the Ecology (2014) study, two of the samples, a flat iron and a clothing iron, had concentrations of total bromine greater that 1% and low levels of TBBPA. This could indicate that the TBBPA was used as a reactive flame retardant. An additional study suggested that halogenated flame retardant TTBP-TAZ is in TV casings at concentrations ranging from less than the limit of detection to 30.7% (Schreder, Peele, & Uding, 2017, not peer reviewed).

Plastic enclosures serve a similar function across a wide range of electric and electronic products. As shown in Table 3, flame retardants are detected in the enclosures of many of these products.

The amount of flame retardant needed to meet the flame retardancy function depends on multiple factors including the chemical structure of the polymer, its properties when heated and when exposed to flames, as well as the type of flame retardants added. Common concentration ranges for halogenated flame retardants are between 2% and 25% (EPA, 2014c; Ministry of Environment and Food of Denmark, 2016).

In addition to chemicals with the function of flame retardants, impurities from these chemicals can also exist. The impurities typically share functional groups and basic structures of the parent compounds. Several such impurities have been identified in electronic enclosures (Ballesteros-Gómez et al., 2014; Jonker et al., 2016).

Estimated volume used in Washington

Most households contain items with enclosed electronic circuitry, such as computers (which, along with laptops, are in 77% of American homes), televisions (96% of homes), monitors, audio or video equipment, gaming consoles, cameras, clothing irons, vacuum cleaners, etc. (Nielson, 2019; U.S. Census Bureau, 2018). An average household has 30 items with enclosed electronic circuitry (Mars & Nafe, 2016), and the average replacement rate is estimated at two per year

(Cleet, 2018). Commercial businesses also own significant (but unknown) numbers of these items.

Washington’s E-Cycle program collects computers, televisions and monitors (Washington Materials Management & Financing Authority, 2018). From 2014 to 2018, between four and six million pounds of plastic enclosures were collected yearly. These items are a major but unknown fraction of all such enclosures, since not all electronics qualify for acceptance to Washington’s E-Cycle program, and not all qualified items are recycled.

Potential for exposure to sensitive populations

People, including sensitive populations such as children and workers, can be exposed to flame retardants used in electronics. Additive chemical flame retardants are not chemically bound to the treated materials and can migrate out of products. A large number of flame retardants have been detected in house dust studies. BTBPE (Brown, Whitehead, Park, Metayer, & Petreas, 2014; Dodson et al., 2012; La Guardia & Hale, 2015; Schreder & La Guardia, 2014) and DBDPE (Brown et al., 2014; Johnson, Stapleton, Mukherjee, Hauser, & Meeker, 2013; Stapleton et al., 2009) have been widely detected in house dust. TPP was detected in indoor dust in studies of homes in North Carolina, Boston, California, and Canada (Knudsen, Hughes, Sanders, Hall, & Birnbaum, 2016; La Guardia, Schreder, Uding, & Hale, 2017; Liu, He, Hites, & Salamova, 2016; Ma, Venier, & Hites, 2012; Zhou et al., 2014). TTBP-TAZ and TBBPA have also been found in house dust (Abdallah, Harrad, & Covaci, 2008; Ballesteros-Gómez et al., 2014; Fromme et al., 2014; Guo et al., 2018; Schreder & La Guardia, 2014).

Flame retardants in electronics can contribute to concentrations in house dust. In 2016, Canadian researchers found that surface wipes of home and office electronics had detectable concentrations of organohalogen and phosphorous-based flame retardants. Concentrations in surface wipes of hard polymer casings were correlated with household dust samples, suggesting that the additive flame retardants used in electronics contribute to household dust concentrations (Abbasi, Saini, Goosey, & Diamond, 2016).

A more recent study from the Netherlands found that time, age, hours-of-operation, and use of the standby function of electronics had significant influences on flame retardant concentrations in house dust (Sugeng, de Cock, Leonards, & van de Bor, 2018). This supports an older study that found decaBDE in dust was associated with the presence of televisions in the bedroom (Allen, McClean, Stapleton, & Webster, 2008). TCP and TPP concentrations in house dust have been associated with proximity to electronics (Brandsma, de Boer, van Velzen, & Leonards, 2014). Similar relationships have also been found for decaBDE and another organohalogen flame retardant, Hexabromocyclododecane (HBCD) (Harrad, Abdallah, & Covaci, 2009; Muenhor & Harrad, 2012).

The Centers for Disease Control (CDC) analyzed stored urine samples, looking for nine organophosphate flame retardants to learn more about PBDE replacements in people six years and older from the 2013 – 2014 NHANES (Ospina, Jayatilaka, Wong, Restrepo, & Calafat, 2018). CDC developed a new method to measure urinary metabolites (the end-products of the chemical after it goes through metabolism in the body) for several other organophosphate flame retardants and found widespread exposure in the general U.S. population. The study detected the TPP metabolite DPHP in more than 80% of the samples analyzed, suggesting widespread exposure to TPP. In general, women had higher levels than men, and children had higher levels than adults. The study comprised a representative sample of the broader U.S. population, and the results are consistent with previous studies that used only convenience samples.

DBDPE was detected in less than 10% of samples of maternal blood and breastmilk collected between 2008 and 2009 in the Sherbrooke region of Canada (Zhou et al., 2014). BTBPE has been detected at low levels in less than 10% of the samples of maternal blood in Norway (Cequier, Marcé, Becher, & Thomsen, 2015).

In U.S. residents, TBBPA has been studied in human blood, breastmilk, and fat tissue. TBBPA was detected in 35% of the 43 breastmilk samples collected in 2004 – 2005 from first-time mothers in Boston, Massachusetts (Carignan et al., 2012). TBBPA was detectable at low levels in 31% of human fat tissues from people who underwent liposuction surgery in New York City (Johnson-Restrepo, Adams, & Kannan, 2008).

The only widespread detection of TBBPA was 95% of an Inuit population in northern Canada that eats marine mammals (Dallaire et al., 2009). TBBPA has also been studied in other countries and detected in human blood (Alberta Health and Wellness, 2008; Kiciński et al., 2012; Kim & Oh, 2014) and breastmilk (Cariou et al., 2008; Pratt et al., 2013; Shi, Wu, Li, Zhao, & Feng, 2009; Shi et al., 2013).

A 2008 study by Abdallah et al. estimated human exposure to TBBPA in adults and toddlers (6 – 24 months) via air inhalation and dust and food ingestion. The study concluded that dust ingestion appears to be the major pathway of exposure to TBBPA for toddlers in the United Kingdom. A toddler weighing ten kg and ingesting 200 mg dust per day with TBBPA at the 95th percentile concentration reported in their study would ingest 100 times more TBBPA via house dust than through their normal diet.

Workers in certain occupations that are exposed to electronics have been found to have higher exposure to some flame retardants than the general public. Occupations most likely to have higher exposures to flame retardants in electronics are office workers, firefighters, and electronics recyclers. While data on occupational exposures to newer flame retardants is scarce, many studies have documented the exposure pathways for PBDEs from electric and electronic products. Sjödin et al. (1999) found elevated PBDEs in computer workers compared to cleaning personnel. Upon further evaluation, Jakobsson et al. (2002) found certain PBDE congeners (distinct PBDE compounds) were approximately five times higher in computer technicians compared to other clerks and cleaning personnel. Other studies did not find a clear relationship between exposure to flame retardant-containing equipment and office workers (Watkins et al., 2011).

Workers recycling electronics are exposed to higher levels of flame retardants. Higher levels of PBDEs were found in recycling workers in China (Qu et al., 2007) and Sweden (Sjödin et al., 1999) compared to control groups. Sjödin et al. (1999) found that levels of PBDEs in workers in an electronics dismantling plant were about five times higher than other workers.

Firefighters are exposed to flame retardants emitted by combusted materials. California has a firefighter biomonitoring program: the Firefighter Occupational Exposure (FOX) project. Compared to the general California population, firefighters were found to have higher levels of several of the eight PBDEs measured in the study (Park et al., 2015). Nearly one third of the firefighters had particularly high blood levels of decaBDE, but the levels are lower than in electronic recyclers.

Potential environmental exposure and exposure to sensitive species

Flame retardants used in electronics have been found in Washington's environment and elsewhere. In Washington, PBDE flame retardants have been found in many different environmental media and appear to be a ubiquitous contaminant in aquatic systems (Ecology, 2006, 2011c, 2011d). Even though some of the most toxic PBDE congeners (pentabromodiphenyl ether (pentaBDE) and octaBDE) have been restricted since 2004, they are environmentally persistent and can still be found in many environments.

DecaBDE may be debrominated to form other congeners (Zhao, Rogers, Ding, & He, 2018). Ecology has detected decaBDE in wastewater treatment plant (WWTP) effluent (Ecology, 2010c), surface water in Puget Sound (Ecology, 2011b), stormwater (Ecology, 2018), sediment (Ecology, 2011c, 2011d, 2016b, 2017, 2018) and freshwater fish (Ecology, 2006, 2012, 2016a, 2019b).

DBDPE is also found in Washington's environment. While it was not detected in WWTP effluent (Schreder & La Guardia, 2014) or sediment (Ecology, 2016c), it has been detected in WWTP influent (Schreder & La Guardia, 2014) and freshwater fish (Ecology, 2016b). TBBPA was not detected in freshwater fish in Washington, but it has been detected in a Swedish river (Gustavsson et al., 2018). BTBPE has been detected in Washington sediment (Ecology, 2016c) and fish (Ecology, 2019c), but was not detected in more recent sediment or stormwater studies (Ecology, 2018). There are no Washington-specific data for TTBP-TAZ.

Flame retardants from electronic enclosures can contribute to environmental flame retardant concentrations. Flame retardants from electronics can contribute to house dust concentrations. Contaminated house dust can be tracked outside (Wild et al., 2015), and can be released into wastewater when we wash textiles with house dust on them (Saini, Thaysen, Jantunen, McQueen, & Diamond, 2016; Schreder & La Guardia, 2014).

Flame retardants in electronic components can continue to be released after their useful life is over and they are disposed of in landfills (Choi, Lee, & Osako, 2009; Kim, Osako, & Sakai, 2006; Osako, Kim, & Sakai, 2004). Electronic products ordinarily have a use life of 5 – 15 years (OECD, 2019). In 2012, EPA estimated that electronic waste comprised 1.4% of the municipal solid waste stream in the U.S. (EPA, 2014d). In Washington, approximately 0.75% of the disposed municipal solid waste is electronic equipment (Ecology, 2016d). The plastic components of electronics (e.g., electronic enclosures) comprise 25 – 30% of the product by weight (Meyer & Katz, 2016). Based on this and the concentrations noted above, as much as 0.1% of municipal solid waste mass could be flame retardants from electronic enclosures.

Many electronic components can be recycled, though EPA (2014b) estimated that less than 30% of electronic products actually are (Meyer & Katz, 2016). Flame retardants can be released into the environment during the recycling process (Matsukami et al., 2015). In fact, TTBP-TAZ has been found in e-waste dust (Guo et al., 2018). When new consumer products are manufactured from recycled material, there is the potential for flame retardants in recycled electronics plastics to contaminate the new items such as toys, food packaging and other single use plastic goods (Turner, 2018). This is an additional path by which flame retardants can be released into the environment when discarded, and through which sensitive populations can be exposed.

Exposure to flame retardants in the environment is a concern for sensitive species. PBDEs, BTBPE, and other brominated flame retardants are persistent in the environment and can be transported long distances. This property has made them ubiquitous in environmental media across the globe and as far away as the Arctic, which is an indicator of persistence and bioaccumulation used by the Stockholm Convention on Persistent Organic Pollutants (de Whit, Herzke, & Vorkamp, 2010).

Certain PBDE congeners, such as PBDE 99, are highly bioaccumulative, which is concerning for species at higher trophic levels. While studies have shown that decaBDE does not bioaccumulate, lower brominated degradation products (including lower substituted PBDEs) bioaccumulate in organisms and concentrate in the environment (EPA, 2009c). Chemicals that bioaccumulate can increase in concentration in animals at higher trophic levels. This process, termed biomagnification, makes orcas and other predators more vulnerable to bioaccumulative chemicals in the environment.

Other brominated flame retardants like TBBPA and BTBPE appear to have lower bioaccumulation potential than PBDEs (La Guardia et al., 2012; Morris et al., 2004). However, modeled K_{ow} values, which are related to how much chemicals partition between water and fat, suggest that many of the replacement brominated flame retardants may still bioaccumulate (Kuramochi, 2014). Indeed, many have been detected in wildlife, though at lower concentrations than PBDEs or HBCD (Covaci et al., 2011).

PBDEs have been found in Southern Resident Orcas (O'Neill et al., 2015; Rayne, Ikonomou, Ross, Ellis, & Barrett-Lennard, 2004; Sloan et al., 2010). Chinook salmon that spend time in Puget Sound, an important orca food source, also have elevated concentrations of PBDEs (O'Neill et al., 2015; Puget Sound Partnership, 2017; Sloan et al., 2010). A similar trend has been observed in Chinook salmon that spend more time near urban areas along the Columbia River (Sloan et al., 2010).

In 2018, Governor Inslee established the Southern Resident Orca Task Force, a 50-member workgroup charged with developing plans for recovering Puget Sound's struggling southern resident orca population. The Task Force identified PBDEs as a primary contaminant of concern.

Existing regulations

Currently, there are no U.S. federal restrictions around flame retardants currently used in electric and electronic enclosures. However, there have been previous restrictions on flame retardants in general. In 1973, polybrominated biphenyls were banned (EPA, 2017b). The manufacturing and importing of pentaBDE was phased out in the U.S. beginning in 2005 (EPA, 2017a). In 2017, the federal Consumer Product Safety Commission (CPSC) granted a petition requiring them to initiate rulemaking to prohibit nonpolymeric, additive organohalogen flame retardants in four product categories including plastic casings surrounding electronics. As of the date of this report, CPSC has not banned organohalogen flame retardants in these products (CPSC, 2019b). In 2019, EPA proposed a rule under TSCA that will ban the processing and distribution in commerce of most uses of decaBDE (EPA, 2019a). TPP and TBBPA are on EPA's proposed list of 20 high priority chemicals to review for the next risk evaluations under the Toxic Substances Control Act (TSCA) (EPA, 2019c).

In the U.S., there are several state regulations to limit exposures to flame retardant chemicals in electric and electronic enclosures. Washington banned the use of PBDEs in all nonedible products, including electronic enclosures, under Chapter 70.76 RCW. This regulation bans all PBDEs in these products as of January 2011.

California, Hawaii, Illinois, Maine, Maryland, Michigan, Minnesota, New York, Oregon, and Vermont also have regulations restricting the use of certain flame retardants in products. While products are not well defined, electronics are not exempted (Safer States, 2020a). In 2017, Washington restricted additive TBBPA and several other flame retardants commonly found in furniture. This restriction includes furniture with electronic enclosures.

A flame retardant ban recently adopted by the European Union prohibits the use of halogenated flame retardants in enclosures and stands of electronic displays after March 1, 2021 (European Commission, 2019b; European Union, 2019).

Availability of safer alternatives

There are several potential alternatives for electric and electronic device casings, including:

- Adding alternative flame retardants to the plastic casings.
- Using more inherently fire-resistant material for casings.
- Design changes to better insulate device casings from the electronic components that can spark a fire.

In addition to additive halogenated flame retardants, several additive organophosphate flame retardants such as TPP and RDP have been used in plastic enclosures to meet flammability standards for some electronic equipment. Other alternatives suitable for some applications include using inherently fire-resistant polymers or non-flammable materials, such as aluminum, or removing the electronic source from the casing.

Several alternatives have been investigated by other organizations (Ecology & Health 2008; EPA, 2014c; Ministry of Environment and Food of Denmark 2016; TCO certified, 2019). Alternative materials and non-halogenated flame retardants may be safer alternatives for some applications. Any alternative solutions would need to meet performance standards and fire safety procedures. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of alternatives in these applications in Phase 3.

Chapter 2: Recreational Polyurethane Foam Products

Overview

Priority product

Recreational foam products are those that are made from polyurethane foam and used as padding in recreational and athletic facilities, such as indoor climbing, gymnastics and athletic gyms, schools and trampoline parks. Examples of recreational foam products include:

- Foam pit cubes.
- Mats and pads—including crash mats, landing mats, training mats, panel mats, martial arts mats, and wall and post pads.

This priority product does not include outdoor playground equipment or padding designed to be worn.

Priority chemical

Organohalogen flame retardants and flame retardants identified under Chapter 70.240 RCW[25] were listed as priority chemicals in RCW 70.365.[26]

Organohalogen flame retardants are chemicals intended to slow ignition and progression of fires. They contain one or more halogen elements bonded to carbon.

The non-halogenated flame retardants identified under Chapter 70.240 RCW are: triphenyl phosphate (TPP), tri-n-butyl phosphate (TNBP), ethylhexyl diphenyl phosphate (EHDPP), tricresyl phosphate (TCP), and isopropylated triphenyl phosphate (IPTPP).

See the electric and electronic enclosures chapter for more information about human health concerns and impacts of environmental release from flame retardants.

Priority product summary

Recreational foam products are a significant source of flame retardants. We considered the criteria in section RCW 70.365.030,[27] and children, workers and people of childbearing age are exposed to flame retardants from recreational foam products. Exposures to flame retardants are higher after spending time in foam pits.

Gymnasts come in frequent contact with recreational foam products and have higher exposures to flame retardants than the general population. Intervention studies have successfully reduced gymnasts' and coaches' exposure to some flame retardants by replacing foam products.

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[25] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240
[26] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365
[27] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Detailed support for our listing of flame retardants in recreational foam products as a priority product is shown below.

Background

Flame retardants are added to recreational foam products to meet flammability standards for materials such as polyurethane foam.

There are three main classes of flame retardants: halogenated (also known as organohalogens), non-halogenated (such as organophosphates), and inorganic based. The most common class of flame retardants used in recreational foam products are the organohalogen flame retardants (Cooper et al., 2016).

Estimated volume of flame retardants in recreational foam products

Historically, the mixture pentaBDE was the most common flame retardant used in flexible polyurethane foam. But with its phase out, a number of alternative flame retardants have replaced its use. This trend has also been found in pit cubes (Carignan et al., 2016). Several studies have screened or tested recreational foam products. In addition to pentaBDE, other flame retardants that have been identified are a mixture of TBB, TBPH, and TPP, TDCPP, V6 or TBPP. Table 4 summarizes several flame retardants that have been measured in recreational foam products above 1% of the total mass.

Table 4. Product testing results with recreational foam products.

Flame retardants	Product types with flame retardants detected above 1% (by weight)	References
Pentabrominated Diphenyl Ether (pentaBDE)	Pit cubes	Carignan et al., 2013; CDC, 2017c; Cooper et al., 2016
Firemaster 550 or 600 (mixture including TBPH, TBB and TPP)	Pit cubes	Carignan et al., 2013; CDC, 2017c; Cooper et al., 2016; Dembsey et al., 2019
TDCPP	Pit cubes	Carignan et al., 2013; CDC, 2017c; Cooper et al., 2016; Dembsey et al., 2019
V6	Pit cubes	Cooper et al., 2016

Additional studies screened products for total bromine using an XRF (Carignan et al., 2013; CDC, 2017c). In these studies, bromine is a surrogate for brominated flame retardants. The recreational foam products that had at least one sample with total bromine greater than 1% (by weight) are summarized in Table 5. This indicates that brominated flame retardants are likely used in all of these product types.

Table 5. Product testing bromine screening results.

Bromine concentration (weight %)	Product types	References
0.0001 – 6.16	Pit cubes	Carignan et al., 2013; CDC, 2017c
0.005 – 3.61	Landing mats	Carignan et al., 2013
0.081 – 1.99	Sting mats	Carignan et al., 2013
0.0001 – 2.9	Floor mat foam	Carignan et al. 2013; CDC, 2017c
0.0001 – 2.9	Floor mat foam	Carignan et al. 2013; CDC, 2017c
0.0029 – 1.3	Hard foam equipment and ring foam	CDC, 2017c

The amount of flame retardant needed to meet the flame retardancy function depends on multiple factors including the chemical structure of the polymer, its properties when heated and when exposed to flames, as well as the type of flame retardants added. In general, additive flame retardants that are intentionally used are frequently found in foam at levels greater than 1%. In one study of recreational foam, the total concentration of flame retardants in foam pits ranged from approximately 2 – 6.5% (Carignan et al., 2013).

Estimated volume used in Washington

We found 21 gymnastics studios in Washington with foam pits pictured on their websites. We estimate that each foam pit is eight feet long, 20 feet wide and four feet deep, and contains approximately 10,800 eight cubic inch foam blocks. These 21 foam pits in the state would house 226,800 eight cubic inch foam blocks. Given that foam pits are found in other locations, and that not all foam pits are advertised on websites, we expect this to be an underestimate. A google search for “indoor trampoline” in Washington revealed 101 additional facilities that may contain foam pits. Even if only half these facilities have foam pits, over 500,000 foam pit blocks would be found in Washington.

In addition to foam pits, flame retardants are also used in mats and padding. Since all gymnastics studios use mats, we took the number of gyms nationally and adjusted it for the population in Washington. There are approximately 3,500 gymnastics studios in the U.S. We estimate that each studio is approximately 20,000 square feet and about 50% covered by mat, leading to approximately 800,000 square feet of mats in Washington’s gyms. This may be an underrepresentation given that mats that may contain flame retardants are used in a number of other types of facilities including traditional gyms and yoga studios. Further, because the thickness of mats varies from a few inches to many feet, we don't know how much foam this translates to.

Potential for exposure to sensitive human populations

Flame retardants migrate out of recreational foam products and contaminate the indoor environment. As foam degrades, flame retardants can accumulate in dust where they can be inhaled, ingested and come into contact with skin. Recreational foam products are designed to be compressed to reduce impact. As they compress, air trapped inside the foam is released. This could also force flame retardants out of the product.

Studies show that gymnastics studios have higher concentrations of flame retardants in dust than homes (Carignan et al., 2013; La Guardia & Hale, 2015). TBPH was one to three orders of magnitude higher in dust from gymnastics studios than from homes (Carignan et al., 2013). Another study found TBB and TDCPP in higher concentrations in gyms than homes, and the total flame retardant burden in gym dust was over eightfold higher than house dust (La Guardia & Hale, 2015).

There is evidence that the flame retardants in gyms come from recreational foam products. Broadwater et al. (2017) characterized flame retardant concentrations in four gymnastics studios and found that flame retardants accumulated in dust, with the highest concentrations in dust from the foam pit. Flame retardants in gyms can also be inhaled. La Guardia and Hale (2015) and Ceballos et al. (2018) found flame retardants on interior window wipes, suggesting that flame retardants in the air accumulated on windows. They also detected flame retardants in respirable particles in gyms (La Guardia & Hale, 2015).

While exposures in gymnastics studios have been the best studied, we expect that the use of similar recreational foam products in other facilities would have the same exposure pathways. Many of these additional recreational facilities are typically marketed toward children, including trampoline, bouncy house, circus, and playdate facilities. These facilities could expose a broader population to flame retardants.

People who spend time in recreational centers are exposed to flame retardants from recreational foam products. A 2013 study found BDE 153 was 4 – 6.5 times higher in collegiate gymnasts than in the general populations (Carignan et al., 2013). This is corroborated by a two- to threefold increase in the concentration of pentaBDE, TBB, and TBPH in hand wipe samples after training in the gym (Carignan et al., 2013). A follow-up 2016 study found that training in the gym led to a 50% increase in the concentration of TPHP metabolites (the end-products of the chemical after it goes through metabolism in the body) in urine (Carignan et al., 2016). In addition to gymnasts, coaches also have elevated exposure while at work. Ceballos et al. (2018) found significantly higher concentrations of nine flame retardants on employees' hands after work.

When foam products with flame retardants are replaced with flame retardant free products, exposures decrease. Dembsey et al. (2019) conducted an intervention study by measuring gymnasts' exposure to flame retardants before and after replacing the foam pit with flame retardant free cubes. They observed a fivefold reduction in the concentration of flame retardants on gymnasts' hands, compared to the pre-intervention samples (Dembsey et al., 2019). A second intervention study found that replacing foam pit cubes reduced the increase in flame retardants on hand wipes typically observed during gymnastics coaches' shifts (Ceballos, Broadwater, Page, Croteau, & La Guardia, 2018). However, the new pit cubes introduced in this study still contained two brominated flame retardants, so the study was only able to show a significant reduction in PBDEs, and not TBB or TBPH.

Exposure to flame retardants from recreational foam is important for sensitive populations, such as workers, children, and people of childbearing age. As discussed previously, gymnastics coaches are exposed to flame retardants at work (Ceballos et al., 2018). Collegiate gymnasts are of childbearing age. Children can start gymnastics before they're even walking, as many facilities offer parent-child classes for babies and toddlers. As children grow older, they spend more time at the gym, increasing exposure periods.

Potential environmental exposure and exposure to sensitive ecological species

Flame retardants used in recreational foam products have been found in Washington's environment. Of the flame retardants found in recreational foam products (Table 4), TPP and V6 have been detected in Washington. TPP has been detected in stormwater (Medlen et al., 2018) and freshwater sediment (Counihan et al., 2014, Mathieu et al., in prep; Medlen et al., 2018), but not freshwater (Alvarez et al., 2015; Mathieu et al., in prep). V6 has been detected in stormwater (Medlen et al., 2018), freshwater, and freshwater sediment (Mathieu et al., in prep; Medlen et al., 2018). TBB and TBPH have been assessed in stormwater and freshwater sediment, but were not detected (Mathieu et al., in prep; Medlen et al., 2018).

Flame retardants from recreational foam products can contribute to environmental flame retardant concentrations. Flame retardants in recreational foam products contribute to gym dust concentrations, which can be tracked outside (Wild et al., 2015) and released into wastewater when we wash textiles with gym dust on them (Saini, Thaysen, Jantunen, McQueen, & Diamond, 2016; Schreder & La Guardia, 2014). Flame retardants in recreational foam products can also contribute to environmental concentrations after their disposal (Lucas et al., 2017).

Exposure to flame retardants in the environment is a concern for sensitive species. Organohalogen flame retardants are persistent in the environment and can be transported long distances. This property has made them ubiquitous in environmental media across the globe and as far away as the Arctic, which is an indicator of persistence and bioaccumulation used by the Stockholm Convention on Persistent Organic Pollutants (de Whit, Herzke, & Vorkamp, 2010). Certain PBDE congeners (distinct PBDE compounds), such as PBDE 99, are highly bioaccumulative, which is concerning for species at higher trophic levels, such as orcas. While PBDEs have largely been phased out of consumer products, their persistence makes contamination a lingering environmental problem. While we have the most data on PBDEs, other persistent flame retardants are expected to pose similar environmental threats.

In 2018, Governor Inslee established the Southern Resident Orca Task Force, a 50-member workgroup charged with developing plans for recovering Puget Sound's struggling southern resident orca population. The Task Force identified PBDEs as a primary contaminant of concern. See the electric and electronic enclosures chapter for more information about the impacts of exposure to PBDEs on sensitive populations.

Existing regulations

There are no specific federal or state regulations pertaining to flame retardants in recreational foam products. However, there are several related to chemicals used on flexible polyurethane foam or products that contain flexible polyurethane foam. The manufacturing and importing of pentaBDE was phased out in the U.S. beginning in 2005 (EPA, 2017) and the chemical is banned in Europe because of its toxicity and environmental persistence (European Parliament, 2003).

In 2017, the federal Consumer Product Safety Commission (CPSC) granted a petition requiring them to initiate rulemaking to prohibit additive, nonpolymeric organohalogen flame retardants in four product categories, including furniture using polyurethane foam. To date, the CPSC has not banned organohalogen flame retardants in these products (CPSC, 2019b), but guidance was developed encouraging manufacturers of several products—including furniture—to refrain from intentionally adding non-polymeric additive halogenated flame retardants to their products (CPSC, 2017).

In 2006, EPA adopted a Significant New Use Rule (SNUR) which requires notification to EPA 90 days prior to U.S. manufacture or import for any use of pentaBDE and octaBDE. In 2012, EPA proposed a rule under TSCA requiring notice to the Agency prior to the manufacture or import of articles to which pentaBDE has been added (EPA, 2019b). Additionally, EPA is proposing a rule to address four flame retardants, including isopropylated phosphate (3:1), a non-halogenated flame retardant. This proposed rule could restrict or prohibit manufacture (including import), processing, and distribution in commerce for many uses of these substances (EPA, 2019e). In addition, TCEP and TPP are on EPA's proposed list of 20 high priority chemicals to review for the next risk evaluations under TSCA (EPA, 2019b).

In addition to federal efforts, several states are limiting exposures to flame retardants in consumer products. They include brominated, organohalogen, and organophosphorous compounds. In Washington, several flame retardants, including TDCPP and TCEP, are restricted in children's products and residential upholstered furniture. There are also reporting requirements for several flame retardants in children's products.

Maine has prohibited the sale of residential upholstered furniture containing more than 0.1% of a flame retardant chemical, or a mixture that includes them (Maine Department of the Environment, 2019). California, Massachusetts, New York, Rhode Island, and the City of San Francisco all have similar laws restricting the sale of products such as residential upholstered furniture and children's products containing flame retardants (Safer States, 2020c). In addition to restrictions, TDCPP is listed as a carcinogen under California's Proposition 65 (OEHHA, 2011).

Availability of safer alternatives

In addition to additive halogenated flame retardants, reactive halogenated flame retardants, and those identified by Ecology under Chapter 70.240 RCW,[28] several non-halogentated flame retardants such as TBPP, ammonium polyphosphate, expandable graphite, and melamine have been investigated in an alternative assessment of flame retardants used in flexible polyurethane foam (Cooper et al., 2016, EPA, 2015).

Other solutions include using foam without flame retardants in conjunction with other fire safety measures, including smoke detectors and sprinkler systems (Dembsey et al., 2018). Any alternative solutions would need to meet Washington building codes and other fire safety standards. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

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[28] http://app.leg.wa.gov/RCW/default.aspx?cite=70.240

Chapter 3: Paints and Printing Inks

Overview

Priority product

Paints and printing inks sold in any form or packaging for personal, commercial, or industrial use. This does not include dyes, toner powder, or painted or printed products.

Priority chemical

Polychlorinated biphenyls (PCBs).

PCBs in consumer products contribute to human and environmental exposure. Exposure is associated with many health problems, including liver toxicity, endocrine disruption, immune toxicity, and impaired reproduction (ATSDR, 2014). Based on their toxicity, persistence in the environment, and ability to accumulate in wildlife and people, PCBs are considered persistent, bioaccumulative, and toxic (Chapter 173-333 WAC[29]).

PCBs were phased out of most consumer products in the late 1970s, but their persistence means they are still problematic in our environment. Reducing current PCB releases from consumer products will help prevent additional long-term environmental consequences.

Priority product summary

Paints and printing inks are a significant source of inadvertently generated PCBs. We made this determination by considering the criteria in RCW 70.365.030.[30] Several bodies of water in the state are considered impaired due to PCB contamination. PCBs have a wide variety of toxic effects, accumulate in people and animals, and contaminate our food supply.

We estimate that colored pigments contained in paints and inks are the largest source of inadvertent PCB (iPCB) presence in consumer goods. Approximately 56 million pounds of printing ink and 30 million gallons of paint are used per year in Washington. Ecology has listed PCBs as persistent, bioaccumulative, and toxic, and has written a Chemical Action Plan (CAP)[31] to address them. Both our CAP and Governor Inslee's Southern Resident Orca Task Force recommended targeting and reducing iPCB production. Detailed support for our listing of PCBs in paints and printing inks as a priority product is shown below.

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[29] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-333
[30] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030
[31] https://fortress.wa.gov/ecy/publications/documents/1507002.pdf

Background

Historically, PCBs have been used intentionally in a wide range of products, such as electronic equipment, caulking, and carbon copy paper. Most intentional uses, often termed “legacy” uses, have been banned in the U.S. since 1979. iPCBs are PCBs that are not intentionally added to products, but are instead produced as an unintended byproduct of the manufacturing process. Although legacy PCBs are still present in some products currently in use and are still the main source of environmental contamination, iPCBs are the predominant source of new PCBs in consumer goods (Ecology & Health, 2015).

There are 209 distinct PCB compounds (known as congeners) which differ depending on the number and location of chlorination on the biphenyl molecule. Different congeners have different physical properties, toxicity, and environmental fates. The general structure for a PCB is shown below, where any of the numbers can contain a chlorine molecule or hydrogen molecule, depending on the congener.

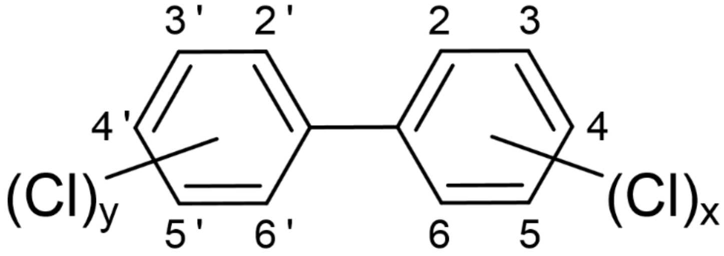

Figure 1. Molecular structure of polychlorinated biphenyls.

Estimated volume of PCBs used in paints and printing inks

iPCBs are widespread in paints and products containing printing inks. Pigments and other compounds can become contaminated with PCBs because of non-specific chlorination processes in many reactions where carbon, chlorine, and heat are involved. Pigments affected include diarylide yellows, phthalocyanines, and titanium dioxide. Pigments added to paints and printing inks can contain iPCBs. Figure 2 shows a list of organic pigments that have been found to contain iPCBs and the PCB congeners that were detected (Heine & Trebilcock, 2018).

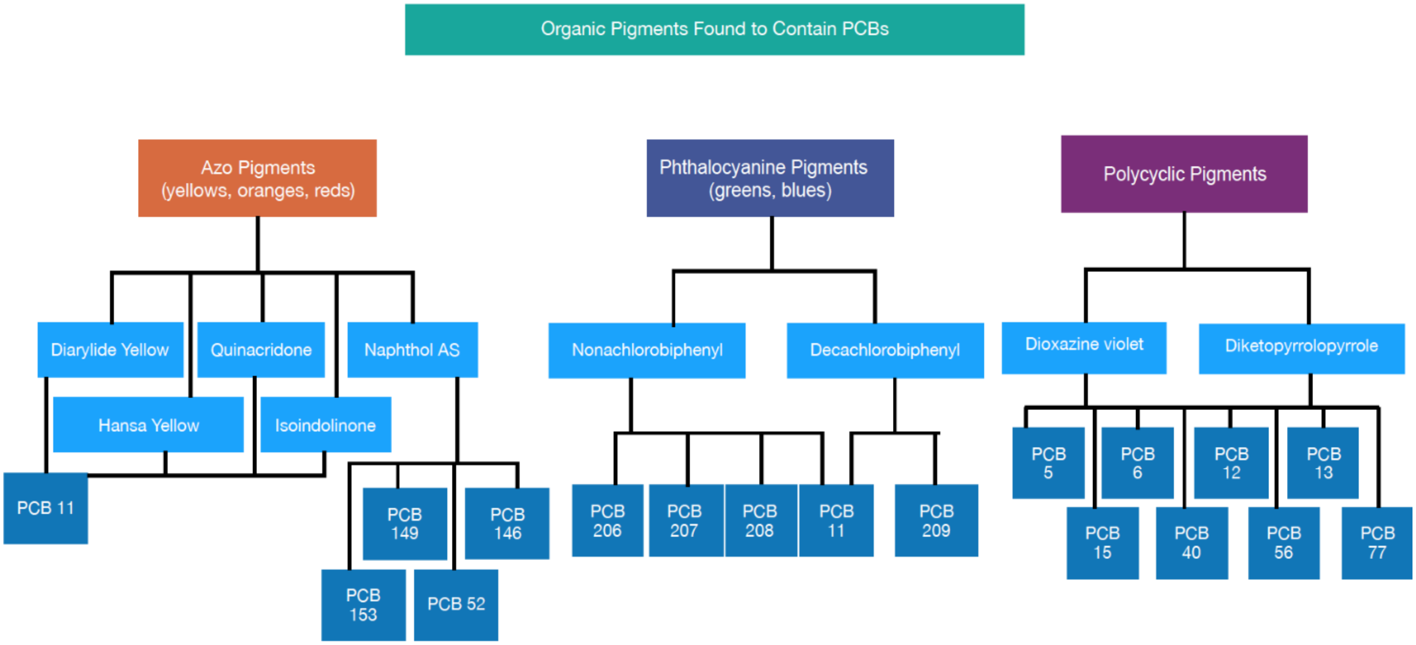

Figure 2. Organic pigments found to contain inadvertent PCBs.

PCB 11 contamination of diarylide yellow pigments is the source that is most associated with pigment manufacturing, but other congeners have been identified in a variety of pigments. Many congeners can be produced during pigment production, but PCB 11 was not a component of any legacy PCB products (such as common PCB mixtures sold under the trade name “Aroclor”), so it is considered a hallmark of inadvertent production. This means that any detections of PCB 11 in sampling, whether in the environment or of products, likely originated from inadvertent production. However, other PCB congeners can also be produced inadvertently, and processes subject to iPCB contamination often contain a large variety of congeners.

One source of iPCBs is from paints and printing inks. This likely occurs mostly in pigment ingredients. However, other ingredients may contribute as well—especially if silicone components are used, as they have been identified as another source of iPCBs (Ecology & Health, 2015). EPA requires reporting of iPCB generation, and pigment and dye manufacturing are the most commonly reported processes that produce iPCBs. Ecology estimated that pigment and dye manufacturing is the current product group whose manufacture contributes the most to PCB levels (Ecology & Health, 2015).

Paints and printing inks vary widely in components, depending on the intended use, but usually contain pigment or dye (provides color), binder (wets and disperses pigment), solvents (dissolve binder and mix components), and excipients (affect properties of ink). Paints and printing inks are the predominant use of pigments, using 50% or more of what is manufactured, with other uses including plastics and toiletries (Chemchain, 2017; Mordor Intelligence, 2019; Nestler, Heine, & Montgomery, 2019; Technavio, 2016).

There are limited studies investigating iPCB concentration in printing inks specifically, but Table 6 shows a summary of studies investigating PCB concentrations in paints, inks, pigments, and printed material from consumer products. For most printed products, unless there is another ingredient likely to cause PCB contamination, we assume that the majority of iPCB content comes from inks used in their production, since that is the most likely source. These results demonstrate that iPCB presence in paints, inks, and printed products is widespread. They also show that in products containing PCB 11, other congeners are often present as well. This suggests that the impact of the products is not limited to PCB 11.

Table 6. Product testing results with inks and pigments.

Year	Product	% of samples with PCB	Total PCB concentration	PCB 11 concentration	Other congeners detected	Reference
2009	Paint	15/33 (45%)	2 – 200 ppb	<LOQ – 16.4 ppb (13/15 samples)	8, 6, 4, 1, 12/13, 2, 3, 209 all in more than 40% of samples	Hu, Dingfei, & Hornbuckle, 2010
2009	Printed paper and plastic	12/12 (100%)	Not tested	0.11 – 38 ppb	Not tested	Rodenburg, Guo, Du, & Cavallo, 2010
2012	Pigments	376/588 (64%)	359 samples <0.5 ppm 17 samples >50 ppm	Not reported	Not reported	METI, 2013a
2013	Yellow pigments	24/24 (100%)	50 ppb – 919 ppm	42 ppb – 918 ppm	52 (4410 ppb max), 28 (1142 ppb), 77 (670 ppb), 126 (33.5 ppb), 153 (90.6 ppb)	Shang et al., 2014
2014	Printed paper and fabric	Not reported	Not tested	1.5 – 86 ppb	Not tested	Guo, Capozzi, Kraeutler, & Rodenburg, 2014

Year	Product	% of samples with PCB	Total PCB concentration	PCB 11 concentration	Other congeners detected	Reference
2015	Municipal products (products used by the City of Spokane)	39/41 (95%)	<LOQ – 2509 ppb	<LOQ – 63.8 ppb in paint	77, 209, 6, 64/72, 50, 28, 31, 118, 52/69 and others above 100 ppb	Spokane, 2015
2019	Paint colorants	8/8 (100%)	0.03 – 284 ppb	0.3 – 1.2 ppb (50% of samples)	209 (256 ppb), 208 (3.7 ppb), 206 (4.8 ppb), 146 (1.3 ppb), 52 (2.4 ppb)	Jahnke & Hornbuckle, 2019
2019	Consumer products including paint	7/16 (44%)	Not reported	<LOQ – 325 ppb	85 (140 ppb), 138 (120 ppb), 121 (100 ppb)	Liu, 2019

Estimated volume used in Washington

Up to 56 million pounds of printing ink are used per year in Washington. One report estimated that the U.S. printing inks market was approximately 2.5 billion pounds in 2017 (Mordor Intelligence, 2018). This estimate would put Washington's share (2.2% by population) at 56 million pounds.

This is supported by data from the Color Pigments Manufacturers Association (CPMA), which estimates that the total amount of phthalocyanine and diarylide pigments imported or manufactured in the U.S. is about 90 million pounds per year (Ecology & Health, 2015). This would mean Washington's share (by population) is around two million pounds of these pigments. Printing inks contain 5 – 30% pigment by weight (PCC, 2018), so if we only consider these two types of pigments, that would amount to approximately 7 – 40 million pounds of printing ink used.

Approximately 30 million gallons of paint and coatings are used in Washington per year. The American Coatings Association reported 2017 production of paints and coatings at 1.28 billion gallons in the U.S., while other sources reported 1.5 billion gallons in 2016 and a projected 1.4 billion gallons in 2020 (EPA, Freedonia Group, 2017; Pilcher, 2018; Wells, 2017). Taking

Washington's share of the U.S. population would give around 30 million gallons of paints in all of these cases, however, not all paints and coatings contain iPCBs.

Ecology's PCB Chemical Action Plan[32] estimated that Washington's share of PCB 11 from yellow pigment is between 0.02 and 31 kg per year, with data from the CPMA pointing toward an annual estimate of 9 – 20 kg (Ecology & Health, 2015).

Potential for exposure to sensitive populations when used

Nearly all people, including infants, are exposed to PCBs. Levels of PCBs in people have declined since the 1980s, but there is still widespread detection of PCBs in the U.S. population, including in infants and children (CDC, 2019a; Ecology & Health, 2015). People are mostly exposed to a mixture of PCBs rather than a single PCB compound. PCBs have been detected in human blood, fat tissue, breastmilk, and in cord blood (ATSDR, 2019). Some PCBs can remain in the body for years after exposure—varying by type of organism and type of PCB congener—and blood levels generally increase with age (Ecology & Health, 2015). Because PCBs are more readily absorbed than excreted, the levels of individual PCB congeners in the body may vary by exposure source and by differences in how bodies process them (CDC, 2017a).

PCBs have been measured and detected in the blood of the U.S. general population over age 11 since the 1999 NHANES survey cycle (CDC, 2019a; CDC, 2019b). PCB congeners 118, 138, 153, and 180 have been found at higher levels in the environment and in human blood samples than other PCB congeners (EPA, 2015c). They were detected in the majority of samples for women aged 16 – 49 from 1999 – 2014 (EPA, 2015c). Measured levels of PCBs in human blood decreased by an estimated 87% from 1973 – 2003 (Schecter et al., 2005; Sjodin et al., 2004). However, PCB 153 was detected in both children and adults more recently, with a median level of 7.4 ng/g lipid in 6 – 9-year-old girls, and 0.21 – 55.6 ng/g lipid in the general population (EPA, 2015c; CDC, 2019a).

In 2013, studies reported the presence of PCB 11, which is associated with iPCB production, in air samples and in the blood of children and mothers (Marek, Thorne, Wang, DeWall, & Hornbuckle, 2013; Zhu et al., 2013). This is an indication that airborne PCBs continue to be an environmental problem affecting large populations. A 2015 study reported PCB congeners 11, 14, 35, 133, and 209 as the most frequently detected non-Aroclor congeners in the blood of participants (Koh, Hornbuckle, & Thorne, 2015). More information on PCB exposures in people is available in the PCB Chemical Action Plan[33] (Ecology & Health, 2015).

People can be exposed to PCBs from paints and inks. Food is the main source of exposure for the general population, and the levels of PCBs in fish we eat are concerning. PCBs occur at the highest concentrations in fatty foods (e.g., dairy products and fish). PCBs from paints and inks

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[32] https://fortress.wa.gov/ecy/publications/documents/1507002.pdf

[33] https://fortress.wa.gov/ecy/publications/documents/1507002.pdf

may enter fish through stormwater and wastewater effluent, thus leading to human dietary exposure (Ecology & Health, 2015).

Paint colorants have the potential to emit PCBs, which can increase in the environment to concentrations of more than 500 pg/m^3 within hours of application (Jahnke & Hornbuckle, 2019). Thus, printing inks can also contribute to airborne PCB concentrations, especially in areas where printing is occurring. PCBs have also been detected in residential environments from indoor air and house dust (Takeuchi, Anezaki, & Kojima, 2017). A study reported concentrations of PCBs in indoor air in homes and schools in East Chicago and Columbus Junction and estimated exposures in mothers and their children (Ampleman et al., 2015). Inhalation exposure was greater in indoor environments than outdoor environments, and included contributions from PCB 11, which the authors attributed to pigments and paint (Ampleman et al., 2015).

PCBs in paints and inks make a significant contribution to exposure. People are exposed to PCBs in air, water, soil, and house dust. Humans can also be exposed to PCBs through direct contact with consumer products that use pigments (Guo, Capozzi, Kraeutler, & Rodenburg, 2014; Rodenburg, Geo, Du, & Cavallo, 2010) as well as from the presence of PCBs in the environment (Vorkamp, 2016).

Biomonitoring studies for PCB congeners unique to dyes and pigments showed that 65% of 85 women in the Midwest had trace levels of PCB 11 in their blood (Marek et al., 2014). Another study demonstrated that on average, about 10% of the total PCBs in study participants came from non-Aroclor PCBs. This study also found that, on average, 50% (but up to 100%) of non-Aroclor PCB concentrations in the participants were likely due to pigment exposure (Koh et al., 2015). PCB 11 concentrations have not decreased since 2004, and in 2007, PCB 11 was found in 91% of air samples taken near 40 Chicago area elementary schools (Hu, Martinez, and Hornbuckle, 2008).

Exposure to PCBs in paints and inks is particularly relevant for infants, children, women of childbearing age, and indigenous populations. Although levels of PCBs in blood for young people have declined over the past three decades, there is still potential for exposure through the use of products that contain these chemicals. Due to the presence of PCBs in consumer products, and their release into the air, dust, and food supply, there is potential for infants and young children to be exposed, especially since they have more contact with house dust (Harrad et al., 2009; Rudel et al., 2003, 2008; Takeuchi et al., 2017). Infants are also exposed *in utero* and via nursing to PCBs that have accumulated in the mother's body over many years. For example, blood levels increased after birth in breastfed infants and then decreased in early adolescence (CDC, 2019a).

A study analyzed blood levels in children and their mothers from urban and rural U.S. communities (n = 200) for 209 PCBs from 2008 – 2010, and found widespread detection of all congeners, including PCB 11. This study reported variability of all PCBs and major metabolites (the end-products of the chemical after it goes through metabolism in the body) in two generations of people, which suggests that short-term exposures to PCBs may be a significant component of what is measured in human blood (Marek et al., 2014).

Local indigenous populations are at more risk from exposure to PCBs than other populations. These indigenous populations were surveyed in fish consumption studies, and they often eat more fish than other communities (Ecology, 2013). Exposure to PCBs has been linked to higher incidence of diabetes and cardiovascular disease in indigenous populations (Nestler et al., 2019). These findings have important health equity and environmental justice implications.

The National Tribal Toxics Council does not consider the 50 ppm allowance for iPCBs in pigments to be sufficiently protective of their population due to increased fish consumption. The Confederated Tribes of the Umatilla Indian Reservation has requested a lower tolerance level, specifically calling out inks and pigments (Nestler et al., 2019).

Potential for exposure to sensitive species when used

PCBs have been found in Washington's environment. PCBs have been found in freshwater, marine water, and sediment in Washington. PCB levels in Washington mussels, an indicator species for detecting local environmental contamination, are found well above national median concentrations (Ecology & Health, 2015). PCBs, including PCB 11, have been detected in atmospheric deposition in the Seattle metro area (Rodenburg, Winstanley, & Wallin, 2019). See Ecology's published Chemical Action Plan[34] (Ecology & Health, 2015) for more details on PCBs in Washington's environment.

The fish tissue equivalent of Washington's human health water quality criterion for PCBs is 5.3 ppb (Ecology & Health, 2015). Washington Department of Fish and Wildlife measured the total PCB concentrations in edible tissues of four marine fish species as compared to this value. All samples of Herring and Chinook, and most (70 – 80%) of the English Sole and Coho exceeded the criterion (Ecology & Health, 2015). There are 158 bodies of water in Washington state that are significantly impacted by PCBs[35] (Ecology & Health, 2015).

PCBs from paints and inks can contribute to environmental PCB concentrations. PCB 11 is considered a hallmark of iPCB contamination, specifically from pigments and dyes, since it is known to be present in many painted and printed materials, and it is not found in legacy PCB products (Heine & Trebilcock, 2018). PCBs have been shown to leach from painted and printed materials when exposed to water (EPA, 2015d; George et al., 2006; Guo et al., 2014). Pigments found in paints and inks are likely sources of PCBs detected in the environment (Andersson, Ottesen & Volden, 2004; Hu, Martinez, & Hornbuckle, 2011; Jartun, Ottesen, Steinnes & Volden, 2009; Jartun, Ottesen, Volden & Lundkvist, 2009; Johnston, George, Richter, Wang &

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[34] https://fortress.wa.gov/ecy/publications/documents/1507002.pdf

[35] https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Assessment-of-state-waters-303d

Wild, 2006; Ruus, Green, Maage & Skei, 2006). They have been directly linked to wastewater discharges with PCB levels above water quality criteria in the City of Spokane (Grossman, 2013).

Limited data are available, but it was estimated that two paper recycling facilities in Washington discharge 28 g of PCBs per year, with 3.8 g being PCB 11, and that the Spokane River Wastewater Treatment Plant (WWTP) was discharging 71 g of PCBs per year (Ecology & Health, 2015). Product testing results suggest that pigments may account for the majority of PCB 11 detected in the environment (Guo et al., 2014), and thus almost certainly contribute other congeners as well.

Exposure to PCBs in the environment is a concern for sensitive species. Although lower weight PCBs such as PCB 11 are less bioaccumulative than other congeners (Rodenburg & Delistraty, 2019), paints and printing inks also contain higher molecular weight PCBs, which bioaccumulate in animals and increase in concentration higher in the food chain. The Southern Resident Orca Task Force noted PCBs as a chemical of concern, and reducing exposure was one of their four recommendations (Southern Resident Orca Task Force, 2018). PCB contamination reduces the health of aquatic organisms throughout the food chain, including forage fish, salmon, and orcas. In addition, a variety of harmful effects have been shown in birds and mammals, such as reduced egg or embryo viability and reduced live births (Ecology & Health, 2015).

PCBs have been detected in many aquatic organisms at potentially harmful levels throughout Washington. Levels of PCBs associated with health impacts are observed in seals in the Strait of Georgia, and most Southern Resident Killer Whales exceed the health effects thresholds for PCB residues (Cullon et al., 2009; Hickie, Ross, Macdonald, & Ford, 2007). PCBs are persistent chemicals, so once released in the environment they can be challenging or impossible to remove, affecting wildlife for years to come. See Ecology's Chemical Action Plan[36] for more details about effects on wildlife health (Ecology & Health, 2015).

Currently the Washington State Department of Health advises human consumption restrictions for specific recreational fish in 14 water bodies of the state due to the high levels of total PCBs measured in fish tissue. This includes the upper, middle and lower Columbia River, the Spokane River, the Yakima River, the Wenatchee River, and Lake Washington (Health, n.d.).

Existing regulations

In 2014, RCW 39.26.280[37] established purchasing policies for Washington state with a preference for products and packaging that do not contain PCBs. In 2018, Governor Jay Inslee signed an executive order directing state agencies to implement nine immediate actions to benefit Southern Resident Orca Whales. In September of 2018, the Task Force published draft recommendations, including accelerating the implementation of the 2014 PCB purchasing law

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[36] https://fortress.wa.gov/ecy/publications/documents/1507002.pdf

[37] https://app.leg.wa.gov/RCW/default.aspx?cite=39.26.280

"to reduce the PCBs entering Puget Sound from products." Product suppliers to the state will provide information about PCBs in their products, and this information can be shared with other purchasers who want to avoid products containing PCBs (Southern Resident Orca Task Force, 2018).

The federal Toxic Substances Control Act (TSCA) bans intentional production and import of intentionally added PCBs and limits iPCBs in pigments to an average of 25 ppm, with a maximum of 50 ppm. This level was set to reduce the impact on industry, except for in the case of diarylide and phthalocyanine pigments, when it was considered feasible to reduce concentrations of iPCBs (Nestler et al., 2019).

The Stockholm Convention does not specify a limit on iPCBs in pigments. However, it dictates that signees develop an action plan to reduce iPCB releases. Some countries party to the Convention, such as Canada, have set a 50 ppm maximum limit and 25 ppm average limit on iPCBs (Nestler et al., 2019), similar to TSCA.

Availability of safer alternatives

Methods of manufacturing are possible to reduce or eliminate—in the case of titanium dioxide—the amount of inadvertent PCB content in pigments and dyes. In addition, inadvertent PCB production is not a problem for non-chlorinated pigments. Some organizations, including HP® and Apple®, have policies in place prohibiting the purchase of ingredients (including paints and inks) with PCBs over 0.1 ppm (Heine & Trebilcock, 2018).

Product testing has shown that paints and printed material with a wide variety of colors do not contain detectable levels of PCBs, indicating that low PCB products are available (Ecology, 2014b, in process of data validation; Ecology & Health, 2015). As part of our Safer Products for Washington process, we will continue to address the feasibility of paints and printing inks with lower iPCB content in Phase 3.

Chapter 4: Carpets and Rugs

Overview

Priority product

Carpets and rugs sold for residential and commercial settings.

Priority chemical

Per- and polyfluoroalkyl substances (PFAS).

PFAS represent a large class of chemicals that are highly persistent and can be bioaccumulative and toxic. Many PFAS have been associated with health problems, including immune system toxicity, reproductive impairment, developmental toxicity, and liver and kidney toxicity (Ecology, 2019a). As more PFAS studies are conducted, we learn that more are toxic.

PFAS are problematic in the environment due to their persistence and ability to move through environmental media. They can be found in most environmental samples. This is especially concerning when they enter drinking water because clean-up efforts are extremely challenging. Some chemicals within this class are considered persistent, bioaccumulative, and toxic (Chapter 173-333 WAC[38]), and chemicals of high concern to children (Chapter 70.240 RCW[39]).

Priority product summary

Carpets and rugs used in residential and commercial settings that have been treated during the manufacturing process for stain and water resistance are significant sources and uses of PFAS. We made this determination by considering the criteria in RCW 70.365.030.[40] Carpet contributes to the amounts of PFAS in our homes, workplaces, and environment, and has the potential to expose infants, young children, and workers.

We estimate that 1,300 – 2,000 metric tons (2.8 – 4.4 million pounds) of PFAS are brought into Washington homes and workplaces in carpet each year—a significant portion of PFAS use in total. Carpets can contribute to human PFAS exposure, particularly for children and infants who spend more time on or near the floor. Children living in homes with treated carpet have higher exposures to PFAS than those without.

The PFAS from our carpets can also be released into the environment. PFAS are environmentally persistent, meaning they break down only very slowly. Due to their water solubility and mobility in soil, PFAS released into the environment can flow into surface water and seep into groundwater. In 2018, Governor Inslee convened the Southern Resident Orca

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[38] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-333
[39] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240
[40] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Task Force to recommend actions to recover the southern orca population. The Task Force named PFAS as chemicals of emerging concern for orcas. Detailed support for our listing of PFAS in carpet as a priority product is shown below.

Background

PFAS are used to treat carpet during the manufacturing process and as aftermarket treatments. This priority product focuses on pretreated carpets and rugs that have had PFAS applied during the manufacturing process. The specific PFAS used to treat carpets and rugs have changed over time. From 1970 to 2002, the largest use of PFOS-derived substances (a subgroup of PFAS) was for carpet applications (48,000 tons globally from 1970 to 2002) (DEPA, 2013; Paul, Jones, & Sweetman, 2009). Currently, another PFAS subgroup—termed fluoropolymers or fluorotelomer-based acrylate polymers—are generally used for carpet stain resistance and carpet care treatments (Bowman, 2018; KEMI 2015). However, other PFAS can also be present as impurities (for example, PFHxA and PFBA) (Bowman, 2018), or can be formed during environmental degradation (FTOHs and PFCAs) (Washington & Jenkins, 2015).

Estimated volume of PFAS used in carpet

We estimate that PFAS are found in over half of carpets in Washington at concentrations that add significantly to the amounts of PFAS in our homes and workplaces. While we don't have data from Washington, a 2008 survey estimated that 56 – 90% of carpets sold in Denmark were treated with fluorinated substances (DEPA, 2013). Given that it is possible Washingtonians purchase more or less treated carpet than Danes, we based our estimate on the low end of the Danish range. In 2009, EPA estimated that the average home had 50 kg of carpet. PFAS concentrations in carpet vary depending on the specific compound—some may be intentionally added and others may be present as breakdown products or contaminants (Table 7).

Table 7. Reported concentrations of PFAS in carpet.

The concentrations vary by PFAS analyte. (* indicates that information on whether the carpet was pre-treated was not available, ^ indicates estimation by KEMI, 2015)

Product	Concentration	PFAS measured	Reference
Pre-treated carpet	484 µg/m^2	Total PFCA (C5 – C12)	EPA, 2009b
Carpet*	57.2 µg/kg	Total PFCA (C4 – C14)	Kotthoff, 2015
Pre-treated carpet	4,010 µg/kg	FTOH/FTS	Liu et al., 2015
Carpet*	73.5 µg/kg	FTOH/FTS	Kotthoff, 2015

Product	Concentration	PFAS measured	Reference
Pretreated carpet	15%^	PTFE and PASF	KEMI, 2015

Estimated volume sold and disposed of in Washington

We estimate that 15,360 metric tons of carpet (over 33 million pounds, which does not include any untreated backings) are sold yearly in Washington. No data is available on the proportion of homes containing carpet in Washington, so this estimate is based on California Carpet Stewardship Program data and adjusted to the population size of Washington. If 56 – 90% of carpets are pretreated to contain approximately 15% PFAS by weight, there are potentially 1,300 – 2,000 metric tons (2.8 – 4.4 million pounds) of PFAS in the carpets sold annually in Washington. The 15% estimate from the Swedish Chemicals Agency is for PTFE and PASF in pretreated carpet that is intentionally added to carpet (KEMI, 2015). Some manufacturing groups claim the KEMI estimation is too high and report use at around 0.1%, which would lead us to estimate that between 90 and 140 metric tons (approximately 200,000 – 300,000 pounds) of PFAS are used in Washington carpets each year. Most studies measure extractable PFAS, such as FTOHs or PFCA, that are present as contaminants and breakdown products. Using the FTOH concentrations from Liu et al. (2015) (4010 μg/kg) and total PFCA (C4 – C14) from EPA (2009b) (73.5 μg/kg), we estimate that up to 60 mg of FTOHs and 1,000 mg of total PFCAs are found as contaminants in carpets sold annually in Washington.

We estimate that 36,000 – 58,000 metric tons of PFAS-treated carpet end up in Washington landfills, and 47 – 76 metric tons of PFAS-treated carpet are illegally dumped each year in Washington. Ecology's draft Chemical Action Plan[41] on PFAS estimated that over 64,000 metric tons (approximately 141 million pounds) of carpet (including both treated fiber and untreated backing) end up in Washington landfills annually (Ecology, 2019a). This estimate is based on a waste characterization study, which involves sampling, sorting and surveying waste material delivered to landfills over a one year period (Ecology, 2016a). If 56 – 90% of carpets disposed of in Washington are treated with PFAS (KEMI, 2015), we estimate that 36,000 – 58,000 metric tons (79.3 – 127.8 million pounds) of PFAS-treated carpet end up in Washington landfills each year. Additionally, in 2004, Ecology estimated that 84 metric tons (approximately 185,000 pounds) of carpet were illegally dumped on Washington roadways (Ecology, 2005). If 56 – 90% of these carpets are treated, 47 – 76 metric tons (approximately 100,000 – 167,000 pounds) of PFAS-treated carpets could be illegally dumped each year.

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[41] https://fortress.wa.gov/ecy/publications/summarypages/1804005.html

Potential for exposure to sensitive populations

Nearly all people, including women of childbearing age, infants, and young children, are exposed to PFAS. A wide range of PFAS subgroups have been frequently detected in humans. The National Health and Nutrition Examination Survey routinely detects PFOA, PFOS, PFNA and PFHxS in blood of nearly all participants (CDC-NHANES, 2015, 2017b). PFBS and PFHpA are less commonly detected in blood, likely due to their relatively quick excretion rates in people (Kubwabo, Kosarac, & Lalonde, 2013), but can be found in indoor dust (de la Torre, Navarro, Sanz, & de los Angeles-Martinez, 2019; Karaskova et al., 2016; Strynar & Lindstrom, 2008).

Blood levels of some PFAS have declined in the U.S. general population due to the phase out of the use of PFOA- and PFOS-related compounds in U.S. production (CDC-NHANES, 2017). Similar results were reported in a study of 610 American Red Cross blood donors from 2000 through 2015 (Olsen et al., 2017). PFPeA, PFHxA, PFHpA, and PFBS have been commonly detected in breastmilk among Korean women (Kang et al., 2016) and PAPS (e.g., 4:2 diPAP, 6:2 diPAP, 8:2 diPAP, and 10:2 diPAP) have been detected in breastmilk among Canadian women (Kubwabo et al., 2013). More information about exposure can be found in the PFAS Chemical Action Plan Health Appendix (Ecology, 2019a).

People, including infants and young children, are exposed to PFAS from treated carpets. As carpet degrades, PFAS can be released into indoor air and accumulate in dust. People inhale and ingest PFAS-contaminated air and dust, leading to human exposure to PFAS. Compared to places without carpet, homes and offices with carpet can have higher concentrations of various PFAS in the indoor environment (Fraser et al., 2013; Gewurtz et al., 2009; Kubwabo, Stewart, Zhu, & Marro, 2005). Karaskova et al. (2016) found that the combined concentration 20 PFAS on carpeted floors was higher than other floor types (Karasova et al., 2016).

Children, including infants, spend more time on or near the floor, and have relatively high respiration rates and frequent hand-to-mouth activity. As such, they are exposed to more contaminated air, carpet, and house dust compared to their body weight than older people. Karaskova et al. (2016), Shoeib et al. (2011), Tian et al. (2016), and Trudel et al. (2008) have found that house dust is an important PFAS exposure route for toddlers. Washburn et al. (2005) estimated that the reasonable maximum exposure scenario for PFOS in carpet was two orders of magnitude higher for infants than adults, meaning infants could be exposed to PFAS at a level that is about 100 times higher than adults.

Studies show that children with carpets in their bedrooms have higher concentrations of PFOS, PFHxS, and Me-PFOSA-AcOH in their bodies than children with other types of bedroom flooring (Harris et al., 2017). Boronow et al. (2019) found that women living in homes with treated carpet had higher exposure to PFAS, PFNA and PFDeA. Fraser et al. (2012) found that office workers in buildings with higher concentrations of FTOH in the air had higher concentrations of PFOA in their blood.

Treated carpet makes a significant contribution to overall exposure. Based on a survey of the concentration of PFAS in 116 consumer articles, EPA concluded that carpet and carpet care products were likely significant sources of exposure to PFCAs (EPA, 2009b). Trudel et al. (2008) found that treated carpet could be a prominent source of consumer product exposure. They estimated that between 5 and 64% of PFOS exposure was related to contact with treated carpet.

Potential environmental exposure and exposure to sensitive species

PFAS are contaminants in Washington's environment. PFAAs have been detected in drinking water, surface waters, groundwater, wastewater effluent, freshwater, sediments, and wildlife in Washington. PFAAs have also been identified above EPA health advisory levels in groundwater used for drinking in Issaquah and in or around four military bases: Joint Base Lewis-McChord, Naval Air Station Whidbey Island, Fairchild Air Force Base, and Navy Base Kitsap-Bangor. (Although, the suspected source of PFAAs in drinking water at these sites is PFAS in firefighting foams.) Environmental monitoring in the state has shown that PFAA concentrations are highest in urban surface water and surface waters receiving minimally diluted WWTP effluent (Ecology, 2016a, 2019a). PFAAs and PFOSA have also been found in Washington wildlife, such as freshwater fish and osprey eggs (Ecology, 2016a, 2019a).

PFAS from carpet can contribute to environmental PFAS concentrations. PFAS from carpet are also released into the environment through disposal of carpet in landfills (Lang, Allred, Peaslee, Field, & Barlaz, 2016). Based on the volume estimates described previously, carpet contributes PFAS in Washington landfills and the environment through illegal dumping.

The persistence of PFAS in the environment heightens concerns about disposal and release. PFAS can be released from landfills into air, groundwater, and as leachate (Hamid & Grace, 2018) and PFAS from illegally-dumped carpet are released directly into the environment. PFAS from indoor carpet can accumulate in house dust (Gewurtz et al., 2009; Knobeloch et al., 2012; Kubwabo et al., 2005) and can be tracked and vented into the outdoor environment (Wild et al., 2015) or released into wastewater through laundering processes (Shoeib, Harner, Webster, & Lee, 2011).

Exposure to PFAS in the environment is a concern for sensitive species. PFAS are environmentally persistent and some are bioaccumulative (Ecology, 2019a). Two PFAS—PDFA and PFOS—have been detected in salmon in Puget Sound (Meador, Yeh, Young, & Gallagher, 2016). Governor Inslee's Southern Resident Orca Task Force identified PFAS as chemicals of emerging concern.

Existing regulations

PFAS are not currently regulated under established environmental laws including the Resource Conservation and Recovery Act, the Comprehensive Environmental Response, Compensation, and Liability Act, the Clean Water Act, the Safe Drinking Water Act, or the Clean Air Act (ITRC, 2018). Several bills are active in Congress to address PFAS—including directing federal agencies to regulate PFAS under the aforementioned laws—and some were included in the Defense Authorization Act, which became law in 2019 (GovTrack, 2019).

In the absence of federal action, states have begun setting drinking water standards and clean up levels to limit exposures to PFAS chemicals (ITRC, 2018). As of the date of this report, there are 23 adopted laws in 11 states pertaining to limits, bans, monitoring, or reporting of PFAS chemicals, and an additional 18 states are considering regulations (Safer States, 2020b). California DTSC has proposed PFAS in carpets as a priority product under their Safer Consumer Products program. This law asks manufacturers to consider alternatives to chemicals of concern in their products and may lead to regulation (DTSC, 2019b).

In Washington, the sale and use of PFAS-containing firefighting foam is restricted, and PFAS in firefighting personal protective equipment must be disclosed to purchasers (Chapter 70.75A RCW[42]). The law provides some limited exemptions from the purchase restrictions for petroleum refineries, oil terminals, chemical plants, and facilities required to use PFAS foams by federal regulations.

In 2018, Washington adopted a restriction (Chapter 70.95G RCW[43]) that will prohibit the use of PFAS in paper or paperboard food packaging if Ecology determines that safer alternatives exist. At the time of this report, Ecology is working to identify whether safer alternatives to PFAS in food packaging are available through the completion of an alternatives assessment.

Health is supporting the State Board of Health (SBOH) in developing state drinking water standards for PFAAs. In 2019, Health published draft recommendations for state action levels for five PFAAs in drinking water: PFOS (15 ppt), PFOA (10 ppt), PFNA (14 ppt), PFHxS (70 ppt), and PFBS (1,300 ppt). The SBOH expects to complete the rulemaking process in 2021 (Ecology, 2019a).

Because PFAS are halogenated organic compounds, they can be regulated under the Washington state Dangerous Waste Regulations (Chapter 173-303 WAC[44]). Halogenated organic compounds are considered dangerous waste when present at or above 100 ppm (0.01 –

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[42] https://app.leg.wa.gov/RCW/default.aspx?cite=70.75A&full=true
[43] https://app.leg.wa.gov/RCW/default.aspx?cite=70.95G&full=true
[44] https://apps.leg.wa.gov/WAC/default.aspx?cite=173-303

1.0%) in a waste stream (code WP02) and extremely hazardous waste when present above 10,000 ppm (1%) in a waste stream (code WP01).

Existing recommendations: PFAS in carpet were identified as a priority in our draft Chemical Action Plan[45] on PFAS.

Availability of safer alternatives

There are a number of ways to meet the function of stain and dirt resistance in carpet. This can be achieved by using PFAS chemistries, non-PFAS "drop in" alternatives, as well as fibers that are inherently stain resistant. Non-PFAS "drop in" solutions include siloxane polymers, polyurethanes, sulfonation, and silicate clay-based repellent. Inherently stain resistant fibers include wool, polypropylene, polyethylene terephthalate, and polytrimethylene terephthalate.

Several alternatives have been investigated by other agencies, such as DTSC (DTSC, 2018a). Manufacturers such as Interface® have stopped production of PFAS-containing carpets (Interface, 2019), and vendors such as Home Depot® and Lowe's® have phased out sale of PFAS-containing carpets (Home Depot, 2019). PFAS-free options and retailers' move to sell only PFAS-free carpets shows promise for the availability and feasibility of safer alternatives. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

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[45] https://fortress.wa.gov/ecy/publications/summarypages/1804005.html

Chapter 5: Leather and Textile Furnishings

Overview

Priority product

Indoor and outdoor leather and textile furnishings used in residential and commercial settings include:

- Table linens.
- Bedding.
- Upholstered furniture.
- Cushions and pillows.
- Curtains, drapes, and awnings.
- Towels.

Carpet is not considered a furnishing in this report. Please see the carpet and rugs chapter for information on carpet.

Priority chemical

Per- and polyfluoroalkyl substances (PFAS).

PFAS represent a large class of chemicals that are highly persistent and can be bioaccumulative and toxic. See the carpet and rugs chapter for more information about human health concerns and impacts of environmental release from PFAS.

Priority product summary

Leather and textile furnishings that have been treated during the manufacturing process for stain, oil, and water resistance are significant sources and uses of PFAS. We made this determination by considering the criteria in RCW 70.365.030.[46] These products contribute to the amounts of PFAS in our homes, workplaces and environment, and have the potential to expose infants, young children, and women of childbearing age.

In 2009, EPA estimated that leather and textile furnishings were one of the most important sources of PFAS in our homes. The PFAS in these materials can also be released into the environment. In 2018, Governor Inslee convened the Southern Resident Orca Task Force to recommend actions to recover the southern resident orca population. The Task Force named PFAS as chemicals of emerging concern for orcas. Detailed support for our listing of PFAS in furnishings as a priority product is shown below.

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[46] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Background

PFAS are applied to a wide variety of leather and textile products to provide stain, oil, and water resistance, both during the manufacturing process and as aftermarket treatments. This priority product focuses on pretreated leather and textile products that have had PFAS applied during the manufacturing process.

The PFAS used to pretreat leather and textiles has changed over time. Side-chain fluorinated polymers are the most common PFAS used in leather and textile furnishings (KEMI, 2015). Side-chain fluorinated polymers have a non-fluorinated carbon backbone connected to short-chain PFAS side-chains such as FTOHs. As treated textiles degrade under normal wear and tear and laundering, the fluorinated side-chains (e.g., FTOHs) can be released. Eventually, FTOHs degrade to PFCAs, which are environmentally persistent. People can inhale or ingest FTOHs and PFCAs. They are the most commonly analyzed PFAS in textiles, and they have often been detected in significant quantities.

Estimated volume of PFAS used in leather and textile furnishings

PFAS are found in a wide range of textile and leather furnishings and contribute to PFAS in the dust and air of our homes and workplaces. PFAS concentrations in leather and textile furnishings vary depending on the specific compound (Table 8). Total organic fluorine estimates are much higher than the estimates of individual types or groups of PFAS. This indicates that we are not typically analyzing for all the specific PFAS present in these products.

In 2015, the Swedish Chemical Agency estimated that textiles can contain 2 – 3% side-chain fluorinated polymers by weight (KEMI, 2015). Multiple studies have found high concentrations of PFCAs and FTOHs in leather and textile furnishings (Table 8).

Based on data from 14 treated home textiles (excluding carpet), with an average total PFCA (C5 – C12) concentration of 336 ng/g, EPA (2009b) estimated that indoor textiles contribute 1.68 mg of PFAS to most American homes. Vestergren et al. (2015) estimated that imported textiles (including furnishings and apparel) contribute 6.59 mg PFOA, 2,130 mg 6:2 FTOH, and 197 mg 8:2 FTOH per person per year, based on Norwegian import data. While these studies present different estimations, both indicate that furnishings contribute PFAS to our homes.

Table 8. Concentrations of PFAS detected in furnishings.

(**Bold concentrations** are used in the volume calculations below.)

PFAS analyzed	Concentration	Product types detected	References
Side-chain fluorinated polymers	**20,000,000 – 30,000,000 ng/g (2 – 3%)**	Leather and textile products.	KEMI, 2015
Total organic fluorine	**246,000 – 933,500 µg/m^2**	Table cloths.	Borg & Ivarsson, 2017
Total PFCA (C5 – C12)	**7 – 1021 ng/g**	Table cloths, mattress pads, microfiber fabric, upholstery (fabric), slip covers, cotton throw.	EPA 2009b; Liu et al., 2015
Total PFCA (C4 – C14)	0.006 – 3.8 µg/m^2	Curtains, sofa pillow, quilts, bed linen, fleece blanket, chair cover, pillow case, towel.	Vestergren et al., 2015
Total PFCA (C4 – C14)	50 – 467 ng/g	Outdoor upholstery, leather and outdoor awning fabric.	Janousek et al., 2019; Kotthoff et al., 2015
Total FTOH	**377 – 42900 ng/g**	Mattress pads, pillows.	Liu et al., 2015
Total FTOH	0.55 – 374 µg/m^2	Table cloths, upholstery (fabric and leather), pillow cases, pillows, towel, curtain, blankets, slip covers, quilts, table cloth, and bed sheet.	Hertzke et al., 2012; Vestergren et al., 2015

Estimated volume of leather and textile furnishings in Washington

We estimate that 15,500 metric tons of treated textile furnishings are found in Washington homes. This is based on the EPA (2009b) estimation that there are 5 kg of treated leather and textile furnishings (excluding carpet) in the average home. According to the U.S. Census, there were 3.1 million households in Washington in 2018. If each household has 5 kg of treated textiles, we estimate that a total of 15,500 metric tons (approximately 34 million pounds) of treated textiles are in Washington homes. This contributes a significant volume of PFAS to homes across Washington. Using the data in Table 8, we estimate that the 15,500 metric tons of leather and textile furnishings contributes 310 – 465 metric tons (approximately 680,000 – 1 million pounds) of side-chain fluorinated polymers, 0.1 – 16 kg total PFCA (C5 – C12), and 6 – 665 kg FTOHs to homes across Washington.

We estimate that up to 1,800 metric tons of PFAS-treated furniture end up in Washington's landfills each year. Our draft Chemical Action Plan[47] on PFAS estimated that 59,842 metric tons (approximately 131 million pounds) of furniture and 167,357 metric tons (approximately 368 million pounds) of other textiles are disposed of each year (Ecology, 2019a). Together, these two sources comprised 5% of Washington's waste (Ecology, 2019a). Not all furnishings are treated with PFAS. In other product categories, such as children's textiles, 3 – 5% are estimated to be treated with PFAS (DEPA, 2015a).

If we assume 3% of furniture is treated, it follows that almost 1,800 metric tons (approximately 3.9 million pounds) of PFAS-treated furniture are disposed of each year in Washington. Similarly, if we assume 3% of textile waste has been treated, about 5,000 metric tons (approximately 11 million pounds) of PFAS-treated textiles may be discarded each year in Washington. Leather and textile furnishings represent a portion of the 5,000 metric tons of PFAS-treated textiles that we estimate are discarded each year.

Potential for exposure to sensitive populations

Nearly all people, including women of childbearing age, infants, and young children, are exposed to PFAS. The National Health and Nutrition Examination Survey routinely detects several PFAS in the blood of nearly all participants (CDC-NHANES, 2015, 2017). See the carpet and rugs chapter for more information about human exposure to PFAS.

People, including infants, young children, and people of childbearing age, are exposed to PFAS from leather and textile furnishings. PFAS from textiles can be released into indoor air and accumulate in dust (Schlummer et al., 2013; Yao et al., 2018). Human exposure to PFAS occurs when people inhale and ingest the contaminated air and dust.

The presence of PFAS in dust from buildings without carpet shows the contribution from other products (Zeng et al., 2020). Children spend more time on or near the floor and have relatively high respiration rates and frequent hand-to-mouth activity. As such, they are exposed to more contaminated air, carpet, and house dust relative to their body weight compared to older poeple. Karaskova et al. (2016), Tian et al. (2016), Shoeib et al. (2011) and Trudel et al. (2008) have found that house dust is an important PFAS exposure route for toddlers.

Babies and children under age 3, who often put objects in their mouths, can ingest PFAS when mouthing textile furnishings, such as table cloths or upholstered furniture. In a 2013 study by the Danish Environmental Protection Agency, PFCAs were found to migrate from textiles into artificial saliva, with the saliva collecting 1% of the concentration found in the textile (DEPA, 2013).

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[47] https://fortress.wa.gov/ecy/publications/SummaryPages/1804005.html

Leather and textile furnishings have been associated with higher exposure to PFAS. Women living in homes with treated carpet or upholstery had higher exposure to PFNA and PFDeA (Boronow et al., 2019). Among 11 sampled indoor air environments, shops that sell textiles (carpet and curtains) had the highest concentrations of PFAS (Yao et al., 2018). Retail shops selling outdoor clothing had among the highest concentrations of PFAS measured in ten indoor environments (Schlummer et al., 2013). It is likely that shops selling outdoor textile furnishings also have high concentrations of PFAS because they can be treated similarly to outdoor clothing.

Treated textiles make a significant contribution to exposure. Based on a survey of the concentration of PFAS in 116 consumer items, EPA concluded that upholstery and other textiles were likely significant sources of exposures to PFCAs (EPA, 2009b). California Department of Toxic Substances Control (DTSC) (2016) identified upholstered furniture, along with carpet and its treatment products, as the largest potential source of significant and widespread PFAS exposures (DTSC, 2016).

Potential environmental exposure and exposure to sensitive species

PFAS are contaminants in Washington's environment. Because PFAS are environmentally persistent, they are typically found in most media sampled. In Washington, PFAAs have been detected in drinking water, surface waters, groundwater, wastewater effluent, freshwater, sediments, and wildlife. See the carpet and rugs chapter for more information about PFAA detection in Washington environmental monitoring studies.

PFAS from leather and textile furnishings can contribute to environmental PFAS concentrations. The persistence and solubility of PFAS in the environment heighten concerns about environmental releases after disposal (Ecology, 2019a). PFAS can be released from landfills into air, groundwater, and as leachate (Hamid & Grace, 2018; Lang et al., 2016, Lang et al., 2017; Wang et al., 2020). If textile furnishings are laundered or subjected to upholstery cleaning machine treatment, PFAS can be released into wastewater (Schellenberger et al., 2019). The Danish Environmental Protection Agency found that an average of 12% of PFCAs in textiles are released during a single wash (DEPA, 2013). Further, PFAS in house dust is tracked into the outdoor environment by foot (Wild et al., 2015).

Outdoor furnishings can degrade and release PFAS directly into the environment. In a study that pumped air through a chamber at a rate of one m^3 every three hours, FTOHs were emitted from some outdoor textiles (clothing) at a rate of 494 ng/hour (Schlummer et al., 2013). This demonstrates the potential for outdoor furnishings to release PFAS directly into the environment.

Exposure to PFAS in the environment is a concern for sensitive species. PFAS are environmentally persistent and some are bioaccumulative (Ecology, 2019a). Two PFAS, PDFA and PFOS, have been detected in salmon in Puget Sound (Meador, Yeh, Young, & Gallagher, 2016). Governor Inslee's Southern Resident Orca Task Force identified PFAS as chemicals of emerging concern.

Existing regulations

PFAS are not currently regulated under established environmental laws including the Resource Conservation and Recovery Act, the Comprehensive Environmental Response, Compensation, and Liability Act, the Clean Water Act, the Safe Drinking Water Act, or the Clean Air Act (ITRC, 2018). Several bills are active in Congress to address PFAS—including directing federal agencies to regulate PFAS under the aforementioned laws—and some were included in the Defense Authorization Act, which became law in 2019 (GovTrack, 2019). The PFAS regulatory landscape is rapidly changing (EPA, 2020).

In the absence of federal action, states have begun setting drinking water levels and clean up levels to limit exposures to PFAS chemicals (ITRC, 2018). California DTSC has proposed PFAS in carpets, treatments for carpets, and other textiles as a priority product under their Safer Consumer Products program. This law asks manufacturers to consider alternatives to chemicals of concern in their products and may lead to regulation (DTSC, 2019). Find more information about state regulations to limit PFAS exposures in the carpet and rugs chapter.

Existing recommendations: PFAS in textiles were identified as a priority in our draft Chemical Action Plan on PFAS (Ecology, 2019a).

Availability of safer alternatives

There are a number of ways to meet the function of stain, oil, and water resistance in textiles and furnishings (Zahid et al., 2019). This can be achieved by using PFAS chemistries, non-PFAS "drop in" alternatives, as well as fibers that are inherently stain resistant. Non-PFAS "drop in" solutions include siloxane polymers, polyurethanes, sulfonation, and silicate clay-based repellent. Inherently stain resistant fibers include wool, polypropylene, polyethylene terephthalate, and polytrimethylene terephthalate.

Several alternatives have been investigated by other agencies, such as the Danish Environmental Protection Agency (DEPA, 2015b) and California DTSC (DTSC, 2018). Businesses such as IKEA® have phased out sales of PFAS-containing textiles (IKEA, 2016), yet they still sell water-resistant outdoor furniture. The Healthier Hospitals program maintains a list of PFAS-free furniture that meets its Healthier Interior goal, and individual health care systems such as Kaiser Permanente have their own policies for PFAS-free furniture (Healthier Hospitals, 2016). The availability of PFAS-free furnishings shows promise for identifying feasible, safer alternatives. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

Chapter 6: Aftermarket Stain and Water Resistance Treatments

Overview

Priority product

Aftermarket stain and water resistance treatments for textile and leather consumer products used in residential and commercial settings and in vehicles.

Treatments may be used on a variety of products by consumers or commercial applicators after their purchase, including carpets, rugs, furniture, home textiles, apparel, and shoes. This does not include products marketed or sold exclusively for use at industrial facilities during the process of carpet, rug, clothing, shoe, or furniture manufacturing.

Priority chemical

Per- and polyfluoroalkyl substances (PFAS).

PFAS represent a large class of chemicals that are highly persistent and can be bioaccumulative and toxic. See the carpet and rugs chapter for more information about human health concerns and impacts of environmental release from PFAS.

Priority product summary

Aftermarket stain and water resistance treatments are applied to textile and leather products in residential and commercial settings and are significant sources and uses of PFAS. We made this determination by considering the criteria in RCW 70.365.030.[48] Stain and water resistance treatments contribute to the amounts of PFAS in our homes, workplaces and environment, and have the potential to expose infants, young children and workers. In 2009, EPA estimated that carpet and carpet care products were the second largest use of some PFAS, with the broadly defined "textiles" category ranking first.

We estimate that 2,300 metric tons (approximately 5 million pounds) of stain and water resistance treatments are used on carpet alone each year in Washington. Additional uses of stain and water resistance treatments include furniture, home textiles, apparel, and shoes. People can be exposed to PFAS while applying stain and water resistance treatments and as the product wears off over time. Since children and infants spend more time on or near the floor, they are disproportionately exposed to PFAS that accumulate in dust. Epidemiological studies show that children living in homes with treated carpet and furniture have higher exposures to PFAS.

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[48] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

The PFAS from stain and water resistance treatments can also be released into the environment. Spills and improper disposal of stain and water resistance treatments can release PFAS directly into environmental media. These PFAS are environmentally persistent and distribute widely in air and water, making clean-up difficult or impossible in some cases. In 2018, Governor Inslee convened the Southern Resident Orca Task Force to recommend actions to recover the southern orca population. The Task Force named PFAS as chemicals of emerging concern for orcas. Detailed support for our listing of PFAS in stain and water resistance treatments as a priority product is shown below.

Background

PFAS have been used in stain and water resistance treatments since the 1950s. They are applied to textile and leather products in homes and workplaces by the general public and professionals to prevent staining, enhance water resistance, and reduce wear and tear. PFAS can be applied during the manufacturing process for pretreated products. Consumers can also purchase treatments to apply themselves. This priority product focuses on treatments applied post-purchase and does not include treatments used by manufacturers.

The specific PFAS used in stain and water resistance treatments have changed over time. From 1970 to 2002, the largest use of PFOS-derived substances was for carpet treatments (48,000 tons globally from 1970 to 2002) (DEPA, 2013; Paul, Jones, & Sweetman, 2009). From the 1950s to the early 2000s, PFOS-based fluorochemicals were used in stain and water resistance treatments, and in 2003, manufacturers moved to PFBS-based chemistries. According to safety data sheets, carpet treatments contain fluorochemicals at concentrations between 3 and 7% (3M, 2018a), and fabric treatments can contain fluorochemicals at concentrations up to 3% (3M, 2018b).

There is no publicly available information on what exact fluorochemicals manufacturers use. However, EPA (2009b), Herzke (2012), and Kothoff et al. (2015) detected PFCAs and FTOHs in impregnation and nanosprays, including carpet treatments (Table 9). Fluorotelomer-based side-chain fluorinated polymers degrade into PFCAs and FTOHs in the environment (Washington & Jenkins, 2015).

Estimated volume of PFAS used in stain and water resistance treatments

Multiple PFAS have been detected in stain and water resistance treatments at concentrations that can add significantly to the concentration of PFAS in our homes. Example concentrations of PFAS in aftermarket treatments are shown in Table 10. Concentrations of PFAS in carpet following aftermarket treatment are shown in Table 11. EPA (2009b) estimated that there are about one kg of household carpet care products in the home. One kg of residential carpet treatments could contribute 953 µg PFCAs to the home, based on the concentration reported in Table 10. If we use the information from safety datasheets that report 3 – 7% fluorochemicals (3M, 2018a), one kg of carpet treatments could contribute between 30,000 and 70,000 mg of fluorochemicals to the average home.

Higher concentrations of PFAS have been found in commercial carpet care products (12,000 µg/kg total PFCA (C5-C12)), relative to household carpet care products (EPA, 2009b). EPA (2009b) estimated that six kg of commercial carpet care products are used during one treatment of a home, suggesting that commercial aftermarket treatments may contribute 72,000 µg PFCAs to the home per treatment. Some manufacturers recommend reapplying these products to carpet every two years.

A case study from 2012 reported high concentrations of PFAS in carpet in a household that regularly used aftermarket carpet treatments (Table 10) (Beesoon, Genuis, Benskin, & Martin, 2012). While the need for reapplication implies degradation over time, it is unlikely that the product wears off completely. Reapplications could lead to increasing concentrations of PFAS in carpet over time.

In addition to carpet, stain and water resistance products can be applied to other household items, such as furniture, textiles, apparel, and shoes. Fabric protectors can contain fluorochemicals at concentrations up to 3% (3M, 2018b). Some manufacturers recommend reapplying furniture treatments every six months.

Table 9. Reported concentrations of PFAS in stain and water resistance treatments.

Ecology estimated the volume in Washington based on the concentration and the product volume estimation described below.

Product	Concentration	PFAS measured	Estimated volume of PFAS in WA per year	Reference
Water proofing agents	29,889 µg/L	Total PFCA	69 Kg	Herzke, Olsson, & Posner, 2012
Household carpet/fabric-care liquids and foams	953 µg/kg	Total PFCA (C5 – C12)	2.2 Kg	EPA, 2009b
Water proofing agents	464,774 µg/L	FTOH/FTS	1076 Kg	Herzke, Olsson, & Posner, 2012
Impregnating sprays (waterproofing)	1,857,300 µg/kg	FTOH/FTS	4307 Kg	Kotthoff, Muller, Jurling, Schlummer, & Fiedler, 2015

Table 10. Concentrations of PFAS in carpet following aftermarket treatment.

Product	Concentration	PFAS measured	Reference
Carpet	112 – 1170 µg/kg	PFOS	Beesoon et al., 2012
Carpet	8 – 153 µg/kg	PFOA	Beesoon et al., 2012
Carpet	12 – 2880 µg/kg	PFHxS	Beesoon et al., 2012

Estimated volume used in Washington

We estimate that up to 2,300 metric tons (approximately 5 million pounds) of aftermarket carpet treatments are used and disposed of in Washington each year. These treatments could contribute 69 kg PFCA and 4307 kg FTOH/FTS to Washington's homes and environment. If we use the 3 – 7% fluorochemicals reported in safety data sheets (3M, 2018), the total fluorochemicals used in Washington may be much higher—between 69 and 161 metric tons (approximately 150,000 – 350,000 pounds)**.**

We based our estimate on two approximations. First, there are 17 million square yards of carpet sold yearly in Washington (based on California Carpet Stewardship Program, adjusted for the population of Washington). Second, there is an aftermarket treatment rate of 1,500 sq ft/gallon every two years after the first five years. We assume carpet care liquids have the same density as water (3M, 2018). To treat the carpet sold annually in Washington, we estimate that 102,000 gallons of treatment would be needed. Since the lifespan of carpet is estimated to be 15 years, we considered carpet purchased over a 15 year period and treated every two years after a five year period. It is unclear what percentage of the carpet in Washington is retreated every two years. Our estimations follow manufacturers' recommendations.

Fabric protectors can be applied to a wide variety of consumer products, such as furniture. Manufacturer directions state that ten ounces can cover one sofa. The U.S. Census reports that there were 3.1 million households in Washington in 2018. If each household in Washington treats their couch twice per year, 1,860 metric tons (approximately 4.1 million pounds) of stain protectors could be used in Washington each year. Even if only 10% of Washingtonians treated their couch twice per year, 186 metric tons (approximately 400,000 pounds) of stain protectors could be used. Manufacturers recommend reapplying to furniture every six months and reapplying to apparel after each wash or dry clean. The need for reapplication after washing is concerning because it implies PFAS is washed off and released into our wastewater systems.

Potential for exposure to sensitive populations

Nearly all people, including infants, children, people of childbearing age, and workers, are exposed to PFAS. The National Health and Nutrition Examination Survey (NHANES) routinely detects several PFAS in the blood of nearly all participants (CDC-NHANES, 2015, 2017b). See the carpet and rugs chapter for more information about human exposure to PFAS.

People, including infants, children, people of childbearing age, and workers are exposed to PFAS from stain and water resistance treatments. People can be exposed to PFAS from stain and water resistance treatment during application, before the product dries, and as it degrades over time. During application, dermal contact and inhalation may occur. When applied to carpet, the exposure pathway during product degradation is similar for aftermarket treatment and pretreated carpet.

However, PFAS in pretreated carpet are polymerized, whereas aftermarket treatments are applied as liquids, making them more likely to migrate from the carpet into dust and air and increasing the need for reapplication. As carpet treatment wears off, PFAS can be released into indoor air and accumulate in dust. Beesoon et al. (2012) reported high blood concentrations of PFHxS, PFOS and PFOA in a family that had their carpet commercially treated approximately every two years for 15 years. PFAS were also detected in their carpet, house dust and indoor air. Homes and offices with carpet can have higher concentrations of various PFAS compared to non-carpeted facilities (Fraser et al. 2013; Gewurtz et al., 2009; Kubwabo, Stewart, Zhu, & Marro, 2005). Karaskova et al. (2016) found that the combined concentration of 20 PFAS on carpeted floors was higher than other floor types.

PFAS from stain and water resistance treatments is applied to consumer products and accumulates in house dust. Because children spend more time on or near the floor, and have relatively high respiration rates and increased hand-to-mouth activity, they have higher exposure to contaminated air, carpet and house dust. Karaskova et al. (2016), Shoeib et al. (2011), Tian et al. (2016), and Trudel et al. (2008) have found that house dust is an important PFAS exposure route for toddlers. Washburn et al. (2005) estimated that the reasonable maximum exposure scenario for PFOS in carpet was two orders of magnitude higher for infants than adults, meaning infants could be exposed to PFAS at a level that is approximately 100 times higher than adults.

Children with carpets in their bedrooms have been found to have higher concentrations of PFOS, PFHxS, and Me-PFOSA-AcOH in their bodies than those with other flooring types (Harris et al., 2017). Boronow et al. (2019) found that women living in homes with treated carpet or upholstery had higher exposure to PFAS, PFNA and PFDeA. Fraser et al. (2012) found that office workers in buildings with higher concentrations of FTOH in the air had higher concentrations of PFOA in their blood.

Apparel can also be treated to increase water resistance. Since manufacturers recommend retreating after every wash, it’s likely that the PFAS used in the product degrade relatively quickly. One study noted that children who wear water-resistant apparel more frequently have higher exposures to PFAS (Wu et al., 2015).

Stain and water resistance treatments make significant contributions to exposure. Based on a survey of the concentration of PFAS in 116 consumer articles, EPA concluded that carpet and carpet care liquids were likely the most significant sources of exposures to PFCAs out of the 13 consumer product categories studied (EPA, 2009b). Trudel et al. (2008) found high levels of PFOA and PFOS in carpet, and determined that treated carpet was the most prominent exposure pathway from consumer products. They estimated that between 5 and 64% of PFOS exposure was related to contact with treated carpet, and concluded that exposure to these chemicals is high for infants, toddlers, and children due to their close contact with carpets (Trudel et al., 2008).

Potential environmental exposure and exposure to sensitive species

PFAS have contaminated Washington's environment. In Washington, PFAAs have been detected in drinking water, surface waters, groundwater, wastewater effluent, freshwater, sediments, and wildlife. See the carpet and rugs chapter for more information about PFAA detection in Washington environmental monitoring studies.

PFAS from stain and water resistance treatments can contribute to environmental PFAS concentrations. PFAS in stain and water resistance treatments can be released into the environment if the product spills or if discarded products are improperly disposed. In the volume estimation section of this chapter, we estimate that up to 2,300 metric tons (approximately 5 million pounds) of aftermarket carpet treatment are used each year. Similarly, 1,860 metric tons (approximately 4.1 million pounds) of fabric protectors could be applied to furniture each year in Washington. Since manufacturers recommend retreating carpets every two years and furniture every six months, we would anticipate the volume applied to furniture would be released from the carpet and furniture into our homes, workplaces, schools, daycares, and ultimately the environment.

Stain and water resistance for mill-treated carpets and rugs lasts approximately five years. Aftermarket treatments do not last as long, and some manufacturers recommend retreating carpet every two years. This recommendation suggests that PFAS in aftermarket treatments may migrate from the carpet more quickly than PFAS in mill-treated carpets.

PFAS from stain and water resistance products applied to consumer products can accumulate in house dust (Beesoon et al., 2008; Kubwabo et al., 2005), and can be tracked into the outdoor environment (Wild et al., 2015) or released into wastewater through the laundering process (Shoeib et al., 2011).

When disposed in landfills, treated carpet can contribute to environmental release as the treatment degrades (Lang, Allred, Peaslee, Field, & Barlaz, 2016). PFAS can be released from landfills into air, groundwater, and as leachate (Hamid & Grace, 2018). The volume of carpet and potential PFAS releases from its disposal are further discussed in the related chapter for pretreated carpet. It addition to the disposal of treated products, there is also potential for partially-used containers of stain and water resistance products to be spilled or improperly disposed.

Exposure to PFAS in the environment is a concern for sensitive species. PFAS are environmentally persistent and some are bioaccumulative (Ecology, 2019a). Environmental monitoring has detected two PFAS (PDFA and PFOS) in salmon in Puget Sound (Meador, Yeh, Young, & Gallagher, 2016). Governor Inslee's Southern Resident Orca Task Force identified PFAS as chemicals of emerging concern.

Existing regulations

PFAS are not currently regulated under established environmental laws, including the Resource Conservation and Recovery Act, the Comprehensive Environmental Response, Compensation, and Liability Act, the Clean Water Act, the Safe Drinking Water Act, and the Clean Air Act (ITRC, 2018). Several bills are active in Congress to address PFAS—including directing federal agencies to regulate PFAS under the aforementioned laws—and some were included in the Defense Authorization Act, which became law in 2019 (GovTrack, 2019).

In the absence of federal action, states have begun setting drinking water levels and clean up levels to limit exposures to PFAS chemicals (ITRC, 2018). California DTSC has proposed PFAS in treatments for textile and leather products as a priority product under their Safer Consumer Products program (DTSC, 2019a). This law asks manufacturers to consider alternatives to chemicals of concern in their products and may lead to regulation (DTSC, 2019b). Find more information about state regulations to limit PFAS exposures in the carpet and rugs chapter.

Existing recommendations: Ecology identified PFAS in aftermarket carpet treatments as a priority in our draft Chemical Action Plan[49] on PFAS (Ecology, 2019a).

Availability of safer alternatives

There are a number ways to meet the function of stain and dirt resistance in textile and leather treatments. Manufacturers achieve this using PFAS chemistries, non-PFAS drop-in alternatives, and by choosing alternative materials that do not require aftermarket stain treatment due to their natural stain resistance. Non-PFAS chemical solutions on the market include silicone dioxide (ProtectME, 2019), and proprietary anionic non-fluorinated polymers (Bridgepoint Systems, 2019; Tri-Plex Technical Services, Ltd., 2019).

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[49] https://fortress.wa.gov/ecy/publications/documents/1804002.pdf

Inherently stain resistant fibers include wool, polypropylene, polyethylene terephthalate, and polytrimethylene terephthalate. Other agencies, including DTSC, have investigated several alternatives (DTSC, 2019a). Alternate materials and PFAS-free chemical alternatives show promise for the availability and feasibility of safer alternatives. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

Chapter 7: Food and Drink Cans

Overview

Priority product

Food and drink cans.

Priority chemical

Phenolic compounds—Bisphenols.

Phenolic compounds, such as bisphenols, are found in consumer products. Exposure to bisphenols is associated with endocrine disruption, which can lead to obesity. Two bisphenols within this class, bisphenol A and bisphenol S, are listed as chemicals of high concern to children (Chapter 70.240 RCW[50]).

Priority product summary

Can linings are a significant source and use of phenolic compounds. We made this determination after considering the criteria in RCW 70.365.030.[51] Metal can linings contribute to bisphenol concentrations in humans and the environment. Approximately 2.5 billion cans are sold each year in Washington, and testing shows that a large proportion of those may contain bisphenol-based can liners.

Dietary exposure to bisphenol A (BPA) is the largest source of exposure to this chemical, and consuming canned food leads to higher biological levels. This is particularly concerning for children and infants, who have the largest exposure to BPA, and who may be more affected by these chemicals due to their sensitive stage in development.

Bisphenols from can linings may also enter the environment. Governor Inslee's Southern Resident Orca Task Force has declared them a chemical of emerging concern for our Puget Sound orca population. Detailed support for our listing of bisphenols in can linings as a priority product is shown below.

Background

Bisphenols, specifically BPA, are commonly used in epoxy resin-based can linings. These linings prevent reaction of the food or beverage with the metal can, maintaining food taste and structural integrity of the can. Resins containing BPA are composed of approximately 85% BPA by weight. The process for forming BPA resins is shown in Figure 3.

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[50] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240
[51] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

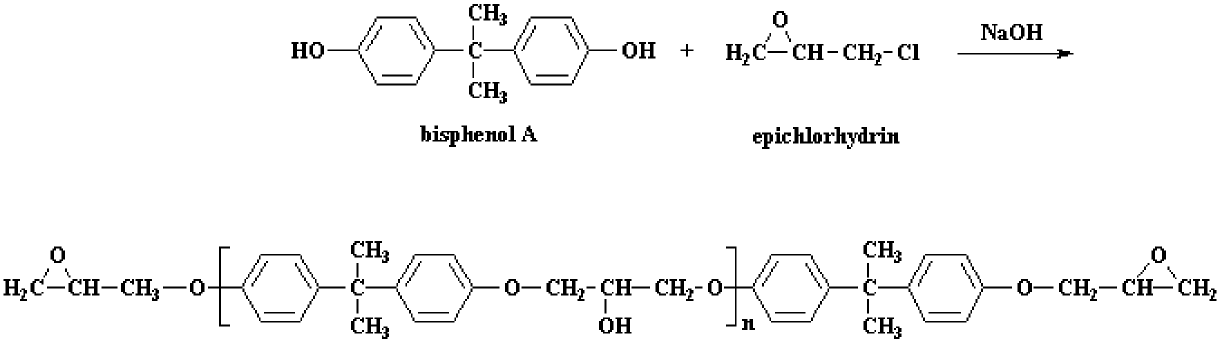

Figure 3. The creation of BPA-based epoxy resin from BPA and epichlorhydrin (PSLC, 2019).

Resins may also contain curing agents, which help improve mechanical, chemical, and heat-resistance properties. The addition of heat or chemicals called accelerators may improve the curing process. Other sealants and polymers may also be used to line cans, and the chemicals used depend on what is contained in the can.

Besides epoxy resins—which are used in many applications other than can linings—bisphenols are used in other applications such as polycarbonate and other plastic production, thermal paper, and as flame retardants. Polycarbonate production is the largest use of bisphenols, but a phase-out of BPA in most food contact materials (excluding food and drink cans) means that this is not anticipated to be a major source of human exposure (EFSA, 2013).

Estimated volume of bisphenols used in can linings

Bisphenols are widely present in can linings and canned food. When studying can linings, researchers often analyze the can contents in order to identify the components of the lining as well as the leaching rate of the lining into its contents. Thus, bisphenol levels in food prior to canning may contribute to testing results, and could provide false positives as to the contents of can linings. However, un-canned food has consistently low levels of BPA—mostly under the limit of detection—suggesting that false positives would be rare (Cao et al., 2011).

Below is a summary of studies investigating bisphenol use in canned food (Table 11). In 2013, EU migration limits for BPA in food contact material were lowered from 600 ng/g to 50 ng/g, so while many of the values below are above the current requirement, they were in compliance when sampled. These results show widespread and well-documented use of bisphenol-based can linings, and that these linings may contaminate the contents of the cans.

Table 11. Summary of studies investigating bisphenol prevalence and concentration in canned food and beverages.

Year	Chemical tested	Prevalence	Concentration	Reference
2004	BPA	25/79 (32%)	11 – 191 ng/g in food	Thomson & Grounds, 2005
2006	BPA	55/97 (57%)	Not Disclosed	Environmental Working Group, 2007
2009	BPA	69/69 (100%)	0.032 – 4.5 µg/L in soft drink	Cao, Corriveau, & Popovic, 2009
2010	BPA	19/22 (86%)	0.32 – 106 ng/g in food	Cao et al., 2011
2010	BPA	17/17 (100%)	0.019 – 0.54 µg/L (ppb) in soft drinks and beer	Bureau of Chemical Safety, 2010
2010	BPA	63/105 (60%)	0.23 – 65.0 ng/g in food	Schecter et al., 2010
2010	BPA	46/50 (92%)	0.7 – 1140 ng/g in food and drink	Chase Wilding et al., 2010
2011	BPA	71/78 (91%)	2.6 – 730 ng/g in food	Noonan, Ackerman, & Begley, 2011
2013	BPA	27/31 (87%)	0.01 – 146 ng/g in food	Liao & Kannan, 2013
2013	BPAF, BPAP, BPB, BPF, BPP, BPS, BPZ	10.5%, 11.2%, 2.62%, 10.1%, 3.37%, 20.9%, 2.25% (food—not all canned)	All with 95th percentile under 0.74 ng/g, suggesting inadvertent contamination. Max of 1130 ng/g BPF found in mustard (which has natural BPF).	Liao & Kannan, 2013
2015	BPA	26/37 (70%)	0.24 – 149.0 ng/g in food	Lorber, Schecter, Shropshire, & Christensen, 2015
2017	BPA	96/252 (32%) of linings	25 – 140 ng/g in food (4 samples)	Cox, 2017

An article published in 2015 stated that “according to coatings specialists, roughly 80%” of epoxy coatings used in can linings are BPA-based (Waldman, 2015). This is supported by a statement from the North American Metal Packaging Alliance which estimated that “75% of canned foods sold in the U.S. are lined with a BPA-based polymer” (Environmental Working Group, 2015).

Another article published in 2018 quoted an internal Can Manufacturers Institute survey which found that at least 90% of can linings do not use BPA-based linings. This could suggest that the industry may be phasing out BPA use (McTigue Pierce, 2018). However, testing data shows that any phase-out of BPA in can linings is incomplete, and other bisphenols besides BPA have not been extensively analyzed, leaving open the possibility of their use—either currently or as a substitute in the future.

Estimated volume used in Washington

Approximately 2.5 billion cans are sold each year in Washington. As shown above, testing results, and even statements from industry, vary widely as to the percentage of cans containing bisphenol-based can liners. According to the 2018 – 2019 Can Manufacturers Institute annual report, about 125 billion cans are produced per year in the U.S. and Canada (CMI, 2019). Taking Washington's population proportion of that would give 2.5 billion cans sold per year in Washington. Depending on which industry group's statement is used (North American Metal Packaging Alliance or Can Manufacturers Institute), BPA-lined cans could consist of as low as 10% or as much as 75% of this figure. This would amount to between 250 million and 1.9 billion BPA-lined cans sold in Washington per year, or 34 – 253 BPA-lined cans per person. (This estimate does not include the alternate bisphenols besides BPA.)

Potential for exposure to sensitive populations when used

Nearly all people are exposed to bisphenols. Findings in the Fourth National Report on Human Exposure to Environmental Chemicals indicate widespread exposure to BPA. CDC scientists found BPA in more than 90% of the urine samples representative of the U.S. population. NHANES data from 2003 to 2014 show a decreasing trend in the urinary BPA concentration for the general U.S. population (CDC, 2019a). Because BPA does not persist for long periods of time in the body, its widespread detection in people indicates that exposures occur frequently. In 2011, the World Health Organization (WHO) estimated the mean dietary daily intake of BPA for adults to be 0.4 – 1.4 µg per kg of body weight (WHO, 2011).

The levels of BPA in humans have changed over time, likely due to their replacement in products. Besides BPA, other bisphenols, such as BPS and BPF, have also been detected in the urine of the U.S. general population, according to NHANES 2013 – 2014 survey. However, there is no evidence that these bisphenols are used in can linings. Exposure to other bisphenols (e.g., BPS) appears to have increased in the U.S. population due to substitution (Ye et al., 2015). However, their levels appear to be lower compared to BPA, and the detection frequency is higher for BPA than other bisphenols (95.7% for BPA, 89.4% for BPS, and 66.5% for BPF) (Lehmler, Liu, Gadogbe, & Bao, 2018).

Due to inaccuracies in historical testing methods, most BPA levels in organisms are likely severely underreported. One study found levels in urine 19 times higher using direct analysis (looking for BPA and metabolites themselves) versus the indirect analysis (looking only for BPA after transforming metabolites enzymatically) used by federal agencies and in the NHANES data set (Gerona, vom Saal, & Hunt, 2019). This has potentially affected risk calculations by suggesting that population exposure levels are much lower than they are in reality.

People can be exposed to bisphenols used in can linings. General population exposure to BPA may occur through the ingestion of foods in contact with BPA-containing materials. According to the FDA, when foods are in direct contact with any packaging material, small but measurable amounts of the packaging materials can migrate into food and can be consumed with it (FDA, 2014).

The FDA is working with the National Toxicology Program (NTP) to answer questions about BPA migration into food from food contact materials. According to a recent study, migration from packaging into beverages and drinking water is as significant as migration to solid food (Russo et al., 2019). Water has been shown to absorb BPA from liners during heating. For example, low levels of BPA were detected in water from all unheated cans, rising from 0.06 to 32 ng/cm^2 after heating at 100 degrees Celsius (Takao, Lee, Kohra, & Arizono, 2002). Another study found BPA from lined cans leaches into water held at 121 degrees Celsius to a much higher extent than water held at 80 degrees Celsius (Sajiki et al., 2007).

Some studies show elevated migration levels of BPA from canned foods. A significant increase in levels of BPA was observed in tomatoes produced in Italy during canning. This testing also showed that the heating process during canning or damage by denting may increase BPA migration levels (Errico et al., 2014). Most migration of BPA (80 – 100%) seems to occur during the can processing step. Storage (up to nine months) did not alter the levels of BPA migration (Goodson, Robin, Summerfield, & Cooper, 2004).

In another study, BPA was detected in packaged meat, but bisphenol metabolites were not. If animals were exposed to BPA before slaughter, one would expect to see evidence of their metabolism. Absence of metabolites (the end-products of the chemical after it goes through metabolism in the body) shows that BPA in these foods was the result of contact with a BPA-containing material, and not from BPA present in the product beforehand (Deceuninck et al., 2019).

Can linings make a significant contribution to exposure to bisphenols. The European Food Safety Authority (EFSA) indicates that eating food and/or drinking water that contains BPA is the largest source of BPA exposure (EFSA, 2015). Consumption of canned food, including some specific types, such as canned vegetables and fruit, canned pasta, and canned soup, is correlated with higher urinary BPA concentrations (Hartle, Navas-Acien, & Lawrence, 2016). In Europe, the consumption of canned meat and vegetables has been identified as the largest source of dietary intake of BPA (Russo, Barbato, Mita, & Grumetto, 2019).

Carwile (2011) characterized exposure and biological processing of dietary BPA in a group of healthy adult humans. The study found that on a controlled diet, canned food items are likely to be significant dietary sources of BPA. Consumption of one serving of canned soup daily over five days was associated with a more than 1000% increase in urinary BPA over soup prepared without canned ingredients (Carwile, 2011). In addition, a study found that urinary BPA concentrations were more than 16 times as high after drinking canned beverages versus drinking glass-bottled beverages (Bae & Hong, 2015).

Exposure to bisphenols in can linings is particularly relevant for infants, children, and certain demographic groups. Children and infants are exposed to bisphenols at a time when their development is especially susceptible (Mao et al., 2020). Due to reduced metabolism of bisphenols in fetuses compared to mothers, fetal exposure through cord blood is 2 – 7 times as high as in mothers (Zhang et al., 2020). By looking at source concentration and use estimations, EFSA scientists calculated projected exposure for different demographics. This analysis found dietary exposure to BPA is highest among children aged six months to ten years (explained by their higher food consumption on a body weight basis), with estimated BPA dietary intake of up to 0.875 µg per kg of body weight per day (EFSA, 2013, 2015).

Concentration of BPA in breastmilk, a possible exposure route to infants, has been correlated with drinking canned beverages (Tateoka, 2015). Women of childbearing age had dietary exposures comparable to men of the same age (up to 0.388 µg per kg of body weight per day) (EFSA, 2015). Bisphenols have been found to transfer between mothers and the womb, with BPA and BPS having similar placental transfer efficiencies and significant correlation between maternal plasma and cord plasma (Pan et al., 2020).

Certain demographic groups may eat more canned food, and thus have higher potential exposure to bisphenols. USDA calculated that in 2004, African Americans were the highest consumers, per person, of canned vegitables. In addition, it was found that people from the southern U.S. spend more on canned vegetables than other regions, while people from the Midwest spend the most on canned fruit. The elderly (over 64) spent the most of any age group (USDA, 2008). Those who receive food assistance through the Supplemental Nutrition Assistance Program for Women, Infant and Children consume an average of 1.3 times as much canned food as the general population (Laatsz, 2012). These findings have important health equity and environmental justice implications.

Potential environmental exposure and exposure to sensitive species

Bisphenols have been found in Washington's environment and elsewhere. Bisphenols are present throughout Washington in various environmental media. The Lower Columbia River Estuary Partnership found BPA in the Columbia river in 2004 (LCREP, 2007). As part of the Puget Sound Toxics Loading Study,[52] Ecology found BPA in WWTP biosolids (Ecology, 2010a), in WWTP effluent (Ecology, 2010b), and in stormwater, with higher levels during rain events (Ecology,

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[52] https://fortress.wa.gov/ecy/publications/documents/1103010.pdf

2011a). BPA was found in Puget Sound at much higher levels than in the relatively uninhabited Barkley Sound, British Columbia, indicating human contributions to environmental concentrations (Keil, Salemme, Forrest, Neibauer, & Logsdon, 2011).

Outside of Washington, BPA is frequently detected, sometimes at concentrations that exceed levels defined as protective of human and environmental health by governmental agencies (Corrales et al., 2015). Other bisphenols are also commonly found in environmental media, such as sediment, water, soil, dust, and throughout the WWTP process (Chen et al., 2016; Hu, Zhu, Yan, Liao, & Jiang, 2019).

Bisphenols from can linings can contribute to environmental bisphenol concentrations. A 2008 risk assessment by the EU found that epoxy resin production is one of the largest sources of BPA release to bodies of water (Aschberger et al., 2008). Other sources include thermal paper recycling and production, PVC use and production, and leaching from polycarbonate bottles. In addition to production of the resin, during the recycling process, cans will be thoroughly washed and water likely released to WWTPs or the environment. Although bisphenols are often removed at greater than 90% efficiency by WWTPs, due to the large volumes of wastewater discharged by recycling facilities, this is still a major source of environmental release (PPRC, 2015, not peer reviewed).

Sewage sludge used to create biosolids for agriculture has the potential to contribute BPA to soil, groundwater, and surface water. (Michałowicz, 2014). Discarded cans also have the potential to contribute bisphenols to the environment. High levels of BPA have been detected in landfill leachate, and groundwater has been contaminated with BPA near waste dumps (Michałowicz, 2014).

Exposure to bisphenols in the environment is a concern for sensitive species. Bisphenols can have a variety of detrimental effects on wildlife (Cioci, Apfelbacher, Strong, & Innes, 2015). There is evidence that BPA is slightly bioaccumulative in some species (Corrales et al., 2015). BPA has been shown to cause embryonic deformities, abnormal behavior, growth inhibition in fish, and reduce the number of offspring in fish (Kang, Aasi, & Katayama, 2007). A wide variety of bisphenols have shown endocrine disrupting, cytotoxic, genotoxic, reprotoxic, and neurotoxic effects (Chen et al., 2016). Growth effects have been shown in fish exposed to water with concentrations as low as 0.078 µg/L (Kang, Aasi, & Katayama, 2007).

While BPA is the most studied bisphenol, there is evidence that other compounds have similar, if not worse, effects on wildlife. For example, BPF, BPS, BPAF, BPB, and BPC have shown estrogenic activity similar to BPA (Chen et al., 2016), and can cause stress effects at low levels in a variety of organisms (Zhou, 2018). Bisphenols were deemed a chemical of emerging concern for the endangered Puget Sound orca population (Southern Resident Orca Task Force, 2018).

Existing regulations

There are no federal regulations pertaining to bisphenols in can linings. In the U.S., there is a state-driven effort to limit exposure to BPA. As of the date of this report, there are 30 adopted laws in 14 states pertaining to limits, bans, monitoring, or reporting of BPA in products, and an additional seven states are considering regulatory actions (Safer States, 2020a). Of the 30 adopted laws, six laws in six states directly ban BPA in infant formula cans and child food containers. California DTSC has listed BPA in food packaging as a proposed priority product under its workplan (DTSC, 2019c). In Washington, BPA is banned from children's food and beverage containers under RCW 70.280.020(1)[53] (Safer States, 2020a).

BPA has been banned from infant feeding bottles across the EU since 2011. It is also banned in other materials that come into contact with food intended for infants and children under three years. France has banned BPA in all food packaging, containers and utensils (ECHA, 2019). In 2018, the European Commission reduced the specific migration limit (SML) of BPA from plastics, coatings and varnishes for metals and other contact sources to food from 0.6 mg/kg to 0.05 mg/kg. The reduced SML coincided with a ban on BPA in plastic bottles and packaging intended for babies and children (ECHA, 2019; PackagingLaw, 2018). The European Chemicals Agency (ECHA) has officially declared that exposure to BPA poses a serious human health concern (ECHA, 2019).

In 2010, EPA proposed rulemaking under the Toxic Substances Control Act (TSCA) to identify BPA as "a substance that may present an unreasonable risk of injury to the environment on the basis of its potential for long-term adverse effects on growth, reproduction and development in aquatic species at concentrations similar to those found in the environment." This proposed rule has been stalled in the Office of Management and Budget for several years (EPA, 2010a).

Availability of safer alternatives

There are a number of ways to store preserved food and beverages, including metal cans with can linings and containers made of different materials. Can linings may be comprised of bisphenols as well as alternative substances such as oleoresins, acrylics, or PET plastic. Other food packaging options include glass, lined cardboard, and PET plastic containers.

Several alternatives have been investigated by NGOs, such as the National Resources Defense Council (Singla, 2016), the Berkeley Center for Green Chemistry (Berkeley, 2016), and the Food Packaging Forum (Geueke, 2016). The availability of bisphenol-free food storage options suggests that safer alternatives are available. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

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[53] https://app.leg.wa.gov/RCW/default.aspx?cite=70.280.020

Chapter 8: Thermal Paper

Overview

Priority product

Thermal paper: Paper coated with a material formulated to change color when exposed to heat.

Priority chemical

Phenolic compounds—Bisphenols.

See the food and drink cans chapter for more information about human health concerns and impacts of environmental release from bisphenols.

Priority product summary

Thermal paper is a significant source and use of phenolic compounds. We made this determination by considering the criteria in RCW 70.365.030.[54] Bisphenol-containing thermal paper contributes to the amounts of bisphenols in our bodies. Approximately 3,300 tons of thermal paper are used every year in Washington, and tests show that much of it contains certain bisphenol derivatives (BPS- or BPA-based chemicals) that function as developers. The European Food Safety Authority (EFSA) estimates that thermal paper use is one of the leading sources of human exposure to BPA.

Multiple industries use thermal paper every day. People absorb the chemicals on thermal paper, significantly raising the concentration of bisphenols in their bodies. Retail workers are especially susceptible to this pathway due to the use of thermal paper for receipt tape. They have much higher exposure than the general population and consistently higher internal levels.

Thermal paper also contributes to bisphenol concentrations in the environment. Recycling of thermal paper is the largest source of BPA to the environment. Governor Inslee's Southern Resident Orca Task Force deemed bisphenols a chemical of emerging concern for our orca population. Detailed support for our listing of bisphenols in thermal paper as a priority product is shown below.

Background

Bisphenols, such as bisphenol A (BPA) and bisphenol S (BPS) (molecular structures below), are utilized as developers in the chemical reaction that provides color when using thermal paper. Bisphenols are organic acid solids which melt when exposed to sufficient heat. When melted, bisphenols will combine with dyes and alter the dye's pH, causing the dye to change color.

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[54] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Other components such as sensitizers and stabilizers can help with performance and reliability of this developer-dye reaction. All of these components are mixed into a thermally-reactive layer and are applied to a wide range of base papers as a coating.

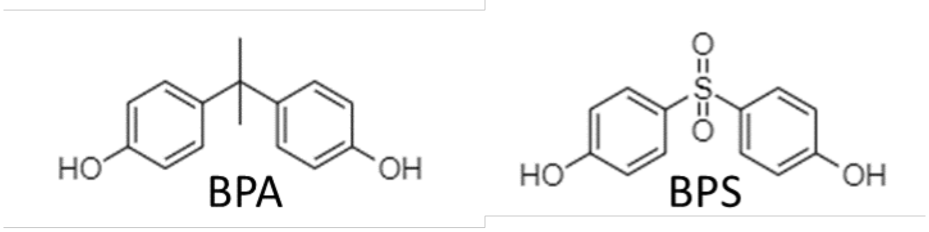

Figure 4. Molecular structures of bisphenol A (BPA) and bisphenol S (BPS).

Besides thermal paper, bisphenols are used in many applications such as epoxy resins, polycarbonate and other plastic production, and as flame retardants. Polycarbonate production is the largest use of bisphenols, but the phase-out of BPA use in most food contact materials means they are not anticipated to be a major source of human exposure (EFSA, 2013).

Estimated volume of bisphenols used in thermal paper

Bisphenols are frequently detected in thermal paper at high concentrations. The following is a summary of bisphenol concentrations found in thermal paper where, for the most part, the chemical is assumed to have been added intentionally (Table 12). Other results not included consist of detections at much lower concentrations (usually less than 100 ppm), which we assume result from inadvertent contamination during manufacturing (such as due to contaminated recycled paper). Bisphenols found in commercially-available thermal paper include BPA and BPS, though other bisphenols have been proposed as alternatives (EPA, 2015a; Pelch et al., 2019).

In the U.S., BPA has been detected in thermal paper at levels as high as 28,000 ppm (2.8%), and BPS at 71,000 ppm (7.1%) (Lunder, Andrews, & Houlihan, 2010). In general, it seems that some thermal paper manufacturers have recently switched from BPA developers to BPS and other alternatives, such as Pergafast 201 and acetic acid, though bisphenols still make up the majority of the developers in receipts (Table 13). These results show the widespread and well-documented use of bisphenols as thermal paper developers.

Table 12. Summary of studies investigating bisphenol concentrations found in thermal paper.

Year	Country	Chemical	Concentration	Reference
2010	U.S.	BPA	300 – 15,400 ppm	Mendum, Stoler, Vanbenschoten, & Warner, 2011

Year	Country	Chemical	Concentration	Reference
2010	U.S. and Japan	BPA	8,000 – 28,000 ppm	Lunder, Andrews, & Houlihan, 2010
2011	Worldwide	BPA	<LOQ – 13,900 ppm (including inadvertent)	Liao & Kannan, 2011
2012	Belgium	BPA	9,000 – 21,000 ppm	Geens, Goeyens, Kannan, Neels, & Covaci, 2012
2012	Worldwide	BPS	0.014 – 22,000 ppm (including inadvertent)	Liao, Liu, & Kannan, 2012
2013	China	BPA	2,580 – 14,700 ppm	Lu, Chang, Sojinu, & Ni, 2013
2014	US	BPA	54 – 79 µg/cm^3	Apfelbachter, Cioci, & Strong, 2014
2014	US	BPS	37 – 75 µg/cm^3	Apfelbachter, Cioci, & Strong, 2014
2015	Brazil	BPA+BPS	1,000 – 43,000 ppm	Rocha, Azevedo, Gallimberti, Campiglia, & Barbosa, 2015
2015 – 2017	Germany	BPA	4,000 – 32,400 ppm	Eckardt & Simat, 2017
2015 – 2017	Germany	BPS	5,000 – 18,000 ppm	Eckardt & Simat, 2017
2017	India	BPA	300 – 6,600 ppm	Rajankar, Mohapatra, & Mathur, 2018
2018	US	BPA	14,500 ppm (78.6 µg/cm^3)	Zaharias, Miller & Olson, 2018
2018	US	BPS	71,000 ppm (311 µg/cm^3)	Zaharias, Miller & Olson, 2018
2019	Worldwide	BPA	<LOQ – 20,270 ppm (including inadvertent)	Molina-Molina et al., 2019
2019	Worldwide	BPS	<LOQ – 13,290 ppm (including inadvertent)	Molina-Molina et al., 2019

Table 13. Summary of studies investigating bisphenol prevalence in thermal paper.

Year	Country	Chemical	Prevalence in thermal paper tested	Reference
2010	U.S.	BPA	8/10 (70%)	Mendum, Stoler, Vanbenschoten, & Warner, 2011
2010	U.S. and Japan	BPA	12/39 (31%)	Lunder et al., 2010
2011	Worldwide	BPA	97/103 (94%) (including inadvertent)	Liao & Kannan, 2011

Year	Country	Chemical	Prevalence in thermal paper tested	Reference
2012	Belgium	BPA	32/44 (73%)	Geens, Goeyens, Kannan, Neels, & Covaci, 2012
2012	Worldwide	BPS	111/111 (100%) (including inadvertent)	Liao, Liu, & Kannan, 2012
2013	China	BPA	42/42 (100%)	Lu, Chang, Sojinu, & Ni, 2013
2014	US	BPA	9/18 (50%)	Apfelbachter, Cioci, & Strong, 2014
2014	US	BPS	9/18 (50%)	Apfelbachter, Cioci, & Strong, 2014
2015	Brazil	BPA+BPS	186/190 (98%)	Rocha, Azevedo, Gallimberti, Campiglia, & Barbosa, 2015
2015 – 2017	Germany	BPA	46.9% – 52.5%	Eckardt & Simat, 2017
2015 – 2017	Germany	BPS	6.1% – 11.4%	Eckardt & Simat, 2017
2016	US	BPA	5/103 (5%)	Lucia, 2016
2016	US	BPS	98/103 (95%)	Lucia, 2016
2017	India	BPA	12/12 (100%)	Rajankar, Mohapatra, & Mathur, 2018
2018	US	BPA	30/167 (18%)	Zaharias, Miller & Olson, 2018
2018	US	BPS	126/167 (75%)	Zaharias, Miller & Olson, 2018
2019	Worldwide	BPA	85/112 (75.9%)	Molina-Molina et al., 2019
2019	Worldwide	BPS	14/112 (12.5%)	Molina-Molina et al., 2019
2019	Worldwide	BPF	0%	Molina-Molina et al., 2019

Estimated volume used in Washington

Approximately 3,300 tons of thermal paper are used per year in Washington. Minnesota Pollution Control Agency estimated in 2015 that national thermal paper use was about 146,000 imperial tons annually (Cioci, Apfelbacher, Strong, & Innes, 2015). Taking the Washington population share of that would give 3,300 tons (6.6 million pounds) of thermal receipt paper used per year.

An EU risk assessment estimated that 1,860 metric tons (approximately 4.1 million pounds) of BPA were used per year for thermal paper in the EU during 2005 – 2006 (Aschberger et al., 2008). Taking the ratio of populations (Washington versus EU) and extrapolating to current time would give an estimated usage of 29 metric tons (nearly 64,000 pounds) of BPA in Washington from thermal paper. This does not take into account BPS use or changes in thermal paper components since 2006.

Potential for exposure to sensitive populations when used

Nearly all people are exposed to bisphenols. Findings in the Fourth National Report on Human Exposure to Environmental Chemicals indicate widespread exposure to BPA. CDC scientists found BPA in more than 90% of the urine samples representative of the U.S. population. NHANES data from 2003 to 2014 show a decreasing trend in the urinary BPA concentration for the general U.S. population (CDC, 2017b). Because BPA does not persist for long periods of time in the body, its widespread detection in people indicates that exposures occur frequently. In 2011, the World Health Organization (WHO) estimated the mean dietary daily intake of BPA for adults to be 0.4 – 1.4 µg per kg of body weight (Hines et al., 2017).

The levels of BPA in humans have changed over time, likely due to their replacement in products. Besides BPA, other bisphenols that may be used in thermal paper, such as BPS and BPF, have also been detected in the urine of the U.S. general population, according to NHANES 2013 – 2014 survey (Lehmler, 2018). Exposure to other bisphenols (e.g., BPS) appear to have increased in the U.S. population due to substitution (Ye et al., 2015). However, their levels appear to be lower compared to BPA, and the detection frequency is much higher for BPA than other bisphenols.

Due to inaccuracies in historical testing methods, most BPA levels in organisms are likely severely underreported. See the food and drink cans chapter for more information about human exposure to bisphenols.

Bisphenols in thermal paper make a significant contribution to exposure. Exposure to bisphenols through thermal paper can occur because bisphenols exist as free molecules in the coating layer. These molecules wear off and can easily be transferred to the skin or anything that touches the paper (Liao & Kannon, 2011). Bisphenols can then be absorbed through the skin or transferred from unwashed hands to food and ingested (Hormann et al., 2014).

Thermal paper is a significant source of BPA for the general population above three years of age (accounting for up to 15% of total exposure in some population groups) (EFSA, 2013). Although exposure is higher in adults from thermal paper, there is still potential for babies and toddlers to be exposed to BPA from thermal paper through recycling of receipts and other routes in addition to direct contact. EFSA (2013) indicates that thermal paper is the second largest source of external BPA exposure after eating food and/or drinking water containing it.

Exposure to BPA for five minutes from a single dermal contact, even when followed by hand washing, leads to detectable urinary levels in people. One study found that after dermal exposure, urinary BPA levels remained elevated for up to nine days, whereas after oral exposure they returned to normal after 10 – 20 hours. A greater proportion of the BPA was detected in urine after dermal exposure than after oral exposure (Liu & Martin, 2017). Thus, it seems likely that dermal exposure to BPA will lead to a higher proportion of BPA in systemic circulation compared to oral exposure.

Several studies have shown that briefly handling receipt papers leads to significant absorption into the body. One study found that over 88% of BPS exposure for most humans comes from handling thermal receipts (Liao et al., 2012). Another study found that BPA transfers readily from receipts to skin and can penetrate the skin to such a depth that it cannot be washed off. This study determined that absorption of BPA into the body increases as much as tenfold when thermal paper is handled with moist or greasy fingers (Biedermann, Tschudin, & Grob, 2010).

Exposure to bisphenols in thermal paper is particularly relevant for children and retail workers. Several studies have calculated estimated daily intake (EDI) for exposure to bisphenols from thermal paper. A study measured BPA in 44 thermal papers collected in Belgium. Exposure levels in the Belgian general population from thermal paper were 445 ng BPA per day (Geens et al., 2012). EFSA estimated that BPA exposure through thermal paper averaged 0.071 µg/kg/day, or 4.4 µg/day for an average 62 kg adult, with the highest exposure group being adolescents aged 10 – 18 years, who had exposure averaging 1.6 times higher than adults, at 0.113 µg/kg/day (EFSA, 2015).

BPA has been detected in household dust, which infants are disproportionately exposed to, suggesting another route of exposure that may be influenced by thermal paper use (Kubwabo et al., 2016). BPA and BPS are able to cross the placenta and can expose the fetus, having similar adverse outcomes to offspring (Mao et al., 2020; Pan et al., 2020). Due to reduced metabolism of bisphenols in fetuses compared to mothers, fetal exposure through cord blood is 2 – 7 times as high as in mothers (Zhang et al., 2020).

The exposure for people who regularly contact thermal paper, such as employees who handle paper receipts, can be much higher than for the general population. For instance, median intake was found at 1.42 µg/day of BPA and BPS in the Brazilian general population, while intake levels for people who were occupationally exposed was 71 µg/day (Rocha et al., 2015). In Shenzhen, China, EDIs were found at 0.69 µg/day for general population and 40.4 µg/day for workers at a supermarket (Lu et al., 2013). At an Italian market, exposures from thermal paper were 0.0625 µg/day for the general population, and 66.8 µg/day for occupationally exposed individuals (Russo, Barbato, & Grumetto, 2017).

The CDC reports that people working in retail industries have 30% more BPA in their bodies than the average U.S. adult (CDC, 2017b). On average, workers in the National Institute for Occupational Safety and Health (NIOSH) study had BPA levels in their urine about 70 times higher than adults in a NHANES general population study (Hines et al., 2017). Handling of thermal receipts has been shown to have a significant impact on urinary BPA levels, and also a correlation with 8-hydroxy-2′-deoxyguanosine, which is a measure of oxidative stress (Lv et al., 2017). NHANES data also showed that overall, workers with potential occupational exposure are more likely to have detectable levels of urinary BPA. Notably, females with potential occupational exposure had significantly higher urinary BPA excretion compared with females with unlikely occupational exposure. However, there was no statistically significant association between occupation and urinary BPA in males (Hehn, 2016).

In cashiers who handled bisphenol-containing receipts, bisphenol levels in urine were significantly higher than in non-cashiers. Post-shift levels (0.54 µg/g) of urinary BPS were significantly higher than pre-shift levels (0.23 µg/g) for the 32 cashiers who handled BPS-containing receipts (Thayer et al., 2016). Other studies found a significant increase in urinary BPA levels for cashiers handling thermal paper daily (Ndaw, Remy, Jargot, & Robert, 2016), and that levels remained elevated 48 hours after handling thermal receipts (Liu et al., 2017).

Potential environmental exposure and exposure to sensitive species when used

Bisphenols have been found in Washington's environment and elsewhere. Bisphenols are present throughout Washington in all forms of environmental media. Outside of Washington, BPA is frequently detected, sometimes at concentrations exceeding levels protective of human and environmental health as defined by governmental agencies (Corrales et al., 2015). See the food and drink cans chapter for more information about the presence of bisphenols in Washington's environment and elsewhere.

Bisphenols from thermal paper can contribute to environmental bisphenol concentrations. A 2008 risk assessment by the EU found that while thermal paper production is one of the smallest industrial uses of BPA, recycling of the paper contributes the largest industrial source of BPA entering the environment (Aschberger et al., 2008). Other sources include PVC use and production, epoxy resin production, and leaching from polycarbonate bottles.

Although BPA is often removed at greater than 90% efficiency by WWTPs, due to the large volumes of wastewater discharged by recycling facilities, this is still a major source of environmental release (PPRC, 2015, not peer reviewed). Sewage sludge used to create biosolids for agriculture has the potential to contribute BPA to soil, groundwater, and surface water. (Michałowicz, 2014). End-of-life thermal paper also has the potential to contribute bisphenols to the environment, either through landfill, recycling, or littering (Wang, Liu, & Liu, 2017). High levels of BPA have been detected in landfill leachate, and groundwater has been contaminated with BPA near landfills (Michałowicz, 2014).

Exposure to bisphenols in the environment are a concern for sensitive species. Bisphenols can have a variety of detrimental effects on wildlife (Cioci et al., 2015). There is evidence that BPA is slightly bioaccumulative in some species (Corrales et al., 2015). BPA has been shown to cause embryonic deformities, abnormal behavior, and growth inhibition, and can also reduce the number of offspring in fish (Kang, Aasi, & Katayama, 2007). Many bisphenols show endocrine disrupting, cytotoxic, genotoxic, reprotoxic, and neurotoxic effects (Chen et al., 2016). Growth effects have been shown in fish exposed to water with concentrations as low as 0.078 µg/L (Kang, Aasi, & Katayama, 2007).

While BPA is the most studied bisphenol, there is evidence that closely related compounds have similar, if not worse, effects on wildlife. For example, BPF, BPS, BPAF, BPB, and BPC have shown estrogenic activity similar to BPA (Chen et al., 2016), and can cause stress effects at low levels in a variety of organisms (Zhou, 2018). Bisphenols were deemed a chemical of concern for the endangered Puget Sound orca population (Southern Resident Orca Task Force, 2018).

Existing regulations

There are no U.S. federal regulations regarding BPA in thermal paper. In the US, there is a state-driven effort to limit exposures to BPA. As of the date of this report, there are 30 adopted laws in 14 states pertaining to limits, bans, monitoring, or reporting of BPA in products, and an additional seven states are considering regulatory actions (Safer States, 2020a). Of the 30 adopted laws, two pertain to paper. In 2011, Connecticut established the first state ban on BPA in thermal receipt paper. In 2019, Illinois prohibited the manufacture, distribution or use of BPA in making banking or business paper (Safer States, 2020a). California DTSC has proposed bisphenols in office machinery consumables as a priority product under their Safer Consumer Products program. This law asks manufacturers to consider alternatives to chemicals of concern in their products and may lead to regulation (DTSC, 2018c).

In 2010, EPA proposed rulemaking under the Toxic Substances Control Act (TSCA) to identify BPA as “a substance that may present an unreasonable risk of injury to the environment on the basis of its potential for long-term adverse effects on growth, reproduction and development in aquatic species at concentrations similar to those found in the environment.” This proposed rule has been stalled in the Office of Management and Budget for several years (EPA, 2010a).

BPA will be banned in thermal paper in the EU starting in 2020 (ECHA, 2016). BPS and BPA are banned in thermal paper in Switzerland (ChemicalWatch, 2019a).

Availability of safer alternatives

Several alternatives to bisphenol-containing thermal paper are available and in circulation. There are a number of automatic paper documentation options including thermal paper and printed paper. For thermal paper, developers include bisphenols, ascorbic acid, Pergafast 201, and urea compounds. Another thermal paper alternative is heat-voided paper. For some applications, no receipt or a digital receipt may be suitable alternatives. Alternatives have been investigated by other agencies, such as EPA (2014a), ECHA (2014a), and MPCA (Cioci et al., 2015).

Several retailers have policies in place prohibiting the use of bisphenols in thermal receipts (ChemicalWatch, 2019b). Some bisphenol-free alternatives may be safer. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of alternatives to bisphenols in thermal paper during Phase 3.

Chapter 9: Laundry Detergent

Overview

Priority product

Laundry detergent.

Priority chemical

Phenolic compounds—Alkylphenol ethoxylates.

Phenolic compounds, such as alkylphenol ethoxylates, are found in consumer products. Exposure to alkylphenol ethoxylates is associated with endocrine disruption, which can lead to obesity.

Some alkylphenol ethoxylates are persistent in the environment and can accumulate in wildlife and people (EPA, 2018b). These factors, combined with their aquatic toxicity, make environmental releases of alkylphenol ethoxylates particularly concerning.

Priority product summary

Laundry detergents are a significant source and use of phenolic compounds. We made this determination after considering the criteria in RCW 70.365.030.[55] Laundry detergents contribute to environmental concentrations of alkylphenol compounds. In 2015, California Department of Toxics Substances Control (DTSC) estimated that institutional cleaners—including laundry detergent—are the largest use of nonylphenol ethoxylates (NPEs).

We estimate that as much as two million pounds of laundry detergent potentially containing alkylphenol ethoxylates (APEs) are used in Washington each year. Laundry detergents containing APEs are disposed of down the drain and make their way through wastewater treatment plants to bodies of water. There, the alkylphenol compounds act as toxicants and can disturb the endocrine systems of aquatic life.

Governor Inslee's Southern Resident Orca Task Force identified alkylphenols as a chemical of emerging concern. Alkylphenols and alkylphenol ethoxylates have been heavily restricted in many other countries, leading to lower concentrations in those areas. Detailed support for our listing of APEs in laundry detergent as a priority product is shown below.

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[55] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Background

APEs are used as surfactants in laundry detergents to help clean clothing and linens. The surfactant reduces the surface tension of liquids, helping them spread and wet materials.

The chemical structure of APEs consists of an alkylated phenol ring with anywhere from one to over 70 ethoxylate groups in a chain (DTSC, 2018b). The most commonly found ethoxylates are nonylphenol ethoxylate (NPE) and octylphenol ethoxylate (OPE).

Manufacturers have largely phased out the production of residential laundry detergents containing APEs, but they are still present in some detergents marketed for commercial or industrial use, especially for use in hospitals, hotels, and nursing homes (also known as on-premises laundries). Large uniform providers, such as Cintas® and Aramark®, have phased out the use of APEs in their detergents. The majority of APE-containing laundry detergents use NPE as the surfactant, so NPE numbers are often used in place of total APEs when considering the class.

Estimated volume of APEs in laundry detergents

Laundry detergents are likely the largest use of APEs in commerce. The percentage of commercial detergents using APEs is unclear, but a market report contracted by California DTSC in 2015 estimated that “institutional cleaners” (including laundry detergents and other cleaning products) was the dominant use of NPEs globally at 39% of use by volume (Figure 4). Other products with less use than cleaners include leather and textiles, paint, oilfield chemicals (fracking fluid), and agrochemicals (pesticide adjuvants).

This is similar to a 2002 finding that 41% of household detergents tested contained NPEs, although residential use of NPEs in detergents has been phased out since that time (Cheng & Ding, 2002). Ecology was unable to find a large quantity of non-laundry detergent cleaning products available on the market containing APEs. This suggests that the “institutional cleaners” category described in DTSC’s market report consists mainly of laundry detergent, which is the most prevalent historical use. DTSC also estimated that cleaning products containing NPEs consisted of concentrations ranging from 10 – 100%, and that over 25% of manufacturers had detergents containing APEs (DTSC, 2018b). A search of laundry detergent safety data sheets (SDSs) found NPE ranges from 1 – 40%.

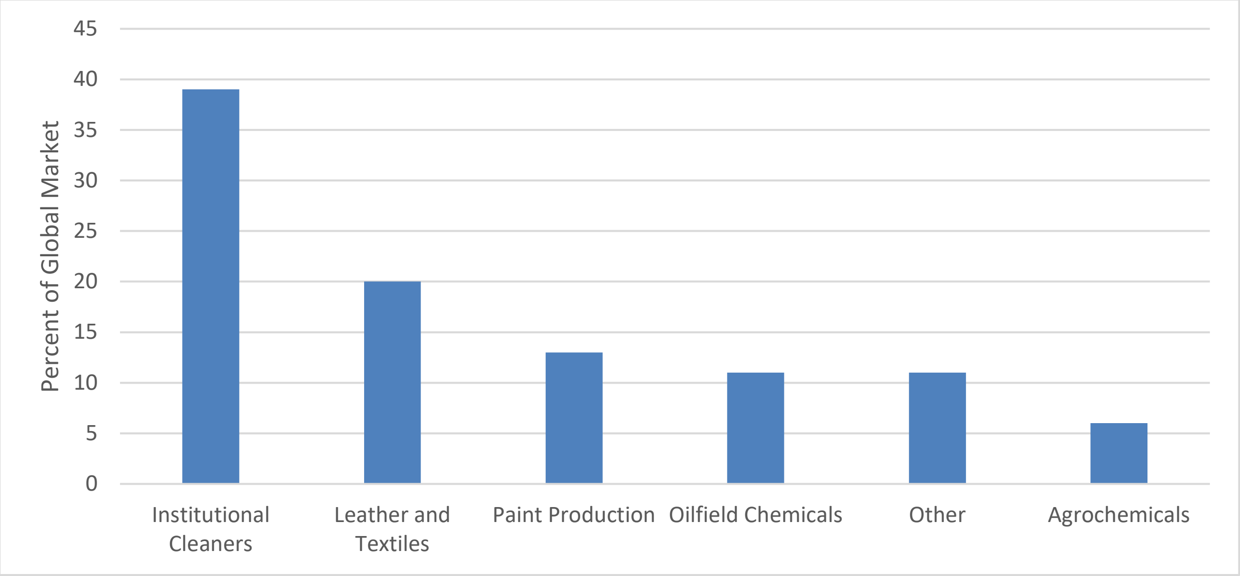

Figure 5. Market use (in %) of nonylphenol ethoxylates globally in 2015, as reported by California DTSC in 2018.

Estimated volume used in Washington

Approximately two million pounds of laundry detergent are used per year in Washington. California DTSC estimated that about two billion pounds of laundry are generated in on-premises launderers in California (DTSC, 2018b). Using a similar method, but substituting Washington numbers for days occupied at facilities, we estimate that on-premises launderers in Washington generate about 370 million pounds of laundry per year (Table 14).

DTSC estimated that eight fluid ounces of 20% NPE-containing liquid is used per 100 pounds of on-premises laundry (DTSC, 2018b). If we use these estimations per 100 pounds of laundry—and make the heaviest use assumption that all on-premises launderers use NPE-containing detergent—this would mean that about two million pounds of laundry detergent containing 370,000 pounds of NPEs would be discharged per year by Washington on-premises laundries.

This figure may be an underrepresentation, as it does not include other industrial type laundry facilities that may use APE-containing laundry detergent, such as cruise ships, cargo ships, and prison facilities. The amount that these facilities would contribute to APE use in Washington is unclear. The daily average prison population in Washington is approximately 37,000 (Jones, 2018), giving an additional 13.5 million room-nights of laundry when a full year is taken into account. This equates to 180 million pounds of laundry if using the same generation rate as hotel rooms. SDSs of laundry detergents marketed for use in detention facilities have been found to contain APEs (Garland C. Norris Company, 2015).

The EPA's Vessel General Permit electronic Notice of Intent database contains 2,102 ships greater than 79 feet in length that submitted a plan to dock in Washington and discharge graywater (which would include laundry wastewater) in 2019. This graywater may or may not be treated prior to release. The quantity of ships that actually entered Washington waters and how much graywater was discharged is unclear.

Table 14. Estimated amount of laundry generated by on-premises launderers in Washington.

Facility type	Units	Millions of units per year	Generation rate (pounds per unit)	% washed in on-premises launderers	Laundry generated (millions of pounds) per year
Hotels and motels	Occupied room nights	24.5	13.25	100%	325
Hospitals	Inpatient days	3.0	15	10%	5
Nursing facilities	Resident days	5.8	7.1	100%	41
Total					**370**

Potential for exposure to sensitive populations when used

Nearly all people, including infants and women of childbearing age, are exposed to APEs. Alkylphenols (AP) are environmental and biological metabolites of APEs. APs are more easily analyzed than APEs due to their varying ethoxylation chain lengths, and they are more persistent. Non-ethoxylated APs are used in much lower quantities than APEs in consumer products, so AP levels are often considered a surrogate of APE presence. In addition, APs have higher aquatic toxicity and endocrine-disrupting activity than APEs, so levels of APs are more relevant from a risk analysis perspective (DTSC, 2018b). For these reasons, APs are more often measured in studies than APEs, and can serve as an indicator of APE exposure and hazard (DTSC, 2018b).

Detection in of APs in urine indicates recent exposures. In a longitudinal study, 4-tert-octylphenol has not been detected in the urine of more than 50% the U.S. general population for more than three survey cycles (2005 – 2006, 2007 – 2008, and 2009 – 2010), so it was no longer reported after 2010. In 2009 – 2010, it was only detected in 5 – 10% of the population CDC, 2019a). This suggests its use in detergents in the U.S. was declining at least until 2010. 4-tert-octylphenol was also detected in the urine of adult Japanese volunteers, though the levels were near or below the detection limit (Inoue et al., 2003; Kawaguchi et al., 2004).

Nonylphenol (NP) has been detected in human breastmilk (Ademollo, Ferrara, Delise, Fabietti, & Funari, 2008), showing exposure in women of childbearing age and a pathway for exposure of newborns. It has also been found in maternal blood (Guenther et al., 2002; Li et al., 2013), umbilical cord blood (Chen et al., 2008; Guenther et al, 2002; Huang et al., 2014), urine, and placenta (Calafat et al., 2005). Both NP and NPEs have been detected in household dust, to which infants are disproportionately exposed (Kuwabo, 2016). In addition, fat tissue collected from surgeries in Italy between 2005 and 2007 contained NP concentrations that ranged from 10 – 226 ng/g (Ferrara et al., 2011).

In 2014, some phenolic compounds, including octylphenol, were measured in the urine of healthy Danish pregnant women. The levels varied significantly among them, suggesting that there were common exposures associated with a specific occupation or environment (Tefre de Renzy-Martin et al., 2014).

People can be exposed to APEs used in laundry detergent. Monitoring data indicate that the general population typically contacts NP via three exposure pathways:

1) Inhalation of ambient air (Rudel et al., 2003; WHO, 2004).

2) Ingestion of dust (Lu et al., 2013), food, and drinking water.

3) Dermal contact with consumer products containing NP (NCBI, n.d.).

People can also be exposed to emissions from the end-of-life phase of products containing low amounts of APEs via air, drinking water, and soil (Health Canada, 2001; EC, 2002; EPA, 2010b). Alkylphenol and alkylphenol ethoxylates are commonly detected in house and office dust (Abafe, 2017; Kuwabo et al., 2016). This indicates that indoor house or office dust may be one of the major routes for human exposure to these compounds. Because laundry detergent is the largest use of APEs, it is likely that use of APE-containing detergents is contributing to all of these exposure pathways.

Laundry detergent makes a significant contribution to APE exposure. The largest source of APE exposure to people is thought to be food, especially fish (CDC, 2019a; EC, 2008). APEs have been detected in wild fish tissue (Ecology, 2016b; Lv et al., 2019). For people who have higher fish consumption, such as indigenous populations (Ecology, 2013), this has important health equity and environmental justice implications.

NPEs in laundry detergent are not completely removed by WWTPs, and therefore may accumulate in fish living in receiving waters via contact with contaminated sediment and water.

Another route of APE exposure to humans may be from drinking water. NP has been detected in bottled water and tap water, and is not removed completely by some water treatment options (Mao et al., 2012). APEs and APs from laundry detergent can be found in WWTP effluent and biosolids that are applied to the land.

Exposure to APEs in laundry detergent is particularly relevant for workers. Workers handling laundry detergent have a higher potential for exposure compared to the general population

(DTSC, 2018b). Occupational exposure to APEs may occur through inhalation and dermal contact with this compound at workplaces where it is produced or used. Workers are exposed to APEs in laundries when they transfer chemicals into washers. These workers can be exposed to either powdered or liquid detergents. Powdered detergents have the greatest potential for inhalation exposure due to breathable dust containing APEs (DTSC, 2018b).

Potential environmental exposure and exposure to sensitive species

AP/APEs have been found in Washington's environment and elsewhere. APEs and APs are not naturally occurring, so any detection in the environment is a consequence of human activity. NP has been found in almost all environmental media in Washington. NP was only detected in 1% of samples from freshwater streams during baseflow and storm events as part of data gathering for Ecology's Puget Sound Toxics Loading Study[56] (Ecology, 2011a), though this is expected since the majority of environmental NP is coming from WWTPs. In two other separate phases of this study, NP was detected at up to 400 ng/L in WWTP influent, at up to 200 ng/L in WWTP effluent (Ecology, 2010b), and found in 6.3% of samples from WWTP discharges. The Puget Sound Toxics Loading Study also predicted that NPE-containing detergents could emit NP directly into the atmosphere (Mackenzie, Mcintyre, Howe, & Israel, 2018). This can lead to atmospheric deposition—the process by which particles from the atmosphere collect on solid surfaces.

A 2007 King County survey of endocrine-disrupting compounds found NP in marine water at levels ranging from at or below the limit of detection (≤ non-detect concentration, ND) up to 0.254 µg/L; in lakes at ND – 0.149 µg/L; in streams at ND – 0.836 µg/L; and in stormwater at up to 44.2 µg/L (KingCounty, 2007). A 2016 study by Ecology found OPE, NP, and NPEs in 48% of freshwater fish tissue, at levels from 445 – 4080 ng/kg (Gottschall et al., 2017).

Globally there has been widespread detection of APEs and APs in water, air, and sediment. Levels were found as high as 743 µg/L and 933 µg/L in surface water in China and 120 µg/L and 258 µg /L in U.S. wastewater (DTSC, 2018b; European Commission, 2002; Salomon et al., 2019; Selvaraj, Shanmugam, Sampath, Joakim Larsson, & Ramaswamy, 2014).

APEs from laundry detergents can contribute to environmental APE concentrations. Facilities that use high levels of APE-containing products have higher levels of APEs in their wastewater compared to facilities that do not use APE-containing products (Nagarnaik, Mills, & Boulanger, 2010). In addition, inadequately treated wastewater can release NP in the aquatic environment, further emphasizing that a large proportion of environmental NP comes from uses that are disposed of into wastewater systems, such as detergents (Mao et al., 2012).

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[56] https://fortress.wa.gov/ecy/publications/documents/1103010.pdf

NP concentrations are much lower in the EU than in Asia. This suggests that NP concentrations in the environment are in part due to NPE/NP use in products in Asian markets that are restricted in the EU, such as laundry detergents. NPE/NP and OPE/OP have been restricted for almost all uses in the EU, including laundry detergent and other cleaning products (Mao et al., 2012). In European WWTPs, NP has significantly decreased in both influent and effluent concentrations since the implementation of restrictions (Höhne & Püttmann, 2008).

During wastewater treatment, NP from laundry detergents can accumulate in biosolids (Mao et al., 2012). Biosolids can then be applied to land for agricultural purposes, which allows for another route of environmental exposure to APs. See the figure on the following page for a diagram of potential pathways for laundry detergents used in on-premises laundries to enter the environment.

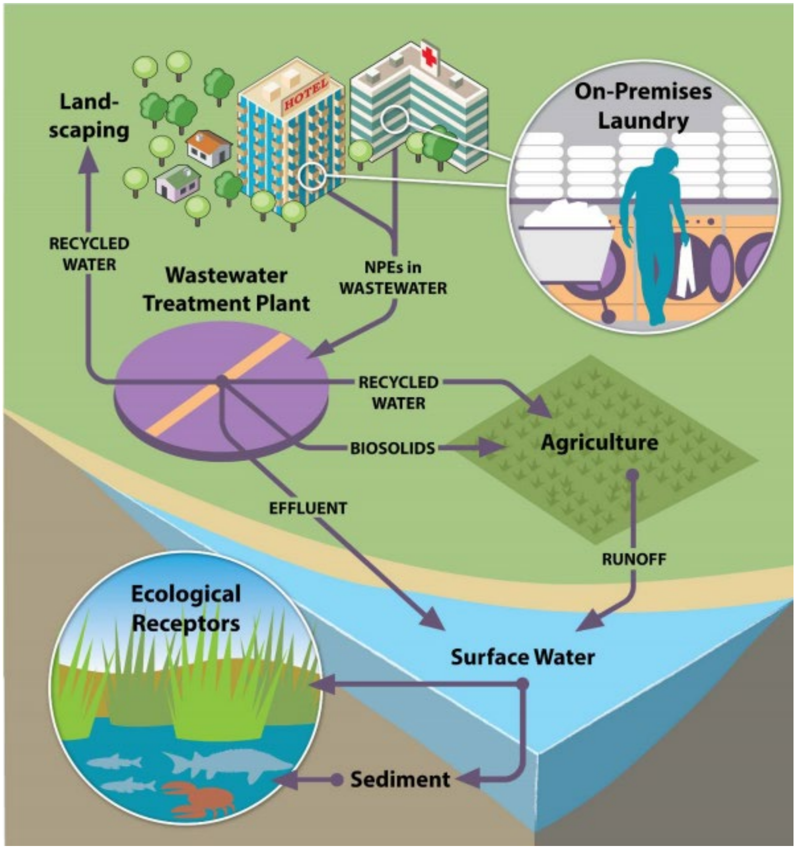

Figure 6. Potential exposure pathways for NPEs from laundry detergents (DTSC, 2018b).

(This figure does not include all possible exposure pathways for NPEs from consumer products to the environment.)

Exposure to APEs in the environment is a concern for sensitive species. Using EPA's standard evaluation procedure, APs and APEs are slightly to very highly toxic to freshwater fish and invertebrates—depending on the ethoxylate chain length. The ability of APEs to damage cell membranes increases with decreasing ethoxylation chain length. Consequently, non-ethoxylated NP and OP are the types most toxic to aquatic organisms (DTSC, 2018b).

In addition to toxicity, NP and OP have shown endocrine-disrupting properties (Celino-Brady, Petro-Sakuma, Breves, Lerner, & Seale, 2019), and therefore are restricted in the EU (ECHA, 2013). NP causes reproductive and growth impairment in a variety of aquatic organisms, and can be immunotoxic to Pacific oysters, even at low concentrations (DTSC, 2018b).

Another concern is the synergistic effect of APEs on toxicity with other compounds, specifically pesticides. This means that when combined with these compounds, such as in stormwater runoff or when applied in agriculture, the resulting mixture is more harmful to wildlife than the sum of the two chemicals separately (DTSC, 2018b).

Existing regulations

Currently there are no federal or state regulations specifically addressing APEs in laundry detergents. As of the date of this report, we are not aware of any pending legislative proposals at the state level related to APEs in detergents. In 2014, EPA proposed a Significant New Use Rule (SNUR) under the Toxic Substances Control Act (TSCA). The rule would require manufacturers to provide at least 90 days notice to EPA before commencing a significant new use or resuming a previous use of the 15 NP/NPEs that are no longer used in commerce. This would give EPA the opportunity to evaluate the intended use and, if warranted, take action to prohibit or limit the activity before it occurs (EPA, 2014b). EPA has also added a NP and NPE category to the Toxics Release Inventory (TRI) Program list of reportable chemicals (EPA, 2018).

The EU restricts NP and NPE via Annex XVII at greater than 0.1% in multiple products including cleaning products. Other APEs are on the European Substance of Very High Concern (SVHC) candidate list (ECHA, 2012), requiring authorization before use.

South Korea’s K-REACH program recently adopted restrictions similar to the European Commission’s to limit the intentional use of NPEs (at greater than or equal to 0.1% by weight) in various products, including domestic, industrial, and institutional cleaning products (DTSC, 2018b).

Availability of safer alternatives

There are a number of surfactants available for laundry detergents. Non-APE chemicals include alcohol ethoxylates, alkyl polyglucosides, and alkyl sulfate esters. Several agencies and NGOs have investigated these alternatives including BizNGO (2014), DTSC (2018), ECHA (2014b), and EPA (2012). The availability of non-APE surfactants suggest that safer alternatives may be available. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

Chapter 10: Vinyl Flooring

Overview

Priority product

Vinyl flooring.

Priority chemical

Phthalates: Synthetic chemical esters of phthalic acid.

Consumer products release phthalates into our homes, schools and workplaces as they degrade. Exposure to phthalates is associated with endocrine disruption, impaired reproduction and prenatal development, neurodevelopmental impacts, and potentially asthma (EPA, 2019f). Some phthalates are listed as chemicals of high concern to children (Chapter 70.240 RCW[57]). While phthalates are not environmentally persistent, their constant release into the environment has led to recontamination of clean-up sites.

Priority product summary

Vinyl flooring is a significant source and use of phthalates. We made this determination after considering the criteria in RCW 70.365.030.[58] We estimate vinyl flooring may contain phthalates at concentrations ranging from 9 – 32% by weight, contributing significant amounts of phthalates to our homes, workplaces, and environment.

While phthalate exposure can come from a variety of sources, the potential for infants and young children to be exposed to phthalates from vinyl flooring is especially concerning. Multiple studies report associations between vinyl flooring in the home and higher concentrations of urinary phthalate metabolites (the end-products of the chemical after it goes through metabolism in the body) in infants, children, and pregnant women.

Vinyl flooring has been associated with higher concentrations of phthalates in air and dust. As vinyl flooring ages, phthalates are released into the environment. In 2011, Ecology estimated that vinyl flooring may contribute 0.1 metric tons (220 pounds) of phthalates to Puget Sound each year. Governor Inslee’s Southern Resident Orca Task Force named phthalates an emerging chemical of concern for orcas and their food sources. Detailed support for our listing of phthalates in vinyl flooring as a priority product is shown below.

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[57] https://app.leg.wa.gov/RCW/default.aspx?cite=70.240

[58] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.030

Background

Phthalates are used in vinyl flooring, also known as polyvinyl chloride (PVC) flooring, to soften plastic and increase flexibility and durability. Historically, ortho-phthalates, such as DEHP and DINP, were the majority of the phthalates used in vinyl flooring. But over time, the industry has moved toward terephthalates. Recently, an NGO study supported retailers' claims that they have stopped selling flooring containing ortho-phthalates (Miller, Belliveau, Walsh, & Shade, 2019, not peer reviewed). During the public comment period for the draft version of this report, eleven flooring manufacturers supported a public comment from the Resilient Flooring and Covering Institute (RFCI) that stated that ortho-phthalates have been removed from the vast majority of flooring (RCFI, 2020).

However, we still don't know how much of the market has moved away from ortho-phthalates. Because children can be exposed to phthalates from vinyl flooring, we want to make sure that the market as a whole has moved away from using ortho-phthalates. We are currently verifying the extent of ortho-phthalate removal from new flooring sold in Washington. Since manufacturers have six months to comply with a data request, new data is not incorporated into this report.

Estimated volume of phthalates used in vinyl flooring

We estimate that over half of vinyl flooring may contain phthalates at concentrations ranging from 9 to 32% by weight. The volume of phthalates used in vinyl flooring has changed over time. In 2011, Washington state estimated that among polyvinyl chloride products, including flooring, 30% are composed of DEHP (Ecology 2011). Afshari et al. (2004) found that 17 – 18.5% of the PVC flooring was comprised of DEHP.

In 2014, a study of 16 types of vinyl flooring found concentrations of phthalates ranging from 9 – 23% of the flooring by weight (Liang & Xu, 2014). In 2016, the California Office of Environmental Health Hazard Assessment estimated that DINP was present in vinyl floors at up to 18.9% in their safe use determination statement (California Office of Environmental Health Hazard Assessment, 2016). A summary of the percent of ortho-phthalates found in vinyl flooring is shown in Table 15. If we assume a home has 150 square feet of vinyl flooring—one 70 square foot kitchen and two 40 square foot bathrooms—weighing approximately 0.9 kg/sq ft, vinyl flooring containing 18% DINP by weight could contribute 24 kg of DINP to the home.

If we assume a home has 150 square feet of vinyl flooring weighing approximately 0.9 kg/sq ft, vinyl flooring containing 18% DINP per weight could contribute 24 kg of DINP to the home.

In 2015, ortho-phthalates were commonly sold in vinyl flooring. An NGO study analyzed 65 vinyl flooring tiles and detected phthalates in 58% of the products (The Ecology Center, 2015, not peer reviewed). It is important to note that in 2018 the same group reassessed phthalates in vinyl flooring sold from retailers advertising "phthalate-free flooring." They analyzed 26 vinyl flooring tiles and did not detect phthalates in any of the products at concentrations above 1% (Miller et al., 2019, not peer reviewed). We are currently confirming the proportion of the market that has removed ortho-phthalates from new flooring sold in Washington.

Table 15. Examples of the percentage of vinyl flooring by weight comprised of phthalates.

(*Based on California's Safe Use Determination)

Phthalate	Major metabolites	Concentration	Reference
DEHP	MEHOP, MEHHP, MEHP, MECPP	17 – 18.5%	Afshari, Gunnarsen, Clausen, & Hansen, 2004
DINP	MINP, MHINP, MOINP	20%	Liang & Xu, 2014
DEHP	MEHOP, MEHHP, MEHP, MECPP	23%	Liang & Xu, 2014
BBP	MBzP	15%	Liang & Xu, 2014
DnBP	MnBP	9%	Liang & Xu, 2014
DINP	MINP, MHINP, MOINP	18.9%*	California, 2016
DEHP	MEHOP, MEHHP, MEHP, MECPP	0.16 – 32.3%	Noguchi & Yamasaki, 2016

Estimated volume used in Washington

We estimate that vinyl flooring sold in Washington each year contributes 4,500 – 16,800 metric tons of phthalates to our homes, workplaces, and schools and 0.15 metric tons of phthalates to the environment. Recent national estimates of the sales of resilient flooring, a category of flooring comprised largely of types of vinyl flooring, range from $3.68 billion in 2016 (Floor Covering Weekly, 2017) to $4.5 billion in 2019 (Resilient Floor Covering Institute, 2019), the lower amount corresponding to 4.27 billion square feet.

Using Washington's population in proportion to the national population, this translates to approximately 100 million square feet (approximately 90,000 metric tons) sold annually in the state. Based on the detection rate of 58% (The Ecology Center, 2015, not peer reviewed) and a concentration range of 9 – 32% phthalates by weight (Table 15), we estimate that 4,500 – 16,800 metric tons (approximately 9.9 million – 37 million pounds) of phthalates are found in vinyl flooring purchased annually in Washington, and that 0.17 metric tons (374 pounds) of phthalates are released to the environment from vinyl flooring.

Potential for exposure to sensitive populations

Nearly all people, including children, people of childbearing age, and workers, are exposed to phthalates. Phthalate exposure is widespread in the U.S. general population. CDC (2019) notes that over 90% of Americans have measurable levels of many phthalate metabolites in their bodies. Ten years of biomonitoring data conducted by the CDC shows widespread exposure to most phthalates in the U.S. population. Between 2001 – 2002 and 2009 – 2010, DEHP exposure has declined about 37%, whereas DINP and DIDP exposure have increased (Zota, Calafat, & Woodruff, 2014). DIDP metabolites were detected in 89% of participants in 2005 – 2006 and 94% of participants in 2009 – 2010 (Zota, Calafat, & Woodruff, 2014). DINP metabolites were detected in 96% of participants in 2005 – 2006 and 98% of participants in 2009 – 2010 (Zota, Calafat, & Woodruff, 2014).

Between 2010 and 2011, the Washington Environmental Biomonitoring Study analyzed urine samples from 422 teens and women of childbearing age for phthalate metabolites. MEOHP, MEHHP, MEHP, and MBZP are some of the major metabolites from phthalates used in vinyl flooring (Table 15). These metabolites were detected in 98% of samples (Health, 2019). This is similar to results from a cohort of 378 pregnant women from Charleston, South Carolina. MBzP, MEHHP, MEOP and MEHP were all detected in greater than 90% of urine samples (Wenzel et al., 2018).

Children show similar exposure patterns as women of childbearing age. MBzP and MEHHP were detected in greater than 96% of samples from children ages 6 – 11 collected between 2007 and 2012 (Odebeatu, Taylor, Fleming, & Osborne, 2018). Human biomonitoring studies have measured parent phthalate in blood (Specht et al., 2014) and their metabolites in human urine (CDC, 2019a; CDC, 2019b; Dong et al., 2017; Health Canada, 2013; Silva et al., 2007; Suzuki et al., 2009; Tefre de Renzy-Martin et al., 2014), semen (Chen et al., 2017; Nassan et al., 2016), saliva (Silva, 2005), and breastmilk (Högberg et al., 2008; Main et al., 2006). Phthalates can cross the placental barrier (Fennel, Krol, Sumner, & Snyder, 2004), and some have been detected in cord blood (Ashley-Martin, 2015) and amniotic fluid (Silva et al., 2004).

These findings demonstrate that exposure to phthalates is widespread and that sensitive populations often have particularly high exposures.

Sensitive populations are exposed to phthalates in vinyl flooring. Epidemiological studies observe higher concentrations of BBP metabolites in people who live in homes with vinyl flooring. Because phthalates are not chemically bound to PVC in vinyl flooring, they can leach out over time, allowing for dermal exposure and contaminating indoor air and house dust (Xu, Cohen-Hubal, Clausen, & Little, 2009). House dust ingestion and inhalation is an important exposure route for infants and young children because they spend more time on or near the floor, have more frequent hand-to-mouth activity, and experience faster respiration rates than adults. A Swedish study of preschool dust observed higher concentrations of DINP in rooms with PVC flooring (Larsson et al., 2017).

Multiple studies have found associations between phthalate metabolites in children's urine and vinyl flooring in the home. A prospective birth cohort of 239 children from New York reported that BBP concentrations in indoor air were significantly higher in rooms with vinyl or linoleum flooring (Just et al., 2015). While linoleum does not typically contain phthalates, the survey question asked about vinyl or linoleum flooring. Children living in homes with vinyl or linoleum floors had higher concentrations of BBP metabolites in their urine (Just et al., 2015). The study questionnaire did not distinguish between vinyl and linoleum flooring. These results are similar to a Swedish study of 110 infants, which found that PVC flooring in the bedroom was associated with higher urinary BBP metabolite concentrations (Carlstedt, Jonsson, & Bornehag, 2012).

Another study shed light on the magnitude of the impact of vinyl flooring on children's urinary BBP metabolites, reporting that children who lived in homes with 100% vinyl flooring had urinary concentrations of BBP metabolites 15 times higher than those of children who lived in homes with no vinyl flooring (Hammel et al., 2019).

Vinyl flooring has also been associated with worsening asthma symptoms, particularly in children. One study found that children living with vinyl flooring in their bedroom were 1.5 times more likely to develop asthma during the following 10-year period as compared with children who live in homes with other types of flooring. The association was strongest for children whose parents had bedrooms with vinyl flooring during pregnancy (Shu, Jönsson, Larsson, Nånberg, & Bornehag, 2014). The impacts of vinyl flooring on asthma may affect low-income and minority populations hardest. Some of these communities already face higher rates of asthma (CDC, 2020), possibly due to increased exposures to other environmental contaminants (Miranda, Edwards, Keating & Paul, 2011; Orellano, Quaranta, Reynoso, Balbi & Vasquez, 2017).

Workers in buildings with vinyl flooring can also have increased exposure. In a Swedish study of personnel in four geriatric hospitals, researchers found asthma symptoms were more common in the two buildings with signs of dampness. They suggested that this was related to phthalate migration from PVC flooring (Norbäck, Wieslander, Nordström, & Wålinder, 2000). A similar study in office buildings also implicated PVC flooring. After employees complained of several respiratory, conjunctival and dermal symptoms, researchers identified degradation of the plastic floor coverings as the source of chemicals in the air (Tuomainen, Seuri, & Sieppi, 2004).

Potential environmental exposure and exposure to sensitive species

Releases of phthalates from vinyl flooring can contribute to environmental concentrations. Phthalates found in vinyl flooring can be released from the product into air and dust (Xu et al., 2009). Following their release from flooring, phthalates can be tracked outside where they can contaminate the environment. They can also be released into our wastewater when we launder dusty items. Phthalates (BBP and DEHP) have been found in both WWTP influent and effluent (Ecology, 2010a).

Phthalates are frequently detected in the environment. In the Puget Sound area, Commencement Bay and the Lower Duwamish Waterway are EPA Superfund sites, partially due to phthalate contamination. Because phthalates are not environmentally persistent, the concentrations observed at these sites are reflective of current release levels. In 2010, a remedial investigation into phthalate levels in the Lower Duwamish Waterway found DEHP in surface sediment, surface water, fish, and invertebrate samples, and BBP in surface sediment (Lower Duwamish Waterway Group, 2010).

A similar investigation into Commencement Bay showed decreasing concentrations of many contaminants between 1999 and 2008. However, during that same period, concentrations of DEHP increased. DEHP was detected in all 30 sediment samples and BBP was detected in 20% of sediment samples (Ecology, 2010b). The City of Tacoma has been monitoring phthalates in stormwater sediment traps from 2001 to 2016, and detected DEHP and BBP (City of Tacoma, 2017). King County has conducted additional monitoring in the Duwamish Estuary and still detected DEHP in sediment, and concluded that juvenile Chinook salmon are exposed to phthalates from sediment in the estuary (King County, 2018).

In 2011, Ecology's Puget Sound Toxics Loading Study[59] estimated the environmental release of phthalates to the Puget Sound area from various sources, including vinyl flooring. Twenty percent of phthalates, seven tons per year, are attributable to PVC products. Of the PVC products, vinyl flooring is estimated to contribute 1.4% of phthalates or 0.1 metric tons of phthalates released into Puget Sound each year (Ecology, 2011b). Expanding this 0.1 metric tons estimate from the Puget Sound region only to the entire population in Washington, we expect that 0.17 metric tons (374 pounds) of phthalates are released to the environment from vinyl flooring.

Phthalates are also an emerging and major source of leachate contaminant from landfills. They are primarily emitted from consumer products and building materials, which include flooring (Ramakrishnan et al., 2015). As phthalates are used as plasticizers, it is likely that PVC materials in the landfill are contributing to the phthalates in leachate (Kalmykova, Bjorklund, Stromvall, & Blom, 2013). There is widespread evidence from worldwide landfill studies that phthalates are leaching, and can become ubiquitous contaminants in the surrounding environment. Disposal of household materials such as flooring is a primary source of phthalates that can contaminate various environmental media (Liu et al., 2010; Paxeus, 2000; Reid, Brougham, Fogarty, & Roche,

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[59] https://fortress.wa.gov/ecy/publications/documents/1103010.pdf

2007; Wowkonowicz & Kijenska, 2017). Phthalate-contaminated leachate can seep into surrounding groundwater to a greater extent than surface water or topsoil (Liu et al., 2010).

Phthalates are a concern for sensitive species. Governor Inslee's Southern Resident Orca Task Force named phthalates as chemicals of emerging concern. Phthalates in the environment can be toxic to both mammalian and non-mammalian aquatic life (Mathieu-Denoncourt, Wallace, de Solla & Langlois, 2015). Similar to concerns in humans, phthalates can cause endocrine disruption and impair reproduction and development in wildlife (Corradetti et al., 2013; Yuen, Qiu & Chen, 2020).

Existing regulations

Currently, no U.S. federal regulations exist for phthalates in vinyl flooring. EPA is evaluating five phthalates under the revised Toxic Substances Control Act (TSCA): DBP, BBP, DEHP, DIBP, and dicyclohexyl phthalate, and slated others for further assessment (EPA, 2019c). Additionally, manufacturers requested that EPA conduct risk evaluations for DIDP and DINP under TSCA Section 6. EPA granted both of these manufacturer requests.

In the U.S., there is a state-driven effort to limit exposures to phthalates. As of the date of this report, there are 30 adopted laws in 14 states pertaining to limits, bans, monitoring, or reporting of phthalate chemicals in products, and an additional seven states are considering regulatory actions (Safer States, 2020a). Of the 30 existing laws, 20 ban phthalates in a variety of food containers, paper products, and children's products. Federal laws (including the federal Consumer Product Safety Improvement Act (CPSIA)) prohibit more than 0.1% of three types of phthalates in children's toys: DEHP, DBP, and BBP (CPSC, 2019a; Safer States, 2020a).

The European Chemicals Agency (ECHA) maintains a list of ten phthalates on their Candidate List of Substances of Very High Concern published under the REACH Regulation (ECHA, 2019). Four phthalates (BBP, DBP, DEHP, and DIBP) are restricted to 0.1% by weight (individually or combined) of the plasticized material in any product, including flooring (QIMA, 2019).

Availability of safer alternatives

There are a number of resilient flooring options including cork, tile, vinyl flooring, linoleum, rubber, and polyolefins. Vinyl flooring may use ortho-phthalate plasticizers or alternatives, such as diisononyl cyclohexandicarboxylate (DINCH), tris (2-ethylhexyl) trimellitate (TOTM), and bis (2-ethylhexyl) terephthalate (DEHT). Several alternatives have been investigated by NGOs, such as Northwest Green Chemistry (Northwest Green Chemistry, 2018), Health Care Without Harm (Lent, Silas, & Vallette, 2009), and Healthy Building Network (Lott, 2014). Many manufacturers have moved away from the use of ortho-phthalates in vinyl flooring and are currently using DEHT, which may be a safer alternative. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

Chapter 11: Personal Care and Beauty Products (Fragrances)

Overview

Priority product

Personal care and beauty products that have fragrances. Examples include:

- Skincare products and body washes.
- Perfumes, colognes, body mists, and toilet waters.
- Eye and facial makeup.
- Face and body paint.
- Hair care products.
- Deodorants.

This priority product focuses on phthalates used in fragrances added to personal care and hygiene products. Products regulated by the Food and Drug Administration as drugs, biological products, or medical devices are excluded.

Priority chemical

Phthalates: Synthetic chemical esters of phthalic acid.

See the vinyl flooring chapter for more information about human health concerns and impacts of environmental release from phthalates.

Priority product summary

Fragrances used in the personal care and hygiene products are a significant source and use of phthalates. We made this determination after considering the criteria in RCW 70.365.020.[60] Fragrances contribute to phthalate exposure in sensitive populations and the environment. Women of childbearing age and pregnant women have higher exposure to the phthalates used in fragrances than men, and this is largely attributed to their use of personal care and hygiene products. Among women, women of color have higher phthalate levels from exposure attributable to fragrances in beauty products. These findings have important health equity and environmental justice implications.

Phthalates are frequently detected in the environment, particularly in urban areas, such as Commencement Bay and the Duwamish Estuary. In 2011, Ecology estimated that fragrances accounted for 33% of the 34 tons of phthalates released into Puget Sound annually. While phthalates are not environmentally persistent, large diffuse sources have led to widespread

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[60] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.020

environmental contamination that has the potential to impact sensitive populations. Governor Inslee's Southern Resident Orca Task Force named phthalates a chemical of concern for orcas and their food sources. Detailed support for our listing of phthalates in personal care and beauty products as a priority product is shown below.

Background

Phthalates serve two functions in fragrances: as solvents to keep ingredients well-blended and as fixatives that help extend the scent lifetime. In order to function in fragrances, phthalates must be able to volatilize, making lower molecular weight phthalates more useful than higher molecular weight phthalates. The most commonly reported phthalate used in fragrances is diethyl phthalate (DEP), but dimethyl phthalate (DMP), di-n-butyl phthalate (DBP), butylbenzyl phthalate (BBP), and di(2-ethylhexyl) phthalate (DEHP) have all been detected at lower concentrations. Fragrances are typically added to cosmetic products to make them smell favorably, however unscented products can still contain fragrances.

Estimated volume of phthalates used in fragrances

Phthalates are frequently detected in fragrances at concentrations that contribute a significant volume to our homes and environment. Since 2002, FDA has been surveying the concentrations of phthalates in personal care and beauty products. While overall their data show a decline in the use of phthalates, DEP was detected in almost half (11 out of 25) of the fragrances sampled in 2010 (FDA, 2013). The concentrations ranged from 480 – 40,000 ppm.

More recent studies suggest phthalates are still widely found in fragrances (Table 16). DEP is the most frequently detected phthalate in fragrances and it is found at mean concentrations between 0.16 and 0.18%, with the maximum concentration reported as 4.4%. In 2015, a study of 47 fragrances detected at least one phthalate in all products analyzed (Al-Saleh, 2016). Of the five phthalates analyzed (DMP, DEP, DBP, BBP, and DEHP), DEP was found in the greatest concentration and present in all 47 products analyzed. This is consistent with a 2013 study of 12 fragrances that detected DEP in all products tested (Guo & Kannan, 2013a). Table 16 shows detection frequencies and concentrations of phthalates in fragrances. These studies support our 2011 estimation that the total phthalate concentration in fragrances is around 1.2% (Ecology, 2011b).

In studies of multiple types of personal care and beauty products, fragrances frequently have the highest concentrations and detection rates for phthalates analyzed. A 2011 study by Koniecki et al. tested 252 products from the Canadian market for 18 phthalates. They detected DEP, DMP, DEBP, DBP, and DEHP. DEP was the most frequently detected phthalate (103 out of 252 products) and the highest concentrations were found in fragrances (Koniecki, Wang, Moody, & Zhu, 2011). Guo and Kannan's (2013a) study of 170 personal care products also found that fragrances had the highest detection frequency and concentration for DEP. Of the 170 personal care products, DEP was detected in 20% of baby care products, 53% of leave-on products, and 24% of rinse-off products (Guo & Kannan, 2013a).

Table 16. Detection frequencies and concentrations of phthalates (detected over 100 ppm) in personal care and hygiene products.

Phthalate	Metabolites	Product type	Detection frequency	Mean (and maximum) concentration	Reference
DEP	MEP	Fragrance	11 out of 25	7,813 ppm (max 44,000 ppm)	FDA, 2013
DEP	MEP	Fragrance	47 out of 47	1,622 ppm (max 23,649 ppm)	Al-Saleh, 2016
DMP	MMP	Fragrance	47 out of 47	30 ppm (max 405 ppm)	Al-Saleh, 2016
BBP	MBZP	Fragrance	47 out of 47	8 ppm (max 187 ppm)	Al-Saleh, 2016
DEHP	MEHOP, MEHHP, MEHP, MECPP	Fragrance	46 out of 47	6 ppm (max 147.5 ppm)	Al-Saleh, 2016
DEP	MEP	Fragrance	12 out of 12	3,420 ppm (max 7,980 ppm)	Guo & Kannan, 2013a
DEP	MEP	Body wash	5 out of 11	270 ppm (max 2,420 ppm)	Guo & Kannan, 2013a
DEP	MEP	Shampoo	3 out of 9	393 ppm (max 3,530 ppm)	Guo & Kannan, 2013a
DEP	MEP	Hair care	5 out of 6	179 ppm (max 897 ppm)	Guo & Kannan, 2013a

In addition to the products listed in the table above, FDA (2013) detected DEP at over 100 ppm in glitter gel, face and body paint, and lotion.

The EPA exposure factors handbook (2011) estimates that 0.65 grams of colognes and toilet waters are used per application and applied between 0.56 and 0.85 times per day, on average. If we estimate that fragrances contain 3,420 ppm DEP and assume an application rate of 0.85 times per day, over the course of a year, a person could contribute 0.7 grams of DEP to their body, home and environment. If the fragrance contained a higher concentration of DEP, such as 44,000 ppm, 8.9 grams of DEP could be released. Based on the studies discussed above, we conclude that phthalates are frequently used in fragrances and contribute to the volume of phthalates found in our bodies, homes, and the environment.

Similarly, the EPA exposure factors handbook (2011) estimates that 10.75 grams of shampoo are used per day. If the shampoo had 393 ppm DEP and the shampoo was applied daily, about

1.5 grams of DEP per person would be used yearly. Using the maximum concentration observed in shampoos (3,530 ppm), almost 14 grams of DEP per person could be used yearly.

Estimated volume used in Washington

We estimate that fragrances are one of the leading contributors of phthalates to Washington's environment, and that 17 tons of phthalates are released into Washington's environment each year from fragrance use. In 2011, Ecology published the Puget Sound Toxics Loading Study,[61] which estimated the contribution various consumer products and chemicals make to Puget Sound. Ecology estimated that there were 13 tons of phthalates contained in the cosmetics and personal care products used yearly in the Puget Sound Region, with 11 tons coming from fragrances. Fragrances released more phthalates to Puget Sound than the other consumer product categories considered, and accounted for more than 30% of total phthalate release (Ecology, 2011b).

Expanding this estimate beyond the Puget Sound region to the entire state, we estimate that 17 tons of phthalates are released from fragrances in personal care products in Washington each year. Based on the continued use of phthalates in fragrances, we anticipate that fragrances still contribute a significant volume of phthalates to Washington's environment.

Potential for exposure to sensitive populations when used

Nearly all people, including infants, children, people of childbearing age, and workers, are exposed to phthalates. Phthalate exposure is widespread in the U.S., with overburdened populations—such as women of color, pregnant women, women of childbearing age, and low-income populations—having higher exposures. CDC notes that over 90% of Americans have measurable levels of phthalate metabolites (the end-products of the chemical after it goes through metabolism in the body) in their urine. CDC also reports that women have higher levels of urinary metabolites than men for phthalates that are associated with personal care and beauty products (CDC, 2019a).

Between 2010 and 2011, the Washington Environmental Biomonitoring Study analyzed urine samples from 422 teens and women of childbearing age for phthalate metabolites. MEP, MBP, MEOHP, MEHHP, MEHP, and MBZP are some of the major metabolites from phthalates used in fragrances (Table 16). These metabolites were detected in 98% of samples. Washington's Low Income Survey and Testing Project analyzed phthalate metabolites in 579 low-income women of childbearing age and teenagers, and observed higher concentrations of MEP, the major metabolite from DEP, compared to the general Washington sample (DOH, n.d.).

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[61] https://fortress.wa.gov/ecy/publications/documents/1103010.pdf

These findings are particularly relevant for pregnant women and women of childbearing age. A prospective study of 446 pregnant women found that lower educational attainment and income were associated with higher concentrations of DBP metabolites in their urine (Polinski et al., 2018). Similar results were observed in a cohort of 378 pregnant women from Charleston, South Carolina. MEP, MMP, MBzP, MEHHP, and MEHP were all detected in greater than 90% of urine samples, with MEP found at the highest concentrations (Wenzel et al., 2018).

Children show similar exposure patterns as women of childbearing age. MEP, MBzP, MEHHP, MEOP were detected in greater than 96% of samples from children ages 6 – 11 collected between 2007 and 2012, with MEP detected at the highest concentration (Odebeatu, Taylor, Flemming, & Osborne, 2019).

These findings demonstrate that exposure to phthalates is widespread and that sensitive populations often have particularly high exposures.

Sensitive populations can be exposed to phthalates in fragrances. Women of childbearing age, pregnant women, and children can be exposed to phthalates when applying fragrances. Phthalates in fragrances can be inhaled, absorbed dermally, ingested, or transferred to house dust. Although phthalates are not expected to bioaccumulate and are not stored in our bodies for a long time, we do have repeated daily exposure to them through their presence in consumer products.

A 2013 analysis of phthalates in personal care products found that “leave on” products, such as lotions and face creams, delivered higher doses of phthalates than “rinse off” products, like shampoo. The analysis also found that DEP contributed the largest exposure of the seven phthalates included in the study (Guo & Kannan, 2013). (While they did not include fragrances in their study, the results may be applicable to fragrances as they are intended to be “leave-on” products and frequently contain higher concentrations of DEP than other types of personal care products.)

This estimation is supported by epidemiological studies of women of childbearing age. Parlett et al. (2013) found that women’s use of fragrances was associated with an increase in the concentration of urinary phthalate metabolites. Similarly, Buckley et al. (2012) found that personal care product use was associated with phthalate metabolites in urine in 50 women age 18 to 38. Perfume and cologne use was associated with higher levels of DEHP metabolites, while use of haircare products was associated with higher concentrations of DEP metabolites.

This is particularly concerning because women of color have been documented to have higher phthalate levels from exposure attributable to fragrances in beauty products (Helm, Nishioka, Brody, Rudel, & Dodson, 2018; Zota & Shamasunder, 2017). Additionally, women who normally intended to buy fragrance-free products had lower concentrations of DEHP metabolites in their urine.

Exposure to phthalates in fragrances is also an occupational concern. A 2017 study of cosmetics and perfume sales clerks found elevated levels of MEP and MEHP in urine after working shifts (Huang, Liao, Chang, Chan, & Lee, 2018).

Children and adolescents are exposed to phthalates in fragrances. A community-led intervention study engaged 100 Latina girls to reduce exposure to phthalates from consumer products. By using products labeled as "phthalate-free," they were able to reduce MEP in urine by over 24% (Harley et al., 2016). This suggests that personal care products could be a significant source of phthalate exposure for girls. Another study reports that urinary phthalate metabolites in infants increase when they are exposed to infant care products that can contain fragrances, such as baby shampoos, baby lotions, and baby powder (Sathyanarayana et al., 2008).

Potential environmental exposure and exposure to sensitive species

Phthalates from fragrances contribute to environmental phthalate concentrations. Phthalates found in fragrances can migrate into various environmental media (Méndez-Díaz, Abel daiem, Rivera-Utrilla, Sanchez-Polo, & Bautista-Toledo, 2012; Wu, Mahmood, Wu, & Zheng, 2008). In the Puget Sound area, Commencement Bay and the Lower Duwamish Waterway are EPA Superfund sites, partially due to phthalate contamination. Because phthalates are not environmentally persistent, the concentrations observed at these sites are reflective of current release levels.

In 2010, a remedial investigation into phthalate levels in the Lower Duwamish Waterway found DEHP in surface sediment, surface water, fish and invertebrates samples and BBP and DMP in surface sediments (Lower Duwamish Waterway Group, 2010). Information regarding the concentrations of other phthalates in Washington's environment can be found in the vinyl flooring chapter.

In 2011, Ecology published the Puget Sound Toxics Loading Study,[62] which estimated the environmental release of a number of toxic chemicals to the Puget Sound area. This report concludes that fragrances are the single largest source of releases and account for about one third (11 out of 34 tons) of total phthalate releases into the Puget Sound Basin. Sources included in this analysis were personal care products, PVC materials, air emissions, and other materials. We expanded this calculation from the Puget Sound region only to evaluate statewide releases, and estimate that 17 tons of phthalates are released into Washington's environment from fragrances annually.

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[62] https://fortress.wa.gov/ecy/publications/documents/1103010.pdf

Although discharge of phthalates from fragrances is primarily assumed to be through publicly-owned treatment works via sanitary sewers and septic systems (Ecology, 2011b), there is the potential for partially-used fragrance containers to be disposed of in landfills. Phthalates are a major source of leachate contaminants from landfills, and are primarily emitted from consumer products and building materials (Ramakrishnan et al., 2015). Phthalates found in fragrances (e.g., DEP, BBP, DEHP) are consistently detected across landfill leachates (Paxeus, 2000).

The degradation of phthalates depends on many factors, including the length of the carbon chain for each phthalate and the surrounding pH, microbes and nutrients, and temperature (Huang, Nkrumah, Appiah-Sefah, 2013; Liu et al., 2010). Studies have also found that landfill leachate contaminated with phthalates can seep into surrounding groundwater (Liu et al., 2010). Phthalates can escape landfills and enter surrounding environmental media.

Exposure to phthalates in the environment is a concern for sensitive species. As discussed in the vinyl flooring chapter, phthalates can be problematic for aquatic life. Governor Inslee's Southern Resident Orca Task Force, a workgroup developing plans to recover Puget Sound's orca population, identified phthalates as chemicals of emerging concern.

Existing regulations

Currently, no federal regulations exist regarding phthalates in personal care and hygiene products. In the U.S. there are a number of state regulations to limit exposures to phthalate chemicals. California is attempting to pass the Toxic-Free Cosmetics Act which would specifically ban phthalates in cosmetics, including fragrances (CA Legislature, 2019; Safer States, 2020a). Find more information about state regulations to limit phthalate exposures in the vinyl flooring chapter.

Federal laws (including the federal Consumer Product Safety Improvement Act (CPSIA)) focus on the removal of phthalates from children's toys (CPSC, 2019a; Safer States, 2020a). EPA is evaluating five phthalates under the revised Toxic Substances Control Act (TSCA): DBP, BBP, DEHP, DIBP, and dicyclohexyl phthalate, and slated others for further assessment (EPA, 2019a). Additionally, manufacturers requested that EPA conduct risk evaluations for DIDP and DINP under TSCA Section 6. EPA granted both of these manufacturer requests.

The European Chemicals Agency maintains a list of ten phthalates on their Candidate List of substances of Very High Concern published under the REACH Regulation (European Chemicals Agency, 2019). Canada banned the use of DEHP in cosmetics and restricted it (as well as DIDP, DINP, DNOP, DBP, and BBP) in children's toys and some child care products (Canadian Health Services, 2019). Two phthalates, dibutyl phthalate (DBP) and bis(2-ethylhexyl) phthalate (DEHP), are banned in cosmetics sold in the European Union (European Commission, 2019a).

Availability of safer alternatives

There are a number of solvents and fixatives used in cosmetic fragrances in addition to phthalates, such as dipropylene glycol and triethyl citrate. Many fragrances are also dissolved in water and ethanol, though these are not compatible with all compounds. In addition, cosmetics could be formulated to be fragrance-free to avoid the use of solvents.

Several alternatives have been investigated by NGOs, such as Northwest Green Chemistry (Northwest Green Chemistry, 2018). Because many phthalate-free personal care products are on the market, safer alternatives may be available. As part of our Safer Products for Washington process, we will continue to address the safety and feasibility of these alternatives in Phase 3.

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Appendix 1: Acronyms

Table 17. Acronyms with definition and CAS number.

This list of CAS numbers is not a comprehensive list of chemicals we are considering for potential regulation.

Acronym	Definition	Chemical Abstracts Service (CAS) number
µg/cm^3	Micrograms per centimeter cubed	
µg/cm^2	Micrograms per centimeter squared	
µg/g	Micrograms per gram	
µg/kg	Micrograms per kilogram	
µg/L	Micrograms per liter	
µg/m^2	Micrograms per meter squared	
2,4,6-TBP	2,4,6-Tribromophenol	118-79-6
ABS	Acrylonitrile butadiene styrene polymers	
AP/APE	Alkylphenol/alkylphenol ethoxylate	
ATSDR	Agency for Toxic Substances and Disease Registry	
BADGE	Bisphenol A diglycidyl ether	Multiple
BBP	Benzyl butyl phthalate	85-68-7
BDE	Brominated diphenyl ether	
BPA	Bisphenol A	80-05-7
BPAF	Bisphenol AF	1478-61-1
BPAP	Bisphenol AP	1571-75-1
BPB	Bisphenol B	77-40-7
BPC	Bisphenol C	79-97-0
BPF	Bisphenol F	620-92-8
BPP	Bisphenol P	2167-51-3
BPS	Bisphenol S	80-09-1
BPZ	Bisphenol Z	843-55-0
BTBPE	1,2-Bis(2,4,6-tribromophenoxy)ethane	37853-59-1
CAP	Chemical Action Plan	
CDC	Centers for Disease Control	
CPSC	Consumer Product Safety Commission	
DBDPE	Decabromodiphenyl ethane	84852-53-9
DBP	Di-n-butyl phthalate	84-74-2
DecaBDE	Decabromodiphenyl ether	1163-19-5
DEHP	Di(2-ethylhexyl) phthalate	117-81-7
DEHT	Bis (2-ethylhexyl) terephthalate	6422-86-2
DEP	Diethyl phthalate	84-66-2
DIBP	Diisobutyl phthalate	84-69-5

Acronym	Definition	Chemical Abstracts Service (CAS) number
DIDP	Diisodecyl phthalate	68515-49-1 and 26761-40-0
DINCH	Diisononyl cyclohexandicarboxylate	474919-59-0 and 166412-78-8
DINP	Diisononyl phthalate	68515-48-0 and 28553-12-0
DMP	Dimethyl phthalate	131-11-3
DnOP	Di-n-octyl phthalate	117-84-0 and 8031-29-6
DPHP	Diphenyl-phosphate	
DTSC	California Department of Toxic Substances Control	
EC	European Commission	
ECHA	European Chemicals Agency	
EDI	Estimated daily intake	
EFSA	European Food Safety Authority	
EHDPP	Ethylhexyl diphenyl phosphate	1241-94-7
EPA	U.S. Environmental Protection Agency	
EU	European Union	
FOX	Firefighter occupational exposure	
FTOH/FTS	Fluorinated telomer alcohol/sulfonates	
GM	Geometric mean	
HIPS	High impact polystyrene	
iPCB	Inadvertent Polychlorinated Biphenyl	Mutiple
IPTPP	Isopropylated triphenyl phosphate	68937-41-7
ITRC	Interstate Technology & Regulatory Council	
kg	Kilogram	
K_{OW}	Octanol—water partition coefficient	
LOQ	Limit of quantitation	
MBzP	Mono-benzyl phthalate	2528-16-7
MECPP	Mono-(2-ethyl-5-carboxypentyl) phthalate	40809-41-4
MEHHP	Mono (2-ethyl-5-hydroxyhexyl) phthalate	40321-99-1
MEHOP	Mono-(2-ethyl-5-oxohexyl) phthalate	40321-98-0
MEHP	Mono(2-ethylhexyl) phthalate	4376-20-9
MEP	Mono-ethyl phthalate	2306-33-4
Me-PFOSA-AcOH	2-(N-methyl-perfluorooctane sulfonamido) acetate	Multiple
mg	Milligram	
mg/kg	Milligrams per killigram	
MHINP	Mono(hydroxyisononyl) phthalate	Multiple
MINP	Monoisononyl phthalate	Multiple

Acronym	Definition	Chemical Abstracts Service (CAS) number
MMP	Mono-methyl phthalate	4376-18-5
MnBP	Mono-n-butyl phthalate	34-74-2 and 131-70-4
MOINP	Mono(oxoisononyl) phthalate	Multiple
MPCA	Minnesota Pollution Control Agency	
ng	Nanogram	
ng/cm^2	Nanograms per centimeter squared	
ng/g	Nanograms per gram	
ng/kg	Nanograms per kilogram	
ng/L	Nanograms per liter	
ng/m^2	Nanograms per meter squared	
ng/mL	Nanograms per milliliter	
NGO	Non-governmental organization	
NHANES	National Health and Nutrition Examination Survey	
NIEHS	National Institute of Environmental Health Sciences	
NIOSH	National Institute for Occupational Safety and Health	
NP/NPE	Nonylphenol/nonylphenol ethoxylate (type of APE)	Multiple
NTP	National Toxicology Program (part of U.S. DHHS)	
OctaBDE	Octabromodiphenyl ether	32536-52-0
OECD	Organisation for Economic Co-operation and Development	
OP/OPE	Octylphenol/octylphenol ethoxylate (type of APE)	Multiple
PAP	Polyfluoroalkyl phosphates	
PASF	Perfluoroalkane sulfonyl fluorides	
PBDEs	Polybrominated diphenyl ethers	Multiple
PC-ABS	Polycarbonate/ABS blends	
PCB	Polychlorinated biphenyl	Multiple
PDFA	Perfluorodecanoic acid	335-76-2
PentaBDE	Pentabromodiphenyl ether	32534-81-9
PET	Polyethylene terephthalate	25038-59-9
PFAA	Perfluoroalkyl acid	
PFAS	Per- and polyfluoroalkyl substances	
PFBA	Perfluorobutanoic acid	375-22-4
PFBS	Perfluorobutane sulfonic acid	375-73-5
PFCA	Perfluoroalkyl carboxylic acid	
PFHpA	Perfluoroheptanoic acid	375-85-9
PFHxA	Perfluorohexanoic acid	307-24-4
PFHxS	Perfluorohexane sulfonic acid	355-46-4
PFNA	Perfluorononanoic acid	375-95-1
PFOA	Perfluorooctanoic acid	335-67-1

Acronym	Definition	Chemical Abstracts Service (CAS) number
PFOS	Perfluorooctane sulfonic acid	1763-23-1
PFPeA	Perfluoropentanoic acid	2706-90-3
PFSA	Perfluoroalkyl sulfonic acid	
ppb	Parts per billion	
ppm	Parts per million	
PVC	Polyvinyl chloride	Multiple
RDP	Resorcinol bis (diphenyl-phosphate)	57583-54-7
RfD	EPA reference dose	
TBBPA	Tetrabromobisphenol A	79-94-7
TCP	Tricresyl phosphate or tris(methylphenyl)phosphate (TMPP)	Multiple
TNBP	Tri-n-butyl phosphate	126-73-8
TOTM	Tris (2-ethylhexyl) trimellitate	3319-31-1
TPP	Triphenyl phosphate	115-86-6
TTBP-TAZ	2,4,6-tris(2,4,6-tribromophenoxy)-1,3,5-triazine	25713-60-4
TV	Television	
UL 94	Standard for Tests for Flammability of Plastic Materials for Parts in Devices and Appliances	
WWTP	Wastewater treatment plant	
XRF	X-ray fluorescence	

Appendix 2: Exemptions

Under the Safer Products for Washington program, Ecology will not identify the following as priority consumer products:

- Plastic shipping pallets manufactured prior to 2012;
- Food or beverages;
- Tobacco products;
- Drug or biological products regulated by the United States food and drug administration;
- Finished products certified or regulated by the federal aviation administration or the department of defense, or both, when used in a manner that was certified or regulated by such agencies, including parts, materials, and processes when used to manufacture or maintain such regulated or certified finished products;
- Motorized vehicles, including on and off-highway vehicles, such as all-terrain vehicles, motorcycles, side-by-side vehicles, farm equipment, and personal assistive mobility devices; and
- Chemical products used to produce an agricultural commodity, as defined in RCW 17.21.020.[63]

Ecology may identify the packaging of products listed above as priority consumer products.

For an electronic product identified by Ecology as a priority consumer product under this section, the department may not make a regulatory determination under RCW 70.365.040[64] to restrict or require the disclosure of a priority chemical in an inaccessible electronic component of the electronic product.

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[63] http://app.leg.wa.gov/RCW/default.aspx?cite=17.21.020

[64] http://app.leg.wa.gov/RCW/default.aspx?cite=70.365.040

Appendix 3: Citation List

Overview

The following citation list was developed to meet the requirements outlined in RCW 70.365.050[65] and 34.05.272.[66] It identifies the peer-reviewed science, studies, reports, and other sources of information used to support our identification of priority consumer products.

It includes only the references that are also listed above in the references section, but also identifies each reference by information source. The following are the types of sources used to support this report:

1. Peer review is overseen by an independent third party.
2. Review is by staff internal to Ecology.
3. Review by persons that are external to and selected by Ecology.
4. Documented open public review process that is not limited to invited organizations or individuals.
5. Federal and state statutes.
6. Court and hearings board decisions.
7. Federal and state administrative rules and regulations.
8. Policy and regulatory documents adopted by local governments.
9. Data from primary research, monitoring activities, or other sources, but that has not been incorporated as part of documents reviewed under other processes.
10. Records of best professional judgment of Ecology employees or other individuals.
11. Sources of information that do not fit into one of the other categories listed.

Citation list

Table 18. References found in this report, categorized by source type.

Citation	Category
3M. (2018a). *Scotchgard™ Fabric Protector (Cat. No. 4101, 4106)*. Safety Data Sheet. Retrieved from https://multimedia.3m.com/mws/mediawebserver?mwsId=SSSSSuUn_zu8l00xMx_14x2GOv70k17zHvu9lxtD7SSSSSS--	11
3M. (2018b). *Scotchgard™ Rug & Carpet Protector (Cat. No. 4406-17, 4406-14)*. Safety Data Sheet. Retrieved from https://multimedia.3m.com/mws/mediawebserver?mwsId=SSSSSuUn_zu8l00xMYtSm8_vOv70k17zHvu9lxtD7SSSSSS--	11

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[65] https://app.leg.wa.gov/RCW/default.aspx?cite=70.365.050

[66] https://app.leg.wa.gov/RCW/default.aspx?cite=34.05.272

Citation	Category
Abafe, O. A., Chokwe, T. B., Okonkwo, J. O., & Martincigh, B. S. (2017). Alkylphenols and alkylphenol ethoxylates in dust from homes, offices and computer laboratories: Implication for personal exposure via inadvertent dust ingestion. *Emerging Contaminants*, *3*, 127–131. https://doi.org/10.1016/j.emcon.2018.01.001	1
Abbasi, G., Saini, A., Goosey, E., & Diamond, M. L. (2016). Product screening for sources of halogenated flame retardants in Canadian house and office dust. *Science of the Total Environment*, *545–546*, 299–307. https://doi.org/10.1016/j.scitotenv.2015.12.028	1
Abdallah, M.A., Harrad, S., & A. Covaci, A. (2008) Hexabromocyclododecanes and tetrabromobisphenol-A in indoor air and dust in Birmingham, U.K: implications for human exposure. *Environ Science & Technology, 42,* 6855-61.	1
Ademollo, N., Ferrara, F., Delise, M., Fabietti, F., & Funari, E. (2008). Nonylphenol and octylphenol in human breast milk. *Environment International*, *34*, 984–987. https://doi.org/10.1016/j.envint.2008.03.001	1
Afshari, A., Gunnarsen, L., Clausen, P. A., & Hansen, V. (2004). Emission of phthalates from PVC and other materials. *Indoor Air*, *14*, 120–128. https://doi.org/10.1046/j.1600-0668.2003.00220.x	1
Agency for Toxic Substances and Disease Registry (ATSDR). (2012). Toxicological Profile for Phosphate Ester Flame Retardants. Agency for Toxic Substances and Disease Registry. 431 pages.	1
Agency for Toxic Substances and Disease Registry (ATSDR). (2014). Polychlorinated Biphenyls - ToxFAQs^{TM}. Retrieved from https://www.atsdr.cdc.gov/toxfaqs/tf.asp?id=140&tid=26	11
Agency for Toxic Substances and Disease Registry (ATSDR). (2019). Toxicological Profile for Polychlorinated Biphenyls (PCBs).	1
Alberta Health and Wellness. (2008). Alberta Biomonitoring Program: Chemicals in Serum of Pregnant women in Alberta, Edmonton.	11
Allen, J. G., McClean, M. D., Stapleton, H. M., & Webster, T. F. (2008). Linking PBDEs in House Dust to Consumer Products using X-ray Fluorescence. *Environmental Science & Technology*, *42*, 4222–4228. https://doi.org/10.1021/es702964a	1
Al-Saleh, I., & Elkhatib, R. (2016). Screening of phthalate esters in 47 branded perfumes. *Environmental Science and Pollution Research*, *23*, 455–468.	1
Alvarez, D., Perkins, S., Nilsen, E., & Morace, J. (2014). Spatial and temporal trends in occurrence of emerging and legacy contaminants in the Lower Columbia River 2008–2010. *Science of The Total Environment*, *484*, 322–330. https://doi.org/10.1016/j.scitotenv.2013.07.128	1

Citation	Category
Ampleman, M. D., Martinez, A., DeWall, J., Rawn, D. F. K., Hornbuckle, K. C., & Thorne, P. S. (2015). Inhalation and Dietary Exposure to PCBs in Urban and Rural Cohorts via Congener-Specific Measurements. *Environmental Science & Technology*, *49*, 1156–1164. https://doi.org/10.1021/es5048039	1
Andersson, M., Ottesen, R., & Volden, T. (2004). Building materials as a source of PCB pollution in Bergen, Norway. *Science of The Total Environment*, *325*, 139–144. https://doi.org/10.1016/j.scitotenv.2003.11.014	1
Apfelbachter, M., Cioci, M., & Strong, P. (2014). BPA and BPS in Thermal Paper: Results of Testing in Minnesota Hospitality Industry. Minnesota Pollution Control Agency, (March), 1–11.	11
Aschberger, K., Munn, S., Olsson, H., Pakalin, S., Pellegrini, G., Vegro, S., & Paya Perez, A. B. (2008). European Union Risk Assessment Report - BPA (4,4’ – Isopropylidenediphenol (Bisphenol – A)) Environment Addendum. In European Commission Joint Research Centre. https://doi.org/10.2788/40195	1
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