SAFER CONSUMER PRODUCTS

Product – Chemical Profile for Food Packaging Containing Perfluoroalkyl or Polyfluoroalkyl Substances

JULY 2020 • DISCUSSION DRAFT

CalEPA California Environmental Protection Agency

DTSC Department of Toxic Substances Control

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Candidate Chemical: A chemical that exhibits a hazard trait and is listed on one or more authoritative lists in the SCP regulations

Product-Chemical Profile: A report generated by DTSC to explain its determination that a proposed Priority Product meets the SCP regulatory criteria for potential significant or widespread adverse impacts to humans or the environment

Priority Product: A product-chemical combination, as identified in regulations by DTSC, that has the potential to contribute to significant or widespread adverse impacts to humans or the environment

ABOUT THIS PROFILE

The Department of Toxic Substances Control (DTSC) identifies product-chemical combinations for consideration as Priority Products in accordance with the process identified in Article 3 of the Safer Consumer Products (SCP) regulations (California Code of Regulations, title 22, Division 4.5, Chapter 55, Article 3). DTSC has determined that food packaging containing any member of the class of perfluoroalkyl and polyfluoroalkyl substances (PFASs) meets the key prioritization criteria (California Code of Regulations. title 22, section 69503.2(a)) for listing a Priority Product:

(1) There must be potential public and/or aquatic, avian, or terrestrial animal or plant organism exposure to the Candidate Chemical(s) in the product; and

(2) There must be the potential for one or more exposures to contribute to or cause significant or widespread adverse impacts.

This Product-Chemical Profile (Profile) demonstrates that the regulatory criteria have been met and serving as the basis for Priority Product rulemaking. The Profile does not provide a comprehensive assessment of all available literature on adverse impacts and exposure for [chemical] or [product]. DTSC will finalize this Profile after considering public comments and may then start the rulemaking process. If this Priority Product regulation is adopted, the responsible entities must follow the reporting requirements pursuant to the SCP regulations (California Code of Regulations. title 22, section 69503.2(a)).

Readers should consider the following:

1. This Profile is not a regulatory document and does not impose any regulatory requirements.
2. The Profile summarizes information compiled by DTSC as of July 2020.
3. DTSC requests that stakeholders provide data on the chemical and product described in this document to assist us in the evaluation process that may lead to our regulatory proposal. Written comments can be submitted using our information management system, CalSAFER, prior to September 13, 2020.
4. By proposing to list this product-chemical combination as a Priority Product containing a Chemical of Concern, DTSC is not asserting that the product cannot be used safely. The proposal indicates only that there is a potential for exposure of people or the environment to the Chemical of Concern in the Priority Product, that such exposure has the potential to cause or contribute to significant or widespread adverse impacts, and that safer alternatives should be explored.

1. RATIONALE FOR PRODUCT-CHEMICAL SELECTION

The Department of Toxic Substances Control (DTSC) proposes to list plant fiber-based food packaging products containing any member of the class of perfluoroalkyl and polyfluoroalkyl substances (PFASs) as a Priority Product. This product-chemical combination meets the identification and prioritization factors outlined in the Safer Consumer Products regulations: (1) there is potential for human and other organism exposure to PFASs from these food packaging products; and (2) the exposure has the potential to contribute to or cause significant and widespread adverse impacts.

PFASs are a class of manufactured chemicals with at least one fully fluorinated carbon atom (Buck et al. 2011; CECBP 2015a). All PFASs are Candidate Chemicals under the Safer Consumer Products (SCP) Program due to their designation on December 22, 2015, as Priority Chemicals under the California Environmental Contaminant Biomonitoring Program.

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DTSC is also taking a class approach to PFASs because:

- PFASs are a wide and varied group of chemicals used in many applications. All PFASs share one common trait – highly stable carbon-fluorine (C-F) bonds that make them or their final degradation products highly persistent in the environment.

- Persistence is a hazard trait identified in the Office of Environmental Health Hazard Assessment's Green Chemistry Hazard Traits regulation 69405.3: "Persistence of a chemical in the environment promotes sustained exposure and contributes to accumulation in the environment."
- Within the overall PFAS class, perfluoroalkyl acids (PFAAs) are the most problematic subclass. They are the most widely and thoroughly characterized subclass and are associated with a number of different health hazards, including endocrine disruption, developmental and reproductive toxicity, and immune dysregulation.
- In the case of PFAAs, their ubiquity in combination with their persistence results in continuous exposure from multiple sources, including contaminated drinking water sources and the food supply. They can accumulate in meat, plants, and drinking water, and ultimately in humans and wildlife.
- The vast majority of PFASs degrade into PFAAs, which typically increases their mobility in the environment, making containment or removal a challenge.

While persistence alone warrants enough concern to include any member of the PFAS class in product prioritization, the health hazards associated with exposure to PFAAs are additionally concerning and underlie our listing.

Potential for exposure to PFASs in plant fiber-based food packaging products

DTSC is concerned about potential human and ecological exposures to PFASs from the use and end-of-life of PFAS-treated food packaging.

Humans are exposed to PFASs through a wide variety of sources and pathways. As a result, PFASs have been detected in the blood serum of over 98 percent of Americans. Estimates vary, but it is thought that the primary sources of human exposure to PFASs are through dietary intake (accounting for up to half of total exposure) and inhalation and ingestion of contaminated indoor air and dust. Human exposure to PFASs through dietary intake can occur via contaminated food and drinking water. Once in the human body, PFASs accumulate in protein-rich tissues and typically have serum half-lives ranging from days to years, depending on their carbon chain lengths. Many studies have shown that PFASs are capable of transfer from pregnant mothers to their fetuses via the placenta during gestation, as well as transfer from nursing mothers to their infants via breastfeeding. These scenarios represent significant periods of PFAS exposure for developing fetuses and children, which may lead to adverse health outcomes. Like humans, wildlife is exposed to PFASs by consuming contaminated water or food. Within aquatic food webs, PFASs were found to increase in concentration from ambient water to plankton and further up the food chain.

Plant fiber-based food packaging products treated with PFASs for grease, oil, or water resistance can expose humans and biota to PFASs during their manufacturing, use, and end-of-life. PFASs can migrate from food packaging into the packaged food, with migration rates dependent on the temperature, acidity, storage time, and fat content of the food. Used PFAS-treated paper, paperboard, and molded fiber food packaging products are sometimes composted, releasing PFASs into the compost. When used food packaging is sent to a landfill, the PFASs can migrate into landfill leachate, contaminating surface waters and the surrounding environment. When applied to soil as fertilizers, biosolids from wastewater treatment plants that treat PFAS-contaminated landfill leachate can contaminate drinking water sources with PFASs, as well as crops such as potatoes, grains, and leafy vegetables. Recycled products made from PFAS-treated paper, paperboard, and molded fiber food packaging can also be a source of PFAS exposure. Harmful PFAS combustion products may also be released when these products are incinerated.

Potential for significant or widespread adverse impacts

All PFASs display at least one of the hazard traits identified in California's Green Chemistry Hazard Traits Regulations (California Code of Regulations, title 22, section 69401 et seq.). PFASs are either extremely persistent in the environment or they degrade into extremely persistent PFASs. Most PFASs are mobile in environmental media such as air and water, and thus are widespread in living organisms and the environment. Several PFASs bioaccumulate significantly in animals or plants, including those consumed by humans as food.

The U.S. Food and Drug Administration (FDA) prohibits the use of certain PFASs in food-contact materials because of their potential to cause adverse human health impacts. These effects, which are well established in animal and human studies, include kidney and testicular cancers, thyroid disease, reduced immune response, and pregnancy-induced hypertension. Evidence from animal, *in vitro*, and modeling studies also links the degradation products of FDA-approved PFASs with multiple toxicological hazard traits, including developmental toxicity, endocrine toxicity, hepatotoxicity, neurodevelopmental toxicity, and reproductive and developmental toxicity.

In general, fetuses, infants, toddlers, and young children experience higher relative exposure levels and are more vulnerable to the effects of environmental toxicants. This is true for PFASs, with children being most at risk of exposure and adverse effects. The American Academy of Pediatrics released a policy statement in 2018 about the risk that food additives pose to children's health, highlighting the potential adverse effects associated with PFASs in food packaging. This statement cites support from the Endocrine Society, a joint 2013 report from the World Health Organization and United Nations Environment Program, and a statement from the International Federation of Gynecology and Obstetrics in 2015, indicating broad consensus on protecting children's health from environmental contaminants, including the PFASs used in food packaging.

Based on the criteria in the Safer Consumer Products regulations, DTSC has determined that plant fiber-based food packaging products containing any member of the class of PFASs have the potential to cause or contribute to significant and widespread adverse impacts to sensitive subpopulations, environmentally sensitive habitats, and threatened and endangered species. Given the known hazard traits, replacing currently used PFASs in food packaging with other members of the PFAS class could constitute a regrettable substitution. Hence, this proposal covers plant fiber-based food packaging products containing any member of the class of PFASs.

DTSC identified several policy goals in its 2018-2020 Priority Product Work Plan, including "to protect Californians from chemicals that migrate into food from food packaging" and "to protect children, women of childbearing age, and pregnant women from exposures to harmful chemicals, especially carcinogens, mutagens, reproductive toxicants, neurotoxicants, developmental toxicants, and endocrine disruptors." Addressing PFASs in food packaging aligns with this overarching goal.

2. PRODUCT DEFINITION AND SCOPE

This section describes the product that forms the basis for the proposed product-chemical combination.

Food packaging is any product used to package hot, cold, frozen, or room-temperature food or beverage items for sale to restaurants and grocery stores or for retail sale to consumers. Food packaging may be used for a variety of purposes including food preservation, transport and delivery to points of retail sale, marketing, tamper resistance, or transport by the consumer (e.g., cups for liquids or wrappers for fast-food items). The principal functions of food packaging are to protect food from external damage and contamination, contain the food for transport, and provide information to the consumer. Food packaging may be made from a variety of materials including, but not limited to, paper, ceramic, plastic, glass, and metals.

This product-chemical profile covers all plant fiber-based food packaging products treated with PFASs to confer oil, grease, or water resistance. These products are made from three main types of materials: paper, paperboard, and molded fiber.

- **Paper** is composed of natural fibers of bleached or unbleached cellulose. It can be derived from virgin or recycled materials. Paper mills transform plant fiber, typically wood fiber, into pulp, using either chemical or mechanical pulping methods that separate the cellulose fibers from the lignin (i.e., the "glue" that binds the tree internally) and other substances present in wood. Chemical pulping relies on pressure-cooking wood chips in the presence of solutions of certain chemicals, while mechanical pulping grinds wood chips via the use of rotating metal discs. The cellulose fibers are then washed with water and drained to form pulp. Next, the pulp is

deposited onto a machine, where it is flattened, dried, and cut into paper sheets and rolls (Casey Printing 2017; American Forest and Paper Association 2020a).

    - Paper food packaging includes, but is not limited to, bakery sleeves and bags, deli liners, fast food wrappers, microwave popcorn bags, butter wraps, baking paper, and paper for dry foods.

- **Paperboard** is typically manufactured from pulp, straw, wastepaper, or a combination of these materials. Paperboard is thicker, heavier, and more rigid than paper because it is made from several layers of pulp. Thicker paperboard is produced by pressing sheets of paper or board together in a process known as laminating (American Forest and Paper Association 2020b; Britannica 2020).

    - Paperboard food packaging includes, but is not limited to, French-fry containers, food trays and boats, takeout boxes and clamshells, ice cream tubs, and some types of paper plates.

- **Molded fiber** is typically made from cellulose fibers, pulp, recycled paperboard, or newsprint. It is manufactured by depositing a water suspension of fibers onto a screened mold. After a vacuum is applied, the fiber structure begins to develop strength. Water is removed by applying pressure to the wet mixture with a matched mold. Once the mass of the solid fraction reaches about 50 percent, the product (called a slurry) is completely dried out in a heated mold or oven (Didone et al. 2017).

    - Molded fiber food packaging includes, but is not limited to, clamshell takeout food containers, food bowls, and egg trays.

The state of Washington recently enacted Engrossed Substitute House Bill 2658 (State of Washington 2018a), which prohibits the sale of food packaging containing PFASs as of January 1, 2022, provided the state Department of Ecology identifies safer alternatives (State of Washington 2018b). Washington's law defines a "food package" as "a package or packaging component that is intended for direct food contact and is comprised, in substantial part, of paper, paperboard, or other materials originally derived from plant fibers" (ECY 2018). The Washington Department of Ecology divides "food package" into three subcategories:

- Paper wraps, liners, bags, and sleeves;
- Dinnerware, such as plates, bowls, and trays; and
- Food service containers, such as takeout cartons for transport or storage.

All the products covered by Washington's law, as listed by the Washington State Department of Ecology, also fall under the scope of DTSC's product-chemical profile and proposed Priority Product listing.

Food packaging products made of materials other than plant fiber are excluded from this proposal, even if they contain PFASs. For instance, food packaging products made of the following materials fall outside the scope of this proposal: metal, plastic, bio-based plastic, and multilaminated materials.

3. CANDIDATE CHEMICAL DEFINITION AND PROPERTIES

This section introduces the Candidate Chemical (or Chemicals) in the proposed product-chemical combination.

This proposal covers all perfluoroalkyl and polyfluoroalkyl substances (PFASs) in current or future production.[1] As of 2015, a few individual PFASs were included on the Safer Consumer Products (SCP)'s Candidate Chemical list. In December of that year, all PFASs were added as Candidate Chemicals under the SCP Program due to their designation as Priority Chemicals under the California Environmental Contaminant Biomonitoring Program (California Biomonitoring Program or CECBP) (CECBP 2015a). The CECBP listing is based on the chemicals' potential for widespread exposures, persistence, bioaccumulation, and emerging evidence for toxicity.

PFASs are a class of fluorinated organic chemicals containing at least one fully fluorinated carbon atom (CECBP 2015a). This class includes more than 3,000 chemicals (KEMI 2015), although fewer may be in current production. The Organisation for Economic Cooperation and Development (OECD) has identified 4,730 Chemical Abstracts Service (CAS) registry numbers related to individual PFASs or commercial PFAS mixtures available on the global market (OECD 2018). PFASs are manufactured and not found naturally in the environment. All PFASs contain carbon and fluorine atoms, in addition to other elements such as oxygen, hydrogen, nitrogen, sulfur, or chlorine. PFASs vary in chain length, i.e., the number of carbon atoms forming the backbone of their molecule, from a chain of two carbons to large molecular-weight polymers.

Based on common characteristics, and for the purposes of this designation, PFASs can be roughly subdivided into four main categories (Buck et al. 2011; Wang et al. 2017b) listed below and depicted in

[1] Note that U.S. EPA and others use the acronym "PFAS" for this class of chemicals. Jn this document, we use the acronym "PFASs" to emphasize that it refers to a class of chemicals, and to make it clear when we refer to multiple members of this class rather than an individual chemical.

Figure 1. However, any other PFAS that exists or will be developed in the future and is used in food packaging products also falls under the scope of this proposal.

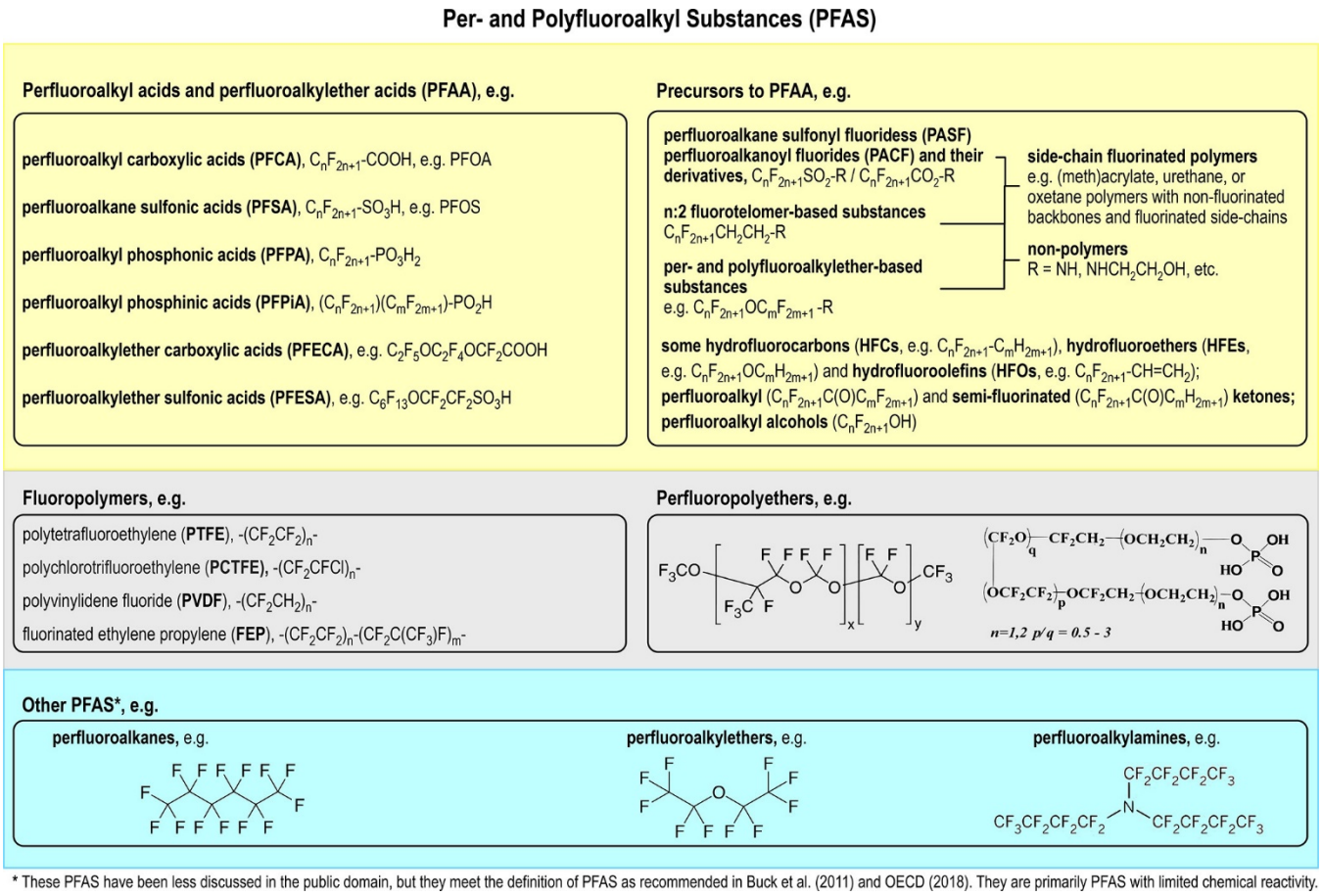

Figure 1: General classification of PFASs. Reprinted with permission Kwiatkowski et al. (2020), available at https://pubs.acs.org/doi/10.1021/acs.estlett.0c00255. Further permissions related to the material excerpted should be directed to the ACS.

1. **Perfluoroalkyl acids (PFAAs)**. These are perfluorinated substances in which fluorine atoms have replaced all hydrogen atoms attached to carbon atoms (except for those associated with functional groups). As a result, these compounds are recalcitrant to degradation and extremely persistent in the environment. This subgroup includes:

    - Perfluoroalkyl carboxylic acids (PFCAs) such as perfluorooctanoate (PFOA);[2]
    - Perfluoroalkyl sulfonic acids (PFSAs) such as perfluorooctane sulfonate (PFOS);[2]
    - Perfluoroalkyl sulfinic acids (PFSiAs);

[2] PFOA and PFOS are referred to as "C8" because they contain eight carbon atoms in their molecules.

- Perfluoroalkyl phosphonic acids (PFPAs);
- Perfluoroalkyl phosphinic acids (PFPiAs);
- Perfluoroether carboxylic acids (PFECAs); and
- Perfluoroether sulfonic acids (PFESAs).

2. **PFAA precursors.** These are mostly polyfluoroalkyl substances, meaning fluorine atoms have replaced all hydrogen atoms attached to at least one (but not all) carbon atoms. Polyfluorinated substances have the potential to degrade into perfluoroalkyl substances, i.e., they are precursors to perfluoroalkyl substances. See examples in Figure 2.

3. **Perfluoropolyethers (PFPEs).** These are perfluoroalkyl substances that are highly persistent in the environment, with large molecular size (oligomers, polymers, and copolymers), and ether linkages. They are unlikely to degrade to PFAAs under typical environmental conditions but may contain PFAA impurities and may release hazardous products during combustion (Nordic Council of Ministers 2020).

4. **Fluoropolymers.** These polymers are materials (as opposed to surface treatments) and are highly persistent in the environment. They cannot degrade to PFAAs under typical environmental conditions, but certain PFAAs have been used in their manufacturing and can occur as impurities. Moreover, fluoropolymers may release PFCAs, including PFOA, during combustion at temperatures between 180 and 800°C (Feng et al. 2015; Schlummer et al. 2015). Examples of fluoropolymers include polytetrafluoroethylene (PTFE), polyvinylidene fluoride (PVDF), and polyvinyl fluoride (PVF).

There are also a few PFASs that do not fall into these categories (Figure 1).

Thus, while PFAAs constitute a small subset (approximately one percent) of PFASs, they are terminal degradation products, manufacturing aids/feedstocks, or impurities of other PFAS class members, which makes their hazard traits relevant to the entire class. PFAAs and some of their precursors are frequently subdivided into longer- and shorter-chain PFASs. The longer-chain PFSAs have six or more perfluorinated carbons; longer-chain PFCAs, PFPAs, and PFPiAs have seven or more perfluorinated carbons (Bowman 2017).

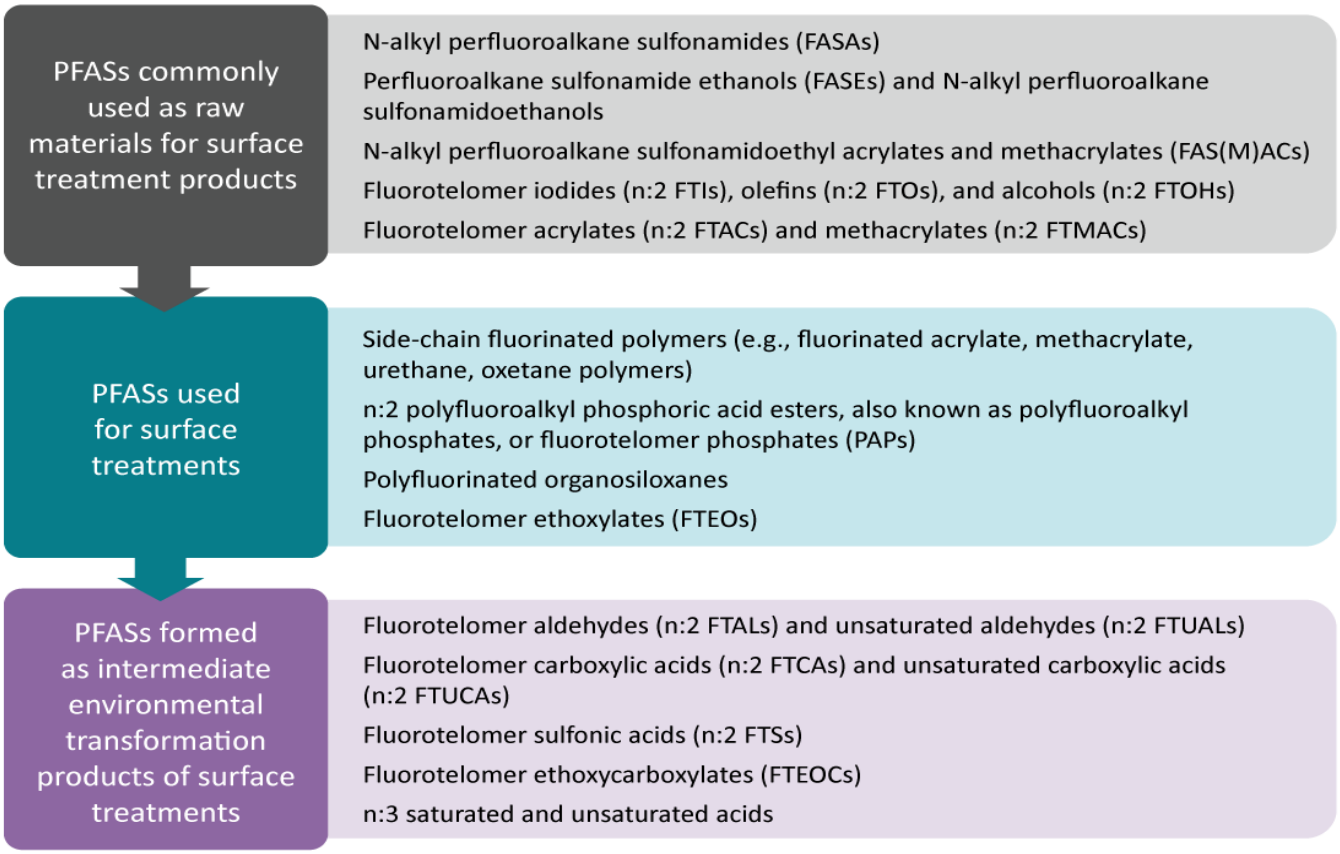

Figure 2: Examples of PFAA precursors, as described in Buck et al. (2011).

Plant fiber-based food packaging is often coated with PFASs to make it resistant to grease, oil, and water (RIVM 2019; FDA 2020a). For grease-resistant paper and paperboard applications, PFASs may be added prior to or after the sheet-forming operation (FDA 2018). Yuan et al. (2016), who measured PFAS food packaging products sampled from China and Ohio, mention that PFASs are also used in printing inks. In molded fiber food packaging products, PFASs are mixed directly into the slurry, and help not only to make the final product grease- and liquid-proof, but also to release the slurry from the mold during the manufacturing process. PFASs may also be used as mold release agents for other types of food packaging products, including some made from plastics (CEH 2018).

Through its Food Contact Notification (FCN) process, the FDA has approved 17 distinct PFAS formulations for use in plant fiber-based food packaging applications. Fifteen of these PFASs are side-chain fluorinated polymers, and two are PFPEs. A total of 28 FCNs were filed for these 17 unique PFAS formulations. Additionally, one PFAS (chromium III complex of *N*- ethyl – *N* – heptadecylfluoro-octane sulfonyl glycine) is allowed to be used in food packaging pursuant to the Code of Federal Regulations list of indirect additives (Title 21, Section 176.160), but the state of Washington has evidence suggesting that compound is no longer used. No fluorotelomer phosphate diesters (diPAPs) are currently approved by FDA for use in food-contact materials.

The PFASs currently approved for use as food-contact substances via FDA's FCN process are:

- Achroma FCN 1493 – copolymer of 2-(dimethylamino) ethyl methacrylate with 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl methacrylate, N-oxide, acetate. Intended use: grease-resistant treatment employed prior to or after the sheet-forming operation for paper and paperboard intended to contact food.
- Achroma FCN 1097 – hexane, 1,6-diisocyanato-, homopolymer, α-[1-[[[3-[[3 (dimethylamino)propyl]amino]propyl]amino]carbonyl]-1,2,2,2-tetrafluoroethyl]-ω-(1,1,2,2,3,3,3-heptafluoropropoxy)poly[oxy[trifluoro(trifluoromethyl)-1,2-ethanediyl]]. Intended use: grease-resistant treatment for paper and paperboard employed either prior to or after the sheet-forming operation.
- Asahi FCN 599/604 – copolymer of perfluorohexylethyl methacrylate, 2-N,N-diethylaminoethyl methacrylate, 2-hydroxyethyl methacrylate, and 2,2'-ethylenedioxydiethyl dimethacrylate, acetic acid salt (CAS 863408-20-2) or malic acid salt (CAS 1225273-44-8). Intended use: oil, grease, and water-resistant treatment for paper and paperboard employed prior to the sheet-forming operation or at the size press.
- Asahi FCN 1676 – 2-propenoic acid, 2-methyl-, 2-hydroxyethyl ester, polymer with 2-propenoic acid and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-methyl-2-propenoate, sodium salt (CAS 1878204-24-0). Intended use: oil-, water-, and grease-proofing agent in paper and paperboard.
- Asahi FCN 1186 – butanedioic acid, 2-methylene-, polymer with 2-hydroxyethyl, 2-methyl-2-propenoate, 2-methyl-2-propenoic acid and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-methyl-2-propenoate, sodium salt (CAS 1345817-52-8). Intended use: oil, grease, and water-resistant treatment for paper and paperboard.
- Chemours[3] FCN 1027/885 – 2-propenoic acid, 2-methyl-, polymer with 2-(diethylamino)ethyl 2-methyl-2-propenoate, 2-propenoic acid and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-methyl-2-propenoate, acetate (CAS 1071022-26-8). Intended use: oil- and grease-resistant treatment for paper and paperboard employed either prior to or after the sheet-forming operation.
- Chemours FCN 940 – hexane, 1,6-diisocyanato-, homopolymer, 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluoro-1-octanol-blocked (CAS 357624-15-8). Intended use: oil- and grease-resistant treatment for paper and paperboard employed either prior to the sheet-forming operation or at the size press.
- Daikin FCN 1451/1360 – 2-Propenoic acid, 2-methyl-, 2-(dimethylamino)ethyl ester, polymer with 1-ethenyl-2-pyrrolidinone and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-propenoate, acetate (CAS 1334473-84-5). Intended use: added at the size press or wet end to impart grease and oil resistance to paper and paperboard.

[3] Chemours has requested for FDA to remove its PFAS-related FCNs (Neltner 2019).

- Daikin FCN 888/827 – 2-propenoic acid, 2-hydroxyethyl ester, polymer with α-(1-oxo-2-propen-1-yl)-ω-hydroxypoly(oxy-1,2-ethanediyl), α-(1-oxo-2-propen-1-yl)-ω-[(1-oxo-2-propen-1-yl)oxy]poly(oxy-1,2-ethanediyl) and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-propenoate (CAS 1012783-70-8). Intended use: at the size press as an oil- and grease-resistant treatment for paper and paperboard.
- Daikin FCN 933 – 2-propenoic acid, 2-methyl-, polymer with 2-hydroxyethyl 2-methyl-2-propenoate, α-(1-oxo-2-propen-1-yl)-ω-hydroxypoly(oxy-1,2-ethanediyl) and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-propenoate, sodium salt (CAS 1158951-86-0). Intended use: oil- and grease-resistant treatment for paper and paperboard at the size press or prior to sheet formation.
- Daikin FCN 1044 – 2-propenoic acid, 2-methyl-, 2-hydroxyethyl ester polymer with 1-ethyenyl-2-pyrrolidinone, 2-propenoic acid and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-propenoate sodium salt (CAS 1206450-10-3). Intended use: added at the size press or prior to sheet formation to impart grease and oil resistance to paper and paperboard.
- Daikin FCN 820 – 2-Propenoic acid, 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl ester, polymer with α-(1-oxo-2-propen-1-yl)-ω-hydroxypoly(oxy-1,2-ethanediyl). Intended use: oil- or grease-resistant treatment for paper and paperboard intended for food-contact use.
- Solenis FCN 518/487/314 – 2-propen-1-ol, reaction products with pentafluoroiodoethane-tetrafluoroethylene telomer, dehydroiodinated, reaction products with epichlorohydrin and triethylenetetramine (CAS 464178-90-3). Intended use: oil- or grease-resistant sizing agent, either prior to the sheet-forming operation or at the size press for paper and paperboard; in microwave heat-susceptor packaging or for single-use applications.
- Solenis FCN 783/746/542 – 2-propen-1-ol, reaction products with 1,1,1,2,2,3,3,4,4,5,5,6,6-tridecafluoro-6-iodohexane, dehydroiodinated, reaction products with epichlorohydrin and triethylenetetramine (CAS 464178-94-7) as manufactured in accordance with the description in the FCN. Intended use: oil- and grease-resistant sizing agent prior to the sheet-forming operation in the manufacture of paper and paperboard.
- Solvay FCN 962/416/195 – diphosphoric acid, polymers with ethoxylated reduced methyl esters of reduced polymerized oxidized tetrafluoroethylene (CAS 200013-65-6). This substance is also known as phosphate esters of ethoxylated perfluoroether, prepared by reaction of ethoxylated perfluoroether diol (CAS 162492-15-1) with phosphorous pentoxide (CAS 1314-56-3) or pyrophosphoric acid (CAS 2466-09-3). Intended use: oil repellant in the manufacture of food-contact paper and paperboard.
- Solvay FCN 538/398 – perfluoropolyether dicarboxylic acid (CAS 69991-62-4), ammonium salt. Intended use: oil and water repellent in the manufacture of food-contact paper and paperboard, either prior to the sheet-forming operation or at the size press.
- Solvay FCN 187 – fluorinated polyurethane anionic resin (CAS 328389-91-9) prepared by reacting perfluoropolyether diol (CAS 88645-29-8), isophorone diisocyanate (CAS 4098-71-9),

2,2-dimethylolpropionic acid (CAS 4767-03-7), and triethylamine (CAS 121-44-8). Intended use: water and oil repellent in the manufacture of paper and paperboard.

While most of the FCNs listed above are for polymer mixtures, some report PFAAs as impurities, including PFOA. 6:2 FTOH is a major impurity in, and a degradation product of, polymers used as stain-, water-, and grease-proof coatings. Migration of the 6:2 FTOH into food is expected as a result of its use in food-contact substances (Kabadi et al. 2020; Rice et al. 2020). Impurity information was not available for many of the currently listed Food Contact Substances (FCS). Those that did contain information are listed in Table 1.

In addition to the FCN process, food-contact substances may also enter the market if they are Generally Recognized As Safe (GRAS) among qualified experts and have been adequately shown to be safe for the intended use. GRAS substances do not undergo the same review process as the FCNs, and GRAS determinations can be made directly by the manufacturer (Neltner et al. 2011). No PFASs are currently included on FDA's list of GRAS substances.

Table 1. Impurities listed in Food Contact Notifications

FCN	Chemical Name	CAS	Known Impurities	Number of Undisclosed Reagents and Impurities
1493	Copolymer of 2-(dimethylamino) ethyl methacrylate with 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl methacrylate, N-oxide, acetate	1440528-04-0	1H,1H,2H,2H-Perfluorooctyl methacrylate and 2-Dimethylaminoethyl methacrylate	10
1186	butanedioic acid, 2-methylene-, polymer with 2-hydroxyethyl, 2-methyl-2-propenoate, 2-methyl-2-propenoic acid and 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl 2-methyl-2-propenoate, sodium salt	Unknown	2-propenoic acid, 2-methyl-, 3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluoro octyl ester; 2-propenoic acid, 2-methyl-, 2-hydroxyethyl ester, 2-methylpropenoic acid, and butanedioic acid, 2-methylene	11
416	diphosphoric acid, polymers with ethoxylated reduced methyl esters of reduced polymerized oxidized tetrafluoroethylene	Unknown	ethoxylated perfluoroether diol, water, phosphorous pentoxide, ethyl acetate, isobutyl alcohol, tetrafluoroethylene, ethylene oxide, 1,4-dioxane, phosphoric acid, ethanol, and acetic acid	0

Relevant physicochemical properties

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(D).

Physicochemical properties can be helpful in predicting a chemical's behavior. A chemical's behavior in humans, wildlife, ecosystems, and the environment may indicate potential adverse public health and environmental impacts.

PFASs possess unique physicochemical characteristics that confer increased stability in extreme temperatures, antistatic properties, and resistance to wettability, staining, and corrosion (Buck et al. 2011; Buck et al. 2012). PFASs are used in a wide variety of applications, including (KEMI 2015; FluoroCouncil 2020).

- Surface treatments for leather and textiles
- Surface treatments for food packaging, processing, and preparation
- Metal (chromium) plating fume suppressants
- Detergents, pharmaceuticals, pesticides
- Aqueous firefighting foams (AFFFs)
- Aerospace, automotive, construction, and electronics manufacturing industries

These unique properties of PFASs also contribute to their environmental persistence, global distribution, and accumulation in water, soils, plants, and animals. Compared to other organic chemicals, PFASs are very persistent in the environment and biota, partly because fluorine (F) forms the strongest single bond with carbon (Kiplinger et al. 1994). Due to its high electronegativity, the F atom pulls the shared electrons of the covalent C-F bond towards itself, which renders it partially negative and the C atom partially positive, creating a polar covalent bond. This bond requires high energy, up to 544 kJ/mol, to dissociate (Lemal 2004), compared to the 337 kJ/mol needed to separate carbon and hydrogen (Dean 1999). Consequently, PFASs are thermally, chemically, and biologically stable, and resistant to abiotic degradation such as atmospheric photo-oxidation, direct photolysis, and hydrolysis (Wang et al. 2016). Perfluoroethers are more thermally stable than other PFASs due to their strong C-O bonds (Ceretta 2013). Polymeric PFASs also have high thermal, chemical, aging, and weather resistance, and are inert to solvents, hydrocarbons, acids and bases (Banks et al. 1994). Even though bacteria and fungi can degrade organic chemicals by utilizing the carbon present in the compounds as an energy source, the F atoms in PFASs surround and shield the carbon backbone, so organisms are largely unable to fully biodegrade these chemicals (Frömel and Knepper 2010). PFASs that can degrade in the environment typically transform into smaller PFASs that do not degrade further under environmental conditions, such as PFAAs (Buck et al. 2011; D'eon and Mabury 2011; Frömel et al. 2010; Krafft and Riess 2015a; Wang et al. 2014a; Washington et al. 2015).

For most PFASs, physicochemical properties data are absent from the public domain. The data available in the literature can range widely depending on the measurement methods or model used for the estimates. Table 2 lists data available for PFAAs. Less is known about PFECAs and PFESAs, but despite structural differences, their physicochemical properties are believed to be similar to those of their PFCA or PFSA counterparts, according to modeling studies (Gomis et al. 2015). In particular, they appear to be similarly persistent and mobile in the environment as the longer-chain PFASs (Gomis et al. 2015).

PFAAs are proteinophilic (protein-binding), accumulating particularly in blood, liver, stomach, kidney, lung, gall bladder, brain, muscle, and yolk sac tissues.

Most PFASs are solid at room temperature. Notable exceptions include 6:2 FTOH and 4:2 FTOH, which exist as a liquid at room temperature, and 8:2 FTOH, which exists as a solid but can sublimate and volatilize at room temperature (Concawe 2016). The boiling points of PFASs are similar to those of the homologous hydrocarbons (i.e., molecules with the same structure but only carbon-hydrogen bonds instead of carbon-fluorine bonds), except for perfluoroethers and perfluoroketones, which boil at significantly lower temperatures compared to their hydrocarbon homologues due to lower intermolecular forces (Ceretta 2013).

Most PFASs exhibit low vapor pressures, but compared to homologous hydrocarbons they tend to be similarly or somewhat more volatile, despite higher molecular weights (Krafft et al. 2015a). Most PFAAs are semi-volatile and can adsorb onto indoor dust (NILU 2015; Dreyer et al. 2015; Ericson et al. 2012; Fraser et al. 2013; Haug et al. 2011; Knobeloch et al. 2012; Liu et al. 2014; Liu et al. 2015a; Strynar and Lindstrom 2008). Some of their precursors however, including FTUCAs and FTOHs, tend to be more volatile and can undergo atmospheric long-range transport (Ding and Peijnenburg 2013). In general, shorter-chain PFASs are more volatile than longer chains, and thus more likely to be released from products (U.S. EPA 2020a).

PFASs display a range of solubility in water, from sparingly soluble to miscible, though models may significantly underestimate PFAS solubility (Xiao 2017). Solubility tends to increase with decreasing carbon chain length and molecular weight (Table 2). PFCAs and PFSAs are relatively more soluble than other PFASs of similar chain length due to the hydrophilicity of the carboxylate and sulfonate groups on these molecules. Solubility of PFCAs and PFSAs tends to decrease with increasing molecular weight due to a concomitant increase in the length of the hydrophobic perfluorinated alkyl chains (Concawe 2016). Their water solubility makes these PFASs mobile in the environment and capable of undergoing long-range transport via ocean currents (Ahrens et al. 2009a; Armitage et al. 2009; Krafft and Riess 2015b; Prevedouros et al. 2006). Model results suggest, however, that PFPiAs are highly hydrophobic; therefore, sediments may be an important sink for PFPiAs in the aquatic environment (Xiao 2017). PFAAs ionize in

water at environmentally relevant pH, according to their negative pK_a (Gomis et al. 2015; Wang et al. 2017b). For PFPAs, full deprotonation (i.e., two negative charges) occurs at a pH of 5.13-7.11 (Xiao 2017).

The bonds between the carbon-fluorine (CF_3 and CF_2) groups present in PFASs create very low critical surface energy, resulting in lipophobic and hydrophobic characteristics. This means PFASs repel both oil and water (Audenaert et al. 1999), making them surface active agents (surfactants) useful for resisting stains from soils, liquids and grease on the surface of carpets, rugs, and other textiles. Due to their low critical surface energy, PFASs tend to accumulate and form micelles at liquid (e.g., water)/air phase boundaries (Concawe 2016; Eftaiha et al. 2012; Mason Chemical Company 2017).

Their hydrophilic and hydrophobic properties pose challenges to determining PFAS partitioning coefficients experimentally, leading to reliance on modeled data and estimates (Shoeib et al. 2004). For example, K_{ow}, which represents a chemical's partitioning behavior between octanol and water, cannot be determined experimentally for PFASs because they form a third phase in addition to octanol and water. All PFAAs listed in Table 2 except for PFBA, PFPeA, PFBS, PFBPA, and PFHxPA have modeled log K_{ow} >4, which means they are considered bioaccumulative according to the California Code of Regulations, Title 22, Division 4.5, Chapter 54, Article 5. However, K_{ow}, a common screening criterion of a neutral compound's ability to partition from water into lipid-rich tissues within an organism (Mackay and Fraser 2000), has limited applicability to PFAAs, due to their unique properties. This is because, in general, PFASs have a low potential to accumulate in adipose (lipid) tissues, unlike most other persistent organic pollutants. PFAAs are proteinophilic (protein-binding), accumulating particularly in blood, liver, stomach, kidney, lung, gall bladder, brain, muscle, and yolk sac tissues (Chen and Guo 2009; Hebert and MacManus-Spencer 2010; Greaves et al. 2012; Jones et al. 2003; Mackay et al. 2000; Pérez et al. 2013). Additionally, the concept of K_{ow} for PFAAs has little to no practical meaning or usefulness because PFAAs exist in anion rather than neutral form at environmentally relevant conditions (Valsecchi et al. 2017).

Data on the physicochemical properties of PFASs are limited to mostly model results, which vary widely because the underlying data set is based on hydrocarbons. Models and experimental data that can adequately characterize PFASs, especially beyond longer-chain PFCAs and PFSAs, are still lacking (Xiao 2017).

Table 2. Physicochemical properties of some PFAAs[4]

Chemical Name	Water Solubility (20 – 25 °C) [g/L]	Melting Point [°C]	Boiling Point [°C]	Vapor Pressure [Pa]	log K_{ow} [-]	log K_{oc} [L/kg]	K_d (pH 7)	Dissociation Constant (pK_a)	Soil Adsorption Coefficient (L/kg)
Perfluorobutanoic acid (PFBA)	Miscible	-18	121	1307	2.8	1.9	--	-0.2 to 0.7	48*
Perfluoropentanoic acid (PFPeA)	113	14*	124	1057	3.4	1.4	--	-0.06	96*
Perfluorohexanoic acid (PFHxA)	22	14	143	457	4.1	1.9	--	-0.13	1,070*
Perfluoroheptanoic acid (PFHpA)	4.2	30	175	158	4.7	2.2	0.4 – 1.1	-0.15	2,110*
Perfluorooctanoic acid (PFOA)	3.4 – 9.5	37 – 60	188 – 192	4 – 1300	5.3	1.3 – 2.4	0 – 3.4	-0.16 to 3.8	1,160*

[4] Data excluding those marked with an asterisk are from peer-reviewed literature as summarized by Concawe (2016) Environmental fate and effects of poly and perfluoroalkyl substances (PFAS). Concawe Soil and Groundwater Taskforce, Brussels, Belgium. Prepared by ARCADIS: T. Pancras, G. Schrauwen, T. Held, K. Baker, I. Ross, H. Slenders, Report # 8/16. Accessed March 15, 2017. Concawe (www.concawe.eu) is a division of the European Petroleum Refiners Association based in Brussels, Belgium. Entries marked with an asterisk (*) were predicted using QSAR models by U.S. EPA (2017a) U.S. Environmental Protection Agency (U.S. EPA): Chemistry Dashboard. Available at: https://comptox.epa.gov/dashboard. Data are calculated and updated regularly by OPERA [OPEn (quantitative) Structure-activity Relationship Application], a standalone free and open source command line application in Matlab (Version 8.2) providing QSAR models predictions. Model validation data set may be found here: http://esc.syrres.com/interkow/EpiSuiteData.htm. Accessed November 6, 2017.

Chemical Name	Water Solubility (20 – 25 °C) [g/L]	Melting Point [°C]	Boiling Point [°C]	Vapor Pressure [Pa]	log K_{ow} [-]	log K_{oc} [L/kg]	K_d (pH 7)	Dissociation Constant (pK_a)	Soil Adsorption Coefficient (L/kg)
Perfluorononanoic acid (PFNA)	9.5	59 – 66	218	1.3	5.9	2.4	2.6 – 5.9	-0.17	2,830*
Perfluorodecanoic acid (PFDA)	9.5	77 – 88	218	0.2	6.5	2.8	2.0 – 31	-0.17	397*
Perfluoroundecanoic acid (PFUnA)	0.004	83 – 101	160 – 230	0.1	7.2	3.3	12 – 103	-0.17	16,700*
Perfluorododecanoic acid (PFDoA)	0.0007	107 – 109	245	0.01	7.8	--	24 – 269	-0.17 to 0.8	85,500*
Perfluorotridecanoic acid (PFTrDA)	0.0002	99*	249*	0.3	8.3	--	--	--	184,000*
Perfluorotetradecanoic acid (PFTeDA)	0.00003	98*	276	0.1	8.9	--	--	--	233,000*
Perfluorobutane sulfonic acid (PFBS)	46 – 57	76 – 84	211	631	3.9	1	--	-6.0 to -5.0	288*
Perfluorohexane sulfonic acid (PFHxS)	2.3	58*	231*	59	5.2	1.8	0.6 – 3.2	-6.0 to -5.0	2,300*
Perfluorooctane sulfonic acid (PFOS)	0.52 – 0.57	54	237* to > 400	6.7	6.4	2.5 – 3.1	0.1 – 97	-6.0 to -2.6	1,460*
Perfluorodecane sulfonic acid (PFDS)	0.002	--	--	0.7	7.7	3.5	--	--	19,200*

Chemical Name	Water Solubility (20 – 25 °C) [g/L]	Melting Point [°C]	Boiling Point [°C]	Vapor Pressure [Pa]	log K_{ow} [-]	log K_{oc} [L/kg]	K_d (pH 7)	Dissociation Constant (pK_a)	Soil Adsorption Coefficient (L/kg)
Perfluorobutyl phosphonic acid (PFBPA)	14260	--	--	0.18	2.2	--	--	--	--
Perfluorohexyl phosphonic acid (PFHxPA)	515	--	--	0.04	3.5	--	--	--	--
Perfluorooctyl phosphonic acid (PFOPA)	25	96*	210*	0.01 5.7e-03*	4.7 4.8*	--	--	--	658*
Perfluorodecyl phosphonic acid (PFDPA)	0.5	87*	210*	0.0002 1.5e-03*	6.0 3.9*	--	--	--	18,100*
Perfluorobutyl phosphinic acid (PFBPiA)	2.3e-06 to 8.2e-03*	-9.9 to 70.1*	208 to 245*	6.8e-07 to 1.2e-02*	2.4 to 11.9*	--	--	--	1,390*
Perfluorohexyl phosphinic acid (PFHxPiA)	3.6e-07 to 3.1e-04*	-25.1 to 106*	213 to 301*	8.5e-11 to 9.6e-03*	4.4 to 15.1*	--	--	--	231,000*
Perfluorooctyl phosphinic acid (PFOPiA)	1.7e-09 to 2.8e-04*	-9.9 to 138*	206 to 353*	9.0e-13 to 1.7e-02*	7.2 to 18.6*	--	--	--	134,000*
Perfluorodecyl phosphinic acid (PFDPiA)	2.9e-08*	10.4*	265*	1.9e-09*	7.6*	--	--	--	132,000*
Perfluorododecyl phosphinic acid (PFDoDPiA)	5.4e-07*	50.7*	290*	2.2e-09*	7.5*	--	--	--	133,000*

Environmental fate

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(E).

Environmental fate describes a chemical's mobility in environmental media, transformation (physical, chemical, or biological), or accumulation in the environment or biota. A chemical's environmental fate in air, water, soil, and living organisms relates to its exposure potential hazard traits, as defined in the California Code of Regulations, Title 22, Chapter 54.

PFASs are widespread in indoor and outdoor environments, humans, and biota. They are found even in raindrops and snowflakes, in high-altitude atmospheric wind currents (Taniyasu et al. 2013), and the deep sea (Sanchez-Vidal et al. 2015; Zhao et al. 2012). The accumulation of PFASs in different environmental compartments and living organisms appears to depend on their partitioning behavior, which depends in part on the perfluoroalkyl chain length and the functional group of the parent compound (EFSA 2011; Wang et al. 2017b).

Numerous factors, including intrinsic chemical properties combined with physical and biogeochemical processes, collectively determine the occurrence and fate of PFASs (ITRC 2020a). Longer-chain PFASs bioaccumulate up the food chain (Concawe 2016). Shorter-chain PFASs are particularly mobile in water and air, can undergo long-range transport, and are difficult to remove from contaminated water (Brendel et al. 2018; Li et al. 2020a). PFAAs are the terminal environmental degradation products of most PFASs, and all are extremely persistent in the environment (Wang et al. 2017b). Shorter-chain PFAAs are distributed globally by oceanic currents and sea spray, or by the degradation of volatile precursors that undergo long-range atmospheric transport (Ateia et al. 2019a). Long-range atmospheric transport followed by wet or dry deposition has led to the ubiquitous presence of PFBA and other PFAAs in snow, water, soil, and sediment samples from a remote alpine nature reserve in China (Wang et al. 2019).

Some studies distinguish between PFAAs entering the environment from direct and indirect sources (Buck et al. 2011; Prevedouros et al. 2006). "Direct" sources refer to PFAA emissions from the life cycle of products containing PFAAs or their derivatives as ingredients, unreacted raw materials, or unintended by-products; "indirect" sources refer to PFAA formation from the degradation of precursors (OECD 2013). Monitoring data suggest that a large number of PFASs are PFAA precursors, meaning they can transform into PFAAs in the environment (Ahrens 2011; ITRC 2020a).

Regardless of their sources, due to their unique physicochemical properties, PFASs released to the environment end up virtually everywhere, including air, dust, wastewater treatment plant (WWTP) effluent, biosolids, soil, inland and ocean waters, drinking water, and food (Lindstrom et al. 2011b; Wang et al. 2016). PFAAs and their precursors are found in the deep ocean and in underground aquifers, in rainwater, snow, ice, glaciers, and pristine Arctic lakes, far from any point source (Kim and Kannan 2007; Kirchgeorg et al. 2016; Muir et al. 2019; Prevedouros et al. 2006; Zhao et al. 2012).

This potential for long-range atmospheric transport followed by deposition onto land and water has resulted in PFAS-contaminated food chains and accumulation in plants and animals, even in remote Arctic and Antarctic regions.

Wildlife and humans are often sinks of environmental PFASs (Cariou et al. 2015; Chang et al. 2014; Fraser et al. 2013; Giesy and Kannan 2001; Kang et al. 2016; Liu et al. 2011a; Monroy et al. 2008; OECD 2013; Olsen et al. 2007a; Tao et al. 2008a; Tao et al. 2008b).

Volatile PFAA precursors, including FTOHs, are distributed globally in the gas phase or adsorbed to airborne particles (Dreyer et al. 2015; Liu et al. 2015a; Wang et al. 2014b), and can re-volatilize back into the atmosphere from water or snow (Xie et al. 2015). PFAAs can become airborne and deposit onto water surfaces (Kim et al. 2007). Global distribution also occurs due to movement of oceanic currents, with PFHxA and PFBS being among the most frequently detected compounds in seawater (Zhao et al. 2012). Strong inverse correlations between salinity and the concentrations of most PFASs measured indicate that ongoing continental discharges are the major sources of PFASs to the marine environment (Zhang et al. 2019c).

Although a considerable number of studies document levels of some PFASs in the environment and various biological media, knowledge regarding the sorption/desorption patterns and mechanisms of transport, transformation, and fate of most PFASs is limited. Relatively few studies are available on PFASs in sediments, as environmental studies worldwide have focused more attention on the quantities of PFASs in biota and water, and less on sediments and their partitioning behavior and fate (Nakata et al. 2006; Zareitalabad et al. 2013). Data on persistence and half-life estimates in biological and environmental media are also limited, and are sometimes conflicting or inaccurate. Half-lives are estimated based on acute exposures rather than on the low-dose chronic exposures experienced by most people and biota. Major data and knowledge gaps remain regarding the sources, transformation, distribution, accumulation, and temporal and spatial trends of various PFASs in wildlife and humans.

The following sections summarize publicly available data on the behavior of PFASs in different environmental compartments and sinks, without following a particular pathway, since the transport and cycling mechanisms of all PFASs in the environment are not fully understood. The available environmental fate data are limited to PFAAs and a few of their precursors. However, since PFAAs are used in the manufacturing of fluoropolymers and PFPEs, the following discussion can be relevant to the life cycle of all PFASs.

PFASs in air

PFASs display a wide range of airborne behaviors. Some PFASs, such as FTOHs, are relatively volatile and detected in indoor air and dust (Ericson et al. 2012; Fraser et al. 2013; Karásková et al. 2016).

Release of these PFASs into air can occur during chemical manufacturing, consumer product manufacturing, product use, around WWTPs, and under landfill conditions after disposal (Ahrens 2011; Ahrens et al. 2011a). The most volatile PFASs are PFAA precursors, which can oxidize to PFAAs in the atmosphere (Ellis et al. 2004; Wallington et al. 2006). PFAAs are semi-volatile and dissociate in water, as the perfluoroalkyl chain forms micelles on the water surface and partitions into the air (Eftaiha et al. 2012; ITRC 2020a; Mason Chemical Company 2017).

Once PFASs enter the air, they can be carried long distances by air currents or adsorb to particulate matter (Barber et al. 2007; Dreyer et al. 2015; Liu et al. 2015a; Shoeib et al. 2010; Wang et al. 2014b), which may be directly inhaled or deposited in rain and snow (Casal et al. 2017; Sammut et al. 2017; Taniyasu et al. 2013). Neutral volatile PFAS precursors, such as FTOHs, are the dominant PFASs found in the gas phase (Ahrens et al. 2012), whereas ionic PFASs, such as PFOA and PFOS, characterized by low vapor pressure and high water solubility, tend to dominate in airborne particulate matter (ITRC 2020a). According to the limited publicly available data, PFASs have long atmospheric half-lives, for instance approximately 20 days for 8:2 FTOH (Ellis et al. 2003).

Due to their potential for long-range atmospheric transport followed by deposition onto land and water, PFASs have contaminated food chains and accumulated in plants and animals, even in remote Arctic and Antarctic regions (Houde et al. 2006; Butt et al. 2010; Casal et al. 2017; Muir et al. 2019; Roscales et al. 2019). Despite the U.S.-wide phaseout of longer-chain PFASs, atmospheric emissions of these compounds from offshore manufacturing may still impact the United States (U.S.). Trans-Pacific atmospheric transport from China leading to adverse impacts on air quality in California has previously been documented for other pollutants (Lin et al. 2014b). Considering the large number of PFASs in commerce today, their continuous air emissions, and their relatively long atmospheric half-lives, atmospheric transport is of significant concern due to the potential for widespread PFAS environmental distribution and contamination impacting humans and other organisms.

PFASs in water

PFASs enter surface water and groundwater through a variety of routes, including industrial discharges from manufacturing sites, WWTP effluent, runoff and leaching from contaminated soil or landfills, deposition of contaminated particulate matter onto water bodies, and glacial meltwater. Their individual solubility and partitioning behavior, combined with the physical and chemical characteristics of receiving waters, determine whether specific PFASs are found dissolved or suspended in the water column, in micelles at the air/water interface (Eftaiha et al. 2012; Mason Chemical Company 2017), or adsorbed to solid particles (Campo et al. 2016). Longer-chain PFAAs tend to partition into suspended particles and sediments, while shorter-chain PFAAs tend to remain in the dissolved phase (Chen et al. 2019).

The ubiquitous presence of PFASs in surface water, groundwater, oceans, sediments, and air has also resulted in drinking water contamination, especially with the more soluble and mobile shorter-chain PFAAs.

PFAAs can contaminate groundwater through groundwater recharge (Liu et al. 2016) or be transported to the oceans where they become globally distributed by ocean currents (Benskin et al. 2012a). Stormwater runoff may contribute significantly to the PFAS load in surface waters (Wilkinson et al. 2017; Zushi and Masunaga 2009). During the treatment of municipal and industrial wastewater, volatile PFASs can be released into the air, while others selectively partition into biosolids and are partially removed from wastewater. WWTP effluent tends to have higher concentrations of PFAAs than the influent because of the transformation of precursors during the treatment process (Appleman et al. 2014; Arvaniti and Stasinakis 2015b; Sinclair and Kannan 2006). PFASs also enter surface water and groundwater via landfill leachate (Ahrens et al. 2016a; Banzhaf et al. 2017).

PFAAs appear to not degrade under environmental conditions, even in activated sewage sludge, and can be more persistent than other legacy persistent organic pollutants such as dichlorodiphenyltricloroethane (DDT), polychlorinated biphenyls (PCBs), and dieldrin (EWG 2003; European Commission 2017). Like other persistent organic pollutants, oceans are considered the ultimate long-term reservoir for PFASs (Prevedouros et al. 2006), including PFAAs and their precursors (González-Gaya et al. 2014). Marine organisms are chronically exposed via contaminated food chains (Ahrens et al. 2009a; Casal et al. 2017; Houde et al. 2011; Xiao 2017). Many physical and biogeochemical processes and factors collectively determine the oceanic occurrence and fate of PFASs, including distance to coastal urban industrial regions, oceanic subtropical gyres, currents, and biogeochemical processes (González-Gaya et al. 2014). PFASs are also transported downwards from the ocean surface mainly by sinking zooplankton fecal pellets and decaying phytoplankton, but these removal mechanisms are slow (González-Gaya et al. 2019).Oceanic currents can transport PFASs over long distances, resulting in PFAS contamination of water in distant pristine regions of the world, far from industrial activities (Prevedouros et al. 2006; Ahrens et al. 2011b).

Snow and ice in the northern latitudes also accumulate PFASs, including the shorter chains (Del Vento et al. 2012). Climate change resulting in snow, ice, and permafrost melt can remobilize PFASs and increase their concentrations in coastal areas (Zhao et al. 2012).

The ubiquitous presence of PFASs in surface water, groundwater, oceans, sediments, and air has also resulted in drinking water contamination, especially with the more soluble and mobile shorter-chain PFAAs (Boone et al. 2019; Gellrich et al. 2013). Shorter-chain PFASs are generally more mobile in groundwater and will leach faster from soil to groundwater (NGWA 2017). Contamination of drinking

water with PFASs and irrigation of food crops with contaminated water is generally considered a major source of human exposure, as discussed in subsequent sections of this document.

PFASs in sediments

Sediments are an important sink and reservoir of PFASs in the aquatic environment (Chen et al. 2016; Pan et al. 2015). Freshwater and marine sediments can become contaminated when PFASs dissolved in water adsorb onto particulate matter and are deposited. PFASs exhibit a range of sediment-water partitioning coefficients that influence their environmental fate. Long-term production and use patterns for PFASs are documented in sediment cores (Codling et al. 2014). Longer-chain PFASs such as PFOS and PFOA are typically the predominant PFASs found in surface sediments (Rankin et al. 2016). Spatial distribution of PFASs in sediments and the occurrence of "hot spots" in sampling areas can provide evidence of specific manufacturing activities and point source discharges (Munoz et al. 2015; White et al. 2015). Monitoring studies (see Chapter 4) suggest that urbanization and industrial activities contribute to PFAS accumulation in sediments, particularly in coastal regions and estuaries near urban centers. PFAS-contaminated coastal waters can be carried to deep sea regions through deep shelf water cascading events (Sanchez-Vidal et al. 2015).

PFASs in soil

PFASs can contaminate the soil via atmospheric transport followed by deposition, or directly through application of biosolids (Sepulvado et al. 2011) , soil conditioners, or other treatments. From soil, they can migrate into surface water or groundwater, including drinking water sources (Lilienthal et al. 2017). The sorption of PFASs to soil is influenced by several factors, including soil organic matter content, clay content, pH, and the presence of cations (Higgins and Luthy 2006; Li et al. 2018). The fate and transport of a specific PFAS in soil depends on its physicochemical properties and is site-specific (Mejia-Avendaño et al. 2020). Longer-chain PFASs tend to partition preferentially into soil (Dalahmeh et al. 2018), while shorter-chain PFASs partition into soil water (Ahrens et al. 2009b; Guo et al. 2015). Longer-chain PFAAs added to an agricultural soil were found to sorb strongly to the soil, whereas the shorter-chain PFAAs were readily transported in water from the soil surface to the root zone and into the groundwater (McLachlan et al. 2019; Muir et al. 2019). Shorter-chain PFCAs have also been observed to migrate out of soil following biosolid application, with disappearance half-lives ranging from 88 to 866 days (Venkatesan and Halden 2014a). According to the study authors, the fate of these PFCAs after loss from soils could include leaching, plant uptake, and volatilization (Zhang et al. 2019a).

Publicly available data on the half-lives of PFASs in soils are scarce and vary widely across models and experiments. For example, one industry study reports a half-life of 1,200 to 1,700 years for the biodegradation of a side-chain fluoroacrylate polymer to PFOA in aerobic soil (Russell et al. 2008). A team of U.S. Environmental Protection Agency (EPA) scientists (Washington et al. 2009) pointed out several issues with that study and arrived at an estimated half-life for fluoroacrylate polymers of 10-17

years based on experimental results and modeling. Subsequently, they evaluated the degradability of two commercial side-chain fluoroacrylate polymers in saturated soil and found half-lives of 33 to 112 years (Washington et al. 2015). Russell et al. (2010) estimated the half-life in soil of a side-chain fluorourethane polymer to be 28 to 241 years. More recently, U.S. EPA scientists calculated an environmental half-life for these polymers between nine and 60 years and concluded that the side-chain fluorinated polymers "now in production might constitute considerable sources to the environment of the new generation of PFASs" (Washington et al. 2019). Dasu and Lee (2016) studied the inherent biotransformation potential of two urethane monomers and found negligible degradation in agricultural soil, but significant degradation in forest soil, with an estimated half-life ranging between three and 22.2 months. Royer et al. (2015) studied the biotransformation of 8:2 FTAC and 8:2 FTMAC to 8:2 FTOH in soils, finding half-lives of five days or less and 15 days, respectively.

PFASs in biota

Plants and animals can bioaccumulate PFASs, with large interspecies differences, particularly between aquatic and terrestrial organisms (Concawe 2016). PFAS uptake has been documented in fruits and vegetables grown in backyard and garden soils irrigated with contaminated water (Scher et al. 2018), agricultural soils spiked with PFASs (McLachlan et al. 2019), potted soils spiked with PFASs (Stahl et al. 2009; Zhang et al. 2019a), and biosolids-amended soil (Blaine et al. 2013; Blaine et al. 2014a; Yoo et al. 2011). Crops grown in agricultural soils without significant PFAS contamination were found to contain small levels of mainly shorter-chain PFASs from rainwater and atmospheric deposition (Eun et al. 2020). Longer-chain PFAAs, especially PFSAs, tend to partition in the roots, whereas the more soluble, shorter-chain PFAAs, especially PFCAs, partition in other parts of the plant (Blaine et al. 2013; Blaine et al. 2014a; Gobelius et al. 2017; Lechner and Knapp 2011; Scher et al. 2018; Stahl et al. 2009; Yoo et al. 2011). Chen et al. (2020) found evidence that the PFECA GenX can bioaccumulate in plants and translocate into different plant tissues, depending on the plant species. Studies have also reported PFAA uptake in tree leaves, possibly from a combination of soil uptake and atmospheric deposition, in an urban environment (Zhang et al. 2015) and near industrial emission sources (Shan et al. 2014). PFAS absorption by plants depends on the PFAS concentration, chain length and functional group; the plant species or cultivar; and the characteristics and amendments of the soil or growth media (Ghisi et al. 2019).

The extent of bioaccumulation is traditionally measured using several different indices. The bioconcentration factor (BCF) is a measure of the amount of a contaminant found in an aquatic organism compared to the amount found in water. The biomagnification factor (BMF) compares the amount of the contaminant in the organism to that in its diet. The bioaccumulation factor (BAF) combines uptake from all sources, including water, diet, and environmental exposures (Arnot and Gobas 2006). Chemicals with BAF or BCF values greater than 1,000 or BMF greater than 1 meet the bioaccumulation hazard trait criteria outlined in the California Code of Regulations, title 22, Division

4.5, Chapter 54, Article 5. However, these traditional measures of bioaccumulation have limited applicability when gauging the potential for PFAS exposures to cause adverse health effects (Cousins et al. 2016). Although the elimination kinetics and toxicity of some single PFASs have been experimentally determined, the ubiquity and extreme persistence of PFASs in the environment lead to continuous internal exposures to mixtures of PFASs, which have not been adequately assessed (Wang et al. 2017b).

Table 3(a-c) summarizes BCFs, BMFs, and BAFs for some PFASs with publicly available data. The BCFs of some longer-chain PFAAs approach 1,000, and a few – PFOS and perfluorohexadecanoic acid (PFHxDA) – exceed it. Animals tend to have greater concentrations of PFASs in their bodies compared to their diets. Earthworms were found to accumulate PFASs, including shorter-chain compounds, from soil, with BAFs ranging from 1.1 to 96 depending on the compound (Munoz et al. 2020). BAFs in aquatic organisms depend on the type and concentration of specific PFASs in water and are generally proportional to the length of the carbon chain or molecular weight, with PFSAs showing higher bioaccumulation than PFCAs of the same chain length – although there is interspecies variation (Hong et al. 2015). However, this relationship between PFAS chain length and bioaccumulation is reversed in aquatic and terrestrial plants. For instance, Atlantic Ocean plankton displays higher BAFs for PFPeA and PFHxA than for PFOA, possibly due to uptake and metabolism of precursor compounds (Zhang et al. 2019c). BAFs for PFCAs in vegetables grown near a fluorochemical industrial park in China also decreased with increasing chain length (Zhang et al. 2020).

Once ingested via food or drinking water, some PFASs remain in the human body for years. Limited studies suggest that serum elimination half-life can range from a few days for shorter-chain PFASs (Chang et al. 2008; Nilsson et al. 2010; Olsen et al. 2007b) to several years for longer-chain PFASs (Bartell et al. 2010; Olsen et al. 2007b; Seals et al. 2011). It can take up to 56 years for total elimination of a chlorinated PFESA (Shi et al. 2016). Reported half-lives for PFAAs and FTOHs in other organisms such as rats, mice, and monkeys are shorter than in humans, typically hours, days, or months (Butenhoff et al. 2004; Chang et al. 2008; Chang et al. 2012; Gannon et al. 2011; Ohmori et al. 2003; Sundström et al. 2012; Tatum-Gibbs et al. 2011).

Table 3a. Examples of bioconcentration factors (BCFs)

Chemical	Organism	BCF	Reference
PFOS	Bluegill	1,866 – 4,312	Drottar (2001)
PFOS	Rainbow trout	1,100 – 5,400	Drottar (2001)
PFOS	Rainbow trout	2,900 (liver), 3,100 (blood)	Martin et al. (2003)

Chemical	Organism	BCF	Reference
PFOA	Water breathing animals	1.8 – 8.0	ECHA (2014)
PFOA	Rainbow trout	12 (liver), 25 (blood)	Martin et al. (2003)
PFHxDA	Carp	4,700 – 4,800	U.S. EPA (2009)
Perfluorooctadecanoic acid (PFODA)	Carp	320 – 430	U.S. EPA (2009)

Table 3b. Examples of bioaccumulation factors (BAFs)

Chemical	Organism	BAF	Reference
PFOS	Zooplankton/water	240	Houde et al. (2008)
PFOS	Mysis/water	1,200	Houde et al. (2008)
PFOS	Sculpin/water	95,000	Houde et al. (2008)
PFOS	Lake trout/water	16,000	Houde et al. (2008)
PFOA	Water breathing animals	0.9 – 266	ECHA (2014)
PFOA	Copepods/water	200 – 500	Munoz et al. (2019)
PFOA	Copepods/suspended solids	251	Munoz et al. (2019)
PFOA	Mysids/water	316	Munoz et al. (2019)
PFOA	Mysids/suspended solids	3.16 – 79.4	Munoz et al. (2019)
PFOA	Shrimp/water	125 – 631	Munoz et al. (2019)
PFOA	Shrimp/suspended solids	2.5 – 31.6	Munoz et al. (2019)
PFHpA	Copepods/water	200 – 500	Munoz et al. (2019)
PFHpA	Copepods/suspended solids	251	Munoz et al. (2019)

Table 3c. Examples of biomagnification factors (BMFs)

Chemical	Organism	BMF	Reference
PFOS	Arctic cod/zooplankton (Western Canadian Arctic)	8.7	Powley et al. (2008)
PFOS	Caribou/lichen (Canada)	2.0 – 9.1	Müller et al. (2011)
PFOS	Wolf/caribou (Canada)	0.8 – 4.5	Müller et al. (2011)
PFOS	Dolphin/seatrout (2 U.S. locations)	0.9	Houde et al. (2006)
PFOS	Seatrout/pinfish (2 U.S. locations)	4.6	Houde et al. (2006)
PFOS	Walrus/clam (Eastern Arctic Food Web)	4.6	Tomy et al. (2004)
PFOS	Narwhal/Arctic cod (Eastern Arctic Food Web)	7.2	Tomy et al. (2004)
PFOS	Beluga/Arctic cod (Eastern Arctic Food Web)	8.4	Tomy et al. (2004)
PFOS	Beluga/redfish (Eastern Arctic Food Web)	4	Tomy et al. (2004)
PFOS	Polar bear/seal (Canadian Arctic)	177	Martin et al. (2004)
PFOA	Water-breathing animals	0.02 – 7.2	ECHA (2014)
PFOA	Caribou/lichen (Canada)	0.9 – 11	Müller et al. (2011)
PFOA	Wolf/caribou (Canada)	0.9 – 3.8	Müller et al. (2011)
PFOA	Walrus/clam (Eastern Arctic Food Web)	1.8	Tomy et al. (2004)
PFOA	Narwhal/Arctic cod (Eastern Arctic Food Web)	1.6	Tomy et al. (2004)
PFOA	Beluga/Arctic cod (Eastern Arctic Food Web)	2.7	Tomy et al. (2004)
PFOA	Beluga/redfish (Eastern Arctic Food Web)	0.8	Tomy et al. (2004)
PFOA	Beluga whale/Pacific herring (Western Canadian Arctic Food Web)	1.3	Tomy et al. (2004)
PFOA	Arctic cod/marine Arctic copepod (Western Canadian Arctic Food Web)	2.2	Tomy et al. (2004)
PFOA	Dolphin/seatrout (2 U.S. locations)	1.8	Houde et al. (2006)
PFOA	Seatrout/pinfish (2 U.S. locations)	7.2	Houde et al. (2006)
PFOA	Polar bears/ringed seal (2 U.S. locations)	45 – 125	Butt et al. (2008)
PFOA	Polar bear/seal (Canadian Arctic)	8.6	Martin et al. (2004)

Degradation, reaction, or metabolic products of concern

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(G).

A Candidate Chemical may degrade, form reaction products, or metabolize into other chemicals that have one or more hazard traits. These metabolites, degradation products, and reaction products (which may or may not be Candidate Chemicals) may cause different adverse impacts from those of the parent chemical. In some cases, a Candidate Chemical's degradation or reaction products or metabolites may have the same hazard trait, and may be more potent or more environmentally persistent, or both, than the parent chemical. In such cases, adverse impacts may be more severe, or may continue long after, the Candidate Chemical's release to the environment.

Of the 4,730 individual PFASs or commercial PFAS mixtures identified by OECD on the global market, 4,186 likely degrade to PFAAs in the environment or biota (OECD 2018). PFASs that degrade into PFAAs are termed PFAA precursors and provide a significant indirect source of PFAAs to the environment (see examples of PFAA precursors in Chapter 3). Degradation to PFAAs can occur, for instance, in the atmosphere, consumer products, landfills, or WWTPs, mainly via hydroxylation or photo-oxidation (Nielsen 2014; Prevedouros et al. 2006; Hamid et al. 2020). The degradation half-life of PFAA precursors can range from days to centuries (Li et al. 2017; Nielsen 2014; Washington et al. 2015).

PFAA precursors can persist in the environment and in biological compartments for various lengths of time, but eventually degrade through biotic and abiotic transformation to PFAAs, via intermediate metabolites suspected of high acute toxicity (Butt et al. 2014; D'eon and Mabury 2007; Rand and Mabury 2012a; Rand and Mabury 2012b; Rand and Mabury 2013; Rand and Mabury 2014b; Rand and Mabury 2017). Fluorotelomer-based compounds degrade (Phillips et al. 2007; Young et al. 2007) or undergo biotransformation into PFCAs (Chen et al. 2017b; Dasu et al. 2016; Kim et al. 2012a; Kim et al. 2014b; Lee et al. 2010; Liu et al. 2010a; Liu et al. 2010b; Rand et al. 2014b; Royer et al. 2015; Ruan et al. 2014; Russell et al. 2015; Zhao and Zhu 2017b; Hamid et al. 2020). The intermediates in this process include saturated and unsaturated fluorotelomer aldehydes (FTALs and FTUALs, respectively), and saturated and unsaturated fluorotelomer carboxylic acids (FTCAs and FTUCAs, respectively) (Frömel et al. 2016; Nilsson et al. 2013; Rand et al. 2014a; Hamid et al. 2020). FTUCAs were detected in all 81 indoor air samples collected in 2015 during a study in China, suggesting the transformation of PFAA precursors such as FTOHs in the indoor environment (Yao et al. 2018). Pharmacokinetic data from rat and human studies on 6:2 FTOH revealed that, of its main metabolites, the 5:3 fluorotelomer carboxylic acid has the highest internal exposure and slowest clearance, which decreases with increasing exposure (Kabadi et al. 2018). The study also found that 5:3 FTCA may reach steady state following repeated exposure to 6:2 FTOH, therefore resulting in biopersistence, with potential for systemic toxicity (Kabadi et al. 2018). In rat plasma, liver, and fat, 5:3 FTCA reaches steady state after approximately one year following repeated oral doses of 6:2 FTOH (Kabadi et al. 2020).

Laboratory studies of fluorotelomer carboxylic acid and aldehyde intermediates indicate they are more acutely toxic than PFCAs, including PFOA. One study found that FTCAs and FTUCAs are one to five orders of magnitude more toxic to freshwater organisms than PFCAs (Phillips et al. 2007). In a human liver cell toxicity assay, FTUALs were found to be up to 200 times more toxic than FTUCAs, FTCAs, and PFCAs (Rand et al. 2014a).

Side-chain fluorinated polymers, which are fluorotelomer-based, can also degrade to FTOHs or other fluorotelomer compounds, with PFCAs as terminal degradation products (Washington et al. 2015; Washington et al. 2019). Because of their potential to be mobilized, interact with their surrounding environment, and degrade (KEMI 2015; Russell et al. 2010), side-chain fluorinated polymers are sometimes referred to as "functionalized oligomers" to distinguish them from true polymers.

Intentional or accidental combustion of PFASs also forms hazardous chemicals. A literature survey regarding the combustion of various fluorinated polymers reported emissions of C3-C14 PFCAs, ozone depleting substances such as chlorofluorocarbons (CFCs), and greenhouse gases such as fluorocarbons when fluoropolymers are combusted at temperatures representative of municipal incinerators (Huber et al. 2009). In a review of how fluoropolymers differ from nonpolymeric PFASs, the authors acknowledged that end-of-life considerations should be further investigated because hazardous substances may arise from the combustion of fluoropolymers (Henry et al. 2018). For instance, during incineration at temperatures above 450°C, PTFE forms additional hazardous substances including the ultra-short-chain PFAA trifluoroacetic acid (TFA) (Huber et al. 2009) and hydrofluoric acid (HF) (Henry et al. 2018), which is also on DTSC's Candidate Chemicals list. An industry-sponsored experimental study in a rotary kiln test facility simulating municipal incinerators found that PTFE polymer pellets begin to decompose at around 500˚C, and by around 650˚C they completely convert to HF gas and F-containing ash, with no significant PFAA emissions (Aleksandrov et al. 2019). Other authors report that at lower temperatures, as could occur during accidental landfill fires, fluoropolymers such as PTFE can break down into PFCAs, including PFOA (Feng et al. 2015; Schlummer et al. 2015).

Thus, while PFAAs constitute a small subset (approximately 1 percent) of PFASs, they are terminal degradation products, manufacturing aids/feedstocks, or impurities of other PFAS class members. This makes their hazard traits relevant to the entire class (see Figure 3).

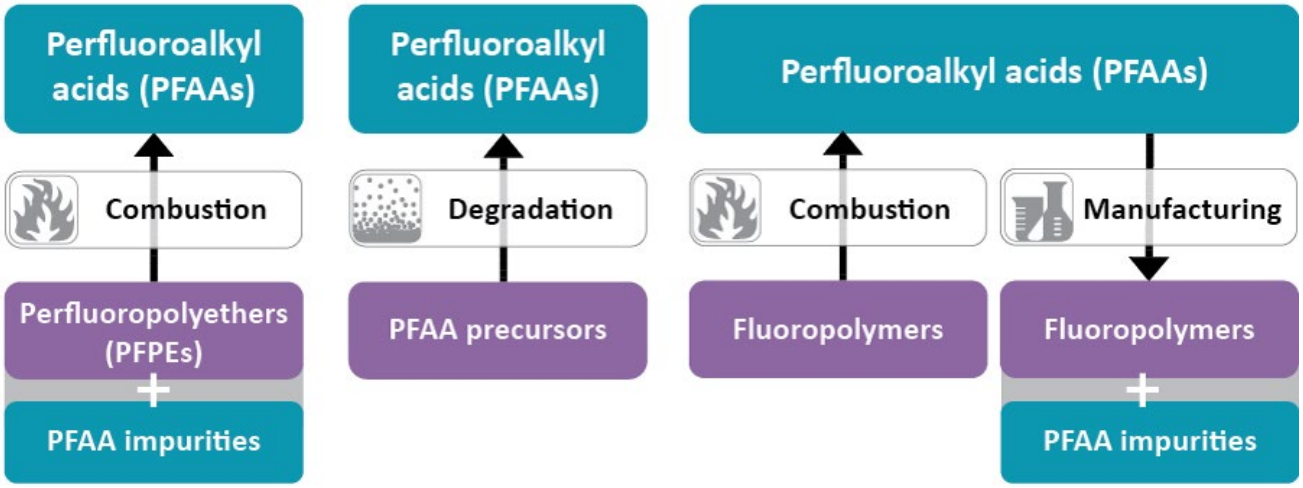

Figure 3: Use of all PFAS subclasses can lead to exposure to PFAAs at some point in the chemicals' life cycle; they either degrade to form PFAAs, release PFAAs if incinerated, or require PFAAs for their manufacture, often resulting in PFAA impurities in the final product. Specifically:

- *fluoropolymers are made using PFAAs*
- *PFPEs and fluoropolymers can contain PFAA impurities*
- *PFAAs can be released from PFPEs and fluoropolymers during combustion*
- *PFAA precursors such as fluorotelomer-based substances, including side-chain fluorinated polymers, degrade to PFAAs*

Hazard traits and environmental or toxicological endpoints

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(A).

The hazard traits and environmental or toxicological endpoints summarized in this section are defined in the SCP regulations sections 69501.1(a)(36) and (33), respectively, both of which refer to the Office of Environmental Health Hazard Assessment's (OEHHA) Green Chemistry Hazard Trait regulations (California Code of Regulations, Title 22, Chapter 54). These include exposure potential, toxicological, and environmental hazard traits.

PFASs display multiple hazard traits according to OEHHA's Green Chemistry Hazard Traits regulations (California Code of Regulations, title 22, section 69401 et seq.). These include toxicological hazard traits, (Articles 2 and 3), environmental hazard traits (Article 4), and exposure potential hazard traits (Article 5).

Exposure potential hazard traits

Environmental persistence

Environmental persistence is one of the hazard traits identified in the Green Chemistry Hazard Traits Regulations. According to OEHHA's Statement of Reasons for these regulations (OEHHA 2012):

"*Persistence of a chemical in the environment promotes sustained exposure and contributes to accumulation in the environment. Because persistence is an inherent property of a chemical in the environment that results in increased exposure to the chemical and consequently potential for health risks, it can appropriately be identified as a hazard trait. Legacy chemicals such as DDT and PCBs remain public health concerns decades after their production was banned because of their ability to persist in the environment*."

PFAAs are extremely persistent in the environment, with the exception of PFPiAs, which degrade into PFPAs and potentially PFCAs (Scheringer et al. 2014; Wang et al. 2016). All other PFAAs degrade significantly only under environmentally irrelevant conditions. PFOS and its salts, perfluorooctane sulfonyl fluoride, PFOA and its salts, and PFOA-related compounds are designated as persistent organic pollutants (POPs) under the Stockholm Convention (UNEP 2020). PFHxS and C11 through C14 PFCAs are listed as very persistent and very bioaccumulative (vPvB) on the European Chemicals Agency (ECHA)'s Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) Candidate List of Substances of Very High Concern (SVHC); PFOA, ammonium perfluorooctanoate (APFO), PFNA, and PFDA are listed as persistent, bioaccumulative, and toxic (PBT) (ECHA 2020). Limited data on some PFECAs and PFESAs suggest resistance to biodegradation (RIVM 2016). PFPEs and fluoropolymers are similarly persistent, whereas PFAA precursors degrade into extremely persistent PFAAs.

Mobility in environmental media

As discussed in Chapter 3, PFAAs are mobile, capable of long-range transport, and have been found in many environmental compartments far away from their sources. Once released into air and water, PFAAs and some of their precursors are highly mobile and undergo long-range transport to distant regions via atmospheric circulation and oceanic currents (Prevedouros et al. 2006; Ahrens et al. 2011b; Muir et al. 2019). Depending on their physicochemical properties, such as water solubility, K_{oc}, and volatility, PFASs can be transported in the dissolved phase (e.g., through soil water, by rivers and oceanic currents), in the particulate phase (e.g., via suspended sediments or airborne particles), or in the atmosphere. Shorter-chain PFAAs have higher aquatic persistence and mobility than the longer chains, and thus are more frequently detected in the aquatic environment (Li et al. 2020a) and preferentially taken up by plants (Blaine et al. 2013), including food crops (Wang et al. 2015b). Shorter-chain PFAAs are especially more mobile in soil and water, hence they accumulate in edible plant parts and are difficult and expensive to remove during water treatment (Brendel et al. 2018; Boone et al. 2019; Ateia et al. 2019a; Crone et al. 2019).

Bioaccumulation

PFAAs tend to bioaccumulate, with elimination half-lives increasing with perfluoroalkyl chain length. In humans, known serum elimination half-lives range from a few days to several years (Bartell et al. 2010; Chang et al. 2008; Olsen et al. 2007b; Nilsson et al. 2010; Seals et al. 2011). PFPiAs may be an exception, since they can biotransform to PFPAs (Joudan et al. 2017; Yeung and Mabury 2016). ECHA's REACH Candidate List of SVHCs designates PFHxS and C11 through C14 PFCAs as very persistent and very bioaccumulative (vPvB), and PFOA, APFO, PFNA, and PFDA as bioaccumulative (PBT) (ECHA 2020). PFAAs differ in their uptake and accumulation in human tissues (Burkemper et al. 2017; Pérez et al. 2013). PFAAs are proteinophilic (protein-binding), accumulating particularly in blood, liver, stomach, kidney, lung, gall bladder, brain, muscle, and yolk sac tissues (Chen et al. 2009; Hebert et al. 2010; Greaves et al. 2012; Jones et al. 2003; Mackay et al. 2000; Pérez et al. 2013). They can cross the brain blood barrier and are present in animal brain tissue (Greaves et al. 2013). PFAAs, especially the shorter-chain ones, tend to bioaccumulate in plants, as evidenced by findings of higher PFAA concentrations in plants compared to environmental levels (Blaine et al. 2013; Blaine et al. 2014a; Gobelius et al. 2017; Lechner et al. 2011; Scher et al. 2018; Yoo et al. 2011; Chen et al. 2020). Longer-chain PFAAs, especially PFSAs, tend to partition in the roots, whereas the more soluble, shorter-chain PFAAs, especially PFCAs, partition in other parts of the plant (Blaine et al. 2013; Blaine et al. 2014a; Gobelius et al. 2017; Lechner et al. 2011; Scher et al. 2018; Stahl et al. 2009; Yoo et al. 2011). The bioaccumulation hazard trait is enhanced by the extreme persistence combined with the high environmental mobility of these chemicals. Biomagnification, which is the increase in contaminant concentration higher in the food chain, has been reported for longer-chain PFAAs in freshwater and marine organisms (Houde et al. 2006). Shorter-chain PFAAs appear not to undergo biomagnification in an Antarctic ecosystem (Gao et al. 2020). Less is known about the potential for bioaccumulation or biomagnification of PFECAs and PFESAs (RIVM 2016) . To our knowledge, the bioaccumulation potential of PFASs other than PFAAs has not been evaluated.

Lactational or transplacental transfer

Transplacental and lactational transfer can be significant routes of exposure for newborns (Cariou et al. 2015; Papadopoulou et al. 2016; Winkens et al. 2017a). PFASs, including PFAAs, nonpolymeric PFAA precursors, and chlorinated polyfluorinated ether sulfonates, undergo transplacental transfer in humans (Midasch et al. 2007; Zhang et al. 2013; Yang et al. 2016; Chen et al. 2017a; Zhao et al. 2017a; Eryasa et al. 2019; Cai et al. 2020). A recent study showed that up to 30.3 percent of PFASs transferred from mother to fetus via the placenta were novel compounds that lack information regarding their toxicity and environmental fate (Li et al. 2020c). This suggests that more PFASs could be undergoing transplacental transfer, beyond those that have been studied to date. The placenta-to-maternal serum ratios of PFOS, PFOA, and PFNA increased during gestation, more so in pregnancies with male fetuses compared to female ones, suggesting bioaccumulation in the placenta and increasing exposure with

fetal age (Mamsen et al. 2019). Lactational transfer has also been documented for PFAAs in humans (Kärrman et al. 2007; Llorca et al. 2010; Mogensen et al. 2015; Mondal et al. 2014; Tao et al. 2008b), but to our knowledge has not been studied for PFAA precursors or other PFASs. A recent toxicokinetic evaluation of five different PFASs found significant differences in the placental and lactational transfer and tissue partitioning behavior of the different PFASs, as well as interspecies differences (Pizzurro et al. 2019).

Global warming potential

Some PFPEs, which are increasingly used as replacements for PFAAs or their precursors, may have high global warming potential.[5] The global warming potential of perfluoropolymethylisopropyl ether, a type of PFPE, ranges from 7,620 over 20 years to 12,400 over 500 years, relative to carbon dioxide (CO_2) (IPCC 2007). Some nonpolymeric fluorinated ethers, which can be used in the production of PFPEs and released to the environment, also have high global warming potential (ranging, relative to CO_2, from 207 to 13,800 over 20 years, from 59 to 14,900 over 100 years, and from 18 to 8,490 over 500 years) (IPCC 2007). Additionally, fluoroform (CHF_3), with a global warming potential of 11,700 to 14,800 relative to CO_2 over 100 years (IPCC 2007), is a degradation product of fluorinated polymers from incomplete combustion (Huber et al. 2009).

Other PFASs ubiquitous in the ocean and transferred onto sea spray aerosols can significantly increase the concentration of cloud condensation nuclei, thus increasing cloud reflectance. This increases the efficiency of solar radiation absorption in the atmosphere, contributing to global warming (MacLeod et al. 2014).

Toxicological hazard traits

Toxicological data are limited to a few PFASs, mostly PFAAs. The toxicities of fluorinated polymers, their intermediate degradation products, and processing aids are poorly understood. Appendix C lists toxicological hazard traits for the PFASs with publicly available data (mostly PFAAs and some precursors). Most toxicological studies evaluate exposure to single PFASs for 90 days or less, even though humans and animals are chronically exposed to PFAS mixtures over a lifetime.

Potential adverse human health effects and risk factors from longer-chain PFAS exposure include:

- Increased serum cholesterol (Skuladottir et al. 2015; Winquist and Steenland 2014)
- Thyroid disease (Winquist and Steenland 2014);

[5] Global warming potential (GWP) is defined by the California Air Resources Board (CARB) at https://ww3.arb.ca.gov/cc/inventory/background/gwp.htm. The GWP value represents the climate forcing of a kilogram of emissions of a substance relative to the same mass of carbon dioxide (CO_2), which has a GWP value of 1. In California, GWPs are calculated over a 100-year time frame.

- Immune dysregulation (Grandjean and Clapp 2014; Grandjean et al. 2016; Grandjean et al. 2017)
- Pregnancy-induced hypertension (C8 Science Panel 2011); and
- Kidney and testicular cancers (C8 Science Panel 2012).

A systematic review of 64 epidemiological studies that assessed children's exposure to PFASs and associations with specific health outcomes found positive correlations between PFAS exposure and suppressed immune system response, dyslipidemia, impaired kidney function, and delayed first occurrence of menstruation.

Epidemiological studies suggest a link to adverse health effects but often entail uncertainties in exposure or confounding factors. A systematic review of 64 epidemiological studies that assessed children's exposure to PFASs and associations with specific health outcomes found positive correlations between PFAS exposure and suppressed immune system response, dyslipidemia, impaired kidney function, and delayed first occurrence of menstruation (Rappazzo et al. 2017). PFOA also appears to be associated with low birth weight in humans (Malits et al. 2017).

The health effects listed above were observed in studies focused on longer-chain PFASs. Possibly due to their shorter observed biological half-lives, shorter-chain PFASs were expected to have lower toxicity compared to the longer chains (DeWitt 2015; Stahl et al. 2011; Wolf et al. 2008). Notable exceptions are PFHxS (C6) (considered a longer-chain due to its high bioaccumulation potential) and PFBS (C4) in some toxicokinetic studies and receptor binding assays (Wolf et al. 2008; Danish Environmental Protection Agency 2015). PFBS was reported to be developmentally toxic in zebrafish (Hagenaars et al. 2011). Delayed pupil response (Butenhoff et al. 2012) and retinal degeneration[6] in rodents (3M 2006) indicate a potential for ocular toxicity in humans from exposure to the ammonium salt of PFBA. Rodent studies have also identified reproductive and developmental (fetal resorption, delayed eye opening) (Das et al. 2008), hematotoxic (reduced red blood cell count, hemoglobin, and hematocrit) (Lieder et al. 2009), and respiratory (nasal degeneration) (Loveless et al. 2009) hazards for the ammonium salt of PFBA. The neurodevelopmental toxicity potential of PFBS has been demonstrated *in vitro* by dose-dependent suppression of neuronal differentiation (Slotkin et al. 2008).

Scientific review of the limited data on the PFECA GenX (a PFOA replacement in some applications) and related perfluorinated ether substances from Chemours (RIVM 2016) indicates potential associations with adverse health effects in laboratory animals, including cancer, body weight gain, changes to the

[6] Unpublished data mentioned in the cover letter accompanying a 3M submission to the TSCA 8(e) docket.

immune system and cholesterol levels, increased weights of kidneys and livers, and liver cell changes (Rae et al. 2015).

When differences in rodent toxicokinetics are taken into consideration, PFECAs and shorter-chain PFAAs may have similar or higher toxic potency than the longer-chain PFAAs they are replacing. Using a toxicokinetic model and existing toxicity data sets, a recent study found that PFBA, PFHxA, and PFOA have the same potency to induce increased liver weight, whereas GenX is more potent (Gomis et al. 2018). The authors concluded that previous findings of lower toxicity of fluorinated alternatives in rats were primarily due to the faster elimination rates and lower distribution to the liver compared to PFOA and other longer-chain PFAAs.

As shown in Appendix C, the evidence of adverse health effects from PFAS exposure in humans is sometimes conflicting (DeWitt 2015). Studies that examine associations between maternal serum, blood, or breast milk PFAS concentrations and child health outcomes sometimes show mixed results (Rappazzo et al. 2017). For example, one study found an association between increased maternal serum PFOS concentrations and increased hyperactivity in children (Høyer et al. 2015), while another reported a null association between PFOS in cord blood and the neurobehavioral symptoms of attention deficit/hyperactivity disorder (Lien et al. 2013). Studies have also suggested detrimental, beneficial, or no effects on metabolism. For instance, PFAS mechanisms of action can both exacerbate insulin resistance and improve insulin sensitivity (Fleisch et al. 2017).

Less is known about the toxicity of most PFAA precursors. Laboratory studies of fluorotelomer carboxylic intermediates indicate that they are more acutely toxic than PFCAs in nonmammalian systems (Phillips et al. 2007; Rand et al. 2014a). Industry studies report increased mortality for 6:2 FTOH (O'Connor et al. 2014), and liver and kidney toxicity for 8:2 FTOH (Ladics et al. 2008), at the highest doses only. Rice et al. (2020) present evidence that PFHxA toxicity and toxicokinetics alone would not adequately inform toxicity characterization of fluorotelomer precursors for which PFHxA is a terminal product (e.g. 6:2 FTOH). Other FTOH studies have found a potential for endocrine disruption, interfering with thyroid and sex steroid hormone systems (Ishibashi et al. 2008; Rosenmai et al. 2016; Weiss et al. 2009).

The mode of action of PFAAs has not been fully characterized. Activation of the nuclear peroxisome proliferation-activated receptor alpha (PPAR-alpha) has been associated with some of the hepatic effects of PFOA and PFOS, although other biological interactions associated with PFAAs' neurodevelopmental and reproductive toxicity have not yet been identified (Guyton et al. 2009; Rappazzo et al. 2017). PFASs have been shown to bind competitively to the human thyroid hormone transport protein (Weiss et al. 2009), and thus may affect thyroid hormone levels and early life brain development (Winkens et al. 2017a).

Environmental hazard traits

Appendix C lists several studies documenting the following environmental hazard traits:

- Phytotoxicity;
- Wildlife developmental impairment;
- Wildlife reproductive impairment; and
- Wildlife survival impairment

Laboratory studies of environmentally relevant species of algae, aquatic plants, terrestrial plants, fish, amphibians, mammals, birds, and important pollinating insects, as well as studies of birds exposed in the wild, show an association between PFAS exposure and several adverse environmental impacts, including developmental, reproductive, or survival impairment (Annunziato et al. 2019; Eggers Pedersen et al. 2016; Liu et al. 2015b; Sanchez et al. 2015; Soloff et al. 2017). Additionally, PFAS toxicity has been reported across a broad range of aquatic organisms (microorganisms, algae, plants, invertebrates, amphibians, fish, and marine mammals) and adverse impacts (e.g., impaired growth, mortality, developmental effects, and reproductive effects), while the environmental effects on terrestrial wildlife are not as well-studied. See Appendix C for details.

A 2011 study testing the structure-activity relationship of PFOA, PFOS, PFBS, and PFBA in fish development found that all four PFASs were teratogens, leading to malformations of the tail and an uninflated swim bladder, causing fish to swim abnormally; exposure to PFBS and PFOS also resulted in fish head malformations (Hagenaars et al. 2011). Swimming and buoyancy are necessary for fish to compete for food and escape predators, ultimately impacting their survival rate. Developmental toxicity was also observed in zebrafish embryos following exposure to 6:2 FTCA (Shi et al. 2017), and to several perfluoropolyether carboxylic acids (Wang et al. 2020). Studies of zebrafish embryos found higher potential for adverse developmental effects for PFASs with sulfonic groups, such as PFBS and PFOS (Ulhaq et al. 2013), and potential endocrine-disrupting effects for 6:2 FTOH (Liu et al. 2009).

Studies show increased acute toxicity in microalgae and zebrafish embryos with increasing chain length (Latala et al. 2009; Ulhaq et al. 2013). A review of fish toxicity data concluded that shorter-chain PFASs have moderate to low acute toxicity at exposures of less than seven days; data on longer exposures were unavailable (Danish Environmental Protection Agency 2015). A 2015 study found that levels of PFCAs and PFSAs, including PFHxA and PFBS, found in various brain regions affected neurochemical markers in Greenland polar bears (Eggers Pedersen et al. 2015).

Structural or mechanistic similarity to chemicals with known adverse impacts

Reference: California Code of Regulations, title 22, section 69503.3(a)(3).

Some chemicals may lack sufficient data to definitively establish presence or absence of harm. In such cases, DTSC may also consider data from other chemicals closely related structurally to the Candidate Chemical to identify potential public health and environmental impacts.

While physicochemical properties and hazard traits for many PFASs are either uncharacterized or unavailable in the public domain, the data available for structurally similar PFASs raise sufficient concerns for DTSC to consider PFASs as a class. In particular, longer- and shorter-chain PFAAs share three key structural and mechanistic properties: their structural similarity to fatty acids (DeWitt et al. 2015), their potential to activate PPAR-alpha (Rosenmai et al. 2018; Wolf et al. 2008; Wolf et al. 2014), and their persistence in the environment (Krafft et al. 2015a). The carbon-fluorine bonds characteristic of the structure of all PFASs endow their extreme environmental persistence.

4. POTENTIAL FOR EXPOSURES TO THE CANDIDATE CHEMICAL IN THE PRODUCT

Reference: California Code of Regulations, title 22, section 69503.3(b).

The SCP regulations direct the Department to evaluate the potential for public or aquatic, avian, or terrestrial animal or plant organism exposure to the Candidate Chemical(s) in the product by considering one or more factors for which information is reasonably available.

Presence of the Candidate Chemical in the Product

Reference: California Code of Regulations, title 22, section 69503.3(b)(2).

This subsection summarizes available information indicating the Candidate Chemical's presence in and release from the product.

PFASs are approved by FDA for use as surface coatings or as additives inside the slurry to impart water, stain, and grease resistance to a wide variety of plant fiber-based food packaging products (see Section 4). According to the existing FCNs, PFASs are approved to be applied to plant fiber-based food packaging products at concentrations ranging from 0.2 to 1.5 percent by weight (FDA 2020a). These polymeric PFAS formulations approved for use in food packaging contain PFAA and nonpolymeric PFAA precursor impurities (RIVM 2019), typically at ng/g levels (Yuan et al. 2016; Schaider et al. 2017; Zabaleta et al. 2017). Because PFASs are extremely persistent and contaminate food packaging manufacturing facilities, even products free of intentionally added PFASs may contain PFAS impurities (Schaider et al. 2017).

Detecting PFASs in food packaging remains difficult. Tests that screen for total fluorine are the fastest, cheapest, and thus most common, but they can only give an indication of the presence or absence of PFASs, not a speciation of individual PFASs and their concentrations (Schultes et al. 2019). Screening methods such as mass spectrometry can be used only to quantify PFASs in a known library and require complicated workflows to accurately identify undiscovered PFASs (Lee et al. 2019).

Recent testing found PFASs, as evidenced by the presence of fluorine, in approximately half of paper and paperboard products tested – including bakery bags, deli wrappers, microwave popcorn bags, French-fry boxes, takeout containers, and pizza boxes (Schaider et al. 2017; Zabaleta et al. 2017). In other recent studies, PFASs were found in all molded fiber food packaging products tested – including bowls, soup containers, clamshells, plates, and food trays (CEH 2018; Fassler 2019). A 2018 study of takeout containers from five of the largest grocery retailers in the U.S., including in California, found levels of total fluorine indicative of likely PFAS treatment in 10 of the 78 samples tested, including takeout containers and bakery/deli papers (Safer Chemicals Healthy Families et al. 2018). A small 2019 testing of products from Albertsons and Safeway supermarkets found PFASs in cake plates and microwave popcorn bags, but not in butcher paper (Safer Chemicals Healthy Families 2019).

Bokkers et al. (2019) identified 22 publications between 2009 and 2018 that reported concentration data for various PFAAs and fluorotelomers in food-contact materials. Prior to the phaseout of longer-chain PFAS in North America, one study identified more than 115 PFASs in products from the European Union, U.S., and China, including several PAPs, polyfluoroalkyl ethoxylates, acrylates, amino acids, sulfonamide phosphates, and thio acids (Trier et al. 2011a). A more recent study analyzed the PFAS composition of 20 food packaging samples collected at fast-food restaurants across the U.S. in 2014-2015 and detected PFOA in six of the samples, despite the 2011 phaseout (FDA 2018), as well as several shorter-chain PFAAs and precursors (e.g., PFHxA, PFBS, 4:2 FTS, and GenX), and several unknown compounds (Schaider et al. 2017).

Concentrations of PFAAs and some precursors measured in three French-fry bags and six microwave popcorn bags collected in Sweden in 2012 ranged from below detection to 2,100 pg/cm^2, accounting for only up to 0.28 percent of the total fluorine in those samples (Schultes et al. 2019). Zabaleta et al. (2017) identified 46 PFAAs and precursors such as FTCAs and FTUCAs in microwave popcorn bags collected in 2015 and 2016 from 12 European Union countries, Mexico, Brazil, the U.S., China, and India. None of the three popcorn bags sampled in the U.S. for this study contained PFAAs, but one of them contained 6:2 FTUCA. Yuan et al. (2016) analyzed 69 food-contact materials collected from commercial retailers in Beijing, China, and online between 2013 and 2015, and 25 food-contact materials collected from Columbus, Ohio, in 2015. The Chinese molded fiber and microwave popcorn samples had the highest levels of PFCAs and FTOHs. In contrast, the molded fiber product sampled in the U.S. contained only 6:2 FTOH (all other FTOHs were below the method quantification limit), at a concentration of 499 ng/g. All the FTOHs on the analyte list, except for 18:2 FTOH, were detected in

the microwave popcorn bags sampled in the U.S., with 6:2 FTOH at highest levels (median concentrations of 485 ng/g) (Yuan et al. 2016).

Market presence and trends

Reference: California Code of Regulations, title 22, sections 69503.3(b)(1)(A-C)

Product market presence information may be used as a surrogate to assess potential exposures to the Candidate Chemical in the product. This information may include statewide sales by volume or number of units, the intended use(s) of the product, and characteristics of the targeted customer base.

After tests in the 1940s found that fluorochemicals imparted resistance to the spreading of liquids such as water and oil on solid surfaces, paper and food producers began applying PFASs to a wide array of packaging products to take advantage of these properties. In 1981, General Mills earned the first patent on the modern microwave popcorn bag (Butler 2019). Popcorn bags are one of the food packaging product types commonly treated with PFASs. PFAS-treated food packages today are ubiquitous in society. Consumers of packaged food span all socioeconomic levels and ethnicities.

The U.S. Census Bureau classifies food packaging products under five different North American Industry Classification System (NAICS) codes. According to data on these five broad NAICS codes,[7] the number of U.S. manufacturing establishments have fallen across each of these sectors by 16 percent on average from 2007 to 2017. Sales revenues by sector fluctuated over this period. For instance, sectors 322130 and 322212 increased substantially and moderately, respectively, in terms of sales, while sectors 322220 and 322299 decreased slightly (U.S. Census Bureau 2007; U.S. Census Bureau 2012; U.S. Census Bureau 2017). The Food Packaging Forum states that paper and paperboard comprise 34 percent of the market share for packaging materials (Muncke 2012). Market data gaps, however, remain substantial. Accessible government, nongovernmental, academic, and corporate data sources reveal only scarce sales information for specific food packaging products.

Foreign imports of food packaging-related goods to the U.S. increased exponentially – by 344 percent – from 2007 to 2018 (U.S. Census Bureau 2019).[8] Additionally, PFOA and PFOS continue to be produced in countries such as China and India and applied to products that enter the global supply chain (SAICM 2019). Approximately 190 companies in the U.S. and abroad supply food packaging products to businesses in California (Thomas Net 2019).

[7] These five NAICS codes – 322130 (paperboard mills), 322220 (paper bags and coated and treated papers), 322212 (folding paperboard boxes), 322219 (other paperboard containers), and 322299 (all other converted paper products) comprise paper-based food packaging and food packaging material manufacturing. Nonfood packaging manufacturing also falls within these codes.

[8] The U.S. imported approximately $845 million in food packaging products during 2018.

Historically, paper and paperboard products were treated with nonpolymeric long-chain PFASs, including PFOS precursors (Trier et al. 2011a; Trier et al. 2011b). Since 3M phased out PFOS and its precursors in 2001, most formulations now are based on polymers such as acrylate polymers with fluorotelomer of sulfonamide alcohol side chains, or PFPEs (Schultes et al. 2019). These are also the types of PFASs currently approved for use in food packaging by FDA (Schaider et al. 2017). Newer polymeric PFAS treatments can contain shorter-chain nonpolymeric PFASs, such as PFAAs, as impurities or degradation products; these unintentionally added PFASs exhibit endocrine activity *in vitro* (Rosenmai et al. 2016).

The most common food-contact applications of PFPEs are as lubricants for food processing equipment and in packaging. For instance, Solvay markets Solvera PFPE for use in pizza boxes, multiwall paper sacks such as those "used for pet food packaging and other oil containing foods like corn meal or batter," fast-food packaging, microwave popcorn packaging, and molded paper applications (Solvay 2020).

Food packaging products containing PFASs are commonly found in homes and places of work. Disposable paper plates and bowls are ubiquitous in many pantries, and countless consumers bring takeout foods home or simply want to avoid wasting leftovers at restaurants. Workers in a wide range of occupations come into contact with food packaging products, and the PFASs they contain, throughout the products' life cycle. These include food preparers and servers, paper goods machine setters, operators, and tenders, and water and wastewater treatment plant workers and system operators.[9] Food preparation and serving occupations (Standard Occupational Code 35-0000) in California employed nearly 1,540,000 workers in 2016. National trends in employment for most of these occupations are projected to move upward through 2028, and employment levels in California for nearly all of these occupations will rise substantially (U.S. Bureau of Labor Statistics 2019). For instance, government analysts estimate that food service manager jobs in California will increase by over 14 percent between 2016 and 2026 (California Employment Development Department 2019). However, not all occupations that involve contact with food packaging follow this trend. Paper goods machine setter, operator, and tender jobs are projected to decline at the national and state levels through this decade (California Employment Development Department 2019; U.S. Bureau of Labor Statistics 2019).

[9] Additional occupations that risk exposure to the PFASs associated with food packaging include food service management, food processing, hazardous materials removal, and septic tank servicing and sewer pipe cleaning. The Standard Occupational Classification Codes associated with the relevant occupations are 11-9051, 35-0000, 47-4041, 47-4071, 51-3000, 51-8031, and 51-9196 (U.S. Bureau of Labor Statistics 2019).

Potential exposures to the Candidate Chemical during the product's life cycle

Reference: California Code of Regulations, title 22, sections 69503.3(b)(3); 69503.3(b)(4)(A-H).

Potential exposures to the Candidate Chemical or its degradation products may occur during various product life cycle stages, including manufacturing, use, storage, transportation, waste, and end-of-life management practices. Information on existing regulatory restrictions, product warnings, or other product use precautions designed to reduce potential exposures during the product's life cycle may also be discussed here.

PFAS-treated food packaging products are pervasive in society, and as such, the general population can be regularly exposed to this class of chemicals. Direct and indirect human exposure to PFASs can occur during several of the life cycle stages of food packaging products: manufacturing, use, and end-of-life. The most potential exposures occur via the use and end-of-life stages.

Exposure to PFASs from food packaging can occur in three primary ways: (1) through ingestion of food or drinks contaminated by PFASs that have migrated from food packaging; (2) through ingestion of food or water contaminated by compost containing PFAS-treated food packaging, by leachate from landfills containing PFAS-treated food packaging, or by effluent from wastewater treatment plants serving landfills or paper recycling facilities; and (3) through exposure to products made of recycled PFAS-containing food packaging materials. These potential exposures are summarized in Figure 4 below.

Figure 4. An overview of the exposure pathways to PFASs used in food packaging

Manufacturing

Volatile PFASs can be released into the environment during the manufacture of food packaging products (Stock et al. 2004). Industrial effluent discharge into WWTPs can contaminate rivers and other waterways (Konwick et al. 2008). Once released into air and water, PFAAs and some of their precursors are highly mobile and undergo long-range transport to distant regions via atmospheric circulation and oceanic currents (Prevedouros et al. 2006; Ahrens et al. 2011b; Muir et al. 2019). Through a Freedom of Information Act request in 2018, the U.S.-based nongovernmental organization (NGO) Environmental Defense Fund obtained four FCNs submitted in 2009 and 2010 by two companies, Daikin America and Chemours, to sell PFASs for paper and paperboard application in food packaging. These four assessments estimated that a "typical" paper mill produces "825 tons of PFAS-coated paper per day and discharges 26 million gallons of water per day." The estimates for the amount of PFAS formulation discharged daily ranged from 95 to 225 pounds, with wastewater concentrations between 43,000 to 103,000 parts per trillion (ppt) (CELA 2019).

PFAAs and their precursors can contaminate soil and sediments via deposition of contaminated particulate matter, use of contaminated irrigation water, and land application of contaminated biosolids (Armitage et al. 2009; Blaine et al. 2014a; Lindstrom et al. 2011a; Sepulvado et al. 2011; Zhang et al. 2016). One study of 262 surface sediment samples from 48 lakes and two reservoirs in China (Qi et al. 2016) found high levels of PFASs (C4 through C14) in sediment correlated with proximity of the sampling locations to industrial activities in food packaging, textile, electroplating, firefighting, semiconductor, precious metals processing, and coating industries.

Use

PFAS impurities and degradation products of the polymeric formulations currently used in food-contact materials can migrate into food items (RIVM 2019). Positive associations between human serum levels of the PFOS precursor N-ethyl perfluorooctane sulfonamidoacetate (N-EtFOSAA) and PFHxS and the use of carpet, nonstick cookware, and food packaging were found in samples from the Chemicals, Health and Pregnancy cohort in Vancouver, Canada, and from the 2005-06 National Health and Nutrition Examination Survey (NHANES), suggesting PFAS migration from these product types into human exposure routes such as diet (Hu et al. 2018).

A recent report (Nordic Council of Ministers 2017) summarizes the studies showing migration of such nonpolymeric PFASs from paper and paperboard used in food packaging applications. They conclude that:

- PFASs that are not covalently bound to the paper (such as the PFAA and FTOH impurities) are more easily released;

- PFASs migrate out of paper and board packaging via hydrolysis, which is accelerated by heating, moisture, and the presence of emulsifiers such as alcohol – this was demonstrated for PAPs, but is expected to also be true for fluoroacrylate ester bound coatings;
- The types of PFASs in the coating matters: what determines the transfer of perfluorinated PFASs is not necessarily true for the polyfluorinated ones;
- Other factors that impact migration include the food composition, the presence of salts, microwaving, the total surface area, and the surface energy of the surfaces.

Other studies have also found correlations between factors such as high heat, acidity, and the presence of emulsifiers, and the migration of PFASs from food packaging. For instance, Still et al. (2013) found that storage in packaging coated with a fluorinated polymer increased the levels of PFAAs and FTOHs in butter, which is an emulsified food. Additionally, to accurately estimate the migration of PFASs from products to food, the octanol-air and octanol-water partition coefficients (K_{oa} and K_{ow}) for the various PFAS structures must be taken into account (Eichler and Little 2020).

Yuan et al. (2016) performed migration tests in water, ethanol solutions, and oil on a molded fiber bowl collected in China. Migration efficiencies into water ranged from 0.005 percent for PFDoA to 18.3 percent for PFBA; i.e., the shorter-chain PFASs tend to migrate more readily. Migration efficiencies significantly increased in ethanol. No PFCAs were detected above method quantitation limits in 10 mL oil. Similarly, the study authors were able to quantify all FTOHs measured, except for 18:2 FTOH, in 10 mL of food simulants, with amounts ranging from 0.01 to 370 ng. Migration efficiencies for FTOHs also increased with decreasing chain length (from 0.004 percent for 16:2 FTOH to 0.24 percent for 6:2 FTOH) and increased with ethanol content.

Note that most of the available studies (including most of those reviewed by Trier et al. 2018) were done on older formulations of PFASs used in food packaging, and many describe the migration potential of PFOA and PFOS, and other longer-chain PFASs. For instance, EFSA (2018) found that transfer from food-contact materials used in food processing and packaging is one of the two main ways PFOS and PFOA get into food (the other being bioaccumulation in aquatic and terrestrial food chains). However, longer-chain PFASs and their precursors have been phased out for use and are not authorized by FDA anymore. As the longer-chain formulations have been phased out and replaced by the shorter-chain formulations, the differences in migration and toxicity remain to be fully evaluated. As noted above, shorter-chain PFASs tend to be more mobile and migrate out of food packaging materials more readily than the longer chains (Nordic Council of Ministers 2017). Their biopersistence and toxicity differ from those of the longer-chain compounds (Rice 2015). However, some newer studies indicate that the intermediate degradation products of the PFAS impurities present in FDA-approved shorter-chain PFAS formulations currently used in food packaging are biopersistent and more toxic than previously thought (Kabadi et al. 2018; Kabadi et al. 2020; Rice et al. 2020).

The best source of information regarding currently used formulations are the FCN documents submitted to FDA, which list numerous PFASs that are expected to migrate out of packaging into food from formulations currently approved for use in the U.S. (Nelter 2018; FDA 2020a).

While several studies have demonstrated that PFASs migrate from food packaging into food items, very few have shown direct links to human exposure. One study that has shown this direct link is a recent publication that analyzed 2003-2014 serum PFAS and dietary recall data from NHANES. Researchers found that consumption of microwave popcorn was associated with significantly higher serum levels of PFOA, PFNA, PFDA, and PFOS, likely as a result of PFAS-treated microwave popcorn packaging (Susmann et al. 2019).

End-of-life

At its end-of-life, plant fiber-based food packaging can be composted, landfilled, recycled, or incinerated.

Composting

U.S. compostability standards, ASTM D6400 and D6868, do not consider PFASs, so PFASs are currently allowed in compostable food packaging (Schaider et al. 2017). Nevertheless, the Biodegradable Products Institute (BPI), the main certifier of compostable U.S. food packaging products, has updated its compostability standard as of March 1, 2019. The new standard incorporates EN 13432, which limits total fluorine to 100 parts per million (ppm) and, as of January 1, 2020, requires a statement that food packaging certified as compostable contains no added fluorine (BPI 2018). However, not all manufacturers choose to get their products certified to the BPI standard, especially since most paper products are considered inherently biodegradable, and thus also compostable without proof of certification.

The PFASs used in food packaging degrade to PFAAs, which are taken up by plants grown in compost-treated soil (Trier et al. 2011a; Zhou et al. 2019; Zhang et al. 2020; Gredelj et al. 2020). Compost samples collected from California and four other U.S. states that included food service packaging had significantly higher PFAA levels than compost that did not include these items, with the shorter-chain PFAAs (PFBA, PFPeA, and PFHxA) being most prevalent (Lee and Trim 2018a). In a recent study (Choi et al. 2019), the majority of PFASs found in compost samples from commercial facilities that accept food packaging contained six or fewer fluorinated carbons in their molecules. These shorter-chain PFASs are extremely persistent in the environment, highly mobile in water, and preferentially taken up by plants (Blaine et al. 2013), including food crops (Wang et al. 2015b). Shorter-chain PFASs generally show increased uptake by leafy plants compared to the longer chains (Blaine et al. 2014a), though longer chains may preferentially accumulate in roots (Zhao et al. 2017a).

The water that comes in contact with the waste materials undergoing composting, known as contact water, can also become contaminated with PFASs. A recent study in Minnesota detected PFAAs more frequently in contact water from composting facilities that accept food packaging compared to those that only accepted yard waste (Wood Environment & Infrastructure Solutions, Inc. 2019).

Landfilling

When disposed of in landfills, plant fiber-based food packaging products can contribute to the observed releases of PFASs from landfills to surface waters via landfill leachate (Allred et al. 2015; Lang et al. 2017). In landfills, the side-chain fluorinated polymers currently used in food packaging can degrade to form first nonpolymeric fluorotelomer-based PFASs such as FTOHs, which then further degrade to intermediates such as FTCAs and FTUCAs, and finally to PFAAs (Hamid et al. 2018; Washington et al. 2015; Hamid et al. 2020). The degradation of these side-chain fluorinated polymers in landfills and other waste stocks can be a significant long-term source (years to even centuries) of PFAAs to the environment (Lang et al. 2017; Li et al. 2017; Washington et al. 2019). Estimated release rates from waste stocks are almost two orders of magnitude higher for shorter-chain PFCAs, due to their greater mobility in water, compared to longer-chain PFCAs (Li et al. 2017; Li et al. 2020a).

Landfills have been identified as a source of PFAS contamination to nearby air and plant leaves (Tian et al. 2018). PFAAs and their precursors from discarded food packaging products can be released into the atmosphere during decomposition in landfills, and become widely dispersed (Ahrens et al. 2011a). PFAAs are also mobilized in landfill leachate (Fuertes et al. 2017; Lang et al. 2016; Shoaeioskouei 2012). Contaminated leachate collected from lined landfills for treatment at WWTPs can result in contaminated waterways, as wastewater treatment does not completely remove PFAAs from the effluent (Hamid and Li 2016; Huset et al. 2011).

PFAAs and their precursors have been detected in landfill leachate all over the world, including Germany (Busch et al. 2010), Norway (Knutsen et al. 2019), Canada (Benskin et al. 2012b), Uganda (Dalahmeh et al. 2018), and Australia (Gallen et al. 2017; Hepburn et al. 2019). In the U.S., an estimated 61.1 million cubic meters of leachate were generated in 2013, containing approximately 600 kg of the sum of 70 PFASs measured (Lang et al. 2017). 5:3 FTCA, an intermediate in the degradation of fluorotelomer-based substances to PFAAs, showed the highest mean concentration in leachate from surveyed landfills, followed by PFHxA; the following PFASs were detected in similar amounts: PFBA, PFPeA, PFHpA, PFOA, and 6:2 FTCA (Lang et al. 2017). In leachate collected from landfills in Florida, U.S., PFHxA and PFHxS were the most abundant species measured, followed by PFBS (Solo-Gabriele et al. 2020). The relatively slow release of PFASs from landfills compared to input rates means that landfilled waste continues to release PFASs for more than a decade or longer (Lang et al. 2017; Washington et al. 2019).

The relatively slow release of PFASs from landfills compared to input rates means that landfilled waste continues to release PFASs for more than a decade or longer.

Recycling

Recycling of postconsumer food packaging is a preferred alternative to landfilling or incineration, but can perpetuate the presence of PFASs in products made from the recycled material (Herzke et al. 2012). Some of the PFASs could also end up in the residual water used for the recycling process (DTSC 2020).

Incineration

If food packaging is incinerated at its end-of-life, the fluorinated polymers can form PFCAs, ozone-depleting compounds, potent greenhouse gases, and other hazardous substances (Henry et al. 2018; Huber et al. 2009; RIVM 2019). While incineration is the only technology proven to destroy PFASs, existing municipal solid waste incinerators often do not reach temperatures high enough to fully break down these compounds (DTSC 2020). For instance, a 2015 study found that, even at temperatures above 400˚C and in the presence of calcium hydroxide, PTFE does not undergo complete mineralization (Wang et al. 2015a).

Aggregate effects

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(B) and sections 69503.3(b)(3).

Multiple sources of exposure to the Candidate Chemical may increase the potential for significant or widespread adverse impacts.

PFASs are found in many frequently used consumer products in Californian households and workplaces, including carpets, rugs, flooring, furniture, clothing, cookware, cleaning products, personal care products, electronics, and food packaging (KEMI 2015), as well as in food (Christensen et al. 2017) and drinking water (Schwanz et al. 2016; Boone et al. 2019; Domingo and Nadal 2019). The indoor environment is a significant source of PFAS exposure due to the presence of PFAS-containing consumer products. The PFASs can evaporate from these products and partition to the indoor air, dust, or surfaces (Zheng et al. 2020). Overall, humans can be exposed to PFASs through a combination of dietary and nondietary pathways, including ingestion and inhalation. To add to this complexity, exposure to PFAAs can be categorized as direct or indirect. Direct exposure occurs when PFAAs enter the body, for example, through inhalation or ingestion. Indirect exposure means exposure to one or more PFAA precursors that are subsequently biotransformed to PFAAs in our bodies (Poothong et al. 2020). Since PFASs are ubiquitously distributed in both indoor and outdoor environments, the general population is exposed to complex PFAS mixtures via multiple pathways simultaneously. This makes it

difficult to model and predict aggregate exposure scenarios for individuals. Better understanding of population-wide aggregate exposures requires both generating more human biomonitoring data and expanding analytical methods to include more PFAS analytes. Nevertheless, DTSC considers all *potential* sources of PFAS exposure under its mission to protect the people and biota of California.

Because of the prevalence of PFAS-treated products in residential and commercial spaces, as well as their prevalence in food and drinking water sources, all Californians are potentially exposed, directly or indirectly. Despite extensive documentation of the widespread environmental PFAS contamination from consumer products such as food packaging, longitudinal studies that evaluate aggregate exposures to these chemicals from all sources through different life stages are still lacking. A lack of comprehensive data on source emissions and the distribution of PFASs through all environmental compartments makes it challenging to accurately estimate aggregate exposures over a lifetime. The frequency, level, and duration of individual or population-level PFAS exposures associated with the use of food packaging products have not been estimated using either deterministic or probabilistic modeling. In assessing aggregate effects, FDA considers other oral exposure sources that fall within the scope of their regulatory authority, but not sources that are outside the agency's purview, like inhalation or dermal exposures from PFAS-treated consumer products found in the indoor environment and exposures from drinking water. Also, FDA does not consider all exposures resulting across the product's life cycle, including from composting, landfilling, incineration, and recycling. These represent key differences between SCP's regulatory approach and FDA's. In evaluating the potential for aggregate exposures to a chemical in a consumer product to cause or contribute to harm, SCP seeks to identify and evaluate significant exposures from any source, via any route, in addition to exposures from food packaging products.

Indicators of potential exposures to the Candidate Chemical

Reference: California Code of Regulations, title 22, section 69503.3(b)(2).

The SCP regulations consider various data that indicate potential for exposure to the Candidate Chemical or its degradation products, including: monitoring data indicating the Candidate Chemical's presence in the indoor and outdoor environment, biota, humans (e.g., biomonitoring studies), human food, drinking water, and other media; and evidence of persistence, bioaccumulation, lactational and transplacental transfer.

PFASs are found ubiquitously in the environment, plants, animals, and humans (Campo et al. 2016; Lindstrom et al. 2011a; Lindstrom et al. 2011b). PFAS-treated food packaging contributes to widespread environmental contamination and exposures, as do other consumer products such as carpets, rugs, furniture, treatments for converted textiles or leathers, cosmetics, and waterproof clothing. Once released to the environment during product manufacture, use, or disposal, PFASs become part of a virtually closed cycle leading to chronic human and ecological exposures. Because

PFAAs and other persistent PFASs lack a natural degradation route, their levels in the environment, humans, and biota may continue to rise for as long as PFASs are used in consumer products.

Between 2009 and 2017, 455 new PFASs have been detected in environmental media and consumer products (Xiao 2017). However, most of the PFASs that contribute to total organic fluorine in the environment, wildlife, and human blood samples remain unidentified due to analytical limitations (Xiao 2017; Spaan et al. 2019). Thus, only a small fraction (sometimes less than 5 percent) of the PFASs in these media are likely reflected in the data summarized in this section. Nakayama et al. (2019) discussed the available analytical methods for measuring PFASs in air, water, sediments, soil, sludge, dust, and biological samples and identified a great need for new methods that expand the suite of PFASs studied, especially in humans and wildlife. As of now, the full extent of the contamination, despite extensive research, remains poorly understood.

Once released to the environment during product manufacture, use, or disposal, PFASs become part of a virtually closed cycle leading to chronic human and ecological exposures. Because PFAAs and other persistent PFASs lack a natural degradation route, their levels in the environment, humans, and biota may continue to rise for as long as PFASs are used in consumer products.

Monitoring data showing that the Candidate Chemical is widespread in the indoor and outdoor environment and accumulates in terrestrial and aquatic organisms

Numerous published studies have found that PFASs are ubiquitous in various types of environmental media, in biota, and in the indoor environment. A whole suite of PFAAs and precursors, including FTOHs, FTOs, FTCAs, FTUCAs, FASAs, FASEs, FTSs, diPAPs, perfluoroalkyl iodides (PFAIs), FTIs, and perfluoroalkyl sulfinates (PFSIs) were found in various matrices such as air, WWTP effluent and sludge, landfill leachate, seawater, rivers, lakes, rain, snow, ice, and soil [see (Frömel et al. 2016) and Table 4 for examples]. Despite the phaseout of longer-chain PFASs that started more than a decade ago, concentrations in environmental samples and biota have not declined (Land et al. 2018). This indicates that ecosystems continue to be exposed to previously released PFASs for a long time, even after their use is discontinued.

Table 4: Monitoring studies found varying levels of PFASs in the environment and biota.

Sample Type	PFAS Type: Concentration	Reference
Indoor air (homes)	6:2 FTOH: 1.8 ng/m³ (median) 8:2 FTOH: 8.7 ng/m³ (median) 10:2 FTOH: 2.5 ng/m³ (median) 8:2 FTAC: 0.27 ng/m³ (median)	Fromme et al. (2015)

Sample Type	PFAS Type: Concentration	Reference
	10:2 FTAC: 0.12 ng/m³ (median)	
Indoor air (homes)	Sum of FTOHs: 3.21 ng/m³ (median) Sum of longer-chain PFCAs: 0.34 ng/m³ (median) Sum of shorter-chain PFCAs: 0.35 ng/m³ (median) Sum of PFSAs: 0.23 ng/m³ (median) Sum of diPAPs: 0.0012 ng/m³ (median)	Yao et al. (2018)
Indoor air (hotels)	Sum of FTOHs: 3.33 ng/m³ (median) Sum of longer-chain PFCAs: 0.21 ng/m³ (median) Sum of shorter-chain PFCAs: 0.26 ng/m³ (median) Sum of PFSAs: 0.12 ng/m³ (median) Sum of FOSE/FOSAs: 0.0053 ng/m³ (median) Sum of diPAPs: 0.0011 ng/m³ (median)	Yao et al. (2018)
Indoor air (schools)	6:2 FTOH: 3.3 ng/m³ (median) 8:2 FTOH: 4.4 ng/m³ (median) 10:2 FTOH: 1.8 ng/m³ (median) 8:2 FTAC: 0.45 ng/m³ (median) 10:2 FTAC: 0.31 ng/m³ (median)	Fromme et al. (2015)
Indoor air (offices)	6:2 FTOH: 1.3 ng/m³ (geometric mean) 8:2 FTOH: 9.9 ng/m³ (geometric mean) 10:2 FTOH: 2.9 ng/m³ (geometric mean)	Fraser et al. (2012)
Indoor dust (homes)	Sum of nine PFCAs (C6 to C14): <1.0 – 37,400 ng/g	Liu et al. (2011b)
Indoor dust (homes)	Sum of longer-chain PFCAs: 59.5 ng/g (median) Sum of shorter-chain PFCAs: 87.4 ng/g (median) Sum of PFSAs: 17.1 ng/g (median) Sum of diPAPs: 1.13 ng/g (median)	Yao et al. (2018)
Indoor dust (homes)	PFOS: 6 ng/g (median) PFDA: 11 ng/g (median) PFHpA: 9 ng/g (median) PFHxA: 6 ng/g (median) PFNA: 8 ng/g (median) PFOA: 10 ng/g (median)	Shin et al. (2020)

Sample Type	PFAS Type: Concentration	Reference
Indoor dust (hotels)	Sum of FTOHs: 149 ng/g (median) Sum of longer-chain PFCAs: 25.8 ng/g (median) Sum of shorter-chain PFCAs: 3.25 ng/g (median) Sum of PFSAs: 10.3 ng/g (median) Sum of FOSE/FOSAs: 2.1 ng/g (median) Sum of diPAPs: 10.2 ng/g (median)	Yao et al. (2018)
Indoor dust (child care centers)	Sum of PFCAs: 8.37 – 386 ng/g Sum of PFSAs: 1.29 – 190 ng/g Sum of 42 PFASs: 46.9 – 6470 ng/g	Wu et al. (2020)
Indoor dust (child care centers)	Sum of PFCAs: 3.4 – 32 ng/g Sum of PFSAs: 0.36 – 37 ng/g Sum of FTS: not detected – 82 ng/g Sum of FTOH: not detected – 3100 ng/g Sum of FOSA/FOSE: not detected – 380 ng/g	Zheng et al. (2020)
Indoor dust (day care center)	PFOA: 31 – 110 ng/g PFOS: 23 – 65 ng/g	Björklund et al. (2009)
Indoor dust (classrooms)	PFOA: <0.05 – 31 ng/g PFNA: <0.05 – 0.71 ng/g PFBS: <0.25 – 49 ng/g PFHxS: <0.1 – 120 ng/g PFOS: <0.1 – 21 ng/g	Harrad et al. (2019)
Indoor dust (cars)	PFOA: <0.05 – 14 ng/g PFNA: <0.05 – 3.1 ng/g PFBS: <0.25 – 170 ng/g PFHxS: <0.1 – 49 ng/g PFOS: <0.1 – 82 ng/g	Harrad et al. (2019)
Outdoor air	FTOHs: 0.06 – 0.19 ng/m^3	Müller et al. (2012)
North American soils	Sum of PFCAs: 145 – 6,080 ng/kg Sum of PFSAs: 35 – 1,990 ng/kg	Rankin et al. (2016)
Sediments (South Bay Area, California)	PFOS: 0.60 – 2.61 ng/g dry weight	Sedlak et al. (2017)

Sample Type	PFAS Type: Concentration	Reference
Sediments (Cape Fear River, North Carolina)	GenX (PFPrOPrA): 3.1 – 21.6 ng/g dry weight	UNCW (2018)
Sediments (aMatikulu and uMvoti estuaries, South Africa)	PFBA: 0.15 – 1 ng/g PFOA: 0.26 – 2.5 ng/g PFOS: 0.05 – 0.99 ng/g PFDoA: 0.03 – 0.63 ng/g PFTrDA: 0.03 – 0.37 ng/g	Fauconier et al. (2019)
South African estuaries (aMatikulu and uMvoti)	aMatikulu – PFOA: 171 – 258 ng/L uMvoti – PFOA: 711 – 788 ng/L	Fauconier et al. (2019)
U.S. river (Cape Fear, North Carolina), used for drinking water	GenX (PFPrOPrA): 631 ng/L (mean)	Sun et al. (2016)
WWTP effluent (San Francisco Bay Area, California)	PFBA: 16 ng/L (mean) PFPeA: 12 ng/L (mean) PFHxA: 26 ng/L (mean) PFBS: 2.7 ng/L (mean) PFHxS: 4.8 ng/L (mean) PFOS: 13 ng/L (mean)	Houtz et al. (2016)
WWTP effluent (Germany)	PFBA: <4.2 ng/L PFPeA: <254 ng/L	Frömel et al. (2016)
Sewage sludge	Sum of 14 PFAAs: 126 – 809 µg/kg	Yan et al. (2012)
Fish (whole fish from the Ohio, Missouri, and Upper Mississippi Rivers)	Sum of 10 PFAAs: 5.9 – 1,270 µg/kg; 53.4 µg/kg (median wet weight)	Ye et al. (2008)
Prey fish (San Francisco Bay, California)	PFOS: 11.8 µg/kg (geometric mean wet weight)	Sedlak et al. (2017)
Cormorant eggs (San Francisco Bay, California)	PFOS: 36.1 – 466 µg/kg (wet weight)	Sedlak et al. (2017)
Harbor seals serum (San Francisco Bay, California)	PFOS: 12.6 – 796 µg/kg	Sedlak et al. (2017)

Sample Type	PFAS Type: Concentration	Reference
Arctic lichen and plants	Sum of six PFCAs (C8 to C13)[10]: 0.02 – 0.26 µg/kg	Müller et al. (2011)
Baltic Sea cod (liver)	PFOS: 6.4 – 62 µg/kg	Kowalczyk et al. (2019)
Eastern Arctic beaked redfish (liver)	PFOS: 0.5 – 2.5 µg/kg	Kowalczyk et al. (2019)
Polar cod (liver)	PFOS: 1.89 – 2.15 µg/kg	Kowalczyk et al. (2019)
Arctic caribou (liver)	Sum of six PFCAs (C8 to C13): 6 – 10 µg/kg	Müller et al. (2011)
Arctic wolf (liver)	Sum of six PFCAs (C8 to C13): 10 – 18 µg/kg	Müller et al. (2011)
Arctic reindeer	PFBA: 0.079-14 µg/kg	Ahrens et al. (2016b)

Air and dust

PFASs are ubiquitous in air and dust, as evidenced by numerous recent studies. All dust samples collected from 18 California child care centers in 2018 contained at least seven PFCAs (PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFTrDA, and PFTeDA), two PFSAs (PFBS and PFOS), and two PFAA precursors (6:2 FTOH and 8:2 FTOH), with 6:2 FTOH and 6:2 FTSA found at highest levels (Wu et al. 2020). Similarly, all dust samples collected from eight child care facilities in other U.S. states (Washington and Indiana) contained PFASs, and 6:2 FTOH was the dominant compound (Zheng et al. 2020). All indoor air and dust samples collected in 2015 from residences, hotels, outdoor equipment stores, curtain stores, and carpet stores in China contained at least one PFAA or precursor (FTOHs, FOSE/FOSAs, or diPAPs) (Yao et al. 2018). A recent study identified 34 emerging PFASs (including PFECAs and short-chain PFAA precursors) in indoor and outdoor airborne particulate matter from five Chinese cities (Yu et al. 2018). PFASs were found in indoor dust in homes sampled on four continents, as well as in British cars, classrooms, and offices (Goosey and Harrad 2011). A review and analysis of literature data published after 2010 concluded that neutral PFASs, mainly FTOHs, FASAs, and FASEs, are most dominant in indoor air due to their low water solubility and high vapor pressure (Jian et al. 2017). 6:2 FTAC and 6:2 FTMAC were also frequently detected, sometimes at high concentrations, in children's bedroom air sampled in Finland during 2014/2015 (Winkens et al. 2017b). 6:2, 8:2, and 10:2 FTOHs have been

[10] C8 to C13 refers to the number of carbon atoms in the PFCA molecule. This range thus covers: PFOA, PFNA, PFDA, PFUnA, PFDoA, PFTrDA.

detected in indoor and outdoor ambient air (Jahnke et al. 2007; Karásková et al. 2016; Kim et al. 2007; Liu et al. 2013b; Shoeib et al. 2011; Tian et al. 2016). The estimated atmospheric lifetime of FTOHs indicates that atmospheric transport can contribute to widespread human exposures and environmental burdens (Ellis et al. 2003). Human exposure to PFAAs, FASAs, FOSEs, and other PFASs can also occur through house dust (Jian et al. 2017), which can have significantly higher PFAS levels than background concentrations in urban soils (Tian et al. 2016; Xu et al. 2013). PFAA precursors in air and dust were observed to correlate with PFAA levels in human serum (Makey et al. 2017). According to a meta-analysis, U.S. household dust can contain hundreds of chemicals (Moschet et al. 2018), including some with hazard traits similar to those of PFASs, such as flame retardants, phthalates, and environmental phenols (Mitro et al. 2016). Because Americans spend, on average, more than 90 percent of their time indoors (Klepeis et al. 2001), incidental ingestion and inhalation of indoor dust represent potentially significant exposure pathways to multiple contaminants along with PFASs.

Surface water and groundwater

PFASs have been detected in creeks and rivers (D'eon et al. 2009), lakes (De Silva et al. 2011), glacial meltwater and snow (Skaar et al. 2018), oceans (Benskin et al. 2012a; González-Gaya et al. 2019; Zhang et al. 2019c), estuaries (Fauconier et al. 2019), and groundwater (Eschauzier et al. 2013; Xiao et al. 2015). Surface waters can contain high levels of PFASs, especially downstream of industrial facilities such as manufacturing sites (Konwick et al. 2008). In fresh water and coastal water, PFHxA and PFHpA have been reported at levels comparable to or higher than PFOA (Cousins et al. 2011). PFPAs have also been detected in surface water (D'eon et al. 2009). In the San Francisco Bay Area of California, PFASs were detected in bay water (Sedlak et al. 2017) and groundwater, with concentrations ranging from 19 to 192 ng/L (PFOS), <LOD to 22 ng/L (PFOA), and <20 ng/L (PFHxS, PFDS, PFDA, and two PFOS precursors) (Plumlee et al. 2008). In the raw water of a drinking water treatment plant in the Cape Fear River watershed, North Carolina, Sun et al. (2016) found the PFECA GenX, a PFOA replacement in fluoropolymer manufacturing, at high levels (631 ng/L on average, with a maximum of approximately 4,500 ng/L). In a recent U.S. EPA – U.S. Geological Survey study, PFBS and PFOA were detected in the source waters of all 25 drinking water treatment plants tested from across the U.S., while PFHxS, PFOS, PFBA, PFPeA, PFHxA, PFHpA, PFNA, and PFDA were detected at over 90 percent of the sites (Boone et al. 2019). Certain PFECAs and PFESAs were detected in surface waters in all countries on three continents surveyed recently, indicating global distribution and contamination (Pan et al. 2018). The ultra-short-chain PFAS TFA was also detected in seawater, rain, and melted snow on several continents (Ateia et al. 2019a). Contaminated surface waters can be ingested by livestock and wild animals, leading to exposure up the food chain.

WWTP effluent

PFASs are commonly found in municipal and industrial wastewaters (Appleman et al. 2014; Arvaniti et al. 2015a; Chen et al. 2017b; Clara et al. 2008; Sun et al. 2012; Coggan et al. 2019). Industrial

wastewater can have PFOA and PFOS levels above 1,000 ng/L (Kim et al. 2012b; Lin et al. 2014a). PFPAs have also been detected in WWTP effluent (D'eon et al. 2009). WWTPs are major point sources for PFAS contamination of aquatic environments (Ahrens 2011; Dalahmeh et al. 2018; Frömel et al. 2016; Gallen et al. 2018; Guerra et al. 2014). Degradation of PFAA precursors, such as FTOHs, within WWTPs can lead to increases in effluent PFAA concentrations (Ahrens 2011; Dalahmeh et al. 2018; Frömel et al. 2016; Gallen et al. 2018; Guerra et al. 2014). Despite the phaseout of PFOS- and PFOA-containing stain- and soil-repellent treatments for converted textiles or leathers, these PFASs are frequently detected in wastewater, indicating their ongoing release into the environment (Gallen et al. 2018; Guerra et al. 2014). A study of effluent from WWTPs in the San Francisco Bay Area detected several PFAAs including PFHxA, PFOA, PFBA, and PFOS, with highest median effluent concentrations of 24, 23, 19 and 15 ng/L, respectively (Houtz et al. 2016). A subsequent study (Sedlak et al. 2017) of samples collected in 2012 from three San Francisco Bay WWTPs found mean PFOA, PFOS, PFHxA, and PFPeA concentrations as high as 39.7, 42.3, 30.1, and 21.2 ng/L, respectively.

Landfill leachate

PFASs are commonly detected in landfill leachate around the world (Benskin et al. 2012b; Fuertes et al. 2017; Gallen et al. 2017; Hamid et al. 2018; Knutsen et al. 2019). A study of 95 leachate samples from 18 U.S. landfills estimated the annual release of PFCAs, FTCAs, and PFSAs and precursors to be 291 kg/yr, 285 kg/yr, and 84 kg/yr, respectively (Lang et al. 2017). A Swedish study of 26 PFASs in samples from groundwater, surface water, WWTP effluent, and landfill leachate found that landfill leachates had the highest average total PFAS concentrations (487 ng/L) (Ahrens et al. 2016a). Shorter-chain PFAAs have become the most abundant PFASs in landfill leachate, reflecting the relatively recent market shift from longer- to shorter-chain compounds (Hamid et al. 2018; Knutsen et al. 2019; Solo-Gabriele et al. 2020). A study of leachate from municipal and industrial landfills in Sweden detected also ultra-short-chain PFASs (with three or fewer carbon atoms) such as TFA, perfluoropropanoic acid (PFPrA), and trifluoromethane sulfonic acid (TFMS) (Björnsdotter et al. 2019).

Soils

PFASs are ubiquitous in soils. A survey of soils around the world, including remote locations such as Antarctica and the Arctic Circle, found PFASs in all samples (Llorca et al. 2012; Rankin et al. 2016; Hanssen et al. 2019). Short-chain PFAAs were found in higher concentrations than PFOS in soil samples collected from residential areas throughout China, reflecting their widespread use (Li et al. 2020b). Side-chain fluorinated polymers were detected in all biosolid-treated soil samples measured in a Canadian study (Chu and Letcher 2017a). Soil and water contaminated with PFASs can lead to plant and soil organism uptake and transfer to grazing animals potentially consumed by humans (Navarro et al. 2017; Yoo et al. 2011). An evaluation of agricultural lands in Kampala, Uganda, found PFASs in the terrestrial food chain, including in plants such as maze, yam, and sugar cane (Dalahmeh et al. 2018). Some plant species show a very high PFAS accumulation potential, with higher BCFs for shorter-chain

PFAAs are found in numerous marine species, including plankton, sea turtles, seals, California sea otters, whales, fish, sharks, polar bears, dolphins, and marine bird eggs; this indicates potential for widespread adverse impacts throughout the marine food web.

PFASs. For example, the BCF (plant/soil ratio) of 6:2 FTSA in birch leaves can be up to 143,700 (Gobelius 2016). Based on their ability to uptake and sequester PFASs in aboveground biomass, some plants have been proposed for use in the phytoremediation of PFAS-contaminated soil (Gobelius 2016).

Sediments

Sediments can also become contaminated. (White et al. 2015) analyzed PFASs in estuarine sediments from the Charleston Harbor and the Ashley and Cooper rivers (n=36) in South Carolina, finding higher concentrations of 11 PFASs than previously reported for U.S. urban areas. The PFASs were dominated by PFOS, followed by PFDA and PFOA (White et al. 2015). Eighteen PFCAs and PFSAs were detected in the archived surface sediments of five major rivers in China (Pan et al. 2014a). Consistent with other studies of sediments from lakes, reservoirs, and estuaries, the most frequently detected PFASs were PFOA and PFOS (detection frequency of 100 and 83 percent, respectively), with greater concentrations in urban areas (Pan et al. 2014b) PFPAs and PFPiAs have been detected in lake sediments, but at lower concentrations compared to other PFAAs (Guo et al. 2016). A recent study found that Cape Fear River sediments appear to act as a repository for GenX, potentially releasing it into the water column (UNCW 2018). Concentrations of PFAAs, including PFHxA and PFBA, have been found at higher concentrations at ocean depths than at the surface, indicating the potential for deep ocean sediments to serve as the ultimate sink for PFAAs that adsorb to particulate matter (Prevedouros et al. 2006; Sanchez-Vidal et al. 2015).

Biota

PFAAs are found in numerous species, including plankton (Casal et al. 2017; Zhang et al. 2019c), sea turtles (Keller et al. 2005; O'Connell et al. 2010; Keller et al. 2012), seals (Routti et al. 2016), California sea otters (Kannan et al. 2006), whales (Hart et al. 2008), fish (Wong et al. 2017), penguins (Llorca et al. 2012) sharks (Kumar et al. 2009), polar bears (Dietz et al. 2008; Liu et al. 2018b; Tartu et al. 2018), dolphins (Adams et al. 2008; Houde et al. 2005; Lynch et al. 2019), seabirds (Escoruela et al. 2018; Roscales et al. 2019), white-tailed eagles (Sun et al. 2019), bald eagle eggs (Wu et al. 2019), and marine bird eggs (Verreault et al. 2007), including PFPiAs in the blood of fish, birds, and dolphins (De Silva et al. 2016). This indicates potential for widespread adverse impacts throughout the marine food web. The Washington State Department of Ecology surveyed regional lakes and analyzed water, fish tissue, and osprey eggs for PFAS contamination. They found that PFOS biomagnified through trophic levels by

several orders of magnitude, but they did not observe any bioaccumulation of shorter-chain PFASs (State of Washington 2017).

As shorter-chain PFAA precursors have gained market share, the concentrations of shorter-chain PFAAs in the marine environment have increased, as evidenced by increasing PFBS levels in cetaceans from 2002-2014 (Lam et al. 2016). The PFBS precursor perfluorobutane sulfonamide (FBSA) also bioaccumulates in fish (Chu et al. 2016). 6:2 Cl-PFESA, currently used as a PFOS substitute in the chrome plating industry, has been detected in a wide range of marine organisms, including several species of gastropods, bivalves, crabs, shrimps, cephalopods, and fish, and tends to magnify along the food chain (Liu et al. 2017). 6:2 Cl-PFESA and hexafluoropropylene oxide trimer acid (a PFOA replacement) accumulate in frogs, particularly in male frog skin, liver, and muscle, and in female frog ovaries, raising concerns regarding potential developmental toxicity (Cui et al. 2018).

PFCAs are found in Arctic seals and Antarctic seabirds, presumably due to long-range transport of precursors such as FTOHs and FASAs via oceanic and atmospheric currents (Llorca et al. 2012; Routti et al. 2016; Roscales et al. 2019). In an Antarctic ecosystem, shorter chains dominated the PFAA content of sediments, algae, gastropods, and fish (Gao et al. 2020). In the Arctic, increased coastal PFAS concentrations can follow ice and snow melt, resulting in an increased exposure potential for marine biota in Arctic coastal regions (Zhao et al. 2012). Numerous studies have documented high concentrations of PFASs in Arctic mammals. For instance, up to 17 PFASs were found in the livers of marine mammals off the coast of Greenland, including killer whales, polar bears, and ringed seals (Gebbink et al. 2016). A long-term study (1968-2015) of archived white-tailed eagle feathers collected in Greenland, Norway, and Sweden showed increasing levels of PFCAs over time, but a decrease in PFOS levels in feathers from Greenland and Norway consistent with the 3M phaseout that took place in 2000 (Sun et al. 2019). A similar study in Norway for the decade of 2008-2017 found decreasing concentrations of PFOS and PFHxS over time, but no significant trends in PFOA and PFNA levels, and a plateauing in the levels of C11 and longer PFCAs in the last four years of the study (Jouanneau et al. 2020). In biological samples taken from Norwegian animals, the highest PFAS levels were measured in arctic foxes, otters, and polar bears; the study also detected PFASs in wolves for the first time in Norway (Hanssen et al. 2019). A Swedish study showed that contamination is exacerbated by point sources such as manufacturing plants and firefighting activities; PFASs were detected in the impacted surface water samples collected near such sources at levels up to 1,920 ng/L and in invertebrates at levels up to 767 ng/g (Koch et al. 2019).

PFASs can be transported across the brain blood barrier and accumulate in animal brain tissue, leading to concerns about potential neurotoxicity (Greaves et al. 2013). A study of 128 young polar bears (between 3 and 5 years old) sampled between 1984 and 2006 estimated that longer-chain PFASs in polar bears will reach levels associated with adverse impacts in lab animals by 2014-2024 (Dietz et al. 2008). PFASs were found to biomagnify throughout the Arctic food chain, with increasing

concentrations in wolves that consume caribou contaminated with PFASs via their lichen-based diet (Müller et al. 2011). Researchers also evaluated the effect of cyclic seasonal emaciation on the distribution and composition of PFASs in Arctic foxes, finding significant differences that could indicate increased potential for health effects from PFASs when foxes have reduced access to food (Aas et al. 2014).

Monitoring data showing that the Candidate Chemical is present in human food

Exposure modeling studies have concluded that the ingestion of contaminated food and drinking water is the largest contributor of PFAAs to the overall human body burden (Trudel et al. 2008). A study of 2013-2014 NHANES data concluded that diet is the dominant and most important source of PFAA exposure for children aged 3 to 11, even more so than for adolescents and adults (Jain 2018).

Contamination of food with PFASs occurs via contact with PFAS-treated food packaging materials (Trier et al. 2011a), as well as from plant and animal uptake from contaminated soil and water (Dalahmeh et al. 2018). Animal feed can also become contaminated when grown in soil treated with PFAS-containing irrigation water or biosolids (Gobelius 2016; Lee et al. 2014; Yoo et al. 2011).

The use of PFAS-contaminated irrigation water has been linked to elevated PFAA concentrations in fruits and vegetables (Blaine et al. 2014b; Zhang et al. 2016; Scher et al. 2018; Zhang et al. 2020). The application of PFAS-contaminated biosolids to land leads to the bioaccumulation of PFAAs in grasses (Yoo et al. 2011) and food crops (Blaine et al. 2014a; Navarro et al. 2017).

The PFASs currently used in food packaging products degrade into shorter-chain PFAAs that show greater bioaccumulation in vegetable crops than the longer-chain PFAAs formerly used in food packaging products (Blaine et al. 2013). Composted biosolid samples from the U.S. and Canada had three times higher levels of PFHxA in 2018 compared to 2016, while PFOA and PFOS remained nearly the same (Kim Lazcano et al. 2019). The study authors interpreted this finding as a reflection of the recent market shift to shorter-chain compounds.

Studies of PFASs in foods (see examples in Table 5) found them in wide-ranging concentrations (Domingo and Nadal 2017; EFSA 2012). Most of the available studies are from Europe and Asia, but levels are expected to be similar in the U.S., including in California. From 2006-2012, the European Food Safety Agency (EFSA 2012) tested 7,560 food samples and found PFASs in most of them. The most prevalent PFAS contaminants were PFOS (29 percent) and PFOA (9 percent). Fifty-three samples collected in various developing countries were analyzed with a novel test method, with limits of detection as low as 3.1 pg/g for PFAAs. PFOS and PFOA were detected in all these samples, and PFHxS in 80 percent of samples (Sadia et al. 2019). PFBS and PFHxA were found to accumulate in food crops, including sugar beets, broccoli, spinach, lettuce, celery, potato, pea, fruit, and green beans (EFSA 2012). A review and analysis of literature data published after 2010 found that shorter-chain PFAAs, including PFBA, PFPeA, and PFHxA, were frequently detected at high concentrations in vegetables,

fruits, and beverages (Jian et al. 2017). PFAAs, including longer and shorter chains, were found to accumulate in the roots and shoots of wheat (Zhao et al. 2018; Zhang et al. 2019b).

Fish appears to be the most frequently PFAS-contaminated food (Yamada et al. 2014a; Yamada et al. 2014b), estimated to contribute the most to dietary exposure (Domingo et al. 2017; EFSA 2012). One study measured PFASs in 21 foods and beverages in Norway, reporting concentrations of PFOS in cod and cod liver that were much higher than in other foods, including other types of fish (Haug et al. 2010). A study in Germany estimated that people who consume 150-300 g of beaked redfish or cod fillet per week are exposed to up to 322 and 198 ng of PFAAs each week, respectively (Kowalczyk et al. 2019). A study of pregnant women and children from six European countries concluded that fish consumption during certain stages of pregnancy is associated with higher concentrations of PFASs in serum (Papadopoulou et al. 2019). Nevertheless, terrestrial animals can also have high PFAS concentrations (EFSA 2012; Haug et al. 2010). For instance, several PFAAs were found in the liver and muscle of beef cattle in China (Wang et al. 2017a).

The PFASs currently used in food packaging degrade into shorter-chain PFAAs that show greater bioaccumulation in vegetable crops than the longer-chain PFAAs formerly used in food packaging products.

A 2012 European Union study estimated that PFOA and PFOS dietary exposure doses were much lower than the tolerable daily intake (TDI) established by EFSA in 2008. Dietary exposure to PFOS in toddlers was estimated to contribute almost 20 percent of the TDI in the most exposed group (EFSA 2012). The TDI was based on a sub-chronic study in Cynomolgus monkeys, which did not consider exposure sources other than dietary, nor exposure to PFOA and PFOS precursors (EFSA 2008). In 2018, however, EFSA proposed to lower its TDI from 150 ng/kg body weight per day to 13 ng/kg body weight per week for PFOS, and from 1,500 ng/kg body weight per day to 6 ng/kg body weight per week for PFOA (Knutsen et al. 2018). According to EFSA, “exposure of a considerable proportion of the population” exceeds the proposed limits for both compounds (Knutsen et al. 2018). TDIs for other PFASs have not been established.

Dietary exposure studies in the U.S. are generally lacking (Domingo et al. 2017), although one study in California found significant positive associations between PFAS serum concentrations for children and adults and consumption of certain foods including butter/margarine, fish, meat products, and microwave popcorn (Wu et al. 2015). Another U.S. study found higher serum levels of PFOA, PFNA, PFDA, and PFOS were associated with consumption of microwave popcorn (Susmann et al. 2019). The globalization of the human food chain, combined with widespread contamination of foods, indicates the potential for lifetime exposure to PFASs for the people of California.

Table 5: Monitoring studies found varying levels of PFASs in human food.

Sample Type	PFAS Type	Concentration	Reference
Potatoes	PFOA	0.07 µg/kg	Schecter et al. (2010)
Vegetables	PFHxA	<LOD – 0.42 µg/kg	EFSA (2012)
Fruit	PFHxA	<LOD – 0.17 µg/kg	EFSA (2012)
Fruit	PFBS	<LOD – 0.067 µg/kg	EFSA (2012)
Fruit	PFHxS	<LOD – 0.20 µg/kg	EFSA (2012)
Meat products	PFOA	<LOD – 0.24 µg/kg	Schecter et al. (2010)
Beef	PFOS	0.0832 µg/kg	Sadia et al. (2019)
Crustaceans	PFOA	<LOD – 8.0 µg/kg	EFSA (2012)
Fish	PFOA	<LOD – 0.30 µg/kg	Schecter et al. (2010)
Fish	PFHxA	<LOD – 23 µg/kg	EFSA (2012)
Fish	PFOA	<LOD – 18.2 µg/kg	EFSA (2012)
Fish	PFOS	<1 ng/g – >100 µg/kg	Berger et al. (2009)
Fish and other seafood[11]	PFOS	<LOD – 310 µg/kg	EFSA (2012)
Fish	PFOS	0.3 - 62 µg/kg	Kowalczyk et al. (2019)
Fish	PFNA	<LOD - 18 µg/kg	Kowalczyk et al. (2019)
Fish	PFOS	0.087 µg/kg	Sadia et al. (2019)
Chicken	PFOS	0.0257 µg/kg	Sadia et al. (2019)
Milk	PFOS	0.0899 µg/kg	Sadia et al. (2019)
Eggs	Sum of 11 PFASs	27 – 160 µg/kg	Wang et al. (2008)
Eggs and products	PFOA	<LOD – 25.5 µg/kg	EFSA (2012)
Eggs and products	PFOS	<LOD – 6.4 µg/kg	EFSA (2012)
Eggs	PFOS	0.212 µg/kg	Sadia et al. (2019)
Home produced eggs (yolks) two km from industrial site	Sum of 12 PFAAs	8.99 – 482 µg/kg	Su et al. (2017)
Butter	PFOA	1.07 µg/kg	Schecter et al. (2010)
Butter	PFOS	0.114 µg/kg	Sadia et al. (2019)

[11] PFOS mean concentrations were “constantly higher” in freshwater fish than in marine fish.

Sample Type	PFAS Type	Concentration	Reference
Margarine	PFOA	0.19 µg/kg	Schecter et al. (2010)
Olive oil	PFOA	1.80 µg/kg	Schecter et al. (2010)
Honey	PFOA	0.00 – 0.47 µg/kg	EFSA (2012)

Drinking water can become contaminated with PFAAs and their precursors from food packaging via several routes, including direct discharge of industrial effluent into waterways; leaching from land-applied compost, wastewater, or biosolids; leachate draining from landfills that contain discarded products; and discharge of WWTP effluent into streams and rivers.

Monitoring data showing that the Candidate Chemical is present in drinking water

PFASs are detected in drinking water around the world, typically in the nanogram per liter (ng/L) range (Domingo et al. 2019). Drinking water can become contaminated with PFAAs and their precursors from food packaging via several routes, including direct discharge of industrial effluent into waterways; leaching of land-applied compost, wastewater, or biosolids; leachate draining from landfills that contain discarded products; and discharge of WWTP effluent into streams and rivers (Lindstrom et al. 2011a). A review and analysis of literature data published after 2010 found that, out of all PFASs measured to date, PFOS, PFHxA, PFHpA, PFNA, PFDA, and PFOA had the highest concentrations and detection frequencies in drinking water (Jian et al. 2017). A study of 133 PFASs in drinking water (bottled and tap) from nine countries, including the U.S., found shorter-chain PFAAs in 64 to 92 percent of the samples (Kaboré et al. 2018).

As part of the third Unregulated Contaminant Monitoring Rule (UCMR 3) program, U.S. EPA (2017a) monitored public water supplies for six PFAAs from 2013 until 2015, reporting their presence in approximately 3 percent of samples tested (Table 6). PFOA and PFOS concentrations equaled or exceeded the lifetime total health advisory of 70 ng/L, combined, in water serving approximately 16.5 million people, with most detects (28 out of 452 samples, or 6.2 percent detection rate) in California (Hu et al. 2016; Hurley et al. 2016). The most significant predictors of observed PFAS detection frequencies and concentrations were: (1) proximity to industrial sites that manufacture or use PFASs; (2) proximity to military fire training areas; and (3) the number of nearby WWTPs (Hu et al. 2016).

The Minimum Reporting Levels (MRLs) for the six PFASs measured by the UCMR 3 program ranged from 10 to 90 ng/L, whereas the method used has a limit of quantification in the 2-5 ng/L range. Eurofins Eaton Analytical (2017), which was responsible for analyzing approximately 40 percent of the

UCMR 3 data, found PFASs in more than a tenth of the samples it analyzed. However, fewer than 2 percent of those samples showed values above the UCMR 3 MRL and were reported to U.S. EPA. This suggests that the UCMR 3 program may have significantly underestimated the prevalence of PFASs in U.S. drinking water. A U.S. EPA-U.S. Geological Survey joint study (Boone et al. 2019) retested 25 of the drinking water treatment plants sampled under UCMR 3. The authors found PFBS, PFHxA, and PFOA in 100 percent of the treated water samples, and PFPeA, PFHpA, PFNA, and PFOS in over 90 percent of those samples. The shorter-chain PFAAs showed no significant decline between the source and the treated water (Boone et al. 2019). A recent U.S. EPA literature review found that, in addition to the six PFASs measured during UCMR3, 23 other PFASs in drinking water were reported to date (Crone et al. 2019).

Drinking water contamination can linger for decades. In August 2017, the town of Belmont, Michigan, measured PFASs in some of its drinking water at levels 540 times above the U.S. EPA health advisory level (32,000 ng/L PFOS and 5,800 ng/L PFOA). The contamination is linked to a hazardous waste site used in the 1960s by a former tannery, despite the fact that no PFASs have been introduced to the site since the 1970s (Chawaga 2017; Ellison 2017).

Table 6: Monitoring studies found varying levels of PFASs in drinking water.

Sample Type	PFAS Type: Concentration	Reference
U.S. drinking water[12]	PFOA: <MRL – 349 ng/L PFOS: <MRL – 1,800 ng/L PFNA: >MRL – 56 ng/L	Hu et al. (2016)
U.S. drinking water	PFBS: <0.032 – 11.9 ng/L PFHxS: ND – 38.4 ng/L PFOS: ND – 36.9 ng/L PFBA: ND – 104 ng/L PFPeA: ND – 514 ng/L PFHxA: 0.09 – 60.8 ng/L PFHpA: ND – 177 ng/L PFOA: <0.56 – 104 ng/L PFNA: ND – 38.6 ng/L PFDA: ND – 24.7 ng/L	Boone et al. (2019)
European drinking water	PFBS: <LOD – 0.24 µg/kg PFHxS: <LOD – 0.011 µg/kg PFOS: <LOD – 0.016 µg/kg	EFSA (2012)

[12] The UCM3 MRLs (Minimum Reporting Levels) for PFASs in U.S. drinking water are 10 – 90 ng/L depending on the PFAS being tested.

Monitoring data showing that the Candidate Chemical is present in California environmental media at levels of concern

Numerous drinking water sources in California are contaminated with PFAAs, according to the UCMR 3 study. Forty percent of the positive UCMR 3 study detections in California are above the U.S. EPA advisory level of 70 ng/L for PFOA and PFOS combined (Hurley et al. 2016). In February 2020, the California State Water Resources Control Board reported data from 600 water system sites. These sites were adjacent to airports with fire training areas or near municipal solid waste landfills (California State Water Resources Control Board 2019). Sampling was conducted in two rounds, the first between April and June 2019 (Q1) and the second between July and September 2019 (Q2). In the Q1 data, 180 of the water system sites sampled exceeded the state's notification level of 6.5 ppt for PFOS, while 149 samples exceeded the notification level of 5.1 ppt for PFOA. In the Q2 samples, the number of sites exceeding notification levels increased to 213 for PFOS and 185 for PFOA. Two shorter-chain PFAAs – PFHxA and PFBS – were detected in 186 and 199 of the Q1 samples, respectively. In the Q2 samples, detections increased: PFHxA was measured in 279 samples and PFBS in 281 samples. In both sampling periods, most samples with positive detections contained more than one PFAS.

California sea otters were found to have PFOA and PFOS levels ranging from less than 5 to 147 ng/g and from less than 1 to 884 ng/g, respectively, and these concentrations were significantly correlated with incidence of disease (Kannan et al. 2006). Historically, San Francisco Bay wildlife has shown some of the highest PFOS levels ever measured (Sedlak and Greig 2012). Harbor seal serum sampled in 2014 still contained PFOS at concentrations ranging from 12.6 to 796 ng/g, more than a decade after the PFOS phaseout (Sedlak et al. 2017).

In 2010, contaminant monitoring of the San Francisco Bay surface water detected several PFAAs, including: PFBS at concentrations up to 7.89 ng/L (average 1.58 ng/L), PFBA at up to 62.20 ng/L (average 12.96 ng/L), and PFHxA ranging from 1.37 ng/L to 221.0 ng/L (SFEI 2010). A 2016 study of PFAAs and precursors in wastewater effluent discharged to San Francisco Bay highlighted the local impact of aqueous film-forming foam (AFFF),[13] and noted that levels of PFBA, PFPeA, and PFHxA in WWTP effluent had increased by 150 to 220 percent since 2009 (Houtz et al. 2016). Due to this widespread contamination, the San Francisco Bay Regional Monitoring Program designated PFOS as "moderate concern" for the San Francisco Bay, and all other PFASs as "possible concerns" (SFEI 2017).

Human biomonitoring data

Nearly all humans show evidence of exposure to some PFASs (Calafat et al. 2007), which can accumulate in human lungs, kidneys, liver, brain, and bone tissue (Pérez et al. 2013). A recent study

[13] AFFF may be the largest direct point source of PFASs (European Commission 2017). Generally, AFFF contains high levels of PFSAs and their precursors, but only low levels of PFCAs (Hu et al. 2016).

found PFOS, PFOA, PFNA, PFDA, PFUnA, and PFHxS in human embryonic and fetal organs, with the highest burden in lung tissue in the first trimester samples, and in the liver in the second and third trimester samples (Mamsen et al. 2019). Biomonitoring studies have been limited to certain PFASs with available analytical methods (see Table 7 for some examples); however humans are exposed to many more PFASs that haven't yet been measured (Yeung et al. 2016).

The Centers for Disease Control and Prevention (CDC), which monitors Americans' exposure to PFASs as part of NHANES, has detected PFASs in the blood of all people tested (Calafat et al. 2007). In NHANES blood samples collected in 2011-2012, average PFHxS, PFOA, and PFOS levels ranged between 1.3 and 6.3 µg/L each (ATSDR 2018a). In a Norwegian blood donor study, PFOS and PFOA were the most commonly detected PFASs, with average concentrations of 5.71 ng/mL and 1.41 ng/mL, respectively, and higher concentrations in older donors (Averina et al. 2020). PFBA and PFHxA were found in 13.3 percent and 22.6 percent, respectively, of 2682 urine samples collected from 2013-2014 NHANES participants (Calafat et al. 2019).

A biomonitoring study of over 65,000 people enrolled in the C8 Health Project, which evaluated exposures to PFASs in drinking water contaminated by DuPont, found PFHxA in the serum of more than 50 percent of the people tested (Frisbee et al. 2009). Shorter-chain PFAAs such as PFHxA, PFPeA, and PFBA often fall below the level of detection in human serum and plasma, but can be found at concentrations higher than most longer-chain PFASs in human whole blood samples (Poothong et al. 2017). They have also been detected in human hair samples at similar or higher levels compared to longer-chain PFAAs (Alves et al. 2015; Ruan et al. 2019; Martín et al. 2019). Shorter-chain PFAAs are also detected in human urine at levels comparable to or higher than PFOA (Hartmann et al. 2017; Kim et al. 2014a; Pérez et al. 2012), and may accumulate in certain organs at levels exceeding those of longer-chain PFAAs. In human autopsy tissues, out of 21 PFAAs studied, PFBA was the dominant compound in kidney and lung (median of 263 and 807 ng/g, respectively) and PFHxA in liver and brain (median of 68.3 and 141 ng/g, respectively), whereas PFOA dominated in bone (median of 20.9 ng/g) (Pérez et al. 2013).

The California Biomonitoring Program, administered jointly by the California Department of Public Health, DTSC, and OEHHA, has measured PFAA levels in several subpopulations (CECBP 2017). The California Teachers Study is a statewide collaborative research project evaluating health-related factors in the development of breast cancer. For more than 20 years, over 133,000 female schoolteachers and employees have participated in the study. As of 2015, the California Biomonitoring Program has found PFHxS, PFOA, and PFOS in more than 99 percent of 1,759 blood samples provided by study participants (CECBP 2015b). Hurley et al. (2016) analyzed blood samples from a subset of 1,333 participants who lived in areas where PFASs in drinking water concentrations were tested. Of these participants, 109 women lived in an area where at least one PFAS was detected in drinking water from public water systems. The analysis showed an association between detectable levels of PFOA and PFOS in drinking

water and the women's blood levels (Hurley et al. 2016). In a subset of 1,257 women aged 40 to 94 from the California Teachers Study, serum levels of longer-chain PFASs decreased from 2011 to 2015 on average by 10 to 20 percent per year – except for PFHxS, which did not change significantly (Hurley et al. 2018). Another study of 139 California households detected PFASs in all study participants, including young children and adults (Wu et al. 2015). Exposure levels correlated with diet and use of consumer products such as fire extinguishers and stain-repellent or waterproof clothing, and residential dust exposure in children.

In North Carolina, a study of 37 young adults detected PFHxS (1.07 – 12.55 ng/mL), PFOS (0.39 – 31.35 ng/mL), PFOA (0.30 – 4.07 ng/mL), and PFNA (0.23 – 4.02 ng/mL) in all serum samples, and PFHxA (ND – 1.00 ng/mL) and PFDA (ND – 1.60 ng/mL) in 83.8 percent and 97.3 percent of serum samples, respectively (Siebenaler et al. 2017). In Europe, biomonitoring of pregnant and nursing women from 1996 to 2010 found that PFBS doubled its blood concentrations every 6.3 years, concurrent with the phaseout of PFOS (Glynn et al. 2012). A U.S. national biomonitoring study of midlife women showed that serum concentrations of legacy PFASs, such as PFOS and PFOA, decreased significantly from 1999 to 2011, while concentrations of emerging compounds such as PFNA, PFDA, and PFUnA significantly increased over the same period (Ding et al. 2020).

Table 7: Human biomonitoring studies found widespread exposure to PFASs.

Population	Sample Type	PFAS Type: Concentration	Reference
Women aged 40 to 94	Serum	PFHpA: 0.084 ng/mL (mean); 1.16 ng/mL (max) PFOA: 3.0 ng/mL (mean); 27.6 ng/mL (max) PFNA: 1.07 ng/mL (mean); 10.4 ng/mL (max) PFDA: 0.28 ng/mL (mean); 3.9 ng/mL (max) PFUnDA: 0.17 ng/mL (mean); 1.3 ng/mL (max) PFHxS: 2.2 ng/mL (mean); 21.8 ng/mL (max) PFOS: 8.5 ng/mL (mean); 99.8 ng/mL (max)	Hurley et al. (2018)
337 California adults	Serum	Me-PFOSA-AcOH: 0.12 ng/mL (geometric mean) PFDA: 0.19 ng/mL (geometric mean) PFHxS: 1.03 ng/mL (geometric mean) PFNA: 0.79 ng/mL (geometric mean) PFOA: 1.49 ng/mL (geometric mean) PFOS: 5.21 ng/mL (geometric mean)	CECBP (2018)

Population	Sample Type	PFAS Type: Concentration	Reference
Children aged 3 to 11	Serum	PFHxS: 0.84 ng/mL (geometric mean) PFNA: 0.79 ng/mL (geometric mean) PFOA: 1.92 ng/mL (geometric mean) PFOS: 3.88 ng/mL (geometric mean) MeFOSAA: 0.110 ng/mL (50^{th} percentile) PFDA: 0.100 ng/mL (50^{th} percentile, girls ages 3-5 years; boys <LOD)	CDC (2018)
Adults	Blood (near contam-inated site)	PFOA: 3.09 ng/mL (mean); 32 ng/mL (max) PFOS: 8.59 ng/mL (mean); 95.6 ng/mL (max) PFHxS: 4.12 ng/mL (mean); 116 ng/mL (max)	NHDHHS (2016)
Adult men	Serum	PFOA: 4.50 ng/mL (mean) PFOS: 20.80 ng/mL (mean) PFHxS: 2.88 ng/mL (mean) PFNA: 1.52 ng/mL (mean)	He et al. (2018)
Adult women	Serum	PFOA: 3.46 ng/mL (mean) PFOS: 14.51 ng/mL (mean) PFHxS: 1.94 ng/mL (mean) PFNA: 1.30 ng/mL (mean)	He et al. (2018)
Pregnant women	Plasma	PFHxS: 0.37 ng/mL (mean) PFHxA: 0.16 ng/mL (mean) PFHpA: 0.18 ng/mL (mean) PFOS: 5.74 ng/mL (mean) PFOA: 2.64 ng/mL (mean) PFNA: 1.35 ng/mL (mean) PFDA: 0.57 ng/mL (mean) PFUnDA: 1.50 ng/mL (mean) PFDoA: 0.21 ng/mL (mean) PFTrDA: 0.35 ng/mL (mean) PFTeDA: 0.12 ng/mL (mean)	Tsai et al. (2018)
Pregnant women	Cord serum	PFOA: 1.59 ng/mL (mean) PFNA: 0.13 ng/L (mean) PFOS: 1.43 ng/L (mean) Sum of 11 PFASs: 3.67 ng/L mean	Cao et al. (2018)

Population	Sample Type	PFAS Type: Concentration	Reference
Newborn	Umbilical cord plasma	PFOA: 7.65 ng/L (mean); 29.97 ng/L (max) PFOS: 2.93 ng/L (mean); 65.61 ng/L (max) PFNA: 0.69 ng/L (mean); 3.29 ng/L (max) PFDA: 0.44 ng/L (mean); 5.73 ng/L (max) PFUnA: 0.45 ng/L (mean); 5.27 ng/L (max) PFDoA: 0.10 ng/L (mean); 1.14 ng/L (max) PFHxS: 0.18 ng/L (mean); 0.85 ng/L (max) PFBS: 0.05 ng/L (mean); 0.46 ng/L (max)	Liu et al. (2018a)
Breastfeeding women	Breast milk	PFPeA: 0.053 ng/mL (geometric mean) PFHxA: 0.047 ng/mL (geometric mean) PFHpA: 0.030 ng/mL (geometric mean) PFOA: 0.071 ng/mL (geometric mean) PFOS: 0.049 ng/mL (geometric mean)	Kang et al. (2016)

Evidence that the Candidate Chemical exhibits certain hazard traits (persistence, bioaccumulation, lactational/transplacental transfer)

PFAAs are highly persistent and tend to accumulate in the environment and biota, due to the extremely strong bond between the carbon and fluorine atoms that does not degrade under typical environmental conditions (Giesy et al. 2010; Liu and Mejia Avendaño 2013a).

PFAAs and nonpolymeric PFAA precursors undergo transplacental transfer in humans (Midasch et al. 2007; Zhang et al. 2013; Yang et al. 2016; Zhao et al. 2017a; Eryasa et al. 2019; Cai et al. 2020). Fluorinated ethers, such as Cl-PFESA, can also transfer from mother to fetus via the placenta (Chen et al. 2017a; Pan et al. 2017; Cai et al. 2020). The placenta-to-maternal serum ratios of PFOS, PFOA, and PFNA were observed to increase during gestation – more so in pregnancies with male fetuses compared to female ones – suggesting bioaccumulation in the placenta and increasing exposure with fetal age (Mamsen et al. 2019). Through transplacental transfer, the developing fetus is exposed to mixtures of PFAAs *in utero* (Bach et al. 2015; Mamsen et al. 2017). Up to 70.1 percent of these transplacental mixtures can come from legacy PFASs (Li et al. 2020c). Gestational diabetes appears to significantly increase the rate of PFAS transplacental transfer (Eryasa et al. 2019).

PFAAs, including the shorter chains, are found in breast milk (Nyberg et al. 2018; Jin et al. 2020), potentially adding to significant cumulative early-life exposures. A 2016 study reported detectable concentrations of PFHxA in over 70 percent of breast milk samples (n=264), and detectable PFOA levels in the breast milk of nearly 99 percent of lactating mothers, associated with exposure to PFASs in consumer products (Kang et al. 2016). A study of lactating mothers in Korea found that PFAS concentrations in breast milk increased significantly after the first month of breastfeeding, likely due to

a change in diet (Lee et al. 2018b). A similar study in Ireland detected four PFASs in breast milk (Abdallah et al. 2020). Breastfed newborns can have higher body burdens of PFAAs compared to those fed only infant formula (Fromme et al. 2010). Kim et al. (2014b) analyzed concentrations of PFASs in the serum and urine of children and found a correlation between longer duration of breastfeeding and serum concentrations.

Transplacental transfer to the fetus and breastfeeding appear to be significant PFAA elimination routes for women. Serum concentrations of PFOA and PFOS are estimated to decrease by 2 to 3 percent per month while breastfeeding (Mondal et al. 2014). Biomonitoring studies of infants and small children suggest that peak concentrations of PFOS and PFOA occur before children turn 2 years old (Winkens et al. 2017a).

5. POTENTIAL FOR SIGNIFICANT OR WIDESPREAD ADVERSE IMPACTS

Reference: California Code of Regulations, title 22, section 69503.2(a).

This section integrates the information provided in the profile to demonstrate how the key prioritization principles, as identified in the SCP regulations, are met.

Two of DTSC's goals listed in the 2018-2020 Priority Product Work Plan are to protect Californians from chemicals that migrate into food from food packaging, and to protect children, women of childbearing age, and pregnant women from exposures to harmful chemicals – especially carcinogens, mutagens, reproductive toxicants, neurotoxicants, developmental toxicants, and endocrine disruptors (DTSC 2018). Numerous stakeholders identified food packaging as an important category and requested that DTSC address it. The use of PFASs in food packaging is of particular concern because of the many hazard traits displayed by this class of chemicals. On January 14, 2020, DTSC organized a workshop about PFASs and their alternatives in food packaging. Panelists, including representatives from academia, industry, NGOs, and other government entities, encouraged DTSC to pursue regulating food packaging products containing PFASs due to the demonstrated potential for exposures, including at end-of-life, and the extreme persistence displayed by these chemicals that could add to potential significant and widespread adverse impacts (DTSC 2020).

There is a broad consensus on the importance of protecting children's health from environmental contaminants such as the PFASs used in food packaging. The American Academy of Pediatrics released a policy statement in 2018 on the risk that food additives pose to children's health, highlighting the potential adverse effects associated with PFASs in food packaging (Trasande et al. 2018). This statement cites support from the Endocrine Society, a joint 2013 report from the World Health

Organization and United Nations Environment Program, and a statement from the International Federation of Gynecology and Obstetrics in 2015.

Food packaging can expose humans and biota to PFASs during product manufacturing, use, and at end-of-life. Exposing humans and other organisms to PFASs from food packaging has the potential to contribute to or cause significant and widespread adverse impacts, based on multiple factors considered by the Safer Consumer Products regulations.

Given (1) the sheer number of PFASs being produced or explored for use worldwide, (2) the potential for irreversible human and ecological exposures to PFASs throughout the life cycle of food packaging, and (3) the potential for these exposures to contribute to or cause significant or widespread adverse impacts, DTSC proposes to list plant fiber-based food packaging containing PFASs as a Priority Product.

Adverse impacts linked to the Candidate Chemical's hazard traits

Reference: California Code of Regulations, title 22, section 69503.3(a).

The SCP regulations direct the Department to evaluate the potential for the Candidate Chemical to contribute to or cause adverse impacts by considering several adverse impact factors for which information is reasonably available.

As discussed in Chapter 3, the PFASs associated with food packaging products show several exposure potential hazard traits, including persistence, mobility, bioaccumulation, and lactational and transplacental transfer. These exposure potential hazard traits stem from the PFASs' unique physicochemical properties and environmental fate (see Chapter 3). PFAAs are extremely persistent, with no known natural degradation pathways under environmental conditions; most other PFASs degrade to PFAAs in the environment and within living organisms, i.e., they are PFAA precursors (Buck et al. 2011; D'eon and Mabury 2010; Krafft et al. 2015a; Wang et al. 2014a; Washington et al. 2015). The extreme environmental persistence of PFAAs, regardless of chain length, leads to their continuous presence in the environment, hence to continuous chronic background exposures for humans and biota (Brendel et al. 2018). Persistence has been called "the most important single criterion affecting chemical exposure and risk via the environment," because persistent chemicals may travel longer distances and affect a larger population, have increased potential for bioaccumulation across food webs, and are very slow to remove from the environment through natural processes or engineered remediation (Mackay et al. 2014). Also, removal of chemicals of concern is only feasible for hotspots and cannot address background environmental contamination (Cousins et al. 2019a).

These extremely persistent chemicals are now global pollutants that can be found in all environmental compartments. Once PFASs enter the environment, there is no practical way to remove them. Their

widespread presence has led to contaminated drinking water and human food chains, resulting in a growing public health concern (APHA 2016). In particular, shorter-chain PFASs are highly mobile in soil and water, accumulate in edible plant parts, and are difficult and expensive to remove during water treatment (Brendel et al. 2018; Boone et al. 2019; Ateia et al. 2019a; Crone et al. 2019). Conventional, aeration, oxidation, and biological treatment are ineffective for removing PFASs from drinking water (Crone et al. 2019). Nanofiltration was the only treatment that efficiently removed PFASs from contaminated drinking water in a recent study (Boiteux et al. 2017). Anion exchange resins and high-pressure membranes such as nanofiltration and reverse osmosis can remove over 90 percent of PFAAs from drinking water, including shorter-chain ones. However, these methods are expensive, perform poorly at removing neutral PFAA precursors, and generate large waste streams with concentrated PFAAs that need to be disposed of (Crone et al. 2019).

Exposing humans and other organisms to PFASs from food packaging has the potential to contribute to or cause significant and widespread adverse impacts, based on multiple factors considered by the Safer Consumer Products regulations.

Furthermore, as discussed in Chapter 3 and detailed in Appendix C, shorter-chain PFAAs, which are the final degradation products of the PFAS formulations used in U.S. food packaging, display several toxicological hazard traits, including developmental toxicity, endocrine toxicity, hematotoxicity, immunotoxicity, neurodevelopmental toxicity, reproductive toxicity, and ocular toxicity. Their environmental hazard traits include phytotoxicity and impairment of wildlife development, reproduction, or survival, however data are limited to a very small number of PFASs and organisms (ITRC 2020b). Longer-chain PFAAs, which may be found in imported products, display the following additional toxicological hazard traits: carcinogenicity, cardiovascular toxicity, hepatotoxicity and digestive system toxicity, nephrotoxicity and other urinary system toxicity, and respiratory toxicity. Studies have also suggested that PFAAs, including PFHxA, may contribute to mixture toxicity and enhance the adverse impacts associated with other hazardous compounds (see Cumulative Effects below). The ubiquitous presence of PFASs in aquatic environments and the limited toxicological knowledge about cumulative impacts to aquatic species have led to increasing concern about the potential for irreversible adverse aquatic impacts (Scheringer et al. 2014).

The PFASs currently used in food packaging may also have other hazardous degradation products. FDA's safe-use determinations for PFASs in food-contact materials are based mainly on data submitted by industry for PFHxA, the final degradation product of those formulations. However, recent research results by FDA scientists suggest that the hazard traits of PFHxA are not reflective of the hazard traits of its precursors. In fact, the researchers found that some intermediate degradation products of PFASs approved for use in food-contact materials display significantly higher bioaccumulation potential and

toxicity compared to PFHxA itself. The fact that the hazards of PFHxA precursors are not accounted for in industry safety assessments (Kabadi et al. 2020; Rice et al. 2020) raises greater concerns for human exposure to PFASs that migrate out of food packaging into food.

As noted in Chapter 4, paper, paperboard, and molded fiber products are sometimes composted at their end-of-life, which releases PFASs into the compost (Lee et al. 2018a). PFASs can be taken up by plants grown in the contaminated soil (Trier et al. 2011a; Blaine et al. 2014a), and can make their way up the food chain, impacting humans and biota. Similarly, when food packaging is landfilled, PFASs can migrate into landfill leachate, contaminating surface waters (Allred et al. 2015; Lang et al. 2017). Recycling and incineration also perpetuate the presence of PFASs in the environment, leading to increased human and ecological exposures. Discharges of PFAS-contaminated WWTP effluents into streams and rivers have led to contamination of water and sediments at concentrations considered potentially hazardous to wildlife and threatening to biodiversity (Rodriguez-Jorquera et al. 2016).

Biosolids, especially from WWTPs that treat leachate from landfills and composting facilities, can be heavily contaminated with PFAAs and precursors that adsorb to organic matter (Arvaniti et al. 2012; Gallen et al. 2018; Kim Lazcano et al. 2019; Sun et al. 2011; Kim Lazcano et al. 2020). Side-chain fluorinated polymers were detected in all biosolid-treated soil samples in a recent Canadian study (Chu and Letcher 2017b), suggesting that when biosolids are applied for land use, the contained PFASs can partition into the soil. Shorter-chain PFAAs have been observed to migrate out of soil following biosolid application, with disappearance half-lives ranging from 88 to 866 days (Venkatesan and Halden 2014b). They can also leach into adjacent waterways (Sepulvado et al. 2011), potentially leading to widespread contamination and ecological impacts. Land application of biosolids contaminated with PFAAs and their precursors can lead to bioaccumulation of PFAAs in grasses (Yoo et al. 2011) and food crops (Blaine et al. 2014a; Lee et al. 2014).

Some studies indicate the potential for shorter-chain PFASs to modulate biological responses, which we recognize is not a robust indicator of toxicological hazard. The relationship between exposure and toxicological outcomes, particularly for PFASs other than PFOA and PFOS, is a major data gap (DeWitt 2015). It remains unclear whether serum concentration or other exposure metrics are useful for estimating exposures and potential adverse impacts. Also missing are early-life longitudinal exposure studies measuring internal and external exposure for multiple pathways and PFASs (Winkens et al. 2017a). Nevertheless, increasing reports of toxicity in humans and experimental animals, sometimes at or below common exposure levels, is leading to growing public health concern (APHA 2016) and expanded regulatory interest (Corsini et al. 2014; Zushi et al. 2012).

Cumulative effects

Reference: California Code of Regulations, title 22, section 69503.3(a)(1)(C).

Cumulative effects occur from exposures to the Candidate Chemical and other chemicals with the same or similar hazard traits or environmental or toxicological endpoints.

PFASs are found in food packaging alongside other chemicals that migrate into food. Some of these other chemicals may have hazard traits similar to those of PFASs; however, mixture toxicity remains poorly understood (Muncke et al. 2020). One effort to estimate cumulative toxicity is presented in the work of RIVM (2018), who used a Relative Potency Factor (RPF) method to compare exposure to a PFAS mixture to exposure to a comparable amount of PFOA. This method is currently limited by the availability of toxicological data for the many different PFASs, as well as by several assumptions about the toxicity mechanism and mode of action. Its calculations rely heavily on subacute and subchronic oral liver toxicity studies conducted in rodents. So far, RPFs have been calculated for only 20 individual PFASs.

Individual PFASs rarely occur in isolation. Instead, people and biota are exposed to mixtures of PFASs and other chemicals, potentially leading to cumulative adverse impacts, yet toxicological endpoints are typically assessed using single PFASs. The adverse human and environmental health effects resulting from exposures to PFAS mixtures have not been well-characterized, and toxicological reference values for PFAS mixtures are lacking. Some have argued that exposure to PFAS mixtures may exert greater toxicity than exposure to a single PFAS (Wang et al. 2011). Although exposure to multiple PFASs may result in additive effects, only a few studies have investigated the toxicity of PFAS mixtures, with varying results. For instance, a cumulative health risk assessment of 17 PFASs in the Swedish population concluded there was a threat of hepatotoxicity or reproductive toxicity in a subpopulation that routinely consumed PFOS-contaminated fish, but not in the general public (Borg et al. 2013).

Evidence of increased mixture toxicity includes results from an *in vitro* study in which binary combinations of PFAAs at low concentrations behaved additively in activating PPAR-alpha (Wolf et al. 2014). A cytotoxicity assay found that a mixture of PFASs altered the cellular lipid pattern of human placental cells at levels below those that induce toxic effects (Gorrochategui et al. 2014). An *in vitro* study of PFOA, PFOS, and PFNA found that toxicity in a human macrophage cell line and acute toxicity in zebrafish were greater for mixtures than individual compounds (Rainieri et al. 2017). The cytotoxicity of individual PFASs and of binary mixtures using an amphibian fibroblast cell line showed mixtures were approximately additive, except with PFOS and PFOA, which were weakly synergistic (Hoover et al. 2019). Another study investigated the toxicity of individual and combined PFASs to human liver cell line (HepG2) and found the binary, ternary, and multi-component combinations of PFOS with PFOA, PFNA, PFDA, PFHxS, and PFHpA displayed synergistic interactions for almost all inhibitory effect levels tested (Ojo et al. 2020).

Mixture toxicity studies of PFASs and other toxicants are limited. The results include:

- Effects on gene expression patterns in zebrafish embryos coexposed to a mixture of PFHxA,

PFOS, and PCB126; lowered oxidative stress response from exposure to PCB126 and PFHxA, and to PCB126, PFOS, and PFHxA (but not from the individual compounds, nor from PCB126 and PFOS), suggesting PFHxA plays a synergistic role, although the exact mechanism remains unknown (Blanc et al. 2017);

- Endocrine disruption in developing rats coexposed to a mixture of PFHxS and 12 endocrine-disrupting chemicals (including bisphenol A, dibutyl phthalate, di-2-ethylhexyl phthalate, and butyl paraben), at doses relevant for human exposure, including doses at which PFHxS alone and the mixture of endocrine disruptors alone showed no or only weak effects (Ramhøj et al. 2018);
- Clinically relevant perturbations of normal cell function of benign breast epithelial cells coexposed to a mixture of PFOA, bisphenol A, and methylparaben at concentrations relevant to human exposure; these effects were not observed when cells from the same lines were exposed to the chemicals individually (Dairkee et al. 2018);
- Cumulative adverse effects *in vitro* of PFOS, PFOA, and several other contaminants (Hg^{2+}, Cd^{2+}, 2,4-D, propylparaben, mitomycin C, and furazolidone) in a bioluminescent cyanobacterial toxicity test (Rodea-Palomares et al. 2012);
- Increased severity of oxidative stress and apoptosis to zebrafish embryos following exposure to PFOS and zinc oxide nanoparticles (Du et al. 2017); and
- Adverse impacts on mitochondrial function in juvenile Chinook salmon after exposure to a mixture of contaminants of emerging concern, including several PFASs, pharmaceuticals, and personal care products, at environmentally relevant concentrations (Yeh et al. 2017).

While the above data suggest a potential for cumulative harm from exposure to mixtures of PFASs or of PFASs and other chemicals, it is unclear whether the outcomes from the cell lines, zebrafish, or juvenile salmon are associated with adverse phenotypes. In spite of the limitations of these approaches, as described by the National Academy of Sciences (NAS 2017), these studies suggest that exposure to certain combinations of PFASs, or co-exposure to certain PFASs and other chemicals, has the potential to contribute to disease by perturbing various pathways or activating some mechanism. In particular, PFAAs may have cumulative impacts with one another and with other hazardous chemicals, but co-exposures that dramatically affect adverse impacts are not known at this time and require more scientific research.

Adverse waste and end-of-life effects

Reference: California Code of Regulations, title 22, sections 69503.2(b)(1)(B) and 69501.1 (a)(8).

This section summarizes findings related to the waste materials and byproducts generated during the life cycle of the product and their associated adverse effects, as described in the SCP regulations. These considerations can form part of the basis for proposing the product-chemical combination.

PFASs are not routinely removed by WWTPs, and removal is expensive (Arvaniti et al. 2015b; Mudumbi et al. 2017). In fact, wastewater treatment typically transforms PFAA precursors into PFAAs, resulting in higher PFAA concentrations in WWTP effluent compared to the influent (Ahrens 2011; Frömel et al. 2016; Guerra et al. 2014). Conventional wastewater treatment systems such as sedimentation, flocculation, coagulation, and biological degradation are ineffective for PFAS removal (Rahman et al. 2014; Ng et al. 2019). Adsorption and ion-exchange have also been recently explored as methods that could increase the speed and efficiency of PFAS removal (Ji et al. 2018; Ateia et al. 2019b). While adsorption and separation technologies combined with destructive technologies show promise for removing a wider range of PFASs, their large-scale application poses several challenges (Horst et al. 2018; Li et al. 2020a), including reliance on chemicals and energy, high cost, and generation of harmful byproducts (Lu et al. 2020). Shorter-chain PFAAs are especially challenging to remove from aquatic environments and to destroy (Li et al. 2020a).

Biosolid treatments are also ineffective at reducing the levels of PFAAs. A study of commercial biosolids from the U.S. and Canada found that thermal hydrolysis had no apparent effect on the PFAA concentration, whereas heat treatment and composting increased PFAA concentrations (especially PFHxA) via the degradation of precursors; only blending with PFAS-free material decreased the concentration of PFAAs in the commercial biosolids, by diluting it (Kim Lazcano et al. 2019).

Removal of PFASs from the waste stream is expensive. For instance, following contamination from carpet manufacturing in Dalton, Georgia water utility Gadsden Water stated that it "has suffered substantial economic and consequential damage, including expenses associated with the future installation and operation of a filtration system capable of removing the chemicals from the water and lost profits and sales" (Chapman 2016). Wastewater effluent can contribute significantly to the PFAA contamination of U.S. drinking water sources (Guelfo and Adamson 2018).

PFASs in landfill leachates, land-applied wastewater, and biosolids can contaminate streams, rivers, and other drinking water sources (Hoffman et al. 2011; Lindstrom et al. 2011a). Stormwater runoff containing PFASs released from consumer products may be a significant pathway for PFAS transport and release to surface waters (Houtz and Sedlak 2012).

Compostability certifiers are beginning to establish maximum concentration thresholds for toxic chemicals in products they certify (BPI 2018). The composting process breaks down the PFASs present in composted food packaging materials to PFAAs, which are environmentally persistent and can be taken up by plants grown in compost-treated soil (Trier et al. 2011a; Zhou et al. 2019; Zhang et al. 2020; Gredelj et al. 2020). Compost samples collected from California and four other U.S. states found significantly higher PFAA levels in samples that included compostable food service ware than compost that did not, with the shorter-chain PFAAs (PFBA, PFPeA, and PFHxA) being most prevalent (Lee et al. 2018a). While they have shorter half-lives in the human body, shorter-chain PFAAs tend to be more

mobile and have preferential uptake in plants, and thus they frequently accumulate in the terrestrial food chain (Choi et al. 2019).

Currently, very few industrial composters in the U.S. accept food and food service packaging. Those that do generally only accept food service packaging from controlled sources (e.g., restaurants that are in contract with private compost companies). A group of composters serving the state of Oregon published a letter in early 2019 titled, "Why we don't want compostable packaging and service ware" (Oregon Composters 2019). One of the nine reasons cited for not accepting compostable food packaging products is that "they may threaten human and environmental health." The letter specifically cites concerns over "packaging designed for water and grease resistance" (i.e., packaging containing intentionally added PFASs). When applied to soil, PFASs in contaminated compost are released to surface waters or taken up by plants and crops, leading to negative health impacts.

Populations that may be adversely impacted

Reference: California Code of Regulations, title 22, sections 69503.3(a)(1)(F) and 69503.3(a)(2).

This section identifies specific populations of humans and environmental organisms that may be harmed if exposed to the Candidate Chemical in the product. Sensitive subpopulations, environmentally sensitive habitats, endangered and threatened species, and impaired environments in California have special consideration as they may be more vulnerable.

Exposures to PFASs occur frequently and from multiple sources over a lifetime, including from placental transfer, breastfeeding, drinking water, food, and contact with consumer products and other environmental media. PFASs have the potential to cause significant and widespread adverse impacts to:

- Sensitive subpopulations (which include fetuses, infants, children, pregnant women, some workers, and people with certain preexisting conditions);
- Environmentally sensitive habitats; and
- Threatened and endangered species.

Fetuses, infants, toddlers, and young children experience higher relative exposure levels and are more vulnerable to the effects of toxicants. Workers in food packaging manufacturing, food preparation, and food service industries can experience high PFAS exposures on the job. Individuals with certain preexisting conditions (e.g., elevated cholesterol, high blood pressure, poor kidney function) may be especially sensitive to PFASs. Pregnant women's exposure to PFASs is of concern due to transplacental and lactational transfer to the offspring. Endangered and threatened species in California are vulnerable to contamination of waterways and food webs with PFASs that originate from food packaging products.

The increasing concentrations of PFASs associated with stain and soil repellents found in environmental media, animal tissues, and biota can contribute to or cause significant or widespread adverse impacts in people and the environment.

Due to their widespread presence in the environment and biota, PFASs may contribute to or cause adverse impacts in all humans and other organisms. A recent review paper (Krafft et al. 2015b) states that:

"Because of ubiquitous PFAS presence, there is no unexposed control population and many studies compare outcome data for high-dose sub-groups with a low-dose sub-group within the same population, a practice that can weaken dose-response correlations."

The potential for adverse impacts of PFASs on aquatic species has been well-documented, with toxic effects noted in aquatic plants (Ding et al. 2013; Ding et al. 2012; Hoke et al. 2012) and widespread bioaccumulation in aquatic animals. Less is known about potential impacts to terrestrial animals and plants, especially at the population level. Shorter-chain PFASs generally show increased uptake by leafy plants compared to the longer chains (Blaine et al. 2014a), though longer chains may preferentially accumulate in roots (Zhao et al. 2017a). PFASs have been shown to bioaccumulate in earthworms, indicating the potential for biomagnification in terrestrial species (Karnjanapiboonwong et al. 2018; Munoz et al. 2020). Wildlife, particularly apex predators, may suffer adverse effects from chronic exposure to PFASs in food and water (Tartu et al. 2017; Tipton et al. 2017). Fish-eating birds, including eagles and ospreys, are especially vulnerable to PFAS exposure via food chain contamination (Giesy et al. 2010). Known impacts to birds from PFAS exposure include reduced embryo survival in chicken (Norden et al. 2016).

Infants, toddlers, and small children comprise a sensitive subpopulation because of their increased ingestion and inhalation rates per unit of body weight, rapid development, immature physiological ability to detoxify environmental contaminants, and behavioral characteristics that predispose them to increased exposures to environmental contaminants (U.S. EPA 2011). This results in a higher body burden of PFASs as compared to adults (Rappazzo et al. 2017). Pregnant women and fetuses are also sensitive subpopulations because of transplacental migration and the vulnerability of the rapidly developing fetus (Slotkin et al. 2008). Breastfed infants are susceptible to increased exposures to PFASs in breast milk, because breastfeeding is a route of PFAS excretion for lactating women (Kang et al. 2016; Kärrman et al. 2007; Mogensen et al. 2015; Mondal et al. 2014; Abdallah et al. 2020). Infants, toddlers, and small children often have increased exposures due to hand-mouth behaviors that can lead to increased incidental ingestion of dust and soil with environmental contaminants, and higher doses relative to body weight compared to adults (U.S. EPA 2011). A study involving over 1,000 children up to seven years old found a negative correlation between plasma concentrations of most

PFAAs and age, indicating higher exposures in younger children (Zhang et al. 2018). Table 8 lists the main routes of PFAS exposure for children.

With more than 3.9 million births in the U.S. in 2016, fetuses and infants constitute a large subpopulation highly vulnerable to exposures to the PFASs released during the treatment of converted textiles or leathers. Because PFASs undergo transplacental transfer (Midasch et al. 2007; Zhang et al. 2013; Yang et al. 2016; Chen et al. 2017a; Zhao et al. 2017a; Eryasa et al. 2019; Cai et al. 2020), fetuses are continuously exposed from conception until birth. PFASs are also frequently detected in breast milk (Nyberg et al. 2018; Jin et al. 2020), with nursing infants exposed daily over many months. The CDC reported that 79 percent of infants born in 2011 were breastfed, with 27 percent still breastfeeding at 12 months of age. The amount of breast milk ingested by newborns averages 76 g, six to eighteen times per day (CDC 2014).

Toddlers can experience increased exposures to PFASs due to behavioral characteristics. Children between the ages of 1 and 3 spend a lot of time in contact with the floor, where dust settles, and engage in indoor hand-mouth behaviors an average of 16 times per day (U.S. EPA 2011). Thus, incidental dust ingestion can be a significant route of exposure for young children (Mercier et al. 2011; Tian et al. 2016; Wu et al. 2020). While adults are estimated to ingest an average of 50 mg/day of soil and dust, children under 6 may ingest 100 mg/day (U.S. EPA 2011). PFAS doses via incidental dust ingestion could be approximately an order of magnitude higher for toddlers than for adults (Karásková et al. 2016). A study of toddlers' cumulative exposure to PFOA and PFOS estimated a daily intake of 53.6 ng/day of PFOA and 14.8 ng/day of PFOS (Tian et al. 2016). A study of U.S. child care environments estimated a median daily intake of 28 PFASs in toddlers via dust ingestion of up to 0.85 ng/kg body weight per day (Zheng et al. 2020). Similarly, a study at California child care centers estimated that the children's daily intake of PFAAs via dust ingestion ranges from 0.023 – 1.9 ng/kg body weight (Wu et al. 2020).

Table 8: Prenatal and early childhood PFAS exposure routes[14]

Exposure Pathway	Basis
Transplacental migration	Exposure to PFASs begins before birth. Transplacental passage is a significant route of human exposure to PFAAs (Kim et al. 2011) and their precursors (Yang et al. 2016). Transfer to the fetus is one of the major PFAS elimination routes for women, particularly for PFOA (Lee et al. 2013).

[14] Protecting children, a sensitive subpopulation, is one of the goals identified in SCP's 2018-2020 Priority Product Work Plan. Childhood is a life stage that all members of a population experience, although children constitute a population subgroup when evaluating exposures at a specific time.

Exposure Pathway	Basis
Ingestion – breast milk	Breast milk ingestion is a significant route of PFAS exposure for infants (Mondal et al. 2014). Infants fed breast milk may have higher PFAS dietary exposure than those fed infant formula (Fromme et al. 2010).
Ingestion – food	Food ingestion is considered a primary route of exposure for the general population, with higher relative exposures in infants, toddlers, and small children because of their low body weight (Egeghy and Lorber 2011). Due to widespread contamination, PFASs are found in a variety of animal and plant foods (Blaine et al. 2014a; Pérez et al. 2014), and can be transferred to food from PFAS-treated packaging (Nordic Council of Ministers 2017). PFASs are readily absorbed after ingestion (ATSDR 2018b; Danish Environmental Protection Agency 2015).
Ingestion – drinking water	Numerous studies conclude that drinking water is a major source of PFAS intake (DeWitt 2015; Trudel et al. 2008). In 2016, U.S. EPA issued drinking water health advisories for PFOA and PFOS of 70 ng/L (combined concentrations), to protect the most sensitive populations – fetuses during pregnancy and breastfed infants (U.S. EPA 2020b). Some states have adopted, or are considering adopting, lower limits.
Inhalation and ingestion of dust	PFASs are widely found in house dust, with higher concentrations in homes with treated carpets (Haug et al. 2011). Exposure via dust is higher in toddlers and small children than adults and can be significant due to children's lower body weights, increased inhalation rate, higher incidental dust ingestion rates, increased floor contact and hand-to-mouth behavior (Mercier et al. 2011; Tian et al. 2016). PFASs are readily absorbed following inhalation and ingestion (ATSDR 2018b).

Studies demonstrate substantial human exposure in workers at fluorochemical manufacturing facilities (Gao et al. 2018) and communities relying on contaminated drinking water sources (Heydebreck et al. 2016). People living near PFAS chemical manufacturers, or in other areas contaminated by PFASs, have higher-than-average levels of PFASs (Fromme et al. 2009). Individuals with biomarkers of susceptibility for certain health conditions may be unusually sensitive to PFAS exposure (ATSDR 2018b).

Residents of low-income communities may be disproportionately impacted by chronic exposures to multiple hazardous industrial chemicals, increasing the likelihood of cumulative adverse health effects (U.S. EPA 2016a). Manufacturing facilities, including those that release PFASs, are often located in

these communities. Environmental justice concerns also arise regarding contamination in areas far from manufacturing sites, such as the remote regions of the Arctic. For instance, the traditional diet of the Inuit in Nunavut, Canada, the Nuuk Inuit in Greenland, and the Faroese includes foods heavily contaminated with PFASs that originated from thousands of miles away (Grandjean et al. 2012; Grandjean et al. 2016; Long et al. 2012; Ostertag et al. 2009).

California's endangered and threatened species could be adversely affected by exposure to PFASs associated with food packaging, especially considering the adverse effects on reproduction and development demonstrated for some PFAAs. This could contribute to the current biodiversity crisis in aquatic ecosystems (Abell 2002; Mora and Sale 2011; Valentini et al. 2016). Threatened and endangered species of fish and marine mammals are particularly vulnerable to population-level adverse impacts if they cannot obtain food or avoid predators. Mammals and birds occupying the highest trophic level, including orcas, wolves, grizzly bears, eagles and condors, are vulnerable to adverse health effects from ingestion of food and water contaminated with PFASs that bioaccumulate and biomagnify (Kannan et al. 2006; Kelly et al. 2009).

Environmentally sensitive habitats in California, including estuaries and other wetlands, can receive surface water contaminated with PFASs via wastewater plant effluent or surface runoff, leading to contaminated water and sediments. These ecosystems are important breeding, spawning, and nesting sites, and feeding grounds for millions of migratory birds – including threatened and endangered species – that transit California during their annual migration.

6. OTHER REGULATORY PROGRAMS

Reference: California Code of Regulations, title 22, section 69503.2(b)(2).

DTSC has assessed all applicable state and federal laws and regulations and international treaties or agreements with the force of domestic law related to the product or the Candidate Chemical in the product. DTSC has determined that these programs do not overlap or conflict with this proposal to list food packaging containing any member of the class of PFASs, nor with any subsequent regulation that may result from such listing.

FDA

FDA regulates chemicals in food packaging that come into contact with food as food additives. Food additives are regulated under several mechanisms: Chemicals may be listed in the Code of Federal Regulations (CFR) Title 21, Chapter 1; they may meet FDA's criteria for substances Generally Recognized as Safe (GRAS); or they may be approved through the FCN process (Schaider et al. 2017).

To our knowledge, no PFASs are currently approved as GRAS, and only one PFAS was approved via an indirect food additive petition under CFR title 21, section 176.160.

The FCN process is the primary method by which FDA regulates food additives that are food-contact substances. Under this program, the manufacturer of a food-contact substance must submit a safety summary and comprehensive toxicological profile to FDA for review, including chemical, toxicological, and exposure information and the identity of degradation products expected to migrate from packaging into food. It is the responsibility of the manufacturer to complete the notification and to fulfill acceptance criteria for an FCN (FDA 2020b). There are currently 28 PFAS-related FCNs for 17 distinct PFAS compositions used in plant-based food-contact surfaces, submitted by six chemical manufacturers (FDA 2020a).

When FDA evaluates a notification, it only considers cumulative exposures and aggregate effects from other dietary exposure routes that it regulates. It does not consider impacts from other sources such as inhalation or dermal exposure from other consumer products found in the indoor environment, or from drinking water sources. Also, FDA does not consider all exposures resulting across the product's life cycle, including from composting, landfilling, incineration, and recycling. These represent key differences between SCP's regulatory approach and FDA's.

One manufacturer, Chemours, recently announced its decision to stop manufacturing PFASs for food packaging uses and has asked FDA to withdraw its approved FCNs, listed in Chapter 3 of this document (Neltner 2019). This will reduce the number of approved FCNs to 25, and the number of distinct PFAS compositions approved for use on plant fiber-based food packaging to 15.

In January 2016, in response to an NGO petition, FDA rescinded approval for three long-chain PFASs compounds for food packaging (Geueke 2016):

1) Diethanolamine salts of mono- and bis (1*H*,1*H*,2*H*,2*H* perfluoroalkyl) phosphates where the alkyl group is even-numbered in the range C_8-C_{18} and the salts have a fluorine content of 52.4 to 54.4 percent as determined on a solids basis;
2) Pentanoic acid, 4,4-bis [(γ-ω-perfluoro-C_{8-20}-alkyl)thio] derivatives, compounds with diethanolamine (CAS 71608-61-2); and
3) Perfluoroalkyl-substituted phosphate ester acids, ammonium salts formed by the reaction of 2,2-bis[([γ], [ω]-perfluoro C_{4-20} alkylthio) methyl]-1,3-propanediol, polyphosphoric acid and ammonium hydroxide.

According to The Nordic Council of Ministers (2017), although these three phased-out PFASs can no longer be applied to food packaging in the U.S., they may still be imported in finished food-contact products.

In November 2016, in response to a petition by 3M, FDA amended the food additive regulations to rescind the use of an additional two long-chain PFASs in response to a petition by 3M, because the chemicals were no longer used for that purpose (FDA 2016).

Due to concern about longer-chain C8 compounds, FDA worked with manufacturers to voluntarily stop distributing C8-containing products as of October 1, 2011. Products existing in the marketplace could still be used until they run out. Participating manufacturers include: BASF Corporation, E. I. DuPont de Nemours & Co., and Clariant Corporation (FDA 2018).

While FDA regulates the use of chemicals in food packaging, it does not consider all cumulative effects, aggregate effects, and end-of-life impacts, and therefore does not address all potential exposures and adverse impacts associated with the use of PFASs in food packaging.

U.S. EPA

Section 5 of the Toxic Substances Control Act (TSCA) authorizes U.S. EPA to issue Significant New Use Rules (SNURs) for new chemicals or existing chemicals used in a significant new way. A SNUR requires companies to notify U.S. EPA at least 90 days prior to manufacturing, importing, or processing substances for a significant new use, and to submit a notification including information about the chemical's identity, physical characteristics, processing and use, and available toxicity data. U.S. EPA has 90 days to evaluate the new use and can request more data, prohibit or limit the manufacture, or allow the use. The following SNURs relate to PFASs:

- December 9, 2002 – SNUR regarding any future manufacture (including imports) of 75 PFASs specifically included in the 2000-2002 voluntary phaseout of PFOS by 3M (U.S. EPA 2002a);
- March 11, 2002 – SNUR regarding any future manufacture (including imports) of 13 PFASs specifically included in the 2000-2002 voluntary phaseout of PFOS by 3M (U.S. EPA 2002b);
- October 9, 2007 – SNUR for 183 PFASs believed to no longer be manufactured, imported, or used in the U.S. (U.S. EPA 2007);
- October 22, 2013 – SNUR requiring companies to report their intent to manufacture certain PFOA-related chemicals to treat carpets, as well as their intent to import carpets containing these PFASs (U.S. EPA 2013);
- (Proposed) January 21, 2015 – proposed SNUR affecting manufacturers (including importers) of PFOA and PFOA-related chemicals, including as part of articles, and processors of these chemicals (U.S. EPA 2015); and
- (Proposed) February 20, 2020 – supplemental to a SNUR issued on certain PFASs in 2015. Amendment includes regulation on imported products that contain certain PFASs used as part of surface coatings (U.S. EPA 2020c);
- June 22, 2020 – final SNUR requiring notice before anyone can resume the use of long-chain PFASs that have been phased out in the U.S., and banning the import of products containing

certain long-chain PFASs as surface coatings and of carpet containing perfluoroalkyl sulfonates (U.S. EPA 2020d).

Since 2000, U.S. EPA has reviewed hundreds of substitutes for PFOA, PFOS, and other longer-chain PFASs, particularly regarding their toxicity, fate, and bioaccumulation under the New Chemicals Program (U.S. EPA 2017b). For many PFASs, U.S. EPA has used TSCA section 5(e) Consent Orders to require testing while allowing production and use. U.S. EPA is also investigating substitutes for certain direct uses of PFOA.

On January 27, 2010, U.S. EPA amended the Polymer Exemption Rule for new chemicals under TSCA to exclude certain side-chain fluorinated polymers due to potential risk to human health or the environment (U.S. EPA 2002b).

The National Defense Authorization Act for Fiscal Year 2020 (NDAA), signed into law on December 20, 2019, contains multiple PFAS-related amendments, including a ban on the use of PFASs in food packaging for military ready-to-eat meals and the phaseout of PFAS-containing firefighting foam (Inhofe 2019). Additionally, section 7321 of the NDAA added several PFASs to the Toxics Release Inventory (TRI) under section 313 of the Emergency Planning and Community Right-to-Know Act of 1986 (EPCRA), effective January 1, 2020. The TRI informs the public about the environmental and safety concerns that may occur from handling and storage of toxic chemicals. The NDAA specifically added 14 PFASs to the TRI, plus any additional PFASs that are subject to a SNUR under either 40 CFR 721.9582 or 721.10536 on or before December 20, 2019, and are identified as active in commerce on the TSCA inventory list published in February 2019. The TRI list currently contains 172 PFASs. Reporting forms are due by July 1, 2021 (U.S. EPA 2020f). Other PFASs will be automatically added to the TRI if any of the following occurs (Inhofe 2019):

1. EPA finalizes a toxicity value for a perfluoroalkyl or polyfluoroalkyl substance or class of PFASs;
2. EPA finalizes a SNUR for a perfluoroalkyl or polyfluoroalkyl substance or class of PFASs;
3. EPA adds a perfluoroalkyl or polyfluoroalkyl substance or class of PFASs to an existing SNUR; or
4. EPA designates a perfluoroalkyl or polyfluoroalkyl substance or class of PFASs as an active chemical substance on the TSCA Inventory.

7. POTENTIAL ALTERNATIVES

Reference: California Code of Regulations, title 22, section 69503.2(b)(3).

This section summarizes information available to DTSC regarding alternatives that may or may not be safer than the Candidate Chemical. DTSC does not need to ensure that these alternatives are safer and may summarize their associated hazards to illustrate readily available information. The sections below may include information such as how readily available an alternative is, product functions addressed by the alternative, and implications for manufacturers using the alternative (e.g., use limitations, product reformulation, different equipment needs).

Based on well-established business cases, The Nordic Council of Ministers (2017) concluded that safer and more sustainable alternatives to PFASs in paper and paperboard food packaging products are available for all intended functional uses and food types. They also found that, except for natural greaseproof paper, which can be more expensive, alternatives are cost-neutral for retailers.

Back in 2018, all molded fiber food packaging products on the market contained PFASs (Clean Production Action 2018). Pursuant to its new compostability standard, effective January 2020, BPI delisted thousands of products due to the presence of intentionally added fluorine or fluorine levels above 100 ppm. Since then, some manufacturers have developed PFAS-free molded fiber food packaging products, demonstrating that it is possible to find functionally acceptable alternatives (DTSC 2020).

Several major U.S. food retailers have shifted, or have committed to shifting, to PFAS-free food packaging. For instance, in December 2018, Whole Foods Market removed PFAS-containing food and bakery packaging from its stores (Schade and Belliveau 2020). In September 2019, Ahold Delhaize, the parent company for grocery chains such as Hannaford, announced that it will restrict intentional use of all PFASs in its private brand products and food packaging beginning in 2020 (Nolan 2019). In January 2020, Taco Bell, the fourth largest fast-food chain in the U.S., committed to globally phasing out all PFASs in "consumer-facing packaging materials" by 2025 (Chemical Watch 2020). Albertsons, the second-largest grocery chain in the U.S., and Panera Bread have initiated efforts to eliminate PFASs from their food packaging (Schade et al. 2020).

Alternatives to PFASs for plant fiber-based food packaging include physical barriers, alternative processing, alternative chemical barriers or coatings, and alternative materials.

Physical barriers can be made of plastic such as polyethylene, polyethylene terephthalate (PET), polyvinyl alcohol, or polylactic acid (PLA), as well as of silicone, aluminum, clay, wax, or biowax such as Clondalkin ECOWAX (Nordic Council of Ministers 2017; Clean Production Action 2018; ECY 2020).

Machine-finished paper uses alternative processing to avoid the use of PFASs. Examples include natural greaseproof paper and vegetable parchment (Nordic Council of Ministers 2017; Clean Production Action 2018; Packaging Europe 2020). Fibers in natural greaseproof paper are refined, which causes them to come into closer contact and bond tightly to each other during manufacture, reducing air permeability and increasing grease resistance. Vegetable parchment is passed through a bath of

concentrated sulfuric acid, which causes the cellulose fibers to react with the acid and almost melt together (Nordic Council of Ministers 2017). Other alternative processing includes mechanical densification and mechanical glazing, which further refine the fibers to create a dense structure that increases grease resistance (Clean Production Action 2018).

Alternative chemical barriers or coatings include starch, carboxymethyl cellulose, aqueous dispersions of copolymers such as styrene and butadiene, aqueous dispersions of waxes, water-soluble hydroxyethylcellulose, chitosan, alkyl ketene dimer, alkenyl succinic anhydride, silicone, and several proprietary coatings of unknown composition (Nordic Council of Ministers 2017). For instance, WestRock EnShield Paperboard claims its chemical coating provides oil and grease resistance and contains no low-density polyethylene and no fluorocarbons (Nordic Council of Ministers 2017). According to a comment submitted to the state of Washington by Scott Seydel, the Pendergrass, Georgia-based Seydel-Woolley & Co. facility produces non-PFAS "coatings for food paper, carton, cup, and linerboard (corrugated) packaging that provide equivalent barriers to oil and water" (Seydel 2018). These coatings include proprietary blends of saturated fats and oils intended to replace petroleum-based waxes, water-dispersible polyester resins, and aqueous polymer blends (The Seydel Companies, Inc. 2018). Solenis produces several PFAS-free chemical barriers marketed under the trade name TopScreen, which are either water-based polymer emulsions or solid biowaxes that are claimed to "repel water and water vapor, hot and cold liquids, and oils and greases" in a wide variety of food packaging products (Solenis 2020). Chitosan grafted with castor oil can also render the coated paper both oil- and water-resistant (Li et al. 2020d).

Alternatives to paper, paperboard, and molded fiber materials include palm leaf, bamboo, and various plastics (Clean Production Action 2018). Examples of bio-based plastics used for food packaging include PLA, polyethylene terephthalate (Bio-PET) and thermoplastic starch, which can be generated from plants, fungi, or organic wastes (Vendries et al. 2020).

Assessing the relative safety of these alternatives is beyond the scope of this document. If one or more plant fiber-based food packaging products containing PFASs are listed as Priority Products in the California Code of Regulations, those entities who wish to continue selling the product in California will have to submit an Alternatives Analysis (AA) to DTSC. The AA process is the mechanism through which DTSC evaluates the need for safer alternatives and ensures that any alternatives chosen are indeed safer for people and the environment.

8. ADDITIONAL CONSIDERATIONS

This section summarizes other relevant information not captured under the adverse impact and exposure factors named in section 69503.3 of the Safer Consumer Products regulations.

Although PFAS manufacturers have been aware of these chemicals' adverse impact potential for decades, national and international action to limit human and ecological exposures has only recently started. In 2015, more than 200 scientists from 40 countries signed The Madrid Statement on Poly- and Perfluoroalkyl Substances (PFASs) (Blum et al. 2015), a scientific consensus on the persistence and potential for harm of this class of chemicals, and a roadmap for international cooperation to prevent further harm.[15]

The essential use concept

In 2019, The Global PFAS Science Panel released a report on the concept of essential and nonessential uses of PFASs and concluded that "non-fluorinated alternatives have been historically available for all applications of paper and board food packaging and the use of fluorinated protective coatings has never been essential" (Cousins et al. 2019b).

Restrictions on PFASs in compostable food packaging

As of March 2019, BPI, the largest U.S. certifier of compostable products, requires manufacturers who seek compostability certification to meet standard EN 13432, which sets a 100-ppm limit for total fluorine. As of January 2020, BPI also requires that manufacturers provide a statement of no intentionally added fluorine.

U.S. EPA research and action plan

In 2006, U.S. EPA developed a 2010/2015 Stewardship Program for reducing emissions of PFOA, its precursors, and related higher homologues (U.S. EPA 2020e). Through a Memorandum of their emissions and products by 2015. Participating companies include: Arkema, Asahi, BASF, Clariant, Daikin, 3M/Dyneon, DuPont, and Solvay Solexis.

On December 4, 2017, U.S. EPA announced a cross-agency effort to address PFASs, including filling data gaps, supporting communities impacted by drinking water contamination, and proactively communicating with states, tribes, partners, and the public about the health effects of PFAS exposure (U.S. EPA 2017c). On March 13, 2018, U.S. EPA also announced a partnership with the National Toxicology Program to develop a tiered testing approach to characterize the toxicity and toxicokinetics of 75 PFASs that represent the entire class (U.S. EPA 2018). In February 2019, U.S. EPA revealed its PFAS Action Plan. In it, EPA commits, among other things, to: (1) a regulatory determination by the end of 2019 whether to set a maximum contaminant level (MCL) for PFOA and PFOS; (2) beginning regulatory work to list PFOA and PFOS as hazardous substances under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA); (3) publishing draft toxicity

[15] One of the authors of this document, Simona Bălan, Ph.D. co-authored the Madrid Statement prior to joining DTSC.

assessments for PFBA, PFHxA, PFHxS, PFNA, and PFDA in 2020; and (4) continuing to improve analytical methods, monitoring, and risk communication (U.S. EPA 2019). In September 2019, U.S. EPA announced it is awarding approximately $6 million to eight grantees to research PFASs in waste streams and identify practical approaches to manage their potential impacts (U.S. EPA 2019).

Laws and regulations under other jurisdictions

In July 2018, San Francisco issued a Food Service and Packaging Waste Reduction Ordinance banning single-use food service ware containing fluorinated chemicals, effective January 1, 2020 (San Francisco 2018; SGS 2018).

In 2019, LD 1433 prohibited the sale of food packaging with intentionally added PFASs in Maine (Safer States 2020). On July 28, 2019, the state of Washington enacted Substitute Senate Bill 5135, which authorizes the state's departments of Ecology and Health to regulate classes of chemicals in consumer products (Washington State Legislature 2019). The law identifies PFASs as one of the five initial priority chemical classes. In March 2018, two PFAS-related bills were signed into law in Washington state. Engrossed Substitute Senate Bill 6413 prohibits the use of PFAS-containing firefighting foam for training purposes as of July 1, 2018, and, as of July 1, 2020, prohibits the sale of firefighting foams with intentionally added PFASs, with exemptions for the military, airports, oil refineries, and chemical plants (State of Washington 2018a). Engrossed Substitute House Bill 2658 prohibits the sale of food packaging containing PFASs as of January 1, 2022, provided the state Department of Ecology identifies safer alternatives by January 1, 2020 (State of Washington 2018b).

PFBS, PFHxS, PFNA, plus a GenX precursor and their salts have been added to the REACH Candidate List of substances of very high concern (SVHCs) for authorization (ECHA 2020). In September 2019, Denmark announced a ban on PFAS-containing cardboard and paper used as food-contact materials, which became effective in July 2020 (Danish Ministry of Environment and Food 2019; Keller and Heckman LLP 2020). On January 30, 2018, South Australia became the first state to ban PFAS-containing firefighting foams (Government of South Australia 2018).

The Stockholm Convention on POPs, which the U.S. has not ratified, is a global treaty to protect human health and the environment from chemicals that remain intact in the environment for long periods, become widely distributed geographically, accumulate in the fatty tissue of humans and wildlife, and have harmful impacts on human health or the environment. In 2009, PFOS, its salts, and perfluorooctane sulfonyl fluoride were added to Annex B of the Stockholm Convention, which restricts the production and use, as well as import and export, of listed POPs (UNEP 2020). Production and use of these chemicals are still allowed for acceptable purposes after Annex B listing, which included use in carpets until 2015. Following the recommendation of the POPs Review Committee (UNEP/POPS/POPRC 2017), the Conference of the Parties decided in the spring of 2019 to list PFOA, its salts, and PFOA-related compounds in Annex A to the Stockholm Convention, with some time-limited exemptions (COP

2019). The POPs Review Committee also agreed in October 2017 with a proposal by Norway to list PFHxS, its salts, and PFHxS-related compounds in Annexes A, B, and/or C to the Convention, and has established a working group to review the proposal further and prepare a draft risk profile (POPRC 2020).

Relevant proposed laws and regulations

Under Senate Bill (SB) 1335, California's Department of Resources Recycling and Recovery (CalRecycle) is proposing a 100-ppm limit for total fluorine in food service packaging sold by food service providers located in state-owned facilities. This would include any facility operating or acting as a concessionaire on state property, as well as facilities contracted to provide food service to state agencies (CalRecycle 2020). As of March 2020, 13 U.S. states are considering policies to eliminate or reduce PFASs in food packaging: Arizona, Connecticut, Illinois, Iowa, Massachusetts, Minnesota, New Hampshire, New Jersey, New York, Rhode Island, Vermont, Virginia, and Wisconsin (Safer States 2020).

Introduced in May 2019, the "Keep Food Containers Safe from PFAS Act of 2019" would amend the Federal Food, Drug, and Cosmetic Act to "deem any [PFAS] used as a food-contact substance to be unsafe and therefore treated as adulterated" (Dingell 2019). If passed, the bill would give FDA until 2022 to enforce the ban on any food-contact items, including packaging, cookware, and food processing equipment. In November 2019, The House Committee on Energy and Commerce proposed a legislative package, the PFAS Action Act, which would require EPA to designate all PFASs as hazardous substances under CERCLA and set a nationwide drinking water standard for PFASs (Stecker and Schultz 2019; NYU Law 2020). In January 2020, several other bills were incorporated into the PFAS Action Act, including: the Protect Drinking Water from PFAS Act, PFAS Right-to-Know Act, PFAS Safe Disposal Act, Protecting Communities from New PFAS Act, Toxic PFAS Control Act, PFAS Testing Act, PFAS Accountability Act, Clean Water Standards for PFAS Act, and the PFAS Transparency Act (NYU Law 2020). Also in January 2020, Senator Bernie Sanders introduced the Prevent Future American Sickness Act of 2020, which addresses several strategies for limiting the presence of PFASs including grants for treating drinking water contamination, a prohibition on food-contact substances, listing PFASs as hazardous air pollutants under the Clean Air Act, and a prohibition of waste incineration (Sanders 2020).

In February 2020, California state Senator Ben Allen introduced SB 1044, which would ban the sale and use of PFASs in firefighting foams and require labeling for PFAS-containing firefighter personal protective equipment as of January 1, 2022 (Allen 2020).

European countries are also taking a closer look at PFASs under REACH and national regulatory frameworks. The German Federal Environment Agency has made the case for designating shorter-chain PFASs as SVHCs and regulating them under REACH as chemicals of equivalent concern to PBT or vPvB substances (Brendel et al. 2018). In April 2017, the trade association Swedish Water, representing

Sweden's municipal water supply companies, called for a ban on PFASs in all consumer products because of the threat to drinking water supplies (Chemical Watch 2017a). In June 2017, 37 Swedish government agencies and research institutions signed a memorandum of understanding to collaboratively study PFASs and reduce their associated risks (Chemical Watch 2017b). Other European Union member countries, including Norway, Denmark, and Germany, have ongoing programs to study and regulate the use of PFASs. On February 1, 2018, the European Commission adopted a proposal for a revised drinking water directive that adds the entire class of PFASs to its list of regulated compounds (European Commission 2018). On May 11, 2020, the national authorities of Germany, the Netherlands, Norway, Sweden, and Denmark initiated a call for evidence to inform a joint REACH restriction proposal to limit the human health and environmental risks associated with the manufacture and use of PFASs. The proposal is expected to be submitted to ECHA in 2022, with restrictions possibly becoming effective as of 2025 (ECHA, 2020c).

Exposure limits

Academic scientists, environmental consulting firms employed by chemical manufacturers, and government agencies around the world have conducted risk assessments using a variety of methods with the goal of determining whether the levels of PFASs found in different environmental media pose a health risk to humans or animals. Contradictory conclusions have been drawn depending on several factors: the type of model used to calculate risk, the assumptions underlying the assignment of exposure factors in exposure dose calculations, and the uncertainties in the data used in the dose-response assessment. This has led to different regulatory levels for allowable concentrations in drinking water.

For instance, to address concerns about potential health effects from cumulative exposures to PFOA and PFOS, U.S. EPA established individual chronic reference doses for PFOA and PFOS of 0.00002 mg/kg/day (U.S. EPA 2016b; U.S. EPA 2016c). These are estimates of daily human exposure expected to be without adverse health effects over a lifetime. In the fall of 2016, after concerns that more than 16 million Americans are exposed to PFOA and PFOS via contaminated drinking water, U.S.EPA reviewed new studies of developmental effects and reduced its 2009 provisional health advisory of 400 parts per trillion (ppt)[16] to a lifetime drinking water health advisory of 70 ppt for PFOA and PFOS combined (U.S. EPA 2020b). In 2018, the Agency for Toxic Substances and Disease Registry (ATSDR) developed MRLs (ATSDR 2018c) that correspond to a drinking water advisory levels of 11 ng/L for PFOA and 7 ng/L for PFOS (Sunderland et al. 2019).

These health advisories are recommendations only and are not legally binding or enforceable. On April 13, 2018, 25 U.S. senators wrote a letter to U.S. EPA Administrator Pruitt asking him to "expeditiously declare an MCL for all PFAS, based on rigorous scientific evidence" (U.S. Senate 2018). Several U.S.

[16] 1 ppt is equivalent to 1 ng/L water.

states have set their own regulatory standards or advisory guidelines for one or more PFASs in one or more environmental media such as drinking water, groundwater, surface water, soil, air, fish, or wildlife (Longsworth 2020).

In California in 2018 OEHHA released a recommendation for interim notification levels of 14 ppt for PFOA and 13 ppt for PFOS in drinking water based on evidence for developmental toxicity, immunotoxicity, cancer, and liver toxicity (OEHHA 2018). In August 2019, OEHHA updated its research and recommended that notification levels be set at the lowest levels at which PFOA and PFOS can be reliably detected in drinking water using currently available and appropriate technologies (OEHHA 2019). As a result, the California State Water Resources Control Board set its drinking water notification levels to 5.1 ppt for PFOA and 6.5 ppt for PFOS (California State Water Resources Control Board 2019). In February 2020, the Board also lowered the response levels for water systems from 70 ppt for PFOA and PFOS combined to 10 ppt for PFOA and 40 ppt for PFOS (California State Water Resources Control Board 2020).

Some scientists have argued that the limit should be even lower to protect sensitive subpopulations. For example, (Grandjean and Clapp 2015) derived a limit of 1 ng/L based on evidence of immunotoxicity in children, combined with uncertainties from incomplete toxicity testing and lack of epidemiological studies.

Addressing PFAS contamination at military sites

On August 1, 2017, a bipartisan group of eight Democrats and six Republicans in the U.S. House of Representatives signed a letter asking the White House to address PFAS contamination caused by Department of Defense firefighting activities (Kildee et al. 2017). Also in August 2017, the New York State Department of Health and state officials from Alaska, Michigan, New Hampshire, Pennsylvania, and Vermont asked ATSDR to launch a longitudinal study of PFAS-impacted communities (Zucker et al. 2017). On December 12, 2017, the Fiscal Year 2018 National Defense Authorization Act was signed into law including a $7 million budget for the CDC and ATSDR to run, in consultation with the Department of Defense, a national study on the health impacts of human exposure to PFASs, especially from contaminated drinking water (U.S. Senate 2017). The final federal budget bill passed by lawmakers in March 2018 included $10 million for the PFAS healthy study, and $43.8 million for Department of Defense cleanup activities (U.S. House of Representatives 2018). In January 2019, a bipartisan Congressional PFAS Task Force was created to address the PFAS contamination on military bases (Office of Congressman Dan Kildee 2019).

9. CONCLUSIONS

DTSC has determined that plant fiber-based food packaging products containing any member of the class of PFASs meet the key prioritization criteria (California Code of Regulations, title 22, section 69503.2(a)) for listing a Priority Product:

(1) There must be potential public and/or aquatic, avian, or terrestrial animal or plant organism exposure to the Candidate Chemical(s) in the product; and

(2) There must be the potential for one or more exposures to contribute to or cause significant or widespread adverse impacts.

The polymeric PFASs approved by FDA for use in food-contact paper, paperboard, and molded fiber products are environmentally persistent, but can eventually degrade into nonpolymeric PFASs, with PFAAs as the final degradation products. Also, according to documents submitted by manufacturers to FDA, these polymeric formulations contain nonpolymeric PFASs as impurities. Nonpolymeric PFASs display one or more hazard traits as defined by the California Code of Regulations.

These nonpolymeric PFASs migrate from the food packaging into food, resulting in exposures. Migration efficiency increases with decreasing PFAS chain length. Also, at the end-of-life of the treated food packaging products, persistent PFASs or other hazardous compounds are released into compost, environmental media, human food, or other consumer products.

While FDA regulates the use of chemicals in food packaging, it does not consider all cumulative effects, aggregate effects, and end-of-life impacts, and therefore does not address all potential exposures and adverse impacts associated with the use of PFASs in food packaging.

Further studies may help inform DTSC's future decision-making. Despite these data gaps, the Department has sufficient information regarding potential exposures and adverse impacts from food packaging containing any member of the class of PFASs to designate this as a Priority Product.

ACRONYMS AND ABBREVIATIONS

PFAS (sub)classes

diPAP	Fluorotelomer phosphate diester
FASA	Perfluoroalkane sulfonamide
FASE	Perfluoroalkane sulfonamide ethanol
FAS(M)AC	Perfluoroalkane sulfonamidoethyl (meth)acrylate
FOSA	Perfluorooctane sulfonamide
FOSE	Perfluorooctane sulfonamide ethanol
FTAC	Fluorotelomer acrylate
FTAL	Saturated fluorotelomer aldehyde
FTCA	Saturated fluorotelomer carboxylic acid
FTEO	Fluorotelomer ethoxylate
FTEOC	Fluorotelomer ethoxycarboxylate
FTMAC	Fluorotelomer methacrylate
FTO	Fluorotelomer olefin
FTOH	Fluorotelomer alcohol
FTS	Fluorotelomer sulfonic acid
FTUAL	Unsaturated fluorotelomer aldehyde
FTUCA	Unsaturated fluorotelomer carboxylic acid
PAP	Polyfluoroalkyl phosphate
PFAA	Perfluoroalkyl acid
PFAI	Perfluoroalkyl iodide
PFAS	Perfluoroalkyl and polyfluoroalkyl substance
PFECA	Perfluoroether carboxylic acid
PFESA	Perfluoroether sulfonic acid
PFCA	Perfluoroalkyl carboxylic acid
PFPA	Perfluoroalkyl phosphonic acid
PFPE	Perfluoropolyether
PFPiA	Perfluoroalkyl phosphinic acid
PFSA	Perfluoroalkyl sulfonic acid
PFSI	Perfluoroalkyl sulfinate
PFSiA	Perfluoroalkyl sulfinic acid

Individual PFASs

APFO	Ammonium perfluorooctanoate
FBSA	Perfluorobutane sulfonamide
N-EtFOSAA	N-ethyl perfluorooctane sulfonamidoacetate
PFBA	Perfluorobutanoic acid
PFBPA	Perfluorobutyl phosphonic acid
PFBPiA	Perfluorobutyl phosphinic acid
PFBS	Perfluorobutane sulfonic acid
PFDA	Perfluorodecanoic acid
PFDS	Perfluorodecane sulfonic acid
PFDoA	Perfluorododecanoic acid
PFDoDPiA	Perfluorododecyl phosphinic acid
PFDPA	Perfluorodecyl phosphonic acid
PFDPiA	Perfluorodecyl phosphinic acid
PFHpA	Perfluoroheptanoic acid
PFHxA	Perfluorohexanoic acid
PFHxDA	Perfluorohexadecanoic acid
PFHxPA	Perfluorohexyl phosphonic acid
PFHxPiA	Perfluorohexyl phosphinic acid
PFHxS	Perfluorohexane sulfonic acid
PFNA	Perfluorononanoic acid
PFOA	Perfluorooctanoate or perfluorooctanoic acid
PFODA	Perfluorooctadecanoic acid
PFOPA	Perfluorooctyl phosphonic acid
PFOPiA	Perfluorooctyl phosphinic acid
PFOS	Perfluorooctane sulfonate or perfluorooctane sulfonic acid
PFPeA	Perfluoropentanoic acid
PFPrA	Perfluoropropanoic acid
PFPrOPrA	Perfluoro-2-propoxypropanoic acid (GenX)
PFTeDA	Perfluorotetradecanoic acid
PFTrDA	Perfluorotridecanoic acid
PFUnA	Perfluoroundecanoic acid
PTFE	Polytetrafluoroethylene
PVDF	Polyvinylidene fluoride
PVF	Polyvinyl fluoride
TFA	Tetrafluoroacetic acid
TFMS	Trifluoromethane sulfonic acid

Other abbreviations used in this document

AA	Alternatives Analysis
AFFF	Aqueous film-forming foam
ATSDR	Agency for Toxic Substances and Disease Registry
BAF	Bioaccumulation factor
BCF	Bioconcentration factor
BMF	Biomagnification factor
BPI	Biodegradable Products Institute
C	Carbon
CalRecycle	California's Department of Resources Recycling and Recovery
CAS	Chemical Abstracts Service
CDC	Centers for Disease Control and Prevention
CECBP	California Environmental Contaminant Biomonitoring Program
CEH	Center for Environmental Health
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFCs	Chlorofluorocarbons
CFR	Code of Federal Regulations
DDT	Dichlorodiplenyltricloroethane
DTSC	Department of Toxic Substances Control
ECHA	European Chemicals Agency
EPA	Environmental Protection Agency
EPCRA	Emergency Planning and Community Right-to-Know Act
F	Fluorine
FCN	Food Contact Notification
FCS	Food Contact Substances
FDA	U.S. Federal Drug and Food Administration
GRAS	Generally Recognized As Safe
HF	Hydrofluoric acid
MCL	Maximum contaminant level
MRL	Minimum Reporting Level
NDAA	National Defense Authorization Act
NGO	Nongovernmental organization
NHANES	National Health and Nutrition Examination Survey
OECD	Organization for Economic Cooperation and Development
OEHHA	Office of Environmental Health Hazard Assessment
PBT	Persistent, bioaccumulative and toxic
PCB	Polychlorinated biphenyl
POPs	Persistent organic pollutants

PPAR	Peroxisome proliferator-activated receptor
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
PET	Polyethylene terephthalate
PLA	Polylactic acid
RPF	Relative Potency Factor
SB	Senate Bill
SCP	Safer Consumer Products
SNUR	Significant New Use Rule
SVHC	Substances of Very High Concern
TDI	Tolerable daily intake
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
UCMR 3	Unregulated Contaminant Monitoring Rule 3
vPvB	Very persistent, very bioaccumulative
WWTP	Wastewater treatment plant

Units

°C	Degrees Celsius
µg	Micrograms
µg/kg	Micrograms per kilogram
g	Grams
g/L	Grams per liter
kg	Kilograms
kg/yr	Kilograms per year
kJ/mol	Kilojoules per mol
L/kg	Liters per kilogram
m	Meters
m^3	Meters cubed
mg/day	Milligrams per day
mg/kg/day	Milligrams per kilogram per day
mg/L	Milligrams per liter
ng/day	Nanograms per day
ng/g	Nanograms per gram
ng/kg	Nanograms per kilogram
ng/L	Nanograms per liter
ng/mL	Nanograms per milliliter
ng/m^2	Nanograms per meter squared

ng/m^3	Nanograms per meter cubed
Pa	Pascal
ppb	Parts per billion
ppm	Parts per million
ppt	Parts per trillion

REFERENCES

Aas CB et al. (2014). Effect of body condition on tissue distribution of perfluoroalkyl substances (PFASs) in Arctic fox (*Vulpes lagopus*). Environmental Science & Technology. 48:pp 11654–11661. doi: 10.1021/es503147n.

Abdallah MAE et al. (2020). Concentrations of perfluoroalkyl substances in human milk from Ireland: Implications for adult and nursing infant exposure. Chemosphere. 246:pp 125724. doi: 10.1016/j.chemosphere.2019.125724.

Abell R. (2002). Conservation biology for the biodiversity crisis: A freshwater follow-up. Conservation Biology. 16:pp 1435–1437. doi: 10.1046/j.1523-1739.2002.01532.x.

Adams J et al. (2008). Land use and the spatial distribution of perfluoroalkyl compounds as measured in the plasma of bottlenose dolphins (*Tursiops truncatus*). Marine Environmental Research. 66:pp 430–437. doi: 10.1016/j.marenvres.2008.07.004.

Ahrens L et al. (2009a). Longitudinal and latitudinal distribution of perfluoroalkyl compounds in the surface water of the Atlantic Ocean. Environmental Science & Technology. 43:pp 3122–3127. doi: 10.1021/es803507p.

Ahrens L. (2011). Polyfluoroalkyl compounds in the aquatic environment: A review of their occurrence and fate. Journal of Environmental Monitoring. 13:pp 20–31. doi: 10.1039/C0EM00373E.

Ahrens L et al. (2011a). Wastewater treatment plant and landfills as sources of polyfluoroalkyl compounds to the atmosphere. Environmental Science & Technology. 45:pp 8098–8105. doi: 10.1021/es1036173.

Ahrens L et al. (2012). Improved characterization of gas-particle partitioning for per- and polyfluoroalkyl substances in the atmosphere using annular diffusion denuder samplers. Environmental Science & Technology. 46:pp 7199–7206. doi: 10.1021/es300898s.

Ahrens L et al. (2016a). Screening of PFASs in groundwater and surface water. Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden, Vatten och miljö: Rapport 2016:2, Sveriges lantbruksuniversitet, Institutionen för vatten och miljö, ISBN 978-91-576-9386-0.

Ahrens L et al. (2011b). Polyfluoroalkyl compounds in the Canadian Arctic atmosphere. Environmental Chemistry. 8:pp 399. doi: 10.1071/EN10131.

Ahrens L et al. (2009b). Partitioning behavior of per- and polyfluoroalkyl compounds between pore water and sediment in two sediment cores from Tokyo Bay, Japan. Environmental Science & Technology. 43:pp 6969–6975. doi: 10.1021/es901213s.

Ahrens L et al. (2016b). Source tracking and impact of per- and polyfluoroalkyl substances at Svalbard. Report prepared by Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden, Norwegian University of Life Sciences (NMBU), Ås, Norway, and University Centre in Svalbard, Longyearbyen, Norway, April 2016. pp 17.

Aleksandrov K et al. (2019). Waste incineration of polytetrafluoroethylene (PTFE) to evaluate potential formation of per- and poly-fluorinated alkyl substances (PFAS) in flue gas. Chemosphere. 226:pp 898–906. doi: 10.1016/j.chemosphere.2019.03.191.

Allen B. (2020). Bill Text - SB-1044 Firefighting equipment and foam: PFAS chemicals. Available at: https://leginfo.legislature.ca.gov/faces/billNavClient.xhtml?bill_id=201920200SB1044. Accessed 25 Feb 2020.

Allred BM et al. (2015). Physical and biological release of poly- and perfluoroalkyl substances (PFASs) from municipal solid waste in anaerobic model landfill reactors. Environmental Science & Technology. 49:pp 7648–7656. doi: 10.1021/acs.est.5b01040.

Alves A et al. (2015). New approach for assessing human perfluoroalkyl exposure via hair. Talanta. 144:pp 574–583. doi: 10.1016/j.talanta.2015.07.009.

American Forest and Paper Association. (2020a). Background on Paper. Available at: https://www.afandpa.org/our-products/paper. Accessed 10 Feb 2020.

American Forest and Paper Association. (2020b). Background on Paperboard. Available at: https://www.afandpa.org/our-products/paper-based-packaging/paperboard. Accessed 10 Feb 2020.

Annunziato KM et al. (2019). Subtle morphometric, behavioral and gene expression effects in larval zebrafish exposed to PFHxA, PFHxS and 6:2 FTOH. Aquatic Toxicology. 208:pp 126–137. doi: 10.1016/j.aquatox.2019.01.009.

APHA. (2016). American Public Health Association (APHA): Reducing human exposure to highly fluorinated chemicals to protect public health. APHA Policy Number 20163. Available at: https://www.apha.org/policies-and-advocacy/public-health-policy-statements/policy-database/2016/12/21/reducing-human-exposure-to-highly-fluorinated-chemicals. Accessed 9 Sep 2017.

Appleman TD et al. (2014). Treatment of poly- and perfluoroalkyl substances in U.S. full-scale water treatment systems. Water Research. 51:pp 246–255. doi: 10.1016/j.watres.2013.10.067.

Armitage JM et al. (2009). Modeling the global fate and transport of perfluorooctane sulfonate (PFOS) and precursor compounds in relation to temporal trends in wildlife exposure. Environmental Science & Technology. 43:pp 9274–9280.

Arnot JA and Gobas FA. (2006). A review of bioconcentration factor (BCF) and bioaccumulation factor (BAF) assessments for organic chemicals in aquatic organisms. Environmental Reviews. 14:pp 257–297. doi: 10.1139/a06-005.

Arvaniti OS et al. (2015a). Reductive degradation of perfluorinated compounds in water using Mg-aminoclay coated nanoscale zero valent iron. Chemical Engineering Journal. 262:pp 133–139. doi: 10.1016/j.cej.2014.09.079.

Arvaniti OS et al. (2012). Occurrence of different classes of perfluorinated compounds in Greek wastewater treatment plants and determination of their solid–water distribution coefficients. Journal of Hazardous Materials. 239–240:pp 24–31. doi: 10.1016/j.jhazmat.2012.02.015.

Arvaniti OS and Stasinakis AS. (2015b). Review on the occurrence, fate and removal of perfluorinated compounds during wastewater treatment. Science of The Total Environment. 524–525:pp 81–92. doi: 10.1016/j.scitotenv.2015.04.023.

Ateia M et al. (2019a). The overlooked short- and ultrashort-chain poly- and perfluorinated substances: A review. Chemosphere. 220:pp 866–882. doi: 10.1016/j.chemosphere.2018.12.186.

Ateia M et al. (2019b). Cationic polymer for selective removal of GenX and short-chain PFAS from surface waters and wastewaters at ng/L levels. Water Research. 163:pp 114874. doi: 10.1016/j.watres.2019.114874.

ATSDR. (2018a). Agency for Toxic Substances and Disease Registry (ATSDR): An overview of perfluoroalkyl and polyfluoroalkyl substances and interim guidance for clinicians responding to patient exposure concerns. Available at: https://stacks.cdc.gov/view/cdc/77114.

ATSDR. (2018b). Agency for Toxic Substances and Disease Registry (ATSDR): Toxicological profile for perfluoroalkyls. Draft for public comment. Available at: https://www.atsdr.cdc.gov/toxprofiles/tp.asp?id=1117&tid=237.

ATSDR. (2018c). Agency for Toxic Substances and Disease Registry (ATSDR): Minimal Risk Levels (MRLs), August 2018. Available at: https://www.atsdr.cdc.gov/mrls/pdfs/atsdr_mrls.pdf.

Audenaert F et al. (1999). Fluorochemical textile repellents—synthesis and applications: A 3M perspective. Journal of the Textile Institute. 90:pp 76–94. doi: 10.1080/00405009908659480.

Averina M et al. (2020). Environmental pollutants in blood donors: The multicentre Norwegian donor study. Transfusion Medicine. doi: 10.1111/tme.12662.

Bach CC et al. (2015). Perfluoroalkyl and polyfluoroalkyl substances and human fetal growth: A systematic review. Critical Reviews in Toxicology. 45:pp 53–67. doi: 10.3109/10408444.2014.952400.

Banks RE, Smart BE and Tatlow JC. (1994). Organofluorine chemistry: Principles and commercial applications. Plenum, New York, ISBN: 978-1-4899-1204-6.

Banzhaf S et al. (2017). A review of contamination of surface-, ground-, and drinking water in Sweden by perfluoroalkyl and polyfluoroalkyl substances (PFASs). Ambio. 46:pp 335–346. doi: 10.1007/s13280-016-0848-8.

Barber JL et al. (2007). Analysis of per- and polyfluorinated alkyl substances in air samples from Northwest Europe. Journal of Environmental Monitoring. 9:pp 530. doi: 10.1039/b701417a.

Bartell SM et al. (2010). Rate of decline in serum PFOA concentrations after granular activated carbon filtration at two public water systems in Ohio and West Virginia. Environmental Health Perspectives. 118:pp 222–228. doi: 10.1289/ehp.0901252.

Benskin JP et al. (2012a). Perfluoroalkyl acids in the Atlantic and Canadian Arctic Oceans. Environmental Science & Technology. 46:pp 5815–5823. doi: 10.1021/es300578x.

Benskin JP et al. (2012b). Per- and polyfluoroalkyl substances in landfill leachate: Patterns, time trends, and sources. Environmental Science & Technology. 46:pp 11532–11540. doi: 10.1021/es302471n.

Berger U et al. (2009). Fish consumption as a source of human exposure to perfluorinated alkyl substances in Sweden – Analysis of edible fish from Lake Vättern and the Baltic Sea. Chemosphere. 76:pp 799–804. doi: 10.1016/j.chemosphere.2009.04.044.

Björklund JA, Thuresson K and de Wit CA. (2009). Perfluoroalkyl compounds (PFCs) in indoor dust: Concentrations, human exposure estimates, and sources. Environmental Science & Technology. 43:pp 2276–2281. doi: 10.1021/es803201a.

Björnsdotter MK et al. (2019). Ultra-short-chain perfluoroalkyl acids including trifluoromethane sulfonic acid in water connected to known and suspected point sources in Sweden. Environmental Science & Technology. doi: 10.1021/acs.est.9b02211.

Blaine AC et al. (2013). Uptake of perfluoroalkyl acids into edible crops via land applied biosolids: Field and greenhouse studies. Environmental Science & Technology. 47:pp 14062–14069. doi: 10.1021/es403094q.

Blaine AC et al. (2014a). Perfluoroalkyl acid distribution in various plant compartments of edible crops grown in biosolids-amended soils. Environmental Science & Technology. 48:pp 7858–7865. doi: 10.1021/es500016s.

Blaine AC et al. (2014b). Perfluoroalkyl acid uptake in lettuce (Lactuca sativa) and strawberry (Fragaria ananassa) irrigated with reclaimed water. Environmental Science & Technology. 48:pp 14361–14368. doi: 10.1021/es504150h.

Blanc M et al. (2017). Mixture-specific gene expression in Zebrafish (*Danio rerio*) embryos exposed to perfluorooctane sulfonic acid (PFOS), perfluorohexanoic acid (PFHxA) and 3,3',4,4',5-pentachlorobiphenyl (PCB126). Science of The Total Environment. 590–591:pp 249–257. doi: 10.1016/j.scitotenv.2017.02.232.

Blum A et al. (2015). The Madrid statement on poly- and perfluoroalkyl substances (PFASs). Environmental Health Perspectives. 123:pp A107-111. doi: 10.1289/ehp.1509934.

Boiteux V et al. (2017). Concentrations and patterns of perfluoroalkyl and polyfluoroalkyl substances in a river and three drinking water treatment plants near and far from a major production source. Science of The Total Environment. 583:pp 393–400. doi: 10.1016/j.scitotenv.2017.01.079.

Boone JS et al. (2019). Per- and polyfluoroalkyl substances in source and treated drinking waters of the United States. Science of The Total Environment. 653:pp 359–369. doi: 10.1016/j.scitotenv.2018.10.245.

Borg D et al. (2013). Cumulative health risk assessment of 17 perfluoroalkylated and polyfluoroalkylated substances (PFASs) in the Swedish population. Environment International. 59:pp 112–123. doi: 10.1016/j.envint.2013.05.009.

Bowman J. (2017). PFASs in carpet and furniture applications. Presentation at the DTSC Public Workshop on Perfluoroalkyl and Polyfluoroalkyl Substances (PFASs) in Carpets, Rugs, Indoor Upholstered Furniture, and Their Care and Treatment Products, January 31, 2017, Sacramento, CA.

BPI (Biodegradable Products Institute). (2018). Position on Fluorinated Chemicals. Available at: https://www.bpiworld.org/Fluorinated-Chemicals. Accessed 13 Dec 2018.

Brendel S et al. (2018). Short-chain perfluoroalkyl acids: Environmental concerns and a regulatory strategy under REACH. Environmental Sciences Europe. 30:pp 9. doi: 10.1186/s12302-018-0134-4.

Britannica. (2020). Background on Paperboard. Available at: https://www.britannica.com/technology/papermaking/Paper-grades#ref625145. Accessed 10 Feb 2020.

Buck RC et al. (2011). Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. Integrated Environmental Assessment and Management. 7:pp 513–541. doi: 10.1002/ieam.258.

Buck RC, Murphy PM and Pabon M. (2012). Chemistry, properties, and uses of commercial fluorinated surfactants. in: Knepper TP, and Lange FT (eds). Handbook of Environmental Chemistry, Volume 17, Polyfluorinated Chemicals and Transformation Products. Springer, New York, pp 1–24. ISBN: 978-3-642-21871-2.

Burkemper JL et al. (2017). Radiosynthesis and biological distribution of 18F-labeled perfluorinated alkyl substances. Environmental Science & Technology Letters. 4:pp 211–215. doi: 10.1021/acs.estlett.7b00042.

Busch J et al. (2010). Polyfluoroalkyl compounds in landfill leachates. Environmental Pollution. 158:pp 1467–1471. doi: 10.1016/j.envpol.2009.12.031.

Butenhoff JL et al. (2004). Pharmacokinetics of perfluorooctanoate in cynomolgus monkeys. Toxicological Sciences. 82:pp 394–406. doi: 10.1093/toxsci/kfh302.

Butenhoff JL et al. (2012). Toxicological evaluation of ammonium perfluorobutyrate in rats: Twenty-eight-day and ninety-day oral gavage studies. Reproductive Toxicology. 33:pp 513–530. doi: 10.1016/j.reprotox.2011.08.004.

Butler S. (2019). A History of Popcorn. in: History Channel. Available at: https://www.history.com/news/a-history-of-popcorn. Accessed 17 Dec 2019.

Butt CM et al. (2010). Levels and trends of poly- and perfluorinated compounds in the Arctic environment. Science of The Total Environment. 408:pp 2936–2965. doi: 10.1016/j.scitotenv.2010.03.015.

Butt CM et al. (2008). Spatial trends of perfluoroalkyl compounds in ringed seals (*Phoca hispida*) from the Canadian Arctic. Environmental Toxicology and Chemistry. 27:pp 542. doi: 10.1897/07-428.1.

Butt CM, Muir DCG and Mabury SA. (2014). Biotransformation pathways of fluorotelomer-based polyfluoroalkyl substances: A review. Environmental Toxicology and Chemistry. 33:pp 243–267. doi: 10.1002/etc.2407.

C8 Science Panel. (2011). Probable link evaluation of pregnancy induced hypertension and preeclampsia. December 5, 2011. Available at: http://www.c8sciencepanel.org/pdfs/Probable_Link_C8_PIH_5Dec2011.pdf. Accessed 11 May 2020.

C8 Science Panel. (2012). Probable link evaluation of cancer. April 15, 2012. Available at: http://www.c8sciencepanel.org/pdfs/Probable_Link_C8_Cancer_16April2012_v2.pdf. Accessed 11 May 2020.

Cai D et al. (2020). High trans-placental transfer of perfluoroalkyl substances alternatives in the matched maternal-cord blood serum: Evidence from a birth cohort study. Science of The Total Environment. 705:pp 135885. doi: 10.1016/j.scitotenv.2019.135885.

Calafat AM et al. (2007). Serum concentrations of 11 polyfluoroalkyl compounds in the U.S. population: Data from the National Health and Nutrition Examination Survey (NHANES). Environmental Science & Technology. 41:pp 2237–2242. doi: 10.1021/es062686m.

Calafat AM et al. (2019). Legacy and alternative per- and polyfluoroalkyl substances in the U.S. general population: Paired serum-urine data from the 2013–2014 National Health and Nutrition Examination Survey. Environment International. 131:pp 105048. doi: 10.1016/j.envint.2019.105048.

California Employment Development Department. (2019). Labor Market Information - Long Term Projections (Ten-years). Available at:

https://www.labormarketinfo.edd.ca.gov/data/employment-projections.html. Accessed 26 Sep 2019.

California State Water Resources Control Board. (2019). Per- and Polyfluoroalkyl Substances (PFAS). Available at: https://www.waterboards.ca.gov/pfas/. Accessed 29 Oct 2019.

California State Water Resources Control Board. (2020). Response levels lowered for water systems statewide as PFAS investigation continues. Available at: https://www.waterboards.ca.gov/press_room/press_releases/2020/pr02062020_pfoa_pfos_response_levels.pdf. Accessed 28 Feb 2020.

CalRecycle. (2020). California Department of Resources Recycling and Recovery (CalRecycle): Public Hearing Notice: Public Meeting Notice: Sustainable Packaging for the State of California Act of 2018 (SB 1335) Formal Rulemaking Public Hearing. Available at: https://www2.calrecycle.ca.gov/PublicNotices/Details/3943. Accessed 26 Mar 2020.

Campo J et al. (2016). Analysis of the presence of perfluoroalkyl substances in water, sediment and biota of the Jucar River (E Spain). Sources, partitioning and relationships with water physical characteristics. Environmental Research. 147:pp 503–512. doi: 10.1016/j.envres.2016.03.010.

Cao W et al. (2018). Perfluoroalkyl substances in umbilical cord serum and gestational and postnatal growth in a Chinese birth cohort. Environment International. 116:pp 197–205. doi: 10.1016/j.envint.2018.04.015.

Cariou R et al. (2015). Perfluoroalkyl acid (PFAA) levels and profiles in breast milk, maternal and cord serum of French women and their newborns. Environment International. 84:pp 71–81. doi: 10.1016/j.envint.2015.07.014.

Casal P et al. (2017). Accumulation of perfluoroalkylated substances in oceanic plankton. Environmental Science & Technology. 51:pp 2766–2775. doi: 10.1021/acs.est.6b05821.

Casey Printing. (2017). Background on Paper. Available at: https://www.caseyprinting.com/blog/how-is-paper-made. Accessed 6 Feb 2020.

CDC. (2018). Centers for Disease Control and Prevention (CDC): Fourth National Report on Human Exposure to Environmental Chemicals Updated Tables, March 2018, Volume One. Available at: https://www.cdc.gov/exposurereport/pdf/FourthReport_UpdatedTables_Volume1_Mar2018.pdf.

CDC. (2014). Centers for Disease Control and Prevention (CDC): Breastfeeding Report Card United States, 2014. Available at: https://www.cdc.gov/breastfeeding/pdf/2014breastfeedingreportcard.pdf. Accessed 9 Oct 2017.

CECBP. (2015a). California Environmental Contaminant Biomonitoring Program (CECBP): Potential Priority Chemicals: Perfluoroalkyl and Polyfluoroalkyl Substances (PFASs). Available at:

https://biomonitoring.ca.gov/sites/default/files/downloads/PotentialPriority_PFASs_111815.pdf.

CECBP. (2017). California Environmental Contaminant Biomonitoring Program (CECBP): Results for perfluorochemicals (PFCs). Available at: https://biomonitoring.ca.gov/results/chemical/154.

CECBP. (2015b). California Environmental Contaminant Biomonitoring Program (CECBP): California Teachers Study (CTS). Sample collection date: 2011 - ongoing. Results last updated on September 22, 2015. Available at: https://biomonitoring.ca.gov/results/projects/411.

CECBP. (2018). California Environmental Contaminant Biomonitoring Program (CECBP): Project results for biomonitoring exposures study (BEST). 2. Expanded. Available at: https://biomonitoring.ca.gov/results/projects/494.

CEH. (2018). Center for Environmental Health (CEH): Avoiding hidden hazards: A purchaser's guide to safer foodware. Available at: https://www.ceh.org/wp-content/uploads/CEH-Disposable-Foodware-Report-final-1.31.pdf.

Ceretta F. (2013). Synthesis and characterization of fluorinated compounds for industrial applications. Ph.D. Dissertation, Università Degli Studi di Padova. pp 127.

Chang ET et al. (2014). A critical review of perfluorooctanoate and perfluorooctanesulfonate exposure and cancer risk in humans. Critical Reviews in Toxicology. 44:pp 1–81. doi: 10.3109/10408444.2014.905767.

Chang SC et al. (2008). Comparative pharmacokinetics of perfluorobutyrate in rats, mice, monkeys, and humans and relevance to human exposure via drinking water. Toxicological Sciences. 104:pp 40–53. doi: 10.1093/toxsci/kfn057.

Chang SC et al. (2012). Comparative pharmacokinetics of perfluorohexanesulfonate (PFHxS) in rats, mice, and monkeys. Reproductive Toxicology. 33:pp 441–451. doi: 10.1016/j.reprotox.2011.07.004.

Chapman. (2016). Alabama Water Utility sues Dalton carpet makers. The Atlanta Journal-Constitution, Sept. 26, 2016. Available at: http://www.ajc.com/news/state--regional/alabama-water-utility-sues-dalton-carpet-makers/Yh6dnovTsKxe9vMij7ahAN/. Accessed 15 Aug 2017.

Chawaga P. (2017). Michigan town uncovers PFC contamination 400 times EPA-advised limits. News Feature in Water Online, September 13, 2017. Available at: https://www.wateronline.com/doc/michigan-town-the-latest-to-uncover-pfc-contamination-0001. Accessed 19 Oct 2017.

Chemical Watch. (2020). US food chain Taco Bell to phase out PFAS, phthalates and BPA. Available at: https://chemicalwatch.com/88548/us-food-chain-taco-bell-to-phase-out-pfas-phthalates-and-bpa. Accessed 19 Feb 2020.

Chemical Watch. (2017a). Swedish water body urges ban on goods potentially containing PFAS. Available at: https://chemicalwatch.com/55406. Accessed 15 Aug 2017.

Chemical Watch. (2017b). Swedish authorities agree to tackle PFAS. Available at: https://chemicalwatch.com/56578/swedish-authorities-agree-to-tackle-pfas. Accessed 15 Aug 2017.

Chen CH et al. (2020). Accumulation and phytotoxicity of perfluorooctanoic acid and 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propanoate in *Arabidopsis thaliana* and *Nicotiana benthamiana*. Environmental Pollution. 259:pp 113817. doi: 10.1016/j.envpol.2019.113817.

Chen F et al. (2017a). Chlorinated polyfluoroalkyl ether sulfonic acids in matched maternal, cord, and placenta samples: a study of transplacental transfer. Environmental Science & Technology. 51:pp 6387–6394. doi: 10.1021/acs.est.6b06049.

Chen H et al. (2017b). Detection, occurrence, and fate of fluorotelomer alcohols in municipal wastewater treatment plants. Environmental Science & Technology. 51:pp 8953–8961. doi: 10.1021/acs.est.7b00315.

Chen H et al. (2019). Multi-compartment distribution of perfluoroalkyl and polyfluoroalkyl substances (PFASs) in an urban catchment system. Water Research. 154:pp 227–237. doi: 10.1016/j.watres.2019.02.009.

Chen H et al. (2016). Reversible and irreversible sorption of perfluorinated compounds (PFCs) by sediments of an urban reservoir. Chemosphere. 144:pp 1747–1753. doi: 10.1016/j.chemosphere.2015.10.055.

Chen YM and Guo LH. (2009). Fluorescence study on site-specific binding of perfluoroalkyl acids to human serum albumin. Archives of Toxicology. 83:pp 255–261. doi: 10.1007/s00204-008-0359-x.

Choi YJ et al. (2019). Perfluoroalkyl acid characterization in U.S. municipal organic solid waste composts. Environmental Science & Technology Letters. 6:pp 372–377. doi: 10.1021/acs.estlett.9b00280.

Christensen KY et al. (2017). Perfluoroalkyl substances and fish consumption. Environmental Research. 154:pp 145–151. doi: 10.1016/j.envres.2016.12.032.

Chu S et al. (2016). A new fluorinated surfactant contaminant in biota: perfluorobutane sulfonamide in several fish species. Environmental Science & Technology. 50:pp 669–675. doi: 10.1021/acs.est.5b05058.

Chu S and Letcher RJ. (2017a). Side-chain fluorinated polymer surfactants in aquatic sediment and biosolid-augmented agricultural soil from the Great Lakes basin of North America. Science of The Total Environment. 607–608:pp 262–270. doi: 10.1016/j.scitotenv.2017.06.252.

Chu S and Letcher RJ. (2017b). Side-chain fluorinated polymer surfactants in aquatic sediment and biosolid-augmented agricultural soil from the Great Lakes basin of North America. Science of The Total Environment. 607–608:pp 262–270. doi: 10.1016/j.scitotenv.2017.06.252.

Clara M et al. (2008). Emissions of perfluorinated alkylated substances (PFAS) from point sources—identification of relevant branches. Water Science and Technology. 58:pp 59–66. doi: 10.2166/wst.2008.641.

Clean Production Action. (2018). A Guide to Purchasing PFAS-Free Food Service Ware. Available at: https://www.cleanproduction.org/resources/entry/pfas. Accessed 11 Dec 2018.

Codling G et al. (2014). Historical trends of inorganic and organic fluorine in sediments of Lake Michigan. Chemosphere. 114:pp 203–209. doi: 10.1016/j.chemosphere.2014.03.080.

Coggan TL et al. (2019). A single analytical method for the determination of 53 legacy and emerging per- and polyfluoroalkyl substances (PFAS) in aqueous matrices. Analytical and Bioanalytical Chemistry. 411:pp 3507–3520. doi: 10.1007/s00216-019-01829-8.

Concawe. (2016). Environmental fate and effects of poly and perfluoroalkyl substances (PFAS). Concawe Soil and Groundwater Taskforce, Brussels, Belgium. Prepared by ARCADIS: T. Pancras, G. Schrauwen, T. Held, K. Baker, I. Ross, H. Slenders, Report # 8/16. Accessed March 15, 2017.

COP. (2019). Conference of the Parties (COP): Ninth Meeting of the Conference of the Parties to the Stockholm Convention. Available at: http://chm.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP9/tabid/7521/Default.aspx. Accessed 11 Mar 2020.

Corsini E et al. (2014). Perfluorinated compounds: Emerging POPs with potential immunotoxicity. Toxicology Letters. 230:pp 263–270. doi: 10.1016/j.toxlet.2014.01.038.

Cousins IT et al. (2016). The precautionary principle and chemicals management: The example of perfluoroalkyl acids in groundwater. Environment International. 94:pp 331–340. doi: 10.1016/j.envint.2016.04.044.

Cousins IT et al. (2019a). Why is high persistence alone a major cause of concern? Environmental Science: Processes & Impacts. 21:pp 781–792. doi: 10.1039/C8EM00515J.

Cousins IT et al. (2019b). The concept of essential use for determining when uses of PFASs can be phased out. pp 27.

Cousins IT, Kong D and Vestergren R. (2011). Reconciling measurement and modelling studies of the sources and fate of perfluorinated carboxylates. Environmental Chemistry. 8:pp 339. doi: 10.1071/EN10144.

Crone BC et al. (2019). Occurrence of per- and polyfluoroalkyl substances (PFAS) in source water and their treatment in drinking water. Critical Reviews in Environmental Science and Technology. 49:pp 2359–2396. doi: 10.1080/10643389.2019.1614848.

Cui Q et al. (2018). Occurrence and tissue distribution of novel perfluoroether carboxylic and sulfonic acids and legacy per/polyfluoroalkyl substances in Black-Spotted Frog (Pelophylax nigromaculatus). Environmental Science & Technology. 52:pp 982–990. doi: 10.1021/acs.est.7b03662.

Dairkee SH et al. (2018). A ternary mixture of common chemicals perturbs benign human breast epithelial cells more than the same chemicals do individually. Toxicological Sciences. 165:pp 131–144. doi: 10.1093/toxsci/kfy126.

Dalahmeh S et al. (2018). Per- and polyfluoroalkyl substances (PFASs) in water, soil and plants in wetlands and agricultural areas in Kampala, Uganda. Science of The Total Environment. 631–632:pp 660–667. doi: 10.1016/j.scitotenv.2018.03.024.

Danish Environmental Protection Agency. (2015). Short-chain polyfluoroalkyl substances (PFAS). A literature review of information on human health effects and environmental fate and effect aspects of short-chain PFAS. Project No. 1707, Copenhagen, Denmark.

Danish Ministry of Environment and Food. (2019). The Minister of Food is ready to ban fluoride substances. Available at: https://mfvm.dk/nyheder/nyhed/nyhed/foedevareministeren-er-klar-til-at-forbyde-fluorstoffer/. Accessed 20 Feb 2019.

Das KP et al. (2008). Effects of perfluorobutyrate exposure during pregnancy in the mouse. Toxicological Sciences. 105:pp 173–181. doi: 10.1093/toxsci/kfn099.

Dasu K and Lee LS. (2016). Aerobic biodegradation of toluene-2,4-di(8:2 fluorotelomer urethane) and hexamethylene-1,6-di(8:2 fluorotelomer urethane) monomers in soils. Chemosphere. 144:pp 2482–2488. doi: 10.1016/j.chemosphere.2015.11.021.

De Silva AO et al. (2011). Detection of a cyclic perfluorinated acid, perfluoroethylcyclohexane sulfonate, in the Great Lakes of North America. Environmental Science & Technology. 45:pp 8060–8066. doi: 10.1021/es200135c.

De Silva AO et al. (2016). Perfluoroalkylphosphinic acids in northern pike (*Esox lucius*), double-crested cormorants (*Phalacrocorax auritus*), and bottlenose dolphins (*Tursiops truncatus*) in relation to other perfluoroalkyl acids. Environmental Science & Technology. 50:pp 10903–10913. doi: 10.1021/acs.est.6b03515.

Dean JA. (1999). Lange's Handbook of Chemistry. McGraw-Hill, New York, NY, ISBN: 978-0-07-016384-3.

Del Vento S et al. (2012). Volatile per– and polyfluoroalkyl compounds in the remote atmosphere of the western Antarctic Peninsula: An indirect source of perfluoroalkyl acids to Antarctic waters. Atmospheric Pollution Research. 3:pp 450–455.

D'eon JC et al. (2009). Perfluorinated phosphonic acids in Canadian surface waters and wastewater treatment plant effluent: Discovery of a new class of perfluorinated acids. Environmental Toxicology and Chemistry. 28:pp 2101. doi: 10.1897/09-048.1.

D'eon JC and Mabury SA. (2011). Exploring indirect sources of human exposure to perfluoroalkyl carboxylates (PFCAs): Evaluating uptake, elimination, and biotransformation of polyfluoroalkyl phosphate esters (PAPs) in the rat. Environmental Health Perspectives. 119:pp 344–350. doi: 10.1289/ehp.1002409.

D'eon JC and Mabury SA. (2007). Production of perfluorinated carboxylic acids (PFCAs) from the biotransformation of polyfluoroalkyl phosphate surfactants (PAPS): Exploring routes of human contamination. Environmental Science & Technology. 41:pp 4799–4805. doi: 10.1021/es070126x.

D'eon JC and Mabury SA. (2010). Uptake and elimination of perfluorinated phosphonic acids in the rat. Environmental Toxicology and Chemistry. pp n/a-n/a. doi: 10.1002/etc.167.

DeWitt JC. (2015). Toxicological effects of perfluoroalkyl and polyfluoroalkyl substances. In: Molecular and Integrative Toxicology. ISBN: 978-3-319-15517-3.

DeWitt JC, Peden-Adams MM and Keil DE. (2015). Immunotoxic effects of perfluoroalkylated compounds: Mechanisms of action. in: Corsini E, and Loveren H van (eds). Molecular Immunotoxicology. Wiley-V C H Verlag Gmbh, Weinheim, pp 263–284. ISBN: 978-3-527-67696-5.

Didone M et al. (2017). Moulded pulp manufacturing: Overview and prospects for the process technology. Packaging Technology and Science. 30:pp 231–249. doi: 10.1002/pts.2289.

Dietz R et al. (2008). Increasing perfluoroalkyl contaminants in East Greenland polar bears (*Ursus maritimus*): A new toxic threat to the Arctic bears. Environmental Science & Technology. 42:pp 2701–2707. doi: 10.1021/es7025938.

Ding G et al. (2012). Toxicity of polyfluorinated and perfluorinated compounds to lettuce (*Lactuca sativa*) and green algae (*Pseudokirchneriella subcapitata*). Archives of Environmental Contamination and Toxicology. 62:pp 49–55. doi: 10.1007/s00244-011-9684-9.

Ding G and Peijnenburg WJGM. (2013). Physicochemical properties and aquatic toxicity of poly- and perfluorinated compounds. Critical Reviews in Environmental Science and Technology. 43:pp 598–678. doi: 10.1080/10643389.2011.627016.

Ding N et al. (2020). Longitudinal trends in perfluoroalkyl and polyfluoroalkyl substances among multiethnic midlife women from 1999 to 2011: The Study of Women's Health Across the Nation. Environment International. 135:pp 105381. doi: 10.1016/j.envint.2019.105381.

Dingell D. (2019). H.R. 2827: Keep Food Containers Safe from PFAS Act of 2019. Available at: https://www.congress.gov/bill/116th-congress/house-bill/2827/text.

Domingo JL and Nadal M. (2019). Human exposure to per- and polyfluoroalkyl substances (PFAS) through drinking water: A review of the recent scientific literature. Environmental Research. 177:pp 108648. doi: 10.1016/j.envres.2019.108648.

Domingo JL and Nadal M. (2017). Per- and polyfluoroalkyl substances (PFASs) in food and human dietary Intake: A review of the recent scientific literature. Journal of Agricultural and Food Chemistry. 65:pp 533–543. doi: 10.1021/acs.jafc.6b04683.

Dreyer A et al. (2015). Particle-size distribution of airborne poly- and perfluorinated alkyl substances. Chemosphere. 129:pp 142–149. doi: 10.1016/j.chemosphere.2014.06.069.

Drottar KR, Van Hoven RL and Krueger HO. (2001). Perfluorooctanesulfonate, potassium salt (PFOS): A flow-through bioconcentration test with the bluegill (*Lepomis macrochirus*). Wildlife International, Ltd., Project No. 454A-134, EPA Docket AR226-1030a042.

DTSC. (2020). Department of Toxic Substances Control (DTSC): PFAS Food Packaging Workshop Part 1 - January 14, 2020. Available at: https://www.youtube.com/watch?v=mkjq6AvmWIY&feature=youtu.be. Accessed 16 Jul 2020.

DTSC. (2018). Department of Toxic Substances Control (DTSC): 2018-2020 Priority Product Work Plan. Available at: https://dtsc.ca.gov/scp/priority-product-work-plan/. Accessed 7 Aug 2019.

Du J et al. (2017). Oxidative stress and apotosis to Zebrafish (*Danio rerio*) embryos exposed to perfluorooctane sulfonate (PFOS) and ZnO nanoparticles. International Journal of Occupational Medicine and Environmental Health. doi: 10.13075/ijomeh.1896.00669.

ECHA. (2014). European Chemicals Agency (ECHA): Annex XV restriction report. Proposal for a restriction: Perfluorooctanoic acid (PFOA), PFOA salts and PFOA-related substances. Version 1.0, October 17, 2014. Available at: https://echa.europa.eu/documents/10162/e9cddee6-3164-473d-b590-8fcf9caa50e7.

ECHA. (2020). European Chemicals Agency (ECHA): Candidate list of substances of very high concern for authorisation. Available at: https://echa.europa.eu/web/guest/candidate-list-table. Accessed 11 May 2020.

ECY. (2018). Washington State Department of Ecology (ECY): PFAS in Food Packaging Alternatives Assessment Project Summary November 6, 2018. Available at: https://www.ezview.wa.gov/Portals/_1962/Documents/PFAS/PFASFoodPackagingAAProjectSummary110618.pdf. Accessed 13 Dec 2018.

ECY. (2020). Washington State Department of Ecology (ECY): PFAS in Food Packaging Alternatives Assessment Update (02/24/2020). Available at: https://www.ezview.wa.gov/site/alias__1962/37610/DesktopDefault.aspx?alias=1962&PageID=37610. Accessed 2 Apr 2020.

EFSA. (2011). The European Food Safety Authority (EFSA): Scientific opinion on the safety evaluation of the substance, 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid], ammonium salt, CAS No. 958445-44-8, for use in food contact materials. EFSA Panel on food contact materials, enzymes, flavourings and processing aids (CEF). EFSA Journal. 9:pp 2182. doi: 10.2903/j.efsa.2011.2182.

EFSA. (2012). The European Food Safety Authority (EFSA): Perfluoroalkylated substances in food: Occurrence and dietary exposure. EFSA Journal. 10: doi: 10.2903/j.efsa.2012.2743.

EFSA. (2008). The European Food Safety Authority (EFSA): Perfluorooctane sulfonate (PFOS), perfluorooctanoic acid (PFOA) and their salts: Scientific opinion of the scientific panel on contaminants in the food chain. EFSA Journal. 653: doi: 10.2903/j.efsa.2008.653.

EFSA Panel on Contaminants in the Food Chain (CONTAM). (2018). Risk to human health related to the presence of perfluorooctane sulfonic acid and perfluorooctanoic acid in food. EFSA Journal. 16: doi: 10.2903/j.efsa.2018.5194.

Eftaiha AF, Brunet SMK and Paige MF. (2012). Influence of film composition on the morphology, mechanical properties, and surfactant recovery of phase-separated phospholipid-perfluorinated fatty acid mixed monolayers. Langmuir. 28:pp 15150–15159. doi: 10.1021/la3026655.

Egeghy PP and Lorber M. (2011). An assessment of the exposure of Americans to perfluorooctane sulfonate: A comparison of estimated intake with values inferred from NHANES data. Journal of Exposure Science & Environmental Epidemiology. 21:pp 150–168. doi: 10.1038/jes.2009.73.

Eggers Pedersen K et al. (2016). Per- and polyfluoroalkyl substances (PFASs) – New endocrine disruptors in polar bears (*Ursus maritimus*)? Environment International. 96:pp 180–189. doi: 10.1016/j.envint.2016.07.015.

Eggers Pedersen K et al. (2015). Brain region-specific perfluoroalkylated sulfonate (PFSA) and carboxylic acid (PFCA) accumulation and neurochemical biomarker responses in east Greenland polar Bears (*Ursus maritimus*). Environmental Research. 138:pp 22–31. doi: 10.1016/j.envres.2015.01.015.

Eichler CMA and Little JC. (2020). A framework to model exposure to per- and polyfluoroalkyl substances in indoor environments. Environmental Science: Processes & Impacts. 22:pp 500–511. doi: 10.1039/C9EM00556K.

Ellis DA et al. (2003). Atmospheric lifetime of fluorotelomer alcohols. Environmental Science & Technology. 37:pp 3816–3820. doi: 10.1021/es034136j.

Ellis DA et al. (2004). Degradation of fluorotelomer alcohols: A likely atmospheric source of perfluorinated carboxylic acids. Environmental Science & Technology. 38:pp 3316–3321. doi: 10.1021/es049860w.

Ellison G. (2017). Tannery chemicals are 540 times above EPA level in Belmont well. Grand Rapids News. Available at: http://www.mlive.com/news/grand-rapids/index.ssf/2017/09/wolverine_house_street_well.html. Accessed 19 Oct 2017.

Ericson JI et al. (2012). Per- and polyfluorinated compounds (PFCs) in house dust and indoor air in Catalonia, Spain: Implications for human exposure. Environment International. 39:pp 172–180. doi: 10.1016/j.envint.2011.09.004.

Eryasa B et al. (2019). Physico-chemical properties and gestational diabetes predict transplacental transfer and partitioning of perfluoroalkyl substances. Environment International. 130:pp 104874. doi: 10.1016/j.envint.2019.05.068.

Eschauzier C et al. (2013). Perfluorinated alkylated acids in groundwater and drinking water: Identification, origin and mobility. Science of The Total Environment. 458–460:pp 477–485. doi: 10.1016/j.scitotenv.2013.04.066.

Escoruela J et al. (2018). Occurrence of per- and polyfluoroalkyl substances in *Calonectris* shearwaters breeding along the Mediterranean and Atlantic colonies. Marine Pollution Bulletin. 131:pp 335–340. doi: 10.1016/j.marpolbul.2018.04.032.

Eun H et al. (2020). Evaluation of perfluoroalkyl substances in field-cultivated vegetables. Chemosphere. 239:pp 124750. doi: 10.1016/j.chemosphere.2019.124750.

Eurofins Eaton Analytical. (2017). PFAS monitoring in a post health advisory world-what should we be doing? Poly- and perfluoroalkyl substances (PFASs). Available at: https://assets.documentcloud.org/documents/4165198/EEA-Study-Documents.pdf. Accessed 15 Nov 2017.

European Commission. (2017). Study for the strategy for a non-toxic environment of the 7th Environment Action Programme, Sub-study d-Very Persistent Chemicals. Prepared by G. Goldenman (Milieu Ltd), with contributions from R. Pedersen, H. Bradley, M. Fernandez, R. Weber, M. Scheringer, and P. Fantke, August 2017. The European Commission, Directorate-General for Environment, Brussels, Belgium. Available at: http://ec.europa.eu/environment/chemicals/non-toxic/pdf/Sub-study%20d%20very%20persistent%20subst.%20NTE%20final.pd. Accessed 22 Nov 2017.

European Commission. (2018). Directive of the European Parliament and of the Council on the quality of water intended for human consumption (recast). Brussels, 1.2.2018, COM(2017) 753 final. Available at: http://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1519210589057&uri=CELEX:52017PC0753. Accessed 11 Apr 2018.

EWG. (2003). Environmental Working Group (EWG): PFCs: Global contaminants: PFCs last forever. Available at: http://www.ewg.org/research/pfcs-global-contaminants/pfcs-last-forever. Accessed 20 Jun 2017.

Fassler J. (2019). The bowls at Chipotle and Sweetgreen are supposed to be compostable. They contain cancer-linked "forever chemicals." Available at: https://newfoodeconomy.org/pfas-forever-chemicals-sweetgreen-chipotle-compostable-biodegradable-bowls/. Accessed 7 Aug 2019.

Fauconier G et al. (2019). Perfluorinated compounds in the aquatic food chains of two subtropical estuaries. Science of The Total Environment. pp 135047. doi: 10.1016/j.scitotenv.2019.135047.

FDA. (2020a). Food and Drug Administration (FDA): Inventory of Effective Food Contact Substance (FCS) Notifications. Available at: https://www.accessdata.fda.gov/scripts/fdcc/?set=FCN. Accessed 2 Mar 2020.

FDA. (2020b). Food and Drug Administration (FDA): Packaging & Food Contact Substances (FCS). in: FDA. Available at: http://www.fda.gov/food/food-ingredients-packaging/packaging-food-contact-substances-fcs. Accessed 28 Feb 2020.

FDA (Food & Drug Administration). (2018). Update on Perfluorinated Grease-proofing Agents. Available at: https://www.fda.gov/Food/IngredientsPackagingLabeling/PackagingFCS/Notifications/ucm308462.htm. Accessed 7 Feb 2019.

FDA (Food and Drug Administration). (2016). Constituent Updates - FDA Removes Approval for the Use of PFCs in Food Packaging Based on the Abandonment. Available at: https://www.fda.gov/Food/NewsEvents/ConstituentUpdates/ucm528911.htm. Accessed 13 Dec 2018.

Feng M et al. (2015). Characterization of the thermolysis products of Nafion membrane: A potential source of perfluorinated compounds in the environment. Scientific Reports. 5: doi: 10.1038/srep09859.

Fleisch AF et al. (2017). Early-life exposure to perfluoroalkyl substances and childhood metabolic function. Environmental Health Perspectives. 125:pp 481–487. doi: 10.1289/EHP303.

FluoroCouncil. (2020). Applications - FluoroTechnology: Critical to Modern Life. Available at: https://fluorocouncil.com/applications/. Accessed 8 May 2020.

Fraser AJ et al. (2013). Polyfluorinated compounds in dust from homes, offices, and vehicles as predictors of concentrations in office workers' serum. Environment International. 60:pp 128–136. doi: 10.1016/j.envint.2013.08.012.

Fraser AJ et al. (2012). Polyfluorinated compounds in serum linked to indoor air in office environments. Environmental Science & Technology. 46:pp 1209–1215. doi: 10.1021/es2038257.

Frisbee SJ et al. (2009). The C8 health project: Design, methods, and participants. Environmental Health Perspectives. 117:pp 1873–1882. doi: 10.1289/ehp.0800379.

Frömel DT et al. (2016). Investigations on the presence and behavior of precursors to perfluoroalkyl substances in the environment as a preparation of regulatory measures. German Environment Agency Report No. (UBA-FB) 002223/ENG. Available at: https://www.umweltbundesamt.de/sites/default/files/medien/378/publikationen/texte_08_2016_investigations_on_the_presence_and_behavior.pdf. Accessed December 20, 2016. pp 272.

Frömel T and Knepper TP. (2010). Biodegradation of fluorinated alkyl substances. In: Whitacre DM, de Voogt P (eds). Reviews of Environmental Contamination and Toxicology. 208: Perfluorinated Alkylated Substances. Springer, New York:pp 161–177.

Fromme H et al. (2015). Neutral polyfluorinated compounds in indoor air in Germany – The LUPE 4 study. Chemosphere. 139:pp 572–578. doi: 10.1016/j.chemosphere.2015.07.024.

Fromme H et al. (2010). Pre- and postnatal exposure to perfluorinated compounds (PFCs). Environmental Science & Technology. 44:pp 7123–7129. doi: 10.1021/es101184f.

Fromme H et al. (2009). Perfluorinated compounds – Exposure assessment for the general population in western countries. International Journal of Hygiene and Environmental Health. 212:pp 239–270. doi: 10.1016/j.ijheh.2008.04.007.

Fuertes I et al. (2017). Perfluorinated alkyl substances (PFASs) in northern Spain municipal solid waste landfill leachates. Chemosphere. 168:pp 399–407. doi: 10.1016/j.chemosphere.2016.10.072.

Gallen C et al. (2017). Australia-wide assessment of perfluoroalkyl substances (PFASs) in landfill leachates. Journal of Hazardous Materials. 331:pp 132–141. doi: 10.1016/j.jhazmat.2017.02.006.

Gallen C et al. (2018). A mass estimate of perfluoroalkyl substance (PFAS) release from Australian wastewater treatment plants. Chemosphere. 208:pp 975–983. doi: 10.1016/j.chemosphere.2018.06.024.

Gannon SA et al. (2011). Absorption, distribution, metabolism, and excretion of [1-14C]-perfluorohexanoate ([14C]-PFHx) in rats and mice. Toxicology. 283:pp 55–62. doi: 10.1016/j.tox.2011.02.004.

Gao K et al. (2020). Occurrence and trophic transfer of per- and polyfluoroalkyl substances in an Antarctic ecosystem. Environmental Pollution. 257:pp 113383. doi: 10.1016/j.envpol.2019.113383.

Gao K et al. (2018). An integrated method for simultaneously determining 10 classes of per- and polyfluoroalkyl substances in one drop of human serum. Analytica Chimica Acta. 999:pp 76–86. doi: 10.1016/j.aca.2017.10.038.

Gebbink WA et al. (2016). Observation of emerging per- and polyfluoroalkyl substances (PFASs) in Greenland marine mammals. Chemosphere. 144:pp 2384–2391. doi: 10.1016/j.chemosphere.2015.10.116.

Gellrich V, Brunn H and Stahl T. (2013). Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in mineral water and tap water. Journal of Environmental Science and Health, Part A. 48:pp 129–135. doi: 10.1080/10934529.2013.719431.

Geueke B. (2016). FDA bans 3 perfluorinated substances | Food Packaging Forum. Available at: https://www.foodpackagingforum.org/news/fda-bans-three-perfluorinated-substances. Accessed 13 Dec 2018.

Ghisi R, Vamerali T and Manzetti S. (2019). Accumulation of perfluorinated alkyl substances (PFAS) in agricultural plants: A review. Environmental Research. 169:pp 326–341. doi: 10.1016/j.envres.2018.10.023.

Giesy JP et al. (2010). Aquatic toxicology of perfluorinated chemicals. Reviews of Environmental Contamination and Toxicology. 202:pp 1–52. doi: 10.1007/978-1-4419-1157-5_1.

Giesy JP and Kannan K. (2001). Global distribution of perfluorooctane sulfonate in wildlife. Environmental Science & Technology. 35:pp 1339–1342. doi: 10.1021/es001834k.

Glynn A et al. (2012). Perfluorinated alkyl acids in blood serum from primiparous women in Sweden: Serial sampling during pregnancy and nursing, and temporal trends 1996-2010. Environmental Science & Technology. 46:pp 9071–9079. doi: 10.1021/es301168c.

Gobelius L. (2016). Uptake of per- and polyfluoroalkyl substances by plants. Degree project. Swedish University of Agricultural Sciences. Available at: https://stud.epsilon.slu.se/9787/1/gobelius_l_161108.pdf. Accessed 16 Jul 2020.

Gobelius L, Lewis J and Ahrens L. (2017). Plant uptake of per- and polyfluoroalkyl substances at a contaminated fire training facility to evaluate the phytoremediation potential of various plant species. Environmental Science & Technology. 51:pp 12602–12610. doi: 10.1021/acs.est.7b02926.

Gomis MI et al. (2018). Comparing the toxic potency in vivo of long-chain perfluoroalkyl acids and fluorinated alternatives. Environment International. 113:pp 1–9. doi: 10.1016/j.envint.2018.01.011.

Gomis MI et al. (2015). A modeling assessment of the physicochemical properties and environmental fate of emerging and novel per- and polyfluoroalkyl substances. Science of The Total Environment. 505:pp 981–991. doi: 10.1016/j.scitotenv.2014.10.062.

González-Gaya B et al. (2014). Perfluoroalkylated substances in the global tropical and subtropical surface oceans. Environmental Science & Technology. 48:pp 13076–13084. doi: 10.1021/es503490z.

González-Gaya B et al. (2019). Vertical transport and sinks of perfluoroalkyl substances in the global open ocean. Environmental Science: Processes & Impacts. doi: 10.1039/C9EM00266A.

Goosey E and Harrad S. (2011). Perfluoroalkyl compounds in dust from Asian, Australian, European, and North American homes and UK cars, classrooms, and offices. Environment International. 37:pp 86–92. doi: 10.1016/j.envint.2010.08.001.

Gorrochategui E et al. (2014). Perfluorinated chemicals: Differential toxicity, inhibition of aromatase activity and alteration of cellular lipids in human placental cells. Toxicology and Applied Pharmacology. 277:pp 124–130. doi: 10.1016/j.taap.2014.03.012.

Government of South Australia. (2018). South Australia Environment Protection (Water Quality) Amendment Policy 2018. Available at: https://www.legislation.sa.gov.au/LZ/V/PO/2018/ENVIRONMENT%20PROTECTION%20(WATER%20QUALITY)%20AMENDMENT%20POLICY%202018_30.1.2018%20P%20521/30.1.2018%20P%20521.UN.PDF. Accessed 7 Feb 2018.

Grandjean P et al. (2016). Serum vaccine antibody concentrations in adolescents exposed to perfluorinated compounds. Environmental Health Perspectives. 125:pp 077018. doi: 10.1289/EHP275.

Grandjean P et al. (2017). Estimated exposures to perfluorinated compounds in infancy predict attenuated vaccine antibody concentrations at age 5-years. Journal of Immunotoxicology. 14:pp 188–195. doi: 10.1080/1547691X.2017.1360968.

Grandjean P et al. (2012). Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA. 307: doi: 10.1001/jama.2011.2034.

Grandjean P and Clapp R. (2014). Changing interpretation of human health risks from perfluorinated compounds. Public Health Reports. 129:pp 482–485. doi: 10.1177/003335491412900605.

Grandjean P and Clapp R. (2015). Perfluorinated alkyl substances: emerging insights into health risks. New Solutions. 25:pp 147–163. doi: 10.1177/1048291115590506.

Greaves AK et al. (2012). Tissue-specific concentrations and patterns of perfluoroalkyl carboxylates and sulfonates in East Greenland polar bears. Environmental Science & Technology. 46:pp 11575–11583. doi: 10.1021/es303400f.

Greaves AK et al. (2013). Brain region distribution and patterns of bioaccumulative perfluoroalkyl carboxylates and sulfonates in East Greenland polar bears (*Ursus maritimus*). Environmental Toxicology and Chemistry. 32:pp 713–722. doi: 10.1002/etc.2107.

Gredelj A et al. (2020). Uptake and translocation of perfluoroalkyl acids (PFAA) in red chicory (*Cichorium intybus* L.) under various treatments with pre-contaminated soil and irrigation water. Science of The Total Environment. 708:pp 134766. doi: 10.1016/j.scitotenv.2019.134766.

Guelfo JL and Adamson DT. (2018). Evaluation of a national data set for insights into sources, composition, and concentrations of per- and polyfluoroalkyl substances (PFASs) in U.S. drinking water. Environmental Pollution. 236:pp 505–513. doi: 10.1016/j.envpol.2018.01.066.

Guerra P et al. (2014). Parameters affecting the formation of perfluoroalkyl acids during wastewater treatment. Journal of Hazardous Materials. 272:pp 148–154. doi: 10.1016/j.jhazmat.2014.03.016.

Guo C et al. (2015). Distribution, source characterization and inventory of perfluoroalkyl substances in Taihu Lake, China. Chemosphere. 127:pp 201–207. doi: 10.1016/j.chemosphere.2015.01.053.

Guo R et al. (2016). Application of a comprehensive extraction technique for the determination of poly- and perfluoroalkyl substances (PFASs) in Great Lakes Region sediments. Chemosphere. 164:pp 535–546. doi: 10.1016/j.chemosphere.2016.08.126.

Guyton KZ et al. (2009). A reexamination of the PPAR-alpha activation mode of action as a basis for assessing human cancer risks of environmental contaminants. Environmental Health Perspectives. 117:pp 1664–1672. doi: 10.1289/ehp.0900758.

Hagenaars A et al. (2011). Structure–activity relationship assessment of four perfluorinated chemicals using a prolonged zebrafish early life stage test. Chemosphere. 82:pp 764–772. doi: 10.1016/j.chemosphere.2010.10.076.

Hamid H and Li L. (2016). Role of wastewater treatment plant (WWTP) in environmental cycling of poly- and perfluoroalkyl (PFAS) compounds. Ecocycles. 2: doi: 10.19040/ecocycles.v2i2.62.

Hamid H, Li LY and Grace JR. (2020). Formation of perfluorocarboxylic acids from 6:2 fluorotelomer sulfonate (6:2 FTS) in landfill leachate: Role of microbial communities. Environmental Pollution. 259:pp 113835. doi: 10.1016/j.envpol.2019.113835.

Hamid H, Li LY and Grace JR. (2018). Review of the fate and transformation of per- and polyfluoroalkyl substances (PFASs) in landfills. Environmental Pollution. 235:pp 74–84. doi: 10.1016/j.envpol.2017.12.030.

Hanssen L et al. (2019). Screening new PFAS compounds 2018. Available at: https://ntnuopen.ntnu.no/ntnu-xmlui/bitstream/handle/11250/2633056/NILU%2breport%2b23-2019.pdf?sequence=1&isAllowed=y.

Harrad S et al. (2019). Perfluoroalkyl substances in drinking water, indoor air and dust from Ireland: Implications for human exposure. Environmental Science & Technology. 53:pp 13449–13457. doi: 10.1021/acs.est.9b04604.

Hart K et al. (2008). Skipjack tuna as a bioindicator of contamination by perfluorinated compounds in the oceans. Science of The Total Environment. 403:pp 215–221. doi: 10.1016/j.scitotenv.2008.05.035.

Hartmann C et al. (2017). Perfluoroalkylated substances in human urine: Results of a biomonitoring pilot study. Biomonitoring. 4:pp 1–10. doi: 10.1515/bimo-2017-0001.

Haug LS et al. (2011). Investigation on per- and polyfluorinated compounds in paired samples of house dust and indoor air from Norwegian homes. Environmental Science & Technology. 45:pp 7991–7998. doi: 10.1021/es103456h.

Haug LS et al. (2010). Levels in food and beverages and daily intake of perfluorinated compounds in Norway. Chemosphere. 80:pp 1137–1143. doi: 10.1016/j.chemosphere.2010.06.023.

He X et al. (2018). PFOA is associated with diabetes and metabolic alteration in US men: National Health and Nutrition Examination Survey 2003–2012. Science of The Total Environment. 625:pp 566–574. doi: 10.1016/j.scitotenv.2017.12.186.

Hebert PC and MacManus-Spencer LA. (2010). Development of a fluorescence model for the binding of medium- to long-chain perfluoroalkyl acids to human serum albumin through a mechanistic evaluation of spectroscopic evidence. Analytical Chemistry. 82:pp 6463–6471. doi: 10.1021/ac100721e.

Henry BJ et al. (2018). A critical review of the application of polymer of low concern and regulatory criteria to fluoropolymers: fluoropolymers PLC. Integrated Environmental Assessment and Management. 14:pp 316–334. doi: 10.1002/ieam.4035.

Hepburn E et al. (2019). Contamination of groundwater with per- and polyfluoroalkyl substances (PFAS) from legacy landfills in an urban re-development precinct. Environmental Pollution. 248:pp 101–113. doi: 10.1016/j.envpol.2019.02.018.

Herzke D, Olsson E and Posner S. (2012). Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in consumer products in Norway – A pilot study. Chemosphere. 88:pp 980–987. doi: 10.1016/j.chemosphere.2012.03.035.

Heydebreck F et al. (2016). Emissions of per- and polyfluoroalkyl substances in a textile manufacturing plant in China and their relevance for workers' exposure. Environmental Science & Technology. 50:pp 10386–10396. doi: 10.1021/acs.est.6b03213.

Higgins CP and Luthy RG. (2006). Sorption of perfluorinated surfactants on sediments. Environmental Science & Technology. 40:pp 7251–7256. doi: 10.1021/es061000n.

Hoffman K et al. (2011). Private drinking water wells as a source of exposure to perfluorooctanoic acid (PFOA) in communities surrounding a fluoropolymer production facility. Environmental Health Perspectives. 119:pp 92–97. doi: 10.1289/ehp.1002503.

Hoke RA et al. (2012). Comparative acute freshwater hazard assessment and preliminary PNEC development for eight fluorinated acids. Chemosphere. 87:pp 725–733. doi: 10.1016/j.chemosphere.2011.12.066.

Hong S et al. (2015). Bioaccumulation characteristics of perfluoroalkyl acids (PFAAs) in coastal organisms from the west coast of South Korea. Chemosphere. 129:pp 157–163. doi: 10.1016/j.chemosphere.2014.06.023.

Hoover G et al. (2019). *In vitro* and *in silico* modeling of perfluoroalkyl substances mixture toxicity in an amphibian fibroblast cell line. Chemosphere. 233:pp 25–33. doi: 10.1016/j.chemosphere.2019.05.065.

Horst J et al. (2018). Water treatment technologies for PFAS: The next generation. Groundwater Monitoring & Remediation. 38:pp 13–23. doi: 10.1111/gwmr.12281.

Houde M et al. (2006). Biological monitoring of polyfluoroalkyl substances: A review. Environmental Science & Technology. 40:pp 3463–3473. doi: 10.1021/es052580b.

Houde M et al. (2011). Monitoring of perfluorinated compounds in aquatic biota: An updated review. Environmental Science & Technology. 45:pp 7962–7973. doi: 10.1021/es104326w.

Houde M et al. (2008). Fractionation and bioaccumulation of perfluorooctane sulfonate (PFOS) isomers in a Lake Ontario food web. Environmental Science & Technology. 42:pp 9397–9403. doi: 10.1021/es800906r.

Houde M et al. (2005). Polyfluoroalkyl compounds in free-ranging bottlenose dolphins (*Tursiops truncatus*) from the Gulf of Mexico and the Atlantic Ocean. Environmental Science & Technology. 39:pp 6591–6598. doi: 10.1021/es0506556.

Houtz EF et al. (2016). Poly- and perfluoroalkyl substances in wastewater: Significance of unknown precursors, manufacturing shifts, and likely AFFF impacts. Water Research. 95:pp 142–149. doi: 10.1016/j.watres.2016.02.055.

Houtz EF and Sedlak DL. (2012). Oxidative conversion as a means of detecting precursors to perfluoroalkyl acids in urban runoff. Environmental Science & Technology. 46:pp 9342–9349. doi: 10.1021/es302274g.

Høyer BB et al. (2015). Anthropometry in 5- to 9-year-old Greenlandic and Ukrainian children in relation to prenatal exposure to perfluorinated alkyl substances. Environmental Health Perspectives. 123:pp 841–846. doi: 10.1289/ehp.1408881.

Hu XC et al. (2018). Can profiles of poly- and perfluoroalkyl substances (PFASs) in human serum provide information on major exposure sources? Environmental Health: A Global Access Science Source. 17:pp 11. doi: 10.1186/s12940-018-0355-4.

Hu XC et al. (2016). Detection of poly- and perfluoroalkyl substances (PFASs) in U.S. drinking water linked to industrial sites, military fire training areas, and wastewater treatment plants. Environmental Science & Technology Letters. 3:pp 344–350. doi: 10.1021/acs.estlett.6b00260.

Huber S et al. (2009). Emissions from incineration of fluoropolymer materials: A literature survey. Norwegian Institue for Air Research (NILU), OR 12/2009.

Hurley S et al. (2016). Preliminary associations between the detection of perfluoroalkyl acids (PFAAs) in drinking water and serum concentrations in a sample of California women. Environmental Science & Technology Letters. 3:pp 264–269. doi: 10.1021/acs.estlett.6b00154.

Hurley S et al. (2018). Time trends in per- and polyfluoroalkyl substances (PFASs) in California women: Declining serum levels, 2011-2015. Environmental Science & Technology. 52:pp 277–287. doi: 10.1021/acs.est.7b04650.

Huset CA et al. (2011). Quantitative determination of fluorochemicals in municipal landfill leachates. Chemosphere. 82:pp 1380–1386. doi: 10.1016/j.chemosphere.2010.11.072.

Inhofe JM. (2019). 116th Congress (2019-2020): S.1790 - National Defense Authorization Act for Fiscal Year 2020. Available at: https://www.congress.gov/bill/116th-congress/senate-bill/1790/text. Accessed 6 Feb 2020.

IPCC. (2007). Intergovernmental Panel on Climate Change (IPCC): Working Group I: The physical science basis, 2.10.2 Direct global warming potentials. IPCC Fourth Assessment Report: Climate Change. Available at: https://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10-2.html.

Ishibashi H et al. (2008). Fluorotelomer alcohols induce hepatic vitellogenin through activation of the estrogen receptor in male medaka (*Oryzias latipes*). Chemosphere. 71:pp 1853–1859. doi: 10.1016/j.chemosphere.2008.01.065.

ITRC. (2020a). Interstate Technology and Regulatory Council (ITRC): PFAS Environmental Fate and Transport. Available at: https://pfas-1.itrcweb.org/5-environmental-fate-and-transport-processes/. Accessed 16 Jul 2020.

ITRC. (2020b). Interstate Technology and Regulatory Council (ITRC): Human and Ecological Health Effects of select PFAS. Available at: https://pfas-1.itrcweb.org/7-human-and-ecological-health-effects-of-select-pfas/. Accessed 11 Jul 2020.

Jahnke A et al. (2007). Urban versus remote air concentrations of fluorotelomer alcohols and other polyfluorinated alkyl substances in Germany. Environmental Science & Technology. 41:pp 745–752. doi: 10.1021/es0619861.

Jain RB. (2018). Contribution of diet and other factors to the observed levels of selected perfluoroalkyl acids in serum among US children aged 3-11 years. Environmental Research. 161:pp 268–275. doi: 10.1016/j.envres.2017.11.018.

Ji W et al. (2018). Removal of GenX and perfluorinated alkyl substances from water by amine-functionalized covalent organic frameworks. Journal of the American Chemical Society. 140:pp 12677–12681. doi: 10.1021/jacs.8b06958.

Jian JM et al. (2017). Global distribution of perfluorochemicals (PFCs) in potential human exposure source–A review. Environment International. 108:pp 51–62. doi: 10.1016/j.envint.2017.07.024.

Jin R et al. (2020). Perfluoroalkyl substances and severity of nonalcoholic fatty liver in Children: An untargeted metabolomics approach. Environment International. 134:pp 105220. doi: 10.1016/j.envint.2019.105220.

Jones PD et al. (2003). Binding of perfluorinated fatty acids to serum proteins. Environmental Toxicology and Chemistry. 22:pp 2639. doi: 10.1897/02-553.

Jouanneau W et al. (2020). Spatiotemporal Analysis of Perfluoroalkyl Substances in White-Tailed Eagle (*Haliaeetus albicilla)* Nestlings from Northern Norway—A Ten-Year Study. Environmental Science & Technology. 54:pp 5011–5020. doi: 10.1021/acs.est.9b06818.

Joudan S, Yeung LWY and Mabury SA. (2017). Biological cleavage of the C-P bond in perfluoroalkyl phosphinic acids in male Sprague-Dawley rats and the formation of persistent and reactive metabolites. Environmental Health Perspectives. 125:pp 117001. doi: 10.1289/EHP1841.

Kabadi SV et al. (2020). Characterizing biopersistence potential of the metabolite 5:3 fluorotelomer carboxylic acid after repeated oral exposure to the 6:2 fluorotelomer alcohol. Toxicology and Applied Pharmacology. 388:pp 114878. doi: 10.1016/j.taap.2020.114878.

Kabadi SV et al. (2018). Internal exposure-based pharmacokinetic evaluation of potential for biopersistence of 6:2 fluorotelomer alcohol (FTOH) and its metabolites. Food and Chemical Toxicology. 112:pp 375–382. doi: 10.1016/j.fct.2018.01.012.

Kaboré HA, Vo Duy S and Munoz G. (2018). Worldwide drinking water occurrence and levels of newly-identified perfluoroalkyl and polyfluoroalkyl substances. Science of The Total Environment. 616–617:pp 1089–1100. doi: 10.1016/j.scitotenv.2017.10.210.

Kang H et al. (2016). Elevated levels of short carbon-chain PFCAs in breast milk among Korean women: Current status and potential challenges. Environmental Research. 148:pp 351–359. doi: 10.1016/j.envres.2016.04.017.

Kannan K, Perrotta E and Thomas NJ. (2006). Association between perfluorinated compounds and pathological conditions in southern sea otters. Environmental Science & Technology. 40:pp 4943–4948. doi: 10.1021/es060932o.

Karásková P et al. (2016). Perfluorinated alkyl substances (PFASs) in household dust in Central Europe and North America. Environment International. 94:pp 315–324. doi: 10.1016/j.envint.2016.05.031.

Karnjanapiboonwong A et al. (2018). Perfluoroalkylsulfonic and carboxylic acids in earthworms (*Eisenia fetida*): Accumulation and effects results from spiked soils at PFAS concentrations bracketing environmental relevance. Chemosphere. 199:pp 168–173. doi: 10.1016/j.chemosphere.2018.02.027.

Kärrman A et al. (2007). Exposure of perfluorinated chemicals through lactation: Levels of matched human milk and serum and a temporal trend, 1996-2004, in Sweden. Environmental Health Perspectives. 115:pp 226–230. doi: 10.1289/ehp.9491.

Keller and Heckman LLP. (2020). Denmark's PFAS Ban in Paper and Cardboard Effective in July 2020. in: The National Law Review. Available at: https://www.natlawreview.com/article/denmark-s-pfas-ban-paper-and-cardboard-effective-july-2020. Accessed 14 Jul 2020.

Keller JM et al. (2005). Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States. Environmental Science & Technology. 39:pp 9101–9108. doi: 10.1021/es050690c.

Keller JM et al. (2012). Perfluoroalkyl contaminants in plasma of five sea turtle species: Comparisons in concentration and potential health risks. Environmental Toxicology and Chemistry. 31:pp 1223–1230. doi: 10.1002/etc.1818.

Kelly BC et al. (2009). Perfluoroalkyl contaminants in an Arctic marine food web: Trophic magnification and wildlife exposure. Environmental Science & Technology. 43:pp 4037–4043. doi: 10.1021/es9003894.

KEMI. (2015). Swedish Chemicals Agency (KEMI): Occurrence and use of highly fluorinated substances and alternatives. Report from a government assignment. Report 7/15. Stockhold, Sweden.

Kildee DT et al. (2017). Letter to President Donald J. Trump, August 1, 2017. Available at: https://dankildee.house.gov/sites/danielkildee.house.gov/files/8-1-17%20-%20PFC%20Contamination%20at%20Military%20Bases%20%28Trump%29.pdf. Accessed 8 Sep 2017.

Kim DH, Lee MY and Oh JE. (2014a). Perfluorinated compounds in serum and urine samples from children aged 5–13 years in South Korea. Environmental Pollution. 192:pp 171–178. doi: 10.1016/j.envpol.2014.05.024.

Kim Lazcano R et al. (2019). Per- and polyfluoroalkyl substances in commercially available biosolid-based products: The effect of treatment processes. Water Environment Research. 91:pp 1669–1677.

Kim Lazcano R et al. (2020). Characterizing and comparing per- and polyfluoroalkyl substances in commercially available biosolid and organic non-biosolid-based products. Environmental Science & Technology. doi: 10.1021/acs.est.9b07281.

Kim MH et al. (2012a). Biodefluorination and biotransformation of fluorotelomer alcohols by two alkane-degrading *Pseudomonas* strains. Biotechnology and Bioengineering. 109:pp 3041–3048. doi: 10.1002/bit.24561.

Kim MH, Wang N and Chu KH. (2014b). 6:2 Fluorotelomer alcohol (6:2 FTOH) biodegradation by multiple microbial species under different physiological conditions. Applied Microbiology and Biotechnology. 98:pp 1831–1840. doi: 10.1007/s00253-013-5131-3.

Kim S et al. (2011). Trans-placental transfer of thirteen perfluorinated compounds and relations with fetal thyroid hormones. Environmental Science & Technology. 45:pp 7465–7472. doi: 10.1021/es202408a.

Kim SK et al. (2012b). Wastewater treatment plants (WWTPs)-derived national discharge loads of perfluorinated compounds (PFCs). Journal of Hazardous Materials. 201–202:pp 82–91. doi: 10.1016/j.jhazmat.2011.11.036.

Kim SK and Kannan K. (2007). Perfluorinated acids in air, rain, snow, surface runoff, and lakes: Relative importance of pathways to contamination of urban lakes. Environmental Science & Technology. 41:pp 8328–8334. doi: 10.1021/es072107t.

Kiplinger JL, Richmond TG and Osterberg CE. (1994). Activation of carbon-fluorine bonds by metal complexes. Chemical Reviews. 94:pp 373–431. doi: 10.1021/cr00026a005.

Kirchgeorg T et al. (2016). Seasonal accumulation of persistent organic pollutants on a high altitude glacier in the Eastern Alps. Environmental Pollution. 218:pp 804–812. doi: 10.1016/j.envpol.2016.08.004.

Klepeis NE et al. (2001). The National Human Activity Pattern Survey (NHAPS): A resource for assessing exposure to environmental pollutants. Journal of Exposure Science & Environmental Epidemiology. 11:pp 231–252. doi: 10.1038/sj.jea.7500165.

Knobeloch L, Imm P and Anderson H. (2012). Perfluoroalkyl chemicals in vacuum cleaner dust from 39 Wisconsin homes. Chemosphere. 88:pp 779–783. doi: 10.1016/j.chemosphere.2012.03.082.

Knutsen H et al. (2019). Leachate emissions of short- and long-chain per- and polyfluoralkyl substances (PFASs) from various Norwegian landfills. Environmental Science: Processes & Impacts. doi: 10.1039/C9EM00170K.

Knutsen HK et al. (2018). Risk to human health related to the presence of perfluorooctane sulfonic acid and perfluorooctanoic acid in food. EFSA Journal. 16: doi: 10.2903/j.efsa.2018.5194.

Koch A et al. (2019). Point source characterization of per- and polyfluoroalkyl substances (PFASs) and extractable organofluorine (EOF) in freshwater and aquatic invertebrates. Environmental Science: Processes & Impacts. 21:pp 1887–1898. doi: 10.1039/C9EM00281B.

Konwick BJ et al. (2008). Concentrations and patterns of perfluoroalkyl acids in Georgia, USA surface waters near and distant to a major use source. Environmental Toxicology and Chemistry. 27:pp 2011. doi: 10.1897/07-659.1.

Kowalczyk J et al. (2019). Perfluoroalkyl substances (PFAS) in beaked redfish (*Sebastes mentella*) and cod (*Gadus morhua*) from arctic fishing grounds of Svalbard. Food Additives & Contaminants: Part B. pp 1–11. doi: 10.1080/19393210.2019.1690052.

Krafft MP and Riess JG. (2015a). Selected physicochemical aspects of poly- and perfluoroalkylated substances relevant to performance, environment and sustainability—Part one. Chemosphere. 129:pp 4–19. doi: 10.1016/j.chemosphere.2014.08.039.

Krafft MP and Riess JG. (2015b). Per- and polyfluorinated substances (PFASs): Environmental challenges. Current Opinion in Colloid & Interface Science. 20:pp 192–212. doi: 10.1016/j.cocis.2015.07.004.

Kumar SK et al. (2009). Perfluorinated organic contaminants in sediment and aquatic wildlife, including sharks, from Georgia, USA. Marine Pollution Bulletin. 58:pp 621–629. doi: 10.1016/j.marpolbul.2008.12.006.

Kwiatkowski CF et al. (2020). Scientific basis for managing PFAS as a chemical class. Environmental Science & Technology Letters. doi: 10.1021/acs.estlett.0c00255.

Ladics GS et al. (2008). 90-day oral gavage toxicity study of 8-2 fluorotelomer alcohol in rats. Drug and Chemical Toxicology. 31:pp 189–216. doi: 10.1080/01480540701873103.

Lam JCW et al. (2016). Perfluoroalkyl substances (PFASs) in marine mammals from the South China Sea and their temporal changes 2002-2014: Concern for alternatives of PFOS? Environmental Science & Technology. 50:pp 6728–6736. doi: 10.1021/acs.est.5b06076.

Land M et al. (2018). What is the effect of phasing out long-chain per- and polyfluoroalkyl substances on the concentrations of perfluoroalkyl acids and their precursors in the environment? A systematic review. Environmental Evidence. 7: doi: 10.1186/s13750-017-0114-y.

Lang JR et al. (2017). National estimate of per- and polyfluoroalkyl substance (PFAS) release to U.S. municipal landfill leachate. Environmental Science & Technology. 51:pp 2197–2205. doi: 10.1021/acs.est.6b05005.

Lang JR et al. (2016). Release of per- and polyfluoroalkyl substances (PFASs) from carpet and clothing in model anaerobic landfill reactors. Environmental Science & Technology. 50:pp 5024–5032. doi: 10.1021/acs.est.5b06237.

Latala A, Nędzi M and Stepnowski P. (2009). Acute toxicity assessment of perfluorinated carboxylic acids towards the Baltic microalgae. Environmental Toxicology and Pharmacology. 28:pp 167–171. doi: 10.1016/j.etap.2009.03.010.

Lechner M and Knapp H. (2011). Carryover of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) from soil to plant and distribution to the different plant compartments studied in cultures of carrots (*Daucus carota ssp. Sativus*), potatoes (*Solanum tuberosum*), and

cucumbers (*Cucumis Sativus*). Journal of Agricultural and Food Chemistry. 59:pp 11011–11018. doi: 10.1021/jf201355y.

Lee H et al. (2014). Fate of polyfluoroalkyl phosphate diesters and their metabolites in biosolids-applied soil: Biodegradation and plant uptake in greenhouse and field experiments. Environmental Science & Technology. 48:pp 340–349. doi: 10.1021/es403949z.

Lee H, D'eon J and Mabury SA. (2010). Biodegradation of polyfluoroalkyl phosphates as a source of perfluorinated acids to the environment. Environmental Science & Technology. 44:pp 3305–3310. doi: 10.1021/es9028183.

Lee JX et al. (2019). Development of screening method for targeted and undiscovered per- and polyfluoroalkyl substances (PFAS) in surface water on Q-TOF mass spectrometer. Available at: https://www.shimadzu.com/an/literature/lcms/apo119056.html.

Lee LS and Trim H. (2018a). Evaluating perfluoroalkyl acids in composts with compostable food serviceware products in their feedstocks. Summary sheet. January 2018 (revised March 9, 2018). Available at: https://zerowastewashington.org/wp-content/uploads/2018/11/one-pager_revised-march-9-2018_lslee_trim.pdf.

Lee S et al. (2018b). Perfluoroalkyl substances (PFASs) in breast milk from Korea: Time-course trends, influencing factors, and infant exposure. Science of The Total Environment. 612:pp 286–292. doi: 10.1016/j.scitotenv.2017.08.094.

Lee YJ et al. (2013). Concentrations of perfluoroalkyl compounds in maternal and umbilical cord sera and birth outcomes in Korea. Chemosphere. 90:pp 1603–1609. doi: 10.1016/j.chemosphere.2012.08.035.

Lemal DM. (2004). Perspective on fluorocarbon chemistry. The Journal of Organic Chemistry. 69:pp 1–11. doi: 10.1021/jo0302556.

Li F et al. (2020a). Short-chain per- and polyfluoroalkyl substances in aquatic systems: Occurrence, impacts and treatment. Chemical Engineering Journal. 380: doi: 10.1016/j.cej.2019.122506.

Li J et al. (2020b). Legacy per- and polyfluoroalkyl substances (PFASs) and alternatives (short-chain analogues, F-53B, GenX and FC-98) in residential soils of China: Present implications of replacing legacy PFASs. Environment International. 135:pp 105419. doi: 10.1016/j.envint.2019.105419.

Li L et al. (2017). Degradation of fluorotelomer-based polymers contributes to the global occurrence of fluorotelomer alcohol and perfluoroalkyl carboxylates: a combined dynamic substance flow and environmental fate modeling analysis. Environmental Science & Technology. 51:pp 4461–4470. doi: 10.1021/acs.est.6b04021.

Li Y et al. (2020c). Transplacental transfer of per- and polyfluoroalkyl substances identified in paired maternal and cord sera using suspect and non-target screening. Environmental Science & Technology. doi: 10.1021/acs.est.9b06505.

Li Y, Oliver DP and Kookana RS. (2018). A critical analysis of published data to discern the role of soil and sediment properties in determining sorption of per and polyfluoroalkyl substances (PFASs). Science of The Total Environment. 628–629:pp 110–120. doi: 10.1016/j.scitotenv.2018.01.167.

Li Z, Rabnawaz M and Khan B. (2020d). Response surface methodology design for biobased and sustainable coatings for water- and oil-resistant paper. ACS Applied Polymer Materials. 2:pp 1378–1387. doi: 10.1021/acsapm.9b01238.

Lieder PH et al. (2009). Toxicological evaluation of potassium perfluorobutanesulfonate in a 90-day oral gavage study with Sprague-Dawley rats. Toxicology. 255:pp 45–52. doi: 10.1016/j.tox.2008.10.002.

Lien GW et al. (2013). Neonatal–maternal factors and perfluoroalkyl substances in cord blood. Chemosphere. 92:pp 843–850. doi: 10.1016/j.chemosphere.2013.04.038.

Lilienthal H et al. (2017). Recent experimental results of effects of perfluoroalkyl substances in laboratory animals – Relation to current regulations and guidance values. International Journal of Hygiene and Environmental Health. 220:pp 766–775. doi: 10.1016/j.ijheh.2017.03.001.

Lin AY. et al. (2014a). Occurrence of perfluorinated compounds in the aquatic environment as found in science park effluent, river water, rainwater, sediments, and biotissues. Environmental Monitoring and Assessment. 186:pp 3265–3275. doi: 10.1007/s10661-014-3617-9.

Lin J et al. (2014b). China's international trade and air pollution in the United States. Proceedings of the National Academy of Sciences. 111:pp 1736–1741. doi: 10.1073/pnas.1312860111.

Lindstrom AB et al. (2011a). Application of WWTP biosolids and resulting perfluorinated compound contamination of surface and well water in Decatur, Alabama, USA. Environmental Science & Technology. 45:pp 8015–8021. doi: 10.1021/es1039425.

Lindstrom AB, Strynar MJ and Libelo EL. (2011b). Polyfluorinated compounds: Past, present, and future. Environmental Science & Technology. 45:pp 7954–7961. doi: 10.1021/es2011622.

Liu B et al. (2015a). Perfluorinated compounds (PFCs) in the atmosphere of Shenzhen, China: Spatial distribution, sources and health risk assessment. Chemosphere. 138:pp 511–518. doi: 10.1016/j.chemosphere.2015.07.012.

Liu C et al. (2011a). Novel perspectives on the bioaccumulation of PFCs – The concentration dependency. Environmental Science & Technology. 45:pp 9758–9764. doi: 10.1021/es202078n.

Liu C et al. (2009). Waterborne exposure to fluorotelomer alcohol 6:2 FTOH alters plasma sex hormone and gene transcription in the hypothalamic–pituitary–gonadal (HPG) axis of zebrafish. Aquatic Toxicology. 93:pp 131–137. doi: 10.1016/j.aquatox.2009.04.005.

Liu H et al. (2018a). Prenatal exposure to perfluoroalkyl and polyfluoroalkyl substances affects leukocyte telomere length in female newborns. Environmental Pollution. 235:pp 446–452. doi: 10.1016/j.envpol.2017.12.095.

Liu H et al. (2015b). Toxic effects of perfluorononanoic acid on the development of Zebrafish (*Danio rerio*) embryos. Journal of Environmental Sciences. 32:pp 26–34. doi: 10.1016/j.jes.2014.11.008.

Liu J et al. (2010a). Aerobic biodegradation of [14C] 6:2 fluorotelomer alcohol in a flow-through soil incubation system. Chemosphere. 80:pp 716–723. doi: 10.1016/j.chemosphere.2010.05.027.

Liu J et al. (2010b). 6-2 Fluorotelomer alcohol aerobic biodegradation in soil and mixed bacterial culture. Chemosphere. 78:pp 437–444. doi: 10.1016/j.chemosphere.2009.10.044.

Liu J and Mejia Avendaño S. (2013a). Microbial degradation of polyfluoroalkyl chemicals in the environment: A review. Environment International. 61:pp 98–114. doi: 10.1016/j.envint.2013.08.022.

Liu W et al. (2013b). Polyfluorinated telomers in indoor air of Japanese houses. Chemosphere. 90:pp 1672–1677. doi: 10.1016/j.chemosphere.2012.09.062.

Liu W et al. (2011b). Analysis of perfluoroalkyl carboxylates in vacuum cleaner dust samples in Japan. Chemosphere. 85:pp 1734–1741. doi: 10.1016/j.chemosphere.2011.09.024.

Liu X et al. (2014). Concentrations and trends of perfluorinated chemicals in potential indoor sources from 2007 through 2011 in the US. Chemosphere. 98:pp 51–57. doi: 10.1016/j.chemosphere.2013.10.001.

Liu Y et al. (2018b). Hundreds of unrecognized halogenated contaminants discovered in polar bear serum. Angewandte Chemie International Edition. 57:pp 16401–16406. doi: 10.1002/anie.201809906.

Liu Y et al. (2017). Chlorinated polyfluoroalkyl ether sulfonic acids in marine organisms from Bohai Sea, China: Occurrence, temporal variations, and trophic transfer behavior. Environmental Science & Technology. 51:pp 4407–4414. doi: 10.1021/acs.est.6b06593.

Liu Z et al. (2016). Risk assessment and source identification of perfluoroalkyl acids in surface and ground water: spatial distribution around a mega-fluorochemical industrial park, China. Environment International. 91:pp 69–77. doi: 10.1016/j.envint.2016.02.020.

Llorca M et al. (2010). Infant exposure of perfluorinated compounds: Levels in breast milk and commercial baby food. Environment International. 36:pp 584–592. doi: 10.1016/j.envint.2010.04.016.

Llorca M et al. (2012). Fate of a broad spectrum of perfluorinated compounds in soils and biota from Tierra del Fuego and Antarctica. Environmental Pollution. 163:pp 158–166. doi: 10.1016/j.envpol.2011.10.027.

Long M, Bossi R and Bonefeld-Jørgensen EC. (2012). Level and temporal trend of perfluoroalkyl acids in Greenlandic Inuit. International Journal of Circumpolar Health. 71:pp 17998. doi: 10.3402/ijch.v71i0.17998.

Longsworth SG. (2020). Processes & considerations for setting state PFAS standards. Available at: https://www.ecos.org/wp-content/uploads/2020/02/Standards-White-Paper-February-2020-FINAL-for-Web.pdf. Accessed 28 Feb 2020.

Loveless SE et al. (2009). Toxicological evaluation of sodium perfluorohexanoate. Toxicology. 264:pp 32–44. doi: 10.1016/j.tox.2009.07.011.

Lu D et al. (2020). Treatment train approaches for the remediation of per- and polyfluoroalkyl substances (PFAS): A critical review. Journal of Hazardous Materials. 386:pp 121963. doi: 10.1016/j.jhazmat.2019.121963.

Lynch KM et al. (2019). Temporal trends in per- and polyfluoroalkyl substances in Bottlenose Dolphins (*Tursiops truncatus*) of Indian River Lagoon, Florida and Charleston, South Carolina. Environmental Science & Technology. 53:pp 14194–14203. doi: 10.1021/acs.est.9b04585.

Mackay D et al. (2014). The role of persistence in chemical evaluations. Integrated Environmental Assessment and Management. 10:pp 588–594. doi: 10.1002/ieam.1545.

Mackay D and Fraser A. (2000). Bioaccumulation of persistent organic chemicals: Mechanisms and models. Environmental Pollution. 110:pp 375–391. doi: 10.1016/S0269-7491(00)00162-7.

MacLeod M et al. (2014). Identifying chemicals that are planetary boundary threats. Environmental Science & Technology. 48:pp 11057–11063. doi: 10.1021/es501893m.

Makey CM et al. (2017). Airborne precursors predict maternal serum perfluoroalkyl acid concentrations. Environmental Science & Technology. 51:pp 7667–7675. doi: 10.1021/acs.est.7b00615.

Malits J et al. (2017). Perfluorooctanoic acid and low birth weight: Estimates of US attributable burden and economic costs from 2003 through 2014. International Journal of Hygiene and Environmental Health. 221:pp 269–275. doi: 10.1016/j.ijheh.2017.11.004.

Mamsen LS et al. (2019). Concentrations of perfluoroalkyl substances (PFASs) in human embryonic and fetal organs from first, second, and third trimester pregnancies. Environment International. 124:pp 482–492. doi: 10.1016/j.envint.2019.01.010.

Mamsen LS et al. (2017). Concentration of perfluorinated compounds and cotinine in human foetal organs, placenta, and maternal plasma. Science of The Total Environment. 596–597:pp 97–105. doi: 10.1016/j.scitotenv.2017.04.058.

Martín J et al. (2019). Exposure assessment to parabens, bisphenol A and perfluoroalkyl compounds in children, women and men by hair analysis. Science of The Total Environment. 695:pp 133864. doi: 10.1016/j.scitotenv.2019.133864.

Martin JW et al. (2004). Identification of long-chain perfluorinated acids in biota from the Canadian Arctic. Environmental Science & Technology. 38:pp 373–380.

Martin JW et al. (2003). Bioconcentration and tissue distribution of perfluorinated acids in rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry. 22:pp 196–204.

Mason Chemical Company. (2017). Specialty Surfactants: Fluorosurfactant - Structure/Function. Available at: http://archive.li/GthbK. Accessed 20 Sep 2019.

McLachlan MS et al. (2019). Fate of a perfluoroalkyl acid mixture in an agricultural soil studied in lysimeters. Chemosphere. 223:pp 180–187. doi: 10.1016/j.chemosphere.2019.02.012.

Mejia-Avendaño S et al. (2020). Sorption of polyfluoroalkyl surfactants on surface soils: Effect of molecular structures, soil properties, and solution chemistry. Environmental Science & Technology. 54:pp 1513–1521. doi: 10.1021/acs.est.9b04989.

Mercier F et al. (2011). Organic contamination of settled house dust, a review for exposure assessment purposes. Environmental Science & Technology. 45:pp 6716–6727. doi: 10.1021/es200925h.

Midasch O et al. (2007). Transplacental exposure of neonates to perfluorooctanesulfonate and perfluorooctanoate: A pilot study. International Archives of Occupational and Environmental Health. 80:pp 643–648. doi: 10.1007/s00420-006-0165-9.

Mitro SD et al. (2016). Consumer product chemicals in indoor dust: A quantitative meta-analysis of U.S. studies. Environmental Science & Technology. 50:pp 10661–10672. doi: 10.1021/acs.est.6b02023.

Mogensen UB et al. (2015). Breastfeeding as an exposure pathway for perfluorinated alkylates. Environmental Science & Technology. 49:pp 10466–10473. doi: 10.1021/acs.est.5b02237.

Mondal D et al. (2014). Breastfeeding: A potential excretion route for mothers and implications for infant exposure to perfluoroalkyl acids. Environmental Health Perspectives. 122:pp 187–192. doi: 10.1289/ehp.1306613.

Monroy R et al. (2008). Serum levels of perfluoroalkyl compounds in human maternal and umbilical cord blood samples. Environmental Research. 108:pp 56–62. doi: 10.1016/j.envres.2008.06.001.

Mora C and Sale P. (2011). Ongoing global biodiversity loss and the need to move beyond protected areas: A review of the technical and practical shortcomings of protected areas on land and sea. Marine Ecology Progress Series. 434:pp 251–266. doi: 10.3354/meps09214.

Moschet C et al. (2018). Household dust as a repository of chemical accumulation: New insights from a comprehensive high-resolution mass spectrometric study. Environmental Science & Technology. 52:pp 2878–2887. doi: 10.1021/acs.est.7b05767.

Mudumbi JBN et al. (2017). Recent developments in polyfluoroalkyl compounds research: A focus on human/environmental health impact, suggested substitutes and removal strategies. Environmental Monitoring and Assessment. 189: doi: 10.1007/s10661-017-6084-2.

Muir D et al. (2019). Levels and trends of poly- and perfluoroalkyl substances in the Arctic environment – An update. Emerging Contaminants. 5:pp 240–271. doi: 10.1016/j.emcon.2019.06.002.

Müller CE et al. (2011). Biomagnification of perfluorinated compounds in a remote terrestrial food chain: Lichen-caribou-wolf. Environmental Science & Technology. 45:pp 8665–8673. doi: 10.1021/es201353v.

Müller CE et al. (2012). Atmospheric fate of poly- and perfluorinated alkyl substances (PFASs): I. Day–night patterns of air concentrations in summer in Zurich, Switzerland. Environmental Pollution. 169:pp 196–203. doi: 10.1016/j.envpol.2012.04.010.

Muncke J. (2012). Food packaging materials. Available at: https://www.foodpackagingforum.org/food-packaging-health/food-packaging-materials. Accessed 18 Mar 2020.

Muncke J et al. (2020). Impacts of food contact chemicals on human health: a consensus statement. Environmental Health. 19: doi: 10.1186/s12940-020-0572-5.

Munoz G et al. (2015). Spatial distribution and partitioning behavior of selected poly- and perfluoroalkyl substances in freshwater ecosystems: A French nationwide survey. Science of The Total Environment. 517:pp 48–56. doi: 10.1016/j.scitotenv.2015.02.043.

Munoz G et al. (2020). Bioaccumulation of zwitterionic polyfluoroalkyl substances in earthworms exposed to aqueous film-forming foam impacted soils. Environmental Science & Technology. 54:pp 1687–1697. doi: 10.1021/acs.est.9b05102.

Munoz G et al. (2019). Temporal variations of perfluoroalkyl substances partitioning between surface water, suspended sediment, and biota in a macrotidal estuary. Chemosphere. 233:pp 319–326. doi: 10.1016/j.chemosphere.2019.05.281.

Nakata H et al. (2006). Perfluorinated contaminants in sediments and aquatic organisms collected from shallow water and tidal flat areas of the Ariake Sea, Japan: Environmental fate of perfluorooctane sulfonate in aquatic ecosystems. Environmental Science & Technology. 40:pp 4916–4921. doi: 10.1021/es0603195.

Nakayama SF et al. (2019). Worldwide trends in tracing poly- and perfluoroalkyl substances (PFAS) in the environment. TrAC Trends in Analytical Chemistry. doi: 10.1016/j.trac.2019.02.011.

NAS. (2017). The National Academies of Sciences, Engineering, and Medicine (NAS): Using 21st century science to improve risk-related evaluations. Available at: https://www.toxicologia.org.ar/wp-content/uploads/2017/02/Risk-Book-2017.pdf.

Navarro I et al. (2017). Uptake of perfluoroalkyl substances and halogenated flame retardants by crop plants grown in biosolids-amended soils. Environmental Research. 152:pp 199–206. doi: 10.1016/j.envres.2016.10.018.

Nelter T. (2018). FDA-approved PFAS and drinking water – Q & A on analytical measurements. Available at: http://blogs.edf.org/health/2018/06/29/pfas-and-drinking-water-analytical-measurements/. Accessed 11 Dec 2018.

Neltner T. (2019). Chemours asks FDA to suspend its approved uses of PFAS in food packaging. Available at: http://blogs.edf.org/health/2019/08/09/chemours-suspend-fda-approved-pfas-food-packaging/?utm_source=mailchimp&utm_campaign=edf-health_none_upd_dmt&utm_medium=email&utm_id=1535389555. Accessed 12 Aug 2019.

Neltner TG et al. (2011). Navigating the U.S. Food Additive Regulatory Program. Comprehensive Reviews in Food Science and Food Safety. 10:pp 342–368. doi: 10.1111/j.1541-4337.2011.00166.x.

Ng A et al. (2019). Fate of environmental pollutants. Water Environment Research. doi: 10.1002/wer.1225.

NGWA. (2017). National Ground Water Association (NGWA): Groundwater and PFAS: State of Knowledge and Practice. Section 4: Fate and Transport. Available at: https://my.ngwa.org/NC__Product?id=a183800000kbKF9AAM.

NHDHHS. (2016). New Hampshire Department of Health and Human Services (NHDHHS): Pease PFC Blood Testing Program: April 2015 – October 2015. Available at: https://www.dhhs.nh.gov/dphs/documents/pease-pfc-blood-testing.pdf.

Nielsen CJ. (2014). University of Oslo: Potential PFOA Precursors. Literature study and theoretical assessment of abiotic degradation pathways leading to PFOA. M-231/2014. Available at: https://www.miljodirektoratet.no/globalassets/publikasjoner/M231/M231.pdf.

Nilsson H et al. (2010). Inhalation exposure to fluorotelomer alcohols yield perfluorocarboxylates in human blood? Environmental Science & Technology. 44:pp 7717–7722. doi: 10.1021/es101951t.

Nilsson H et al. (2013). Biotransformation of fluorotelomer compound to perfluorocarboxylates in humans. Environment International. 51:pp 8–12. doi: 10.1016/j.envint.2012.09.001.

NILU. (2015). Norwegian Institute for Air Research (NILU): PFASs in house dust. Available at: https://www.miljodirektoratet.no/globalassets/publikasjoner/M430/M430.pdf.

Nolan J. (2019). Ahold Delhaize USA brands announce commitment to sustainable chemistry, transparent products and packaging. Available at: https://saferchemicals.org/2019/09/19/grocer-ahold-delhaize-to-restrict-toxic-chemicals-in-food-packaging-and-beauty-products/. Accessed 19 Feb 2020.

Norden M, Berger U and Engwall M. (2016). Developmental toxicity of PFOS and PFOA in great cormorant (*Phalacrocorax carbo sinensis*), herring gull (*Larus argentatus*) and chicken (*Gallus gallus domesticus*). Environmental Science and Pollution Research. 23:pp 10855–10862. doi: 10.1007/s11356-016-6285-1.

Nyberg E et al. (2018). Inter-individual, inter-city, and temporal trends of per- and polyfluoroalkyl substances in human milk from Swedish mothers between 1972 and 2016. Environmental Science: Processes & Impacts. 20:pp 1136–1147. doi: 10.1039/C8EM00174J.

NYU Law. (2020). PFAS Federal Legislation. Available at: https://www.law.nyu.edu/centers/state-impact/press-publications/research-reports/pfas-federal-legislation. Accessed 19 Feb 2020.

O'Connell SG et al. (2010). Temporal and spatial trends of perfluorinated compounds in juvenile loggerhead sea turtles (*Caretta caretta*) along the East Coast of the United States. Environmental Science & Technology. 44:pp 5202–5209. doi: 10.1021/es9036447.

O'Connor JC et al. (2014). Evaluation of the reproductive and developmental toxicity of 6:2 fluorotelomer alcohol in rats. Toxicology. 317:pp 6–16. doi: 10.1016/j.tox.2014.01.002.

OECD. (2018). Organisation for Economic Cooperation and Development (OECD): Toward a new comprehensive global database of per- and polyfluoroalkyl substances (PFASs): Summary report on updating the OECD 2007 list of per- and polyfluoroalkyl substances (PFASs). Available at: https://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV-JM-MONO(2018)7&doclanguage=en.

OECD. (2013). Organisation for Economic Cooperation and Development (OECD): Synthesis paper on per- and polyfluorinated chemicals (PFCs). Available at: https://www.oecd.org/env/ehs/risk-management/PFC_FINAL-Web.pdf.

OEHHA. (2012). Office of Environmental Health Hazard Assessment (OEHHA): Amended Final Statement of Reasons, Division 4.5, Title 22, Cal. Code of Regulations, Chapter 54, Green

Chemistry Hazard Traits. Available at: https://oehha.ca.gov/media/downloads/risk-assessment/gcfsor011912.pdf. Accessed 11 May 2020.

OEHHA. (2018). Office of Environmental Health Hazard Assessment (OEHHA): Recommendation for interim notification levels for perfluorooctanoic acid (PFOA) and perfluorooctone sulfornate (PFOS). A memorandum to the Deputy Director, Division of Drinking Water, California State Water Resources Control Board, June 26, 2018. Available at: https://www.waterboards.ca.gov/drinking_water/certlic/drinkingwater/documents/pfos_and_pfoa/OEHHA_Recommended_Int_NL_Jun_26_2018.pdf.

OEHHA. (2019). Office of Environmental Health Hazard Assessment (OEHHA): Notification level recommendations for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS). Available at: https://oehha.ca.gov/water/notification-level/notification-level-recommendations-perfluorooctanoic-acid-pfoa. Accessed 29 Oct 2019.

Office of Congressman Dan Kildee. (2019). Repres. Kildee, Fitzpatrick launch congressional PFAS task force. January 23, 2019 Press Release. Available at: https://dankildee.house.gov/media/press-releases/reps-kildee-fitzpatrick-launch-congressional-pfas-task-force. Accessed 4 Mar 2019.

Ohmori K et al. (2003). Comparison of the toxicokinetics between perfluorocarboxylic acids with different carbon chain length. Toxicology. 184:pp 135–140. doi: 10.1016/S0300-483X(02)00573-5.

Ojo AF, Peng C and Ng JC. (2020). Combined effects and toxicological interactions of perfluoroalkyl and polyfluoroalkyl substances mixtures in human liver cells (HepG2). Environmental Pollution (Barking, Essex : 1987). 263:pp 114182. doi: 10.1016/j.envpol.2020.114182.

Olsen GW et al. (2007a). Preliminary evidence of a decline in perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA) concentrations in American Red Cross blood donors. Chemosphere. 68:pp 105–111. doi: 10.1016/j.chemosphere.2006.12.031.

Olsen GW et al. (2007b). Half-life of serum elimination of perfluorooctanesulfonate, perfluorohexanesulfonate, and perfluorooctanoate in retired fluorochemical production workers. Environmental Health Perspectives. 115:pp 1298–1305. doi: 10.1289/ehp.10009.

Oregon Composters. (2019). A message from composters serving Oregon: Why we don’t want compostable packaging and serviceware. Available at: https://www.thompsonsanitary.com/wp-content/uploads/2019/04/message_from_oregon_composters_on_no_serviceware.pdf. Accessed 27 Mar 2020.

Ostertag SK et al. (2009). Historic dietary exposure to perfluorooctane sulfonate, perfluorinated carboxylates, and fluorotelomer unsaturated carboxylates from the consumption of store-bought and restaurant foods for the Canadian population. Journal of Agricultural and Food Chemistry. 57:pp 8534–8544. doi: 10.1021/jf9014125.

Packaging Europe. (2020). Zanders launches “100% natural” barrier paper for food packaging. Available at: https://packagingeurope.com/api/content/1bcd9e56-4be0-11ea-8c24-1244d5f7c7c6/. Accessed 10 Mar 2020.

Pan CG et al. (2015). Spatial distribution of perfluoroalkyl substances in surface sediments of five major rivers in China. Archives of Environmental Contamination and Toxicology. 68:pp 566–576. doi: 10.1007/s00244-014-0113-8.

Pan CG et al. (2014a). Contamination profiles of perfluoroalkyl substances in five typical rivers of the Pearl River Delta region, South China. Chemosphere. 114:pp 16–25. doi: 10.1016/j.chemosphere.2014.04.005.

Pan CG et al. (2014b). Spatiotemporal distribution and mass loadings of perfluoroalkyl substances in the Yangtze River of China. Science of The Total Environment. 493:pp 580–587. doi: 10.1016/j.scitotenv.2014.06.033.

Pan Y et al. (2018). Worldwide distribution of novel perfluoroether carboxylic and sulfonic acids in surface water. Environmental Science & Technology. 52:pp 7621–7629. doi: 10.1021/acs.est.8b00829.

Pan Y et al. (2017). Novel chlorinated polyfluorinated ether sulfonates and legacy per-/polyfluoroalkyl substances: placental transfer and relationship with serum albumin and glomerular filtration rate. Environmental Science & Technology. 51:pp 634–644. doi: 10.1021/acs.est.6b04590.

Papadopoulou E et al. (2016). Exposure of Norwegian toddlers to perfluoroalkyl substances (PFAS): The association with breastfeeding and maternal PFAS concentrations. Environment International. 94:pp 687–694. doi: 10.1016/j.envint.2016.07.006.

Papadopoulou E et al. (2019). Diet as a source of exposure to environmental contaminants for pregnant women and children from six European countries. Environmental Health Perspectives. 127:pp 107005. doi: 10.1289/EHP5324.

Pérez F et al. (2013). Accumulation of perfluoroalkyl substances in human tissues. Environment International. 59:pp 354–362. doi: 10.1016/j.envint.2013.06.004.

Pérez F et al. (2012). Automated analysis of perfluorinated compounds in human hair and urine samples by turbulent flow chromatography coupled to tandem mass spectrometry. Analytical and Bioanalytical Chemistry. 402:pp 2369–2378. doi: 10.1007/s00216-011-5660-5.

Pérez F et al. (2014). Assessment of perfluoroalkyl substances in food items at global scale. Environmental Research. 135:pp 181–189. doi: 10.1016/j.envres.2014.08.004.

Phillips MM et al. (2007). Fluorotelomer acids are more toxic than perfluorinated acids. Environmental Science & Technology. 41:pp 7159–7163. doi: 10.1021/es070734c.

Pizzurro DM et al. (2019). Interspecies differences in perfluoroalkyl substances (PFAS) toxicokinetics and application to health-based criteria. Regulatory Toxicology and Pharmacology. 106:pp 239–250. doi: 10.1016/j.yrtph.2019.05.008.

Plumlee MH, Larabee J and Reinhard M. (2008). Perfluorochemicals in water reuse. Chemosphere. 72:pp 1541–1547. doi: 10.1016/j.chemosphere.2008.04.057.

Poothong S et al. (2020). Multiple pathways of human exposure to poly- and perfluoroalkyl substances (PFASs): From external exposure to human blood. Environment International. 134:pp 105244. doi: 10.1016/j.envint.2019.105244.

Poothong S et al. (2017). Distribution of novel and well-known poly- and perfluoroalkyl substances (PFASs) in human serum, plasma, and whole blood. Environmental Science & Technology. 51:pp 13388–13396. doi: 10.1021/acs.est.7b03299.

POPRC. (2020). Persistent Organic Pollutants Review Committee (POPRC): POPRC recommendations for listing chemicals. Available at: http://chm.pops.int/tabid/243/Default.aspx. Accessed 11 May 2020.

Powley CR et al. (2008). Polyfluorinated chemicals in a spatially and temporally integrated food web in the Western Arctic. Chemosphere. 70:pp 664–672. doi: 10.1016/j.chemosphere.2007.06.067.

Prevedouros K et al. (2006). Sources, fate and transport of perfluorocarboxylates. Environmental Science & Technology. 40:pp 32–44. doi: 10.1021/es0512475.

Qi Y et al. (2016). Spatial distribution and source apportionment of PFASs in surface sediments from five lake regions, China. Scientific Reports. 6:pp 22674. doi: 10.1038/srep22674.

Rae CJM et al. (2015). Evaluation of chronic toxicity and carcinogenicity of ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate in Sprague–Dawley rats. Toxicology Reports. 2:pp 939–949. doi: 10.1016/j.toxrep.2015.06.001.

Rahman MF, Peldszus S and Anderson WB. (2014). Behaviour and fate of perfluoroalkyl and polyfluoroalkyl substances (PFASs) in drinking water treatment: A review. Water Research. 50:pp 318–340. doi: 10.1016/j.watres.2013.10.045.

Rainieri S et al. (2017). Toxic effects of perfluorinated compounds at human cellular level and on a model vertebrate. Food and Chemical Toxicology. 104:pp 14–25. doi: 10.1016/j.fct.2017.02.041.

Ramhøj L et al. (2018). Perfluorohexane sulfonate (PFHxS) and a mixture of endocrine disrupters reduce thyroxine levels and cause anti-androgenic effects in rats. Toxicological Sciences. 163:pp 579–591. doi: 10.1093/toxsci/kfy055.

Rand AA et al. (2014a). Cellular toxicity associated with exposure to perfluorinated carboxylates (PFCAs) and their metabolic precursors. Chemical Research in Toxicology. 27:pp 42–50. doi: 10.1021/tx400317p.

Rand AA and Mabury SA. (2012a). In vitro interactions of biological nucleophiles with fluorotelomer unsaturated acids and aldehydes: Fate and consequences. Environmental Science & Technology. 46:pp 7398–7406. doi: 10.1021/es3008485.

Rand AA and Mabury SA. (2012b). Assessing the structure–activity relationships of fluorotelomer unsaturated acids and aldehydes with glutathione: Reactivity of glutathione with fluorotelomer unsaturated acids and aldehydes. Cell Biology and Toxicology. 28:pp 115–124. doi: 10.1007/s10565-012-9211-4.

Rand AA and Mabury SA. (2013). Covalent binding of fluorotelomer unsaturated aldehydes (FTUALs) and carboxylic acids (FTUCAs) to proteins. Environmental Science & Technology. pp 130109073401005. doi: 10.1021/es303760u.

Rand AA and Mabury SA. (2014b). Protein binding associated with exposure to fluorotelomer alcohols (FTOHs) and polyfluoroalkyl phosphate esters (PAPs) in rats. Environmental Science & Technology. pp 140205115442006. doi: 10.1021/es404390x.

Rand AA and Mabury SA. (2017). Is there a human health risk associated with indirect exposure to perfluoroalkyl carboxylates (PFCAs)? Toxicology. 375:pp 28–36. doi: 10.1016/j.tox.2016.11.011.

Rankin K et al. (2016). A North American and global survey of perfluoroalkyl substances in surface soils: Distribution patterns and mode of occurrence. Chemosphere. 161:pp 333–341. doi: 10.1016/j.chemosphere.2016.06.109.

Rappazzo K, Coffman E and Hines E. (2017). Exposure to perfluorinated alkyl substances and health outcomes in children: A systematic review of the epidemiologic literature. International Journal of Environmental Research and Public Health. 14:pp 691. doi: 10.3390/ijerph14070691.

Rice PA et al. (2020). Comparative analysis of the toxicological databases for 6:2 fluorotelomer alcohol (6:2 FTOH) and perfluorohexanoic acid (PFHxA). Food and Chemical Toxicology. 138:pp 111210. doi: 10.1016/j.fct.2020.111210.

Rice PA. (2015). C6-perfluorinated compounds: The new greaseproofing agents in food packaging. Current Environmental Health Reports. 2:pp 33–40. doi: 10.1007/s40572-014-0039-3.

RIVM. (2019). National Institute for Public Health and the Environment: Dutch National Institute for Public Health and the Environment (RIVM): Per- and polyfluoroalkyl substances (PFASs) in food contact materials. Available at: https://rivm.openrepository.com/bitstream/handle/10029/623023/2018-0181.pdf?sequence=1&isAllowed=y.

RIVM. (2016). Dutch National Institute for Public Health and the Environment (RIVM): Evaluation of substances used in the GenX technology by Chemours. Dordrecht. Available at: https://www.rivm.nl/bibliotheek/rapporten/2016-0174.pdf.

RIVM. (2018). Dutch National Institute for Public Health and the Environment (RIVM): Mixture exposure to PFAS: A Relative Potency Factor approach. Available at: http://rivm.openrepository.com/rivm/handle/10029/622164.

Rodea-Palomares I et al. (2012). Toxicological interactions of perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) with selected pollutants. Journal of Hazardous Materials. 201–202:pp 209–218. doi: 10.1016/j.jhazmat.2011.11.061.

Rodriguez-Jorquera IA et al. (2016). Footprints of urban micro-pollution in protected areas: Investigating the longitudinal distribution of perfluoroalkyl acids in wildlife preserves. Plos One. 11:pp e0148654. doi: 10.1371/journal.pone.0148654.

Roscales JL et al. (2019). Spatial and interspecies heterogeneity in concentrations of perfluoroalkyl substances (PFASs) in seabirds of the Southern Ocean. Environmental Science & Technology. 53:pp 9855–9865. doi: 10.1021/acs.est.9b02677.

Rosenmai AK et al. (2016). Fluorinated alkyl substances and technical mixtures used in food paper-packaging exhibit endocrine-related activity in vitro. Andrology. 4:pp 662–672. doi: 10.1111/andr.12190.

Rosenmai AK et al. (2018). Relationship between peroxisome proliferator-activated receptor alpha activity and cellular concentration of 14 perfluoroalkyl substances in HepG2 cells. Journal of Applied Toxicology. 38:pp 219–226. doi: 10.1002/jat.3515.

Routti H et al. (2016). Spatial and temporal trends in perfluoroalkyl substances (PFASs) in ringed seals (*Pusa hispida*) from Svalbard. Environmental Pollution. 214:pp 230–238. doi: 10.1016/j.envpol.2016.04.016.

Royer LA et al. (2015). Microbial transformation of 8:2 fluorotelomer acrylate and methacrylate in aerobic soils. Chemosphere. 129:pp 54–61. doi: 10.1016/j.chemosphere.2014.09.077.

Ruan T et al. (2014). 6:2 Fluorotelomer iodide in vitro metabolism by rat liver microsomes: Comparison with [1,2-14C] 6:2 fluorotelomer alcohol. Chemosphere. 112:pp 34–41. doi: 10.1016/j.chemosphere.2014.02.068.

Ruan Y et al. (2019). Assessing exposure to legacy and emerging per- and polyfluoroalkyl substances via hair – The first nationwide survey in India. Chemosphere. 229:pp 366–373. doi: 10.1016/j.chemosphere.2019.04.195.

Russell MH et al. (2008). Investigation of the biodegradation potential of a fluoroacrylate polymer product in aerobic soils. Environmental Science & Technology. 42:pp 800–807. doi: 10.1021/es0710499.

Russell MH et al. (2010). Evaluation of PFO formation from the biodegradation of a fluorotelomer-based urethane polymer product in aerobic soils. Polymer Degradation and Stability. 95:pp 79–85. doi: 10.1016/j.polymdegradstab.2009.10.004.

Russell MH, Himmelstein MW and Buck RC. (2015). Inhalation and oral toxicokinetics of 6:2 FTOH and its metabolites in mammals. Chemosphere. 120:pp 328–335. doi: 10.1016/j.chemosphere.2014.07.092.

Sadia M, Yeung LWY and Fiedler H. (2019). Trace level analyses of selected perfluoroalkyl acids in food: Method development and data generation. Environmental Pollution. pp 113721. doi: 10.1016/j.envpol.2019.113721.

Safer Chemicals Healthy Families. (2019). Toxic food packaging at the grocery store. Available at: https://saferchemicals.org/2019/05/13/toxic-food-packaging-at-the-grocery-store/. Accessed 11 Mar 2020.

Safer Chemicals Healthy Families, Toxic-Free Future and Mind the Store. (2018). Take Out Toxics. PFAS Chemicals in Food Packaging. Available at: www.toxicfreefuture.org/take-out-toxics.

Safer States. (2020). Bill Tracker: PFAS. in: Safer States. Available at: http://www.saferstates.org/toxic-chemicals/pfas/. Accessed 4 Mar 2020.

SAICM. (2019). Strategic Approach to International Chemicals Management (SAICM): Criteria for elevation of obligations to progress SAICM Issues of Concern (IoCs) in the post 2020 multilateral regime: The case of per and poly fluoroalkyl substances (PFASs). Available at: http://www.saicm.org/Portals/12/documents/meetings/IP3/stakeholders/IPEN-Case-PFAS.pdf.

Sammut G et al. (2017). Perfluoroalkyl substances in the Maltese environment – (I) Surface water and rain water. Science of The Total Environment. 589:pp 182–190. doi: 10.1016/j.scitotenv.2017.02.128.

San Francisco. (2018). Resolution in Support of the Single-Use Food Ware Plastics, Toxics, and Litter Reduction Ordinance. in: Sfenvironment.Org - Our Home. Our City. Our Planet. Available at: https://sfenvironment.org/policy/resolution-in-support-of-the-singleuse-food-ware-plastics-toxics-and-litter-reduction-ordinance. Accessed 13 Dec 2018.

Sanchez D et al. (2015). Transcriptional and cellular responses of the green alga *Chlamydomonas reinhardtii* to perfluoroalkyl phosphonic acids. Aquatic Toxicology. 160:pp 31–38. doi: 10.1016/j.aquatox.2014.12.002.

Sanchez-Vidal A et al. (2015). Delivery of unprecedented amounts of perfluoroalkyl substances towards the deep-sea. Science of The Total Environment. 526:pp 41–48. doi: 10.1016/j.scitotenv.2015.04.080.

Sanders B. (2020). S. 3227 - Prevent Future American Sickness Act of 2020. Available at: https://www.congress.gov/bill/116th-congress/senate-bill/3227/all-info.

Schade M and Belliveau M. (2020). More Major Retailers Are Saying 'Forever Chemicals No More.' Available at: https://sustainablebrands.com/read/chemistry-materials-packaging/more-major-retailers-are-saying-forever-chemicals-no-more. Accessed 4 Mar 2020.

Schaider LA et al. (2017). Fluorinated compounds in U.S. fast food packaging. Environmental Science & Technology Letters. 4:pp 105–111. doi: 10.1021/acs.estlett.6b00435.

Schecter A et al. (2010). Perfluorinated compounds, polychlorinated biphenyls, and organochlorine pesticide contamination in composite food samples from Dallas, Texas, USA. Environmental Health Perspectives. 118:pp 796–802. doi: 10.1289/ehp.0901347.

Scher DP et al. (2018). Occurrence of perfluoroalkyl substances (PFAS) in garden produce at homes with a history of PFAS-contaminated drinking water. Chemosphere. 196:pp 548–555. doi: 10.1016/j.chemosphere.2017.12.179.

Scheringer M et al. (2014). Helsingør Statement on poly- and perfluorinated alkyl substances (PFASs). Chemosphere. 114:pp 337–339. doi: 10.1016/j.chemosphere.2014.05.044.

Schlummer M et al. (2015). Emission of perfluoroalkyl carboxylic acids (PFCA) from heated surfaces made of polytetrafluoroethylene (PTFE) applied in food contact materials and consumer products. Chemosphere. 129:pp 46–53. doi: 10.1016/j.chemosphere.2014.11.036.

Schultes L et al. (2019). Total fluorine measurements in food packaging: How do current methods perform? Environmental Science & Technology Letters. doi: 10.1021/acs.estlett.8b00700.

Schwanz TG et al. (2016). Perfluoroalkyl substances assessment in drinking waters from Brazil, France and Spain. Science of The Total Environment. 539:pp 143–152. doi: 10.1016/j.scitotenv.2015.08.034.

Seals R, Bartell SM and Steenland K. (2011). Accumulation and clearance of perfluorooctanoic acid (PFOA) in current and former residents of an exposed community. Environmental Health Perspectives. 119:pp 119–124. doi: 10.1289/ehp.1002346.

Sedlak MD et al. (2017). Per- and polyfluoroalkyl substances (PFASs) in San Francisco Bay wildlife: Temporal trends, exposure pathways, and notable presence of precursor compounds. Chemosphere. 185:pp 1217–1226. doi: 10.1016/j.chemosphere.2017.04.096.

Sedlak MD and Greig DJ. (2012). Perfluoroalkyl compounds (PFCs) in wildlife from an urban estuary. Journal of Environmental Monitoring. 14:pp 146–154. doi: 10.1039/C1EM10609K.

Sepulvado JG et al. (2011). Occurrence and fate of perfluorochemicals in soil following the land application of municipal biosolids. Environmental Science & Technology. 45:pp 8106–8112. doi: 10.1021/es103903d.

Seydel S. (2018). Comment submitted regarding Washington State's Department of Ecology PFAS Alternatives Assessment. Available at: https://commentinput.com/attachments/projectID_1002/10350/merged//040AByONM6.pdf?v=CRK9EBDHQ. Accessed 24 Dec 2018.

SFEI. (2017). San Francisco Estuary Institute (SFEI): Contaminants of emerging concern in San Francisco Bay: A strategy for future investigations. November, 2017 revision. Regional Monitoring Program for Water Quality in San Francisco Bay (RMP). Available at: http://www.sfei.org/sites/default/files/biblio_files/RMP%20CEC%20Strategy%20Nov%202017_0.pdf.

SGS. (2018). Safe Guards (SGS): San Francisco becomes the first city in the US to prohibit PFAS chemicals in single-use food service ware. This provision in the new law will become effective on January 1, 2020. Available at: https://www.sgs.com/en/news/2018/08/safeguards-12418-san-francisco-bans-pfas-chemicals-in-single-use-food-service-ware. Accessed 19 Feb 2020.

Shan G et al. (2014). Concentration profiles and spatial distribution of perfluoroalkyl substances in an industrial center with condensed fluorochemical facilities. Science of The Total Environment. 490:pp 351–359. doi: 10.1016/j.scitotenv.2014.05.005.

Shi G et al. (2017). 6:2 Chlorinated polyfluorinated ether sulfonate, a PFOS alternative, induces embryotoxicity and disrupts cardiac development in zebrafish embryos. Aquatic Toxicology. 185:pp 67–75. doi: 10.1016/j.aquatox.2017.02.002.

Shi Y et al. (2016). Human exposure and elimination kinetics of chlorinated polyfluoroalkyl ether sulfonic acids (Cl-PFESAs). Environmental Science & Technology. 50:pp 2396–2404. doi: 10.1021/acs.est.5b05849.

Shin H et al. (2020). Measured concentrations of consumer product chemicals in California house dust: Implications for sources, exposure, and toxicity potential. Indoor Air. 30:pp 60–75. doi: 10.1111/ina.12607.

Shoaeioskouei S. (2012). Perfluorinated compounds in landfill leachate from discarded carpets. M.S. Thesis, Faculty of Graduate Studies (Civil Engineering), the University of British Columbia, Vancouver. June 2012.

Shoeib M et al. (2004). Indoor and outdoor air concentrations and phase partitioning of perfluoroalkyl sulfonamides and polybrominated diphenyl ethers. Environmental Science & Technology. 38:pp 1313–1320. doi: 10.1021/es0305555.

Shoeib M et al. (2010). Survey of polyfluorinated chemicals (PFCs) in the atmosphere over the northeast Atlantic Ocean. Atmospheric Environment. 44:pp 2887–2893. doi: 10.1016/j.atmosenv.2010.04.056.

Shoeib M et al. (2011). Indoor sources of poly- and perfluorinated compounds (PFCs) in Vancouver, Canada: Implications for human exposure. Environmental Science & Technology. 45:pp 7999–8005. doi: 10.1021/es103562v.

Siebenaler R et al. (2017). Serum perfluoroalkyl acids (PFAAs) and associations with behavioral attributes. Chemosphere. 184:pp 687–693. doi: 10.1016/j.chemosphere.2017.06.023.

Sinclair E and Kannan K. (2006). Mass loading and fate of perfluoroalkyl surfactants in wastewater treatment plants. Environmental Science & Technology. 40:pp 1408–1414. doi: 10.1021/es051798v.

Skaar JS et al. (2018). Elucidation of contamination sources for poly- and perfluoroalkyl substances (PFASs) on Svalbard (Norwegian Arctic). Environmental Science and Pollution Research. 26:pp 7356–7363. doi: 10.1007/s11356-018-2162-4.

Skuladottir M et al. (2015). Examining confounding by diet in the association between perfluoroalkyl acids and serum cholesterol in pregnancy. Environmental Research. 143:pp 33–38. doi: 10.1016/j.envres.2015.09.001.

Slotkin TA et al. (2008). Developmental neurotoxicity of perfluorinated chemicals modeled *in vitro*. Environmental Health Perspectives. 116:pp 716–722. doi: 10.1289/ehp.11253.

Solenis. (2020). Barrier coatings for paper and board. Available at: https://solenis.com/en/solutions/pulp-paper-solutions/barrier-coatings/?gclid=EAIaIQobChMIosTp_OCS6AIVhtdkCh0hIAO4EAAYASAAEgKRZ_D_BwE. Accessed 11 Mar 2020.

Soloff AC et al. (2017). Environmental perfluorooctane sulfonate exposure drives T cell activation in bottlenose dolphins: PFOS-induced T cell activation in dolphins. Journal of Applied Toxicology. 37:pp 1108–1116. doi: 10.1002/jat.3465.

Solo-Gabriele HM et al. (2020). Waste type, incineration, and aeration are associated with per- and polyfluoroalkyl levels in landfill leachates. Waste Management. 107:pp 191–200. doi: 10.1016/j.wasman.2020.03.034.

Solvay. (2020). Solvera® PFPE. Available at: https://www.solvay.com/en/brands/solvera-pfpe. Accessed 18 Mar 2020.

Spaan KM et al. (2019). Mammals from the Northern Hemisphere. ChemRxiv. pp 41.

Stahl T et al. (2009). Carryover of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) from soil to plants. Archives of Environmental Contamination and Toxicology. 57:pp 289–298. doi: 10.1007/s00244-008-9272-9.

Stahl T, Mattern D and Brunn H. (2011). Toxicology of perfluorinated compounds. Environmental Sciences Europe. 23:pp 38. doi: 10.1186/2190-4715-23-38.

State of Washington. (2018a). Engrossed Substitute Senate Bill 6413, 2018 Regular Session, 65th Legislature. Enacted: 2018. Available at: http://lawfilesext.leg.wa.gov/biennium/2017-18/Pdf/Bills/Session%20Laws/Senate/6413-S.SL.pdf?q=20200708201908.

State of Washington. (2018b). House Bill 2658, 2018 Regular Session, 65th Legislature. Enacted: 2018. Available at: http://lawfilesext.leg.wa.gov/biennium/2017-18/Pdf/Bills/House%20Bills/2658.pdf.

State of Washington. (2017). Survey of per- and poly-fluoroalkyl substances (PFASs) in rivers and lakes, 2016, Publication No. 17-03-021, September 2017. Available at: https://fortress.wa.gov/ecy/publications/SummaryPages/1703021.html.

Stecker T and Schultz D. (2019). House Committee approves far-reaching PFAS package. Available at: https://news.bloombergenvironment.com/environment-and-energy/house-committee-approves-far-reaching-pfas-package.

Still M et al. (2013). Impact of Industrial Production and Packaging Processes on the Concentration of Per- and Polyfluorinated Compounds in Milk and Dairy Products. Journal of Agricultural and Food Chemistry. 61:pp 9052–9062. doi: 10.1021/jf4020137.

Stock NL et al. (2004). Polyfluorinated telomer alcohols and sulfonamides in the North American troposphere. Environmental Science & Technology. 38:pp 991–996. doi: 10.1021/es034644t.

Strynar MJ and Lindstrom AB. (2008). Perfluorinated compounds in house dust from Ohio and North Carolina, USA. Environmental Science & Technology. 42:pp 3751–3756. doi: 10.1021/es7032058.

Su H et al. (2017). Home produced eggs: An important pathway of human exposure to perfluorobutanoic acid (PFBA) and perfluorooctanoic acid (PFOA) around a fluorochemical industrial park in China. Environment International. 101:pp 1–6. doi: 10.1016/j.envint.2017.01.016.

Sun H et al. (2012). Perfluoroalkyl compounds in municipal WWTPs in Tianjin, China—Concentrations, distribution and mass flow. Environmental Science and Pollution Research. 19:pp 1405–1415. doi: 10.1007/s11356-011-0727-6.

Sun H et al. (2011). Long-chain perfluorinated chemicals in digested sewage sludges in Switzerland. Environmental Pollution. 159:pp 654–662. doi: 10.1016/j.envpol.2010.09.020.

Sun J et al. (2019). White-tailed eagle (*Haliaeetus albicilla*) body feathers document spatiotemporal trends of perfluoroalkyl substances in the northern environment. Environmental Science & Technology. 53:pp 12744–12753. doi: 10.1021/acs.est.9b03514.

Sun M et al. (2016). Legacy and emerging perfluoroalkyl substances are important drinking water contaminants in the Cape Fear River watershed of North Carolina. Environmental Science & Technology Letters. 3:pp 415–419. doi: 10.1021/acs.estlett.6b00398.

Sunderland EM et al. (2019). A review of the pathways of human exposure to poly- and perfluoroalkyl substances (PFASs) and present understanding of health effects. Journal of Exposure Science & Environmental Epidemiology. 29:pp 131–147. doi: 10.1038/s41370-018-0094-1.

Sundström M et al. (2012). Comparative pharmacokinetics of perfluorohexanesulfonate (PFHxS) in rats, mice, and monkeys. Reproductive Toxicology. 33:pp 441–451. doi: 10.1016/j.reprotox.2011.07.004.

Susmann HP et al. (2019). Dietary habits related to food packaging and population exposure to PFASs. Environmental Health Perspectives. 127: doi: 10.1289/EHP4092.

Taniyasu S et al. (2013). Does wet precipitation represent local and regional atmospheric transportation by perfluorinated alkyl substances? Environment International. 55:pp 25–32. doi: 10.1016/j.envint.2013.02.005.

Tao L et al. (2008a). Biomonitoring of perfluorochemicals in plasma of New York State personnel responding to the World Trade Center disaster. Environmental Science & Technology. 42:pp 3472–3478. doi: 10.1021/es8000079.

Tao L et al. (2008b). Perfluorinated compounds in human milk from Massachusetts, U.S.A. Environmental Science & Technology. 42:pp 3096–3101. doi: 10.1021/es702789k.

Tartu S et al. (2018). Choose your poison-Space-use strategy influences pollutant exposure in Barents Sea polar bears. Environmental Science & Technology. 52:pp 3211–3221. doi: 10.1021/acs.est.7b06137.

Tartu S et al. (2017). Multiple-stressor effects in an apex predator: Combined influence of pollutants and sea ice decline on lipid metabolism in polar bears. Scientific Reports. 7:pp 16487. doi: 10.1038/s41598-017-16820-5.

Tatum-Gibbs K et al. (2011). Comparative pharmacokinetics of perfluorononanoic acid in rat and mouse. Toxicology. 281:pp 48–55. doi: 10.1016/j.tox.2011.01.003.

The Seydel Companies, Inc. (2018). Coatings for paper and corrugated board. Available at: http://www.seydel.com/products/paper-coatings/. Accessed 24 Dec 2018.

Thomas Net. (2019). List of Suppliers. Available at: https://www.thomasnet.com/?_ga=2.4361397.786548292.1581720133-537871987.1579733717. Accessed 5 Dec 2019.

CELA. (2019). Canadian Environmental Law Association (CELA): Scoping per- and polyfluoroalkly substances releases from the recycling of paper and textiles and their implications for the Great Lakes-St Lawrence River ecosystem. Available at: https://cela.ca/wp-content/uploads/2019/10/Report-PFAS-Sept-2019.pdf.

Tian Y et al. (2018). Occurrence and phase distribution of neutral and ionizable per- and polyfluoroalkyl substances (PFASs) in the atmosphere and plant leaves around landfills: A case study in Tianjin, China. Environmental Science & Technology. 52:pp 1301–1310. doi: 10.1021/acs.est.7b05385.

Tian Z et al. (2016). Human exposure to per- and polyfluoroalkyl substances (PFASs) via house dust in Korea: Implication to exposure pathway. Science of The Total Environment. 553:pp 266–275. doi: 10.1016/j.scitotenv.2016.02.087.

Tipton JJ et al. (2017). Analysis of PFAAs in American alligators part 1: Concentrations in alligators harvested for consumption during South Carolina public hunts. Journal of Environmental Sciences. 61:pp 24–30. doi: 10.1016/j.jes.2017.05.045.

Tomy GT et al. (2004). Fluorinated organic compounds in an eastern Arctic marine food web. Environmental Science & Technology. 38:pp 6475–6481. doi: 10.1021/es049620g.

Trasande L et al. (2018). Food additives and child health. Pediatrics. 142:pp e20181408. doi: 10.1542/peds.2018-1408.

Nordic Council of Ministers. (2017). PFAS in paper and board for food contact: Options for risk management of poly- and perfluorinated substances. Available at: https://backend.orbit.dtu.dk/ws/portalfiles/portal/149769110/Rapport_PFAS_in_paper_and_board_for_food_contact_Options_for_risk_management_of_poly_and_perfluorina.pdf.

Trier X, Granby K and Christensen JH. (2011a). Polyfluorinated surfactants (PFS) in paper and board coatings for food packaging. Environmental Science and Pollution Research. 18:pp 1108–1120. doi: 10.1007/s11356-010-0439-3.

Trier X, Nielsen NJ and Christensen JH. (2011b). Structural isomers of polyfluorinated di- and tri-alkylated phosphate ester surfactants present in industrial blends and in microwave popcorn bags. Environmental Science and Pollution Research. 18:pp 1422–1432. doi: 10.1007/s11356-011-0488-2.

Trudel D et al. (2008). Estimating consumer exposure to PFOS and PFOA. Risk Analysis. 28:pp 251–269. doi: 10.1111/j.1539-6924.2008.01017.x.

Tsai MS et al. (2018). Determinants and temporal trends of perfluoroalkyl substances in pregnant women: The Hokkaido study on environment and children's health. International Journal of Environmental Research and Public Health. 15:pp 989. doi: 10.3390/ijerph15050989.

Ulhaq M et al. (2013). Comparison of developmental toxicity of seven perfluoroalkyl acids to zebrafish embryos. Environmental Toxicology and Pharmacology. 36:pp 423–426. doi: 10.1016/j.etap.2013.05.004.

UNCW. (2018). University of North Carolina at Wilmington (UNCW): Report to the Environmental Review Commission from the University of North Carolina at Wilmington regarding the implementation of section 20.(a)(2) of House Bill 56 (S.L. 2017-209). Available at: https://www.ncleg.net/documentsites/committees/ERC/ERC%20Reports%20Received/2018/CFPUA%20and%20UNC-W/2018-April%20HB%2056%20UNCW%20Rpt.pdf.

UNEP. (2020). United Nations Environment Programme (UNEP): All POPs listed in the Stockholm Convention. Available at: http://chm.pops.int/TheConvention/ThePOPs/AllPOPs/tabid/2509/Default.aspx. Accessed 12 May 2020.

UNEP/POPS/POPRC. (2017). United Nations Environment Programme, Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee (UNEP/POPS/POPRC): Highlights of the thirteenth meeting of the Persistent Organic Pollutants Review Committee (POPRC.13), Rome, Italy, from 17 to 20 October 2017. Available at: http://www.pops.int/Default.aspx?tabid=5965. Accessed 10 Nov 2017.

U.S. Bureau of Labor Statistics. (2019). Employment Projections. Available at: https://data.bls.gov/projections/occupationProj. Accessed 26 Sep 2019.

U.S. Census Bureau. (2007). Economic Census. Available at: https://www.census.gov/data/tables/2007/econ/census/manufacturing-reports.html. Accessed 25 Sep 2019.

U.S. Census Bureau. (2012). Economic Census. Available at: https://www.census.gov/data/tables/2012/econ/census/manufacturing-reports.html. Accessed 25 Sep 2019.

U.S. Census Bureau. (2017). Economic Census. Available at: https://data.census.gov/cedsci/table?hidePreview=true&table=EC1731BASIC&tid=ECNBASIC2017.EC1731BASIC&lastDisplayedRow=647&q=EC1731BASIC%3A%20Manufacturing%3A%20Summary%20Statistics%20for%20the%20U.S.,%20States,%20and%20Selected%20Geographies%3A%202017. Accessed 25 Sep 2019.

U.S. Census Bureau. (2019). USA Trade Online. Available at: https://usatrade.census.gov. Accessed 19 Dec 2019.

U.S. EPA. (2020a). U.S. Environmental Protection Agency (U.S. EPA): Chemistry Dashboard. Available at: https://comptox.epa.gov/dashboard.

U.S. EPA. (2009). U.S. Environmental Protection Agency (U.S. EPA): Long-chain perfluorinated chemicals (PFCs) action plan. December 30, 2009. Available at: https://www.epa.gov/sites/production/files/2016-01/documents/pfcs_action_plan1230_09.pdf. Accessed 23 Aug 2017.

U.S. EPA. (2017a). U.S. Environmental Protection Agency (U.S. EPA): Occurrence Data for the Unregulated Contaminant Monitoring Rule. Available at: https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule. Accessed 9 Nov 2017.

U.S. EPA. (2011). U.S. Environmental Protection Agency (U.S. EPA): Exposure Factors Handbook, 2011 Edition (Final Report). U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-

09/052F, 2011. Available at: https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=236252. Accessed 22 Aug 2017.

U.S. EPA. (2020b). U.S. Environmental Protection Agency (U.S. EPA): Drinking water health advisories for PFOA and PFOS. Available at: https://www.epa.gov/ground-water-and-drinking-water/drinking-water-health-advisories-pfoa-and-pfos. Accessed 11 May 2020.

U.S. EPA. (2016a). U.S. Environmental Protection Agency (U.S. EPA): EJ 2020 Action Agenda. The U.S. Environmental Protection Agency's 2016 – 2020 Strategic Plan. Available at: https://www.epa.gov/sites/production/files/2016-05/documents/052216_ej_2020_strategic_plan_final_0.pdf.

U.S. EPA. (2002a). U.S. Environmental Protection Agency (U.S. EPA): 40 CFR Part 721 Perfluoroalkyl Sulfonates; Significant New Use Rule. Federal Register, Vol. 67, No. 236, December 9, 2002. Available at: https://www.gpo.gov/fdsys/pkg/FR-2002-12-09/pdf/02-31011.pdf. Accessed 11 May 2020.

U.S. EPA. (2002b). U.S. Environmental Protection Agency (U.S. EPA): 40 CFR Part 721 Perfluoroalkyl Sulfonates; Significant New Use Rule; Final Rule and Supplemental Proposed Rule. Federal Register, Vol. 67, No. 47, March 11, 2002. Available at: https://www.gpo.gov/fdsys/pkg/FR-2002-03-11/pdf/02-5746.pdf. Accessed 11 May 2020.

U.S. EPA. (2007). U.S. Environmental Protection Agency (U.S. EPA): 40 CFR Part 721. Perfluoroalkyl Sulfonates, Significant New Use Rule. Federal Register, Vol. 72, No. 194, October 9, 2007. Available at: https://www.gpo.gov/fdsys/pkg/FR-2007-10-09/pdf/E7-19828.pdf. Accessed 11 May 2020.

U.S. EPA. (2013). U.S. Environmental Protection Agency (U.S. EPA): 40 CFR Parts 9 of 721 Perfluoroalkyl Carboxylate Substances; Final Significant New Use Rule. U.S. EPA, Washington, DC. Available at: https://www.regulations.gov/contentStreamer?documentId=EPA-HQ-OPPT-2012-0268-0034&contentType=pdf. Accessed 11 May 2020.

U.S. EPA. (2015). U.S. Environmental Protection Agency (U.S. EPA). Long-chain perfluoroalkyl carboxylate and perfluoroalkyl sulfonate chemical substances; Significant New Use Rule. Available at: https://www.federalregister.gov/documents/2015/01/21/2015-00636/long-chain-perfluoroalkyl-carboxylate-and-perfluoroalkyl-sulfonate-chemical-substances-significant.

U.S. EPA. (2020c). U.S. Environmental Protection Agency (U.S. EPA): EPA continues to act on PFAS, proposes to close import loophole and protect American consumers. Available at: https://www.epa.gov/newsreleases/epa-continues-act-pfas-proposes-close-import-loophole-and-protect-american-consumers. Accessed 24 Feb 2020.

U.S. EPA. (2020d). U.S. Environmental Protection Agency (U.S. EPA): Risk Management for Per- and Polyfluoroalkyl Substances (PFASs) under TSCA. Available at: https://www.epa.gov/assessing-

and-managing-chemicals-under-tsca/risk-management-and-polyfluoroalkyl-substances-pfass. Accessed 11 May 2020.

U.S. EPA. (2017b). U.S. Environmental Protection Agency (U.S. EPA): Reviewing New Chemicals under the Toxic Substances Control Act (TSCA). Available at: https://www.epa.gov/reviewing-new-chemicals-under-toxic-substances-control-act-tsca. Accessed 17 May 2017.

U.S. EPA. (2020e). U.S. Environmental Protection Agency (EPA). 2010/2015 PFOA Stewardship Program. Available at: https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/and-polyfluoroalkyl-substances-pfass-under-tsca#tab-3.

U.S. EPA. (2017c). U.S. Environmental Protection Agency (U.S. EPA): EPA launches cross-agency effort to address PFAS. U.S. Environmental Protection Agency (U.S. EPA): News Releases from U.S. EPA Office of Research and Development (ORD), 12/04/2017. Available at: https://www.epa.gov/newsreleases/epa-launches-cross-agency-effort-address-pfas. Accessed 5 Jan 2018.

U.S. EPA. (2018). U.S. Environmental Protection Agency (U.S. EPA): EPA toxicologists focus innovative research on PFAS compounds. March 13, 2018. Available at: https://www.epa.gov/sciencematters/epa-toxicologists-focus-innovative-research-pfas-compounds. Accessed 11 Apr 2018.

U.S. EPA. (2019). U.S. Environmental Protection Agency (U.S. EPA): EPA awards $6 million to research potential environmental impacts of PFAS substances in waste streams. Available at: https://www.pressreleasepoint.com/print/1583225. Accessed 26 Sep 2019.

U.S. EPA. (2016b). U.S. Environmental Protection Agency (U.S. EPA): Health effects support document for perfluorooctanoic acid (PFOA). Office of Water (4304T), Health and Ecological Criteria Division, Washington, DC, EPA Document Number 822-R-16-003, May 2016. Available at: https://www.epa.gov/sites/production/files/2016-05/documents/pfoa_hesd_final_508.pdf. Accessed 28 Nov 2017.

U.S. EPA. (2016c). U.S. Environmental Protection Agency (U.S. EPA): Health effects support document for perfluorooctane sulfonate (PFOS). Office of Water (4304T), Health and Ecological Criteria Division, Washington, DC, EPA Document Number 822-R-16-002, May 2016. Available at: https://www.epa.gov/sites/production/files/2016-05/documents/pfos_hesd_final_508.pdf. Accessed 28 Nov 2017.

U.S. EPA. (2020f). U.S. Environmental Protection Agency (U.S. EPA): List of PFAS added to the TRI by the NDAA. in: US EPA. Available at: https://www.epa.gov/toxics-release-inventory-tri-program/list-pfas-added-tri-ndaa. Accessed 17 Mar 2020.

U.S. House of Representatives. (2018). Text of the House Amendment to the Senate Amendment to H.R. 1625, March 2018. Available at: http://docs.house.gov/billsthisweek/20180319/BILLS-115SAHR1625-RCP115-66.pdf. Accessed 9 Apr 2018.

U.S. Senate. (2018). A letter from 25 Senators to the Honorable Scott Pruitt, Administrator of the U.S. Environmental Protection Agency concerning groundwater contamination from per- and polyfluoroalkyl substances (PFAS) in communities and on Department of Defense (DOD) installations across their states, April 13, 2018. Available at: https://drive.google.com/file/d/1LgpWUVI-wfvSW90LtTzjymSNm_BAZTj1/view. Accessed 9 May 2018.

U.S. Senate. (2017). S.1519 - National Defense Authorization Act for Fiscal Year 2018 (Report No. 115–125). Available at: https://www.congress.gov/bill/115th-congress/senate-bill/1519/text. Accessed 5 Jan 2018.

Valentini A et al. (2016). Next-generation monitoring of aquatic biodiversity using environmental DNA metabarcoding. Molecular Ecology. 25:pp 929–942. doi: 10.1111/mec.13428.

Valsecchi S et al. (2017). Deriving environmental quality standards for perfluorooctanoic acid (PFOA) and related short chain perfluorinated alkyl acids. Journal of Hazardous Materials. 323:pp 84–98. doi: 10.1016/j.jhazmat.2016.04.055.

Vendries J et al. (2020). The significance of environmental attributes as indicators of the life cycle environmental impacts of packaging and food service ware. Environmental Science & Technology. 54:pp 5356–5364. doi: 10.1021/acs.est.9b07910.

Venkatesan AK and Halden RU. (2014a). Loss and in situ production of perfluoroalkyl chemicals in outdoor biosolids–soil mesocosms. Environmental Research. 132:pp 321–327. doi: 10.1016/j.envres.2014.04.024.

Venkatesan AK and Halden RU. (2014b). Loss and in situ production of perfluoroalkyl chemicals in outdoor biosolids–soil mesocosms. Environmental Research. 132:pp 321–327. doi: 10.1016/j.envres.2014.04.024.

Verreault J, Berger U and Gabrielsen GW. (2007). Trends of perfluorinated alkyl substances in herring gull eggs from two coastal colonies in northern Norway: 1983-2003. Environmental Science & Technology. 41:pp 6671–6677. doi: 10.1021/es070723j.

Wallington TJ et al. (2006). Formation of C7F15COOH (PFOA) and other perfluorocarboxylic acids during the atmospheric oxidation of 8:2 fluorotelomer alcohol. Environmental Science & Technology. 40:pp 924–930. doi: 10.1021/es051858x.

Wang C et al. (2019). Assessing the contribution of atmospheric transport and tourism activities to the occurrence of perfluoroalkyl acids (PFAAs) in an Alpine Nature Reserve. Science of The Total Environment. 697:pp 133851. doi: 10.1016/j.scitotenv.2019.133851.

Wang F et al. (2015a). Effectiveness and mechanisms of defluorination of perfluorinated alkyl substances by calcium compounds during waste thermal treatment. Environmental Science & Technology. 49:pp 5672–5680. doi: 10.1021/es506234b.

Wang G et al. (2017a). Occurrence, distribution, and risk assessment of perfluoroalkyl acids (PFAAs) in muscle and liver of cattle in Xinjiang, China. International Journal of Environmental Research and Public Health. 14:pp 970. doi: 10.3390/ijerph14090970.

Wang T et al. (2011). Hydrophobicity-dependent QSARs to predict the toxicity of perfluorinated carboxylic acids and their mixtures. Environmental Toxicology and Pharmacology. 32:pp 259–265. doi: 10.1016/j.etap.2011.05.011.

Wang Y et al. (2008). Perfluorooctane sulfonate and other fluorochemicals in waterbird eggs from south China. Environmental Science & Technology. 42:pp 8146–8151. doi: 10.1021/es8006386.

Wang Z et al. (2017b). A never-ending story of per- and polyfluoroalkyl substances (PFASs)? Environmental Science & Technology. 51:pp 2508–2518. doi: 10.1021/acs.est.6b04806.

Nordic Council of Ministers. (2020). Per- and polyfluoroalkylether substances: Identity, production and use. Available at: http://urn.kb.se/resolve?urn=urn:nbn:se:norden:org:diva-5872.

Wang Z et al. (2016). Comparative assessment of the environmental hazards of and exposure to perfluoroalkyl phosphonic and phosphinic acids (PFPAs and PFPiAs): Current knowledge, gaps, challenges and research needs. Environment International. 89–90:pp 235–247. doi: 10.1016/j.envint.2016.01.023.

Wang Z et al. (2014a). Global emission inventories for C4–C14 perfluoroalkyl carboxylic acid (PFCA) homologues from 1951 to 2030, Part I: Production and emissions from quantifiable sources. Environment International. 70:pp 62–75. doi: 10.1016/j.envint.2014.04.013.

Wang Z et al. (2015b). Hazard assessment of fluorinated alternatives to long-chain perfluoroalkyl acids (PFAAs) and their precursors: Status quo, ongoing challenges and possible solutions. Environment International. 75:pp 172–179. doi: 10.1016/j.envint.2014.11.013.

Wang Z et al. (2014b). Atmospheric concentrations and gas/particle partitioning of neutral poly- and perfluoroalkyl substances in northern German coast. Atmospheric Environment. 95:pp 207–213. doi: 10.1016/j.atmosenv.2014.06.036.

Washington JW et al. (2015). Decades-scale degradation of commercial, side-chain, fluorotelomer-based polymers in soils and water. Environmental Science & Technology. 49:pp 915–923. doi: 10.1021/es504347u.

Washington JW et al. (2009). Degradability of an acrylate-linked, fluorotelomer polymer in soil. Environmental Science & Technology. 43:pp 6617–6623. doi: 10.1021/es9002668.

Washington JW et al. (2019). Determining global background soil PFAS loads and the fluorotelomer-based polymer degradation rates that can account for these loads. Science of The Total Environment. 651:pp 2444–2449. doi: 10.1016/j.scitotenv.2018.10.071.

Washington State Legislature. (2019). Substitute Senate Bill 5135. Available at: http://lawfilesext.leg.wa.gov/biennium/2019-20/Pdf/Bills/Session%20Laws/Senate/5135-S.SL.pdf. Accessed 26 Sep 2019.

Weiss JM et al. (2009). Competitive binding of poly- and perfluorinated compounds to the thyroid hormone transport protein transthyretin. Toxicological Sciences. 109:pp 206–216. doi: 10.1093/toxsci/kfp055.

White ND et al. (2015). Elevated levels of perfluoroalkyl substances in estuarine sediments of Charleston, SC. Science of The Total Environment. 521–522:pp 79–89. doi: 10.1016/j.scitotenv.2015.03.078.

Wilkinson JL et al. (2017). Spatial distribution of organic contaminants in three rivers of Southern England bound to suspended particulate material and dissolved in water. Science of The Total Environment. 593–594:pp 487–497. doi: 10.1016/j.scitotenv.2017.03.167.

Winkens K et al. (2017a). Early life exposure to per- and polyfluoroalkyl substances (PFASs): A critical review. Emerging Contaminants. 3:pp 55–68. doi: 10.1016/j.emcon.2017.05.001.

Winkens K et al. (2017b). Perfluoroalkyl acids and their precursors in indoor air sampled in children's bedrooms. Environmental Pollution. 222:pp 423–432. doi: 10.1016/j.envpol.2016.12.010.

Winquist A and Steenland K. (2014). Perfluorooctanoic acid exposure and thyroid disease in community and worker cohorts. Epidemiology. 25:pp 255–264. doi: 10.1097/EDE.0000000000000040.

Wolf CJ et al. (2008). Activation of mouse and human peroxisome proliferator-activated receptor alpha by perfluoroalkyl acids of different functional groups and chain lengths. Toxicological Sciences. 106:pp 162–171. doi: 10.1093/toxsci/kfn166.

Wolf CJ et al. (2014). Evaluating the additivity of perfluoroalkyl acids in binary combinations on peroxisome proliferator-activated receptor-α activation. Toxicology. 316:pp 43–54. doi: 10.1016/j.tox.2013.12.002.

Wong YM et al. (2017). The measurement of bisphenol A and its analogues, perfluorinated compounds in twenty species of freshwater and marine fishes, a time-trend comparison and human health based assessment. Marine Pollution Bulletin. 124:pp 743–752. doi: 10.1016/j.marpolbul.2017.05.046.

Wood Environment & Infrastructure Solutions, Inc. (2019). Investigation of per- and polyfluoroalkyl substances (PFAS) at select source separated organic material and yard waste sites, Minnesota. Available at: https://www.pca.state.mn.us/sites/default/files/w-sw4-37.pdf.

Wu X (May) et al. (2015). Serum concentrations of perfluorinated compounds (PFC) among selected populations of children and Adults in California. Environmental Research. 136:pp 264–273. doi: 10.1016/j.envres.2014.09.026.

Wu Y et al. (2020). Per- and polyfluoroalkyl substances in paired dust and carpets from childcare centers. Chemosphere. pp 126771. doi: 10.1016/j.chemosphere.2020.126771.

Wu Y et al. (2019). Novel and legacy per- and polyfluoroalkyl substances in bald eagle eggs from the Great Lakes region. Environmental Pollution. pp 113811. doi: 10.1016/j.envpol.2019.113811.

Xiao F. (2017). Emerging poly- and perfluoroalkyl substances in the aquatic environment: A review of current literature. Water Research. 124:pp 482–495. doi: 10.1016/j.watres.2017.07.024.

Xiao F et al. (2015). Perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA) in soils and groundwater of a U.S. metropolitan area: Migration and implications for human exposure. Water Research. 72:pp 64–74. doi: 10.1016/j.watres.2014.09.052.

Xie Z et al. (2015). Neutral poly-/perfluoroalkyl substances in air and snow from the Arctic. Scientific Reports. 5:pp 8912. doi: 10.1038/srep08912.

Xu Z et al. (2013). Human exposure to fluorotelomer alcohols, perfluorooctane sulfonate and perfluorooctanoate via house dust in Bavaria, Germany. Science of The Total Environment. 443:pp 485–490. doi: 10.1016/j.scitotenv.2012.10.089.

Yamada A et al. (2014a). Dietary exposure to perfluoroalkyl acids of specific French adult sub-populations: High seafood consumers, high freshwater fish consumers and pregnant women. Science of The Total Environment. 491–492:pp 170–175. doi: 10.1016/j.scitotenv.2014.01.089.

Yamada A et al. (2014b). Perfluoroalkyl acid contamination and polyunsaturated fatty acid composition of French freshwater and marine fishes. Journal of Agricultural and Food Chemistry. 62:pp 7593–7603. doi: 10.1021/jf501113j.

Yan H et al. (2012). Short- and long-chain perfluorinated acids in sewage sludge from Shanghai, China. Chemosphere. 88:pp 1300–1305. doi: 10.1016/j.chemosphere.2012.03.105.

Yang L et al. (2016). Human placental transfer of perfluoroalkyl acid precursors: Levels and profiles in paired maternal and cord serum. Chemosphere. 144:pp 1631–1638. doi: 10.1016/j.chemosphere.2015.10.063.

Yao Y et al. (2018). Per- and polyfluoroalkyl substances (PFASs) in indoor air and dust from homes and various microenvironments in China: Implications for human exposure. Environmental Science & Technology. 52:pp 3156–3166. doi: 10.1021/acs.est.7b04971.

Ye X et al. (2008). Perfluorinated compounds in whole fish homogenates from the Ohio, Missouri, and Upper Mississippi Rivers, USA. Environmental Pollution. 156:pp 1227–1232. doi: 10.1016/j.envpol.2008.03.014.

Yeh A et al. (2017). Effect of contaminants of emerging concern on liver mitochondrial function in Chinook salmon. Aquatic Toxicology. 190:pp 21–31. doi: 10.1016/j.aquatox.2017.06.011.

Yeung LWY and Mabury SA. (2016). Are humans exposed to increasing amounts of unidentified organofluorine? Environmental Chemistry. 13:pp 102. doi: 10.1071/EN15041.

Yoo H et al. (2011). Quantitative determination of perfluorochemicals and fluorotelomer alcohols in plants from biosolid-amended fields using LC/MS/MS and GC/MS. Environmental Science & Technology. 45:pp 7985–7990. doi: 10.1021/es102972m.

Young CJ et al. (2007). Perfluorinated acids in Arctic snow: New evidence for atmospheric formation. Environmental Science & Technology. 41:pp 3455–3461. doi: 10.1021/es0626234.

Yu N et al. (2018). Non-target and suspect screening of per- and polyfluoroalkyl substances in airborne particulate matter in China. Environmental Science & Technology. 52:pp 8205–8214. doi: 10.1021/acs.est.8b02492.

Yuan G et al. (2016). Ubiquitous occurrence of fluorotelomer alcohols in eco-friendly paper-made food-contact materials and their implication for human exposure. Environmental Science & Technology. 50:pp 942–950. doi: 10.1021/acs.est.5b03806.

Zabaleta I et al. (2017). Screening and identification of per- and polyfluoroalkyl substances in microwave popcorn bags. Food Chemistry. 230:pp 497–506. doi: 10.1016/j.foodchem.2017.03.074.

Zareitalabad P et al. (2013). Perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) in surface waters, sediments, soils and wastewater – A review on concentrations and distribution coefficients. Chemosphere. 91:pp 725–732. doi: 10.1016/j.chemosphere.2013.02.024.

Zhang D, Zhang W and Liang Y. (2019a). Distribution of eight perfluoroalkyl acids in plant-soil-water systems and their effect on the soil microbial community. Science of The Total Environment. 697:pp 134146. doi: 10.1016/j.scitotenv.2019.134146.

Zhang H et al. (2015). Uptake of perfluoroalkyl acids in the leaves of coniferous and deciduous broad-leaved trees. Environmental Toxicology and Chemistry. 34:pp 1499–1504. doi: 10.1002/etc.2968.

Zhang L et al. (2019b). Uptake mechanisms of perfluoroalkyl acids with different carbon chain lengths (C2-C8) by wheat (*Triticum acstivnm L.*). Science of The Total Environment. 654:pp 19–27. doi: 10.1016/j.scitotenv.2018.10.443.

Zhang M et al. (2020). Bioaccumulation and human exposure of perfluoroalkyl acids (PFAAs) in vegetables from the largest vegetable production base of China. Environment International. 135:pp 105347. doi: 10.1016/j.envint.2019.105347.

Zhang R et al. (2018). Plasma concentration of 14 perfluoroalkyl acids (PFAAs) among children from seven cities in Guangdong, China. Science of The Total Environment. 616–617:pp 1469–1476. doi: 10.1016/j.scitotenv.2017.10.167.

Zhang T et al. (2013). Distribution of poly- and perfluoroalkyl substances in matched samples from pregnant women and carbon chain length related maternal transfer. Environmental Science & Technology. 47:pp 7974–7981. doi: 10.1021/es400937y.

Zhang X, Lohmann R and Sunderland EM. (2019c). Poly- and perfluoroalkyl substances in seawater and plankton from the Northwestern Atlantic margin. Environmental Science & Technology. 53:pp 12348–12356. doi: 10.1021/acs.est.9b03230.

Zhang Y et al. (2016). Perfluorinated compounds in greenhouse and open agricultural producing areas of three provinces of China: Levels, sources and risk assessment. International Journal of Environmental Research and Public Health. 13:pp 1224. doi: 10.3390/ijerph13121224.

Zhao H, Guan Y and Qu B. (2018). PFCA uptake and translocation in dominant wheat species (*Triticum aestivum* L.). International Journal of Phytoremediation. 20:pp 68–74. doi: 10.1080/15226514.2017.1337066.

Zhao L et al. (2017a). Isomer-specific transplacental efficiencies of perfluoroalkyl substances in human whole blood. Environmental Science & Technology Letters. 4:pp 391–398. doi: 10.1021/acs.estlett.7b00334.

Zhao S and Zhu L. (2017b). Uptake and metabolism of 10:2 fluorotelomer alcohol in soil-earthworm (*Eisenia fetida*) and soil-wheat (*Triticum aestivum* L.) systems. Environmental Pollution. 220:pp 124–131. doi: 10.1016/j.envpol.2016.09.030.

Zhao Z et al. (2012). Distribution and long-range transport of polyfluoroalkyl substances in the Arctic, Atlantic Ocean and Antarctic coast. Environmental Pollution. 170:pp 71–77. doi: 10.1016/j.envpol.2012.06.004.

Zheng G et al. (2020). Indoor exposure to per- and polyfluoroalkyl substances (PFAS) in the childcare environment. Environmental Pollution. 258:pp 113714. doi: 10.1016/j.envpol.2019.113714.

Zhou J et al. (2019). Insights into uptake, translocation, and transformation mechanisms of perfluorophosphinates and perfluorophosphonates in wheat (*Triticum aestivum L.*). Environmental Science & Technology. doi: 10.1021/acs.est.9b05656.

Zucker HA et al. (2017). Letter to Brenda Fitzgerald, M.D., Director, Centers for Disease Control and Prevention, Administrator, Agency for Toxic Substances and Disease Registry, US Department of Health & Human Services, August 24, 2017. Available at: https://www.health.ny.gov/environmental/investigations/drinkingwaterresponse/docs/2017-08-24_cdc_study_letter.pdf. Accessed 15 Sep 2017.

Zushi Y, Hogarh JN and Masunaga S. (2012). Progress and perspective of perfluorinated compound risk assessment and management in various countries and institutes. Clean Technologies and Environmental Policy. 14:pp 9–20. doi: 10.1007/s10098-011-0375-z.

Zushi Y and Masunaga S. (2009). Identifying the nonpoint source of perfluorinated compounds using a geographic information system based approach. Environmental Toxicology and Chemistry. 28:pp 691–700.

APPENDIX A: POTENTIAL RELEVANT FACTORS

Non-exhaustive list of adverse impact factors that may be relevant to this proposed Priority Product

Relevant Factors are used in SCP's Alternatives Analysis (AA) to make a focused and meaningful comparison of adverse impacts during the product's lifecycle between the Priority Product and alternative. This Profile has identified adverse impacts in the following categories:

- Adverse environmental impacts
- Adverse public health impacts
- Adverse waste and end-of-life effects
- Environmental fate
- Physicochemical properties
- Associated exposure pathways and life cycle segments, including:
    - Manufacture
    - Use
    - Waste generation and management
    - Reuse and recycling
    - End-of-life disposal

At a minimum, all AAs submitted for this product-chemical combination must include a discussion of these impacts and how they compare between the Priority Product and whatever alternative(s) have been identified at the appropriate point in the lifecycle. This list is not intended to be comprehensive. Also, alternatives evaluated in the AA report will likely have additional adverse impacts that don't apply to the Priority Product; these will also need to be assessed in the AA report. Product performance and economics are generally not evaluated in the Profile.

APPENDIX B: REPORT PREPARATION*

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* This document is based on information presented in the Product-Chemical Profile for Carpets and Rugs Containing Perfluoroalkyl or Polyfluoroalkyl Substances and the Product-Chemical Profile for Treatments Containing Perfluoroalkyl or Polyfluoroalkyl Substances.

APPENDIX C – PFAS HAZARD TRAITS

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Carcinogenicity	Limited evidence in humans for the carcinogenicity of perfluorooctanoic acid (PFOA). A positive association was observed for cancers of the testis and kidney. Overall evaluation: Perfluorooctanoic acid (PFOA) is possibly carcinogenic to humans (Group 2B). Review by authoritative body.	Positive	IARC (2016)
	Suggestive evidence of carcinogenic potential for PFOA. Epidemiology studies demonstrate an association of serum PFOA with kidney and testicular tumors among highly exposed members of the general population. Two chronic bioassays of PFOA support a positive finding for its ability to be tumorigenic in one or more organs of rats, including the liver, testes, and pancreas. Review by authoritative body.	Positive	U.S. EPA (2020)
	Suggestive evidence of carcinogenic potential for PFOS. In a chronic oral toxicity and carcinogenicity study of PFOS in rats, liver and thyroid tumors, and mammary fibroadenomas were identified. Review by authoritative body.	Positive	U.S. EPA (2016)
	Positive association of kidney cancer with PFOA exposure as observed in high PFOA exposure cohorts in occupational and community settings.	Positive	Barry et al. (2013); C8 Science Panel (2012); Steenland and Woskie (2012); Vieira et al. (2013)
	Positive association of testicular cancer with PFOA exposure in high PFOA exposure cohort.	Positive	Barry et al. (2013); C8 Science Panel (2012)
	Increased risk of thyroid cancer in occupational cohort exposure to PFOA.	Positive	Barry et al. (2013)
	Possible increase in prostate cancer mortality in workers with 10 years occupational exposure to PFOA.	Positive	Gilliland and Mandel (1993)
	Higher PFOA serum levels associated with testicular, kidney, prostate and ovarian cancer, and non-Hodgkin lymphoma. (High PFOA exposure cohort)	Positive	Lundin et al. (2009); Vieira et al. (2013)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Higher risk of prostate cancer in individuals with PFAS levels above median and a first degree relative with prostate cancer. Positive of genetic susceptibility. PFHxS showed a statistically significant interaction.	Positive	Hardell et al. (2014)
	Weak positive association of breast cancer risk with PFOS exposure in Danish National Birth Cohort.	Positive	Bonefeld-Jørgensen et al. (2014)
	Tumor triad in male rats (liver, Leydig cells, pancreatic) dosed with APFO.	Positive	Biegel et al. (2001)
	Ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate was tested for chronic toxicity and carcinogenicity in Sprague–Dawley rats. Level of evidence sufficient to show that the findings of liver, pancreas, testicular tumors are relevant for humans.	Positive	RIVM et al. (2016); Rae et al. (2015)
	General Danish population study did not indicate increased incidence of prostate, bladder, pancreatic, liver cancers.	Negative	Eriksen et al. (2009)
	No association between PFOA levels and breast cancer risk in mothers from Danish National Birth Cohort.	Negative	Bonefeld-Jørgensen et al. (2014)
	No increased prostate cancer risk compared to controls (Gleason score and prostate-specific antigen (PSA) levels).	Negative	Hardell et al. (2014)
	No association of occupational APFO exposure with liver, pancreatic or testicular cancers.	Negative	Lundin et al. (2009)
	Study reports lack of concordance between community exposures and occupational exposures one or two magnitudes higher than those for the general population. The discrepant findings across the study populations were described as likely due to chance, confounding, and/or bias.	Negative	Chang et al. (2014); (3M-funded independent tox/epi review)
	No evidence of genotoxicity from NaPFHx in bacterial mutagenicity assays (333-5000 µg/mL), nor chromosome aberrations in human peripheral lymphocytes (with or without metabolic activation) (5-3860 µg/mL).	Negative	Loveless et al. (2009)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	No evidence of carcinogenicity in either male or female Sprague-Dawley rats treated with PFHxA when administered orally daily for 2 years.	Negative	Klaunig et al. (2015)
	No observed association between occupational APFO exposure and the risk of dying or developing liver, pancreatic, testicular, kidney, prostate, breast, bladder, and thyroid, cancers.	Negative	Raleigh et al. (2014)
	PFASs modestly produce reactive oxygen species and DNA damage in the HepG2 cell line. Increase in ROS production was not concentration-dependent and the compounds did not generate DNA damage that could be detected by the alkaline comet assay.	Negative	Eriksen et al. (2010)
Developmental Toxicity	PFOA and PFOS are "known to the State of California to cause reproductive toxicity", based on formal identification by an authoritative body (U.S. EPA). Review by authoritative body.	Positive	OEHHA (2017)
	The EPA Office of Water selected a RfD of 0.00002 mg/kg-day for PFOA based on reduced ossification and accelerated puberty (in male mice) (Lau et al. 2006).	Positive	U.S. EPA (2016a)
	EPA derived a reference dose for PFOS of 0.00002 mg/kg-day based on decreased neonatal rat body weight from the 2-generation study (Luebker et al. 2005).	Positive	U.S. EPA (2016)
	Treatment of pregnant CD-1 mice with NH4+ PFBA by oral gavage is associated with delayed eye opening in pups and delayed vaginal opening in 175 or 350 mg/kg-day dose groups.	Positive	Das et al. (2008)
	Decreased birth weight with increased maternal/cord blood PFOA levels in general population.	Positive	Apelberg et al. (2007);Fei et al. (2007); Johnson et al. (2014); Maisonet et al. (2012)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	No association between PFOA levels and birth weight in high-exposure C8 community population.	Negative	Darrow et al. (2013); Nolan et al. (2009); Savitz et al. (2012a); Savitz et al. (2012b); Stein et al. (2009)
	K+ PFHxS given to Sprague Dawley rats by oral gavage. No treatment-related effects (reproductive, developmental, neurological) in dams or offspring compared with controls.	Negative	Butenhoff et al. (2009)
	2,3,3,3-Tetrafluoro-2-(heptafluoropropoxy)propanoate (a PFOA replacement technology) in rats at 100 and 1000 mg/kg-day decreases fetal weights and produces early delivery of offspring. Offspring were alive, with no increase in resorptions. Reproductive effects were observed at dose levels also inducing maternal toxicity, which "do not normally warrant classification."	Negative	RIVM et al. (2016)
Reproductive Toxicity	Decreased fertility in humans with increased concentrations of PFOS, PFOA and PFHxS in maternal plasma.	Positive	Fei et al. (2009); Velez et al. (2015)
	Probable link between PFOA exposure and pregnancy-induced hypertension in humans.	Positive	C8 Science Panel (2012)
	PFOS, PFDoA, PFNA, and PFOA are cytotoxic in JEG-3 human placental cells (EC50s range from 107 - 647 µM). PFOS, PFOA, and PFBS are potential CYP19 aromatase inhibitors (IC50s 57 - 80 µM). Notable inhibitory effect of PFBS with relatively low uptake in cells.	Positive	Gorrochategui et al. (2014)
	Treatment of pregnant CD-1 mice with PFBA by oral gavage was associated with increased incidence of fetal resorption and increased offspring liver weight at postnatal day (PND) 1 (but not by PND 10). Delayed eye opening in pups was observed in all dose groups (35, 175 and 350 mg/kg-day) and delayed puberty in highest two dose groups.	Positive	Das et al. (2008)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Increased ovarian follicular fluid levels of PFHxS were associated with reduced baseline follicle count and post-fertilization success in women receiving assistive reproductive technology.	Positive	McCoy et al. (2017)
Cardiovascular Toxicity	Probable link between PFOA exposure and elevated cholesterol.	Positive	C8 Science Panel (2012)
	No increase in relative risk of heart disease. Significant 'trend' for increased risk in heart disease after 10-year lag in PFOA exposure.	Positive	Sakr et al. (2009)
	Positive correlation of serum cholesterol with PFOA, PFOS and PFHxS exposure.	Positive	Fisher et al. (2013); Costa et al. (2009); Nelson et al. (2010); Sakr et al. (2007a); Skuladottir et al. (2015); Steenland et al. (2010); Wang et al. (2012); Winquist and Steenland (2014)
	No indication of increased serum lipids/cholesterol in humans with elevated PFOS/PFOA serum levels.	Negative	Emmett et al. (2006) (exposed residents); Olsen et al. (2000); Olsen and Zobel (2007)
	No electrocardiogram (EKG) alterations in PFOA-exposed workers (5-9550 ng/mL serum PFOA).	Negative	Sakr et al. (2007b); Sakr et al. (2007a)
	No increased mortality ratio in exposed workers for heart disease.	Negative	Lau et al. (2007); Lundin et al. (2009); Steenland and Woskie (2012)
Endocrine Toxicity	Probable link between PFOA exposure and thyroid disease in human based on occupational and community cohort studies.	Positive	C8 Science Panel (2012); Lopez-Espinosa et al. (2012); Melzer et al. (2010); Winquist and Steenland (2014b)
	Increased risk of gestational diabetes development with serum PFOA	Positive	Zhang et al. (2015)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Increased mortality from diabetes in occupational exposure cohorts. Authors specify that mortality may not be a good surrogate for incidence.	Positive	Lau et al. (2007); Leonard et al. (2008); Lundin et al. (2009) (SMR = 197); Steenland and Woskie (2012) (SMR = 1.90)
	Positive association of serum estradiol and testosterone with serum PFOA in men. Authors indicated circadian variations (collection time) may confound data.	Positive	Sakr et al. (2007b); Sakr et al. (2007a)
	Decreased free and total T4 in PFOA-treated Cynomolgus monkeys.	Positive	Butenhoff et al. (2002)
	PFBS (IC50 = 68 µM) and PFHxS are aromatase inhibitors in placental cells. Inhibitory effect of PFBS important despite low uptake in cells.	Positive	Gorrochategui et al. (2014)
	Activation of estrogen receptor in reporter gene assay for 4:2, 6:2 and 8:2 FTOH. C4+ PFCAs all activate PPAR-alpha; no PFBA activation of PPAR-gamma.	Positive	Rosenmai et al. (2016)
	PPAR-alpha activities are induced at lower cellular concentrations for short-chain homologs relative to long-chains.	Positive	Rosenmai et al. (2016)
	PFHxSK inhibits 11 -HSD2 in human and rat kidney microsomes. PFHxSK is more potent in human than rat microsomes.	Positive	Zhao et al. (2010)
	Non-cytotoxic levels of PFHxDI (dodecafluoro-1,6-diiodohexane) significantly promote the proliferation of MCF-7 breast cancer cells. PFHxDI's preferential binding affinity for estrogen receptor α and β isoforms mediated this cellular response.	Positive	Song et al. (2018)
	PFHxS can disrupt the thyroid system of Wistar rats, as measured by T4 levels in dams and offspring. PFHxS potentiated the antiandrogenic effect of an endocrine disruptor mixture in male offspring.	Positive	Ramhøj et al. (2018)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Competitive binding to transthyretin (for TH transport) in binding assay, reducing free thyroxine in blood. Competitive binding of PFHxS > PFOS/PFOA > PFHpA. Binding of PFASs 12-300 times lower than thyroxine.	Positive	Weiss et al. (2009)
	No effect from PFBA and FTOHs.	Negative	
	PFBA/PFBS had lowest potency of activation of PPAR-alpha in COS-1 cells in multiple chain lengths tested.	Negative	Wolf et al. (2008)
	The potassium salts of PFBS and PFHxS has no effect on 3β or 17β-HSD activity in human or rat testes microsomes, even at high concentrations.	Negative	Zhao et al. (2010)
	No evidence of altered thyroid function with occupational exposure, serum thyroid stimulating hormone (TSH), T3, T4.	Negative	Mundt et al. (2007); Olsen et al. (1998); Olsen et al. (2003); Olsen and Zobel (2007); Sakr et al. (2007a); Sakr et al. (2007a)
	No association between serum PFOA and Type 2 diabetes incidence in general or worker populations.	Negative	MacNeil et al. (2009); Steenland et al. (2015)
	No association between serum PFOA and measures of metabolic syndrome in general or worker populations.	Negative	Lin et al. (2009)
Hemato Toxicity	Reduced red blood cell (RBC) count, hemoglobin and hematocrit in male rats (90 day PFBS dosage).	Positive	Lieder et al. (2009)
	No correlation in hematological parameters (including hematocrit, hemoglobin, red blood cells, white blood cells, platelets) and serum PFOS in plant workers.	Negative	Olsen et al. (2003); Olsen et al. (1998); Olsen and Zobel (2007)
	No alterations in blood counts in workers exposure to PFOA.	Negative	Sakr et al. (2007b); Sakr et al. (2007a)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Hepatotoxicity and Digestive System Toxicity	Positive minor association of serum PFOA with serum liver enzyme levels (AST, ALT, GGT).	Positive	Costa et al. (2009); Gallo et al. (2012); Olsen et al. (2000); Olsen et al. (2003); Olsen and Zobel (2007); Sakr et al. (2007b); Sakr et al. (2007a)
	Probable link between PFOA exposure and ulcerative colitis.	Positive	C8 Science Panel (2012); Steenland et al. (2015)
	Small, increased liver weight in Cynomolgous monkeys (PFOA capsules).	Positive (low N, 1-2% increase)	Butenhoff et al. (2002)
	Hepatonecrosis, hepatic tumors in Cynomolgous monkeys. At termination of dosing, the mean PFOS serum level corresponding to the LOAEL is 171,000 ng/mL in females and 173,000 ng/mL in males (K-PFOS capsules).	Positive (low N)	Seatcat et al. (2002)
	In rats, PFHxS is more toxic to liver than PFOS and PFBS.	Positive	Danish Environmental Protection Agency (2015)
	In a 90-day study, PFHxA was administered to rats daily by oral gavage. The NOAEL is 50 mg/kg/day for males and >200 mg/kg/day for females based on increased liver weight, peroxisome beta oxidation activity and centrilobular hepatocyte hypertrophy.	Positive	Chengelis et al. (2009)
	Reversible increased liver weight at 30 and 150 mg/kg-day APFB, slight hepatocellular hypertrophy and decreased serum total cholesterol in male S-D rats at 150 mg/kg-day for 28 days APFB.	Positive	Butenhoff et al. (2012)
	Various repeated dose studies of varying duration (but 90 days or less) and dose in rodents of FRD-902, ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propanoate revealed increased liver and kidney weights, hepatocellular hypertrophy and alterations in blood chemistries. Chronic feeding study revealed various liver pathologies.	Positive	RIVM et al. (2016)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Increased incidence of focal cystic degeneration, and centrilobular hepatocellular hypertrophy and necrosis in male Sprague-Dawley rats given 50 mg/kg-day ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate orally for 2 years. Similar results in female rats given 500 mg/kg-day. Clinical chemistry evaluations among males receiving 50 mg/kg-d showed increases in enzymes indicative of liver injury (alkaline phosphatase, alanine aminotransferase, aspartate aminotransferase, and sorbitol dehydrogenase).	Positive	Rae et al. (2015)
	K+ PFHxS to Sprague Dawley rats by oral gavage. Effects in parental males include reduced serum cholesterol (all doses), increased liver-to-body and liver-to brain weight ratio, centrilobular hypertrophy, hypertrophy AND hyperplasia of thyroid follicular cells, decreased hematocrit (3 and 10 mg/kg b.w. per day), decreased triglycerides and increased albumin, urea nitrogen, alkaline phosphatase, Ca2+ and albumin/globulin ratio (10 mg/kg b.w. per day)	Positive	Butenhoff et al. (2009)
	4:2, 6:2 and 8:2 FTOHs were incubated in freshly isolated rat hepatocytes, causing cell and mitochondrial toxicity. Cytotoxicity is related to cytochrome P450-mediated bioactivation. LC50s 0.66, 3.7 and 1.4 mM, respectively.	Positive	Martin et al. (2009)
	Compared with PFOA and PFOS, long-chain PFAS alternatives 6:2 chlorinated polyfluorinated ether sulfonate, hexafluoropropylene trimer acid, HFPO tetramer acid, and 6:2 fluorotelomer sulfonic acid show greater cytotoxicity on human liver HL-7702 cells.	Positive	Sheng et al. (2017)
	6:2 chlorinated polyfluorinated ether sulfonate (6:2 Cl-PFESA) and hexafluoropropylene oxide (HFPO) show unique binding modes and higher binding affinity to human liver fatty acid binding protein (hl-FABP), and higher cytotoxicity in human liver cells, than PFOA and PFOS.	Positive	Sheng et al. (2018)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	PPAR-alpha induction in human hepatocellular carcinoma cells is similar for C4 and C5 ("short-chain") PFCAs as for C12 and C14 ("long-chain"), but is induced at lower cellular concentrations by the C4 and C5 compounds tested. Other PFCAs tested (C6-C10) induce PPAR-alpha greater than 2.5-fold.	Positive	Rosenmai et al. (2018)
	No significant relationship between serum PFOA and liver function in residents of community water district.	Negative	Emmett et al. (2006)
	No association of PFOA exposure with liver disease. The observed U-shaped dose-response for serum bilirubin may explain inverse responses in occupational cohorts.	Negative	C8 Science Panel (2012)
	No reported gastric ulcers or colon polyps in workers exposed to PFOS.	Negative	Grice et al. (2007)
Immunotoxicity	Several PFASs were associated with an increased number of respiratory tract infections in the first 10 years of life, suggesting immunosuppressive effects.	Positive	Impinen et al. (2018)
	Depressed antibody production post-vaccination, or increased odds of, with increasing PFOA, PFOS and PFHxS serum concentrations.	Positive	Grandjean et al. (2012); Grandjean et al. (2016); Granum et al. (2013); Looker et al. (2014)
	In human peripheral blood leukocytes, PFBS inhibits the release of TNF-α and IL-10, but IL-6 and IFN-γ are unaffected. In THP-1 cells, PFBS also inhibits NF-κB activation (by inhibiting LPS-induced phosphorylation of P65, necessary for NF-κB transcription), and prevents I-κB degradation.	Positive	Corsini et al. (2012)
	Probable link between PFOA exposure and ulcerative colitis.	Positive	C8 Science Panel (2012); Steenland et al. (2015)
	Prenatal exposure to PFOA, PFDA, PFDoA and PFHxS increases the risk of childhood atopic dermatitis in female children during the first 24 months of life in a prospective birth cohort study.	Positive	Chen et al. (2018)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	Oral administration of 2,3,3,3-Tetrafluoro-2-(Heptafluoropropoxy)-Propanoate in mice for 28 days. Evaluated for T-cell dependent antibody response (TDAR) and splenic lymphocyte subpopulations 5 days post-antigen treatment. TDAR is suppressed in females at 100 mg/kg. T lymphocyte numbers are increased in males at 100 mg/kg.	Positive	Rushing et al. (2017)
Urinary System Toxicity	Increased mortality from renal disease (SMR 3.11) in workers at PFOA facility. PFOA exposure estimated based on job history and data from biomonitoring survey. Significant positive trend in nonmalignant kidney disease.	Positive	Steenland and Woskie (2012)
Nephrotoxicity and other	No association of kidney function, (measured blood urea nitrogen and serum creatinine) with occupational PFOS/PFOA exposure.	Negative	Costa et al. (2009); Mundt et al. (2007); Olsen et al. (2003); Olsen et al. (1998)
Neurodevelopmental Toxicity	Positive association with Attention Deficit Hyperactivity Disorder (ADHD) in children in a highly PFOA-exposed community.	Positive	Hoffman et al. (2010)
	A single exposure to PFHxS (9.2 mg/kg) on postnatal day 10 affected the cholinergic system and altered adult spontaneous behavior and cognitive function in male and female mice.	Positive	Viberg et al. (2013)
	PFBS uniquely suppressed differentiation of Ach and Dopa phenotypes in vitro PC12 (rat neuronal) cells. Concentration-dependent decrease in expression of tyrosine hydroxylase and choline acetyltransferase. Unlikely for one shared mechanism of toxicity across perfluorinated chemicals.	Positive	Slotkin et al. (2008)
	No finding of adverse association between PFOA exposure and neuropsychological test performance in children.	Negative	Stein et al. (2013)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Neuro Toxicity	PFASs such as PFOA and PFOS may induce neurobehavioral effects, particularly in developmentally exposed animals.	Positive	Mariussen (2012)
	Neurobehavioral endpoints were not affected in Sprague Dawley rats following exposure to PFHxA (0, 20, 100 or 500 mg/kg b.w. per day NaPFHx) for 90 days.	Negative	Loveless et al. (2009)
Respiratory Toxicity	90-day evaluation of NaPFHxA in Sprague Dawley rats following oral gavage, indicates a NOAEL of 20 and 100 mg/kg-day in males and females, respectively, based on mild to minimal degeneration of the nasal epithelium.	Positive	Loveless et al. (2009)
	Three cases of young healthy adults, following occupational inhalation of a new waterproofing formulation containing an acrylate fluoropolymer, developed a rapidly progressive dyspnea; two had hypoxemia and flu-like reactions within 1-2 hours of exposure.	Positive	Lazor-Blanchet et al. (2004)
	A commercial nanofilm spray product containing a perfluorosilane induced a concentration-dependent decrease of the tidal volume, significant increases of protein level in bronchoalveolar lavage fluid, reduced body weight, atelectasis, emphysema, and lung hemorrhages in BALB/cJ mice compared to a formulation containing alkylsilane.	Positive	Nørgaard et al. (2010)
	Normal pulmonary function in fluoropolymer plant workers with serum PFOA.	Negative	Sakr et al. (2007b); Sakr et al. (2007a)
Hazard Traits	Increased PPAR-alpha activation in human-transfected COS-1 monkey kidney cells by PFHxA, PFBA, PFHxS, and PFBS. PFDA inactive.	Positive	Wolf et al. (2008)
	Chlorinated polyfluorinated ether sulfonates (6:2 Cl-PFAES and 8:2 Cl-PFAES) bound to PPAR-α, -β, and -γ with affinity higher than PFOS and has agonistic activity toward PPAR signaling pathways with similar or greater potency.	Positive	Li et al. (2018)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Other Toxicological	28-day 150 mg/kg-d APFB repeat-dose oral study in S-D rats. Delay in pupillary reflex response. Outer retinal degeneration and loss of photoreceptors. Decrease in inner retinal, nuclear and synaptic terminal layer thicknesses. No mortality.	Positive	3M (2006); Butenhoff et al. (2012)
	Ammonium 4,8-dioxa-3H-perfluorononanoate (ADONA) toxicity was evaluated in acute and repeat-dose studies of up to 90-days duration. It is a mild skin irritant and a moderate to severe eye irritant in rabbits.	Positive	Gordon (2011)
	Survival rate of female Sprague-Dawley rats significantly decreased at presumed maximally tolerated dose of PFHxA (200 mg/kg-d) in 2-year study.	Positive	Klaunig et al. (2015)
	Experimental and QSAR modeling results showed PFAS mixture cytotoxicity in amphibian fibroblasts was approximately additive, except with PFOS and PFOA, which were weakly synergistic.	Positive	Hoover et al. (2019)
	Dopaminergic neurodegeneration in nematodes (*C. elegans*) exposed to PFOS.	Positive	Sammi et al. (2019)
	No generation of reactive oxygen species (ROS) nor DNA damage by PFBS and PFHxA in HepG2 cells.	Negative	Eriksen et al. (2010)
Phytotoxicity	Aquatic toxicity of PFHxA, PFHpA, PFOA and PFNA to several species of marine microalgae increases with increasing chain length: for every extra perfluoromethylene group in the alkyl chain, toxicity increases twofold. EC50 ranges from 0.28 mM to 12.84 mM.	Positive	Latala et al. (2009)
	Toxicity of PFBS, PFOS, PFHxA, PFOA, PFDoA and PFTeA on the membrane system of the freshwater algae *Scenedesmus obliquus* increased with chain length. PFOS, PFDoA and PFTeA inhibits algal growth in a concentration-dependent manner.	Positive	Liu et al. (2008)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
	A broad range of acute and chronic toxicity responses in algae and aquatic plants have been observed, with stronger effects associated with increased perfluoroalkyl chain length.	Positive	Beach et al. (2006); Ding and Peijnenburg (2013); Environment Canada (2006); Environment Canada (2012); Giesy et al. (2010); UNEP/POPS/POPRC (2016a); UNEP/POPS/POPRC (2016b); UNEP/POPS/POPRC (2016c); UNEP/POPS/POPRC (2016d); Ding et al. (2012a); Ding et al. (2012b); Hoke et al. (2012);Latala et al. (2009); Liu et al. (2008)
	The aquatic toxicity of seven PFASs was investigated on the root elongation of lettuce (*L. sativa*) seeds and photosynthesis of green algae (*P. subcapitata*). The toxic effects on lettuce seeds and green algae were found to be similar in a relative sense and were shown to have a good relationship with the fluorinated carbon-chain length. The toxicity of these chemicals increases with increasing fluorinated carbon chain length. PFBA was more toxic than expected, likely due to acidification of the test solution. Because the unicellular *P. subcapitata* is sensitive to the pH value of the test solution, the actual EC50 value of PFBA is lower and deviates from the relationship between log EC50 value and chain length.	Positive	Ding et al. (2012a)
	Tail deformation and uninflated swim bladders in zebrafish exposed to PFBA. Lowered heart rates and cardiac output. Malformations of the head in zebrafish exposed to PFBS. No structure-activity relationship for some endpoints. Exposure concentrations 50-3000 mg/L.	Positive	Hagenaars et al. (2011)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Wildlife Developmental Impairment	Adverse impacts on embryonic development, hatchability, and time to hatch of Japanese medaka (*O. latipes*) eggs from exposure to PFOA and PFOS	Positive	Ji et al. (2008)
	Developmental toxicity observed in zebrafish embryos exposed to 6:2 FTCA. Observed effects include: decreased hatching and survival percentages, reduced heart rate, increased malformations (especially pericardial edema), decreased erythrocyte number, and disrupted erythroid differentiation during zebrafish embryonic development.	Positive	Shi et al. (2017a)
	Delayed development (time to metamorphosis) in northern leopard frog (Rana pipiens) tadpoles exposed to 10 ug/L PFHxS for 40 days.	Positive	Hoover et al. (2017)
	F–53B [6:2 chlorinated polyfluorinated ether sulfonate] exposure (1.5 - 12 mg/L) induced developmental toxicity, including delayed hatching, increased occurrence of malformations, and reduced survival. Malformations, including pericardial and yolk sac edemas, abnormal spines, bent tails, and uninflated swim bladders increased with time course and dose. Continuous exposure resulted in high accumulation levels in zebrafish embryos, suggesting an inability for embryos to eliminate this compound and a high cumulative risk to fish. F–53B induced cardiac toxicity and reduced heart rate.	Positive	Shi et al. (2017b)
	Dopamine decreased significantly in the brains of Northern Leopard frogs treated with PFOA and PFOS. Significant increases in dopamine turnover also resulted from PFOA and PFOS treatment.	Positive	Foguth et al. (2019)
	PFECAs (e.g., perfluoro (3,5,7-trioxaoctanoic) acid (PFO3OA), perfluoro (3,5,7,9-tetraoxadecanoic) acid (PFO4DA), and perfluoro (3,5,7,9,11-pentaoxadodecanoic) acid (PFO5DoDA)) tested on zebrafish embryos damaged swim bladder mesothelium and significantly lowered thyroid hormone levels in larvae. Exogeneous T3 or T4 supplementation partly rescued PFECA-induced posterior swim bladder malformation.	Positive	Wang et al. (2020)

Health Hazard Category	Summary of Findings	Adverse Impact Finding	Reference
Wildlife Reproductive Impairment	Decreased fecundity of female Japanese medaka (*O. latipes*) after exposure to PFOS and PFOA.	Positive	Ji et al. (2008)
	Egg concentrations of PFASs are negatively correlated with hatching success in wild and laboratory birds.	Positive	Custer et al. (2012); Custer et al. (2014); Nordén et al. (2016)
	PFOS has adverse impacts on the reproduction of bumblebees (*Bombus terrestris*), including reduced ovarian size.	Positive	Mommaerts et al. (2011)
Wildlife Survival Impairment	Decreased hatching and survival percentages observed in zebrafish embryos exposed to 6:2 FTCA and to F–53B [6:2 chlorinated polyfluorinated ether sulfonate].	Positive	Shi et al. (2017a) Shi et al. (2017b)
	Adverse impacts on fecundity, embryonic development, hatchability, and time to hatch of Japanese medaka (*O. latipes*) eggs from exposure to PFOA and PFOS.	Positive	Ji et al. (2008)
	Survival rate of female Sprague-Dawley rats significantly decreased at presumed maximally tolerated dose of PFHxA (200 mg/kg-d) in two-year study.	Positive	Klaunig et al. (2015)

APPENDIX D – APPENDIX REFERENCES

3M. (2006). Final Report: 28-Day Oral Study in Rats. ToxDocs Study Number 08-047, MTDID 8391. TSCA 8(E) Supplemental Submission: Docket No. 8EHQ-06-16638, 3M Strategic Toxicology Laboratory, St. Paul, MN, August 19, 2006.

Apelberg BJ et al. (2007). Cord serum concentrations of perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA) in relation to weight and size at birth. Environmental Health Perspectives. 115:pp 1670–1676. doi: 10.1289/ehp.10334.

Barry V, Winquist A and Steenland K. (2013). Perfluorooctanoic acid (PFOA) exposures and incident cancers among adults living near a chemical plant. Environmental Health Perspectives. 121:pp 1313–1318. doi: 10.1289/ehp.1306615.

Beach SA et al. (2006). Ecotoxicological evaluation of perfluorooctanesulfonate (PFOS). Reviews of Environmental Contamination and Toxicology. 186:pp 133–174.

Biegel LB et al. (2001). Mechanisms of extrahepatic tumor induction by peroxisome proliferators in male CD rats. Toxicological Sciences. 60:pp 44–55. doi: 10.1093/toxsci/60.1.44.

Bonefeld-Jørgensen EC et al. (2014). Breast cancer risk after exposure to perfluorinated compounds in Danish women: A case–control study nested in the Danish National Birth Cohort. Cancer Causes & Control. 25:pp 1439–1448. doi: 10.1007/s10552-014-0446-7.

Butenhoff J. (2002). Toxicity of ammonium perfluorooctanoate in male cynomolgus monkeys after oral dosing for 6 months. Toxicological Sciences. 69:pp 244–257. doi: 10.1093/toxsci/69.1.244.

Butenhoff JL et al. (2009). Evaluation of potential reproductive and developmental toxicity of potassium perfluorohexanesulfonate in Sprague Dawley rats. Reproductive Toxicology. 27:pp 331–341. doi: 10.1016/j.reprotox.2009.01.004.

Butenhoff JL et al. (2012). Toxicological evaluation of ammonium perfluorobutyrate in rats: Twenty-eight-day and ninety-day oral gavage studies. Reproductive Toxicology. 33:pp 513–530. doi: 10.1016/j.reprotox.2011.08.004.

C8 Science Panel. (2012). Probable link evaluation of cancer. April 15, 2012. Available at: http://www.c8sciencepanel.org/pdfs/Probable_Link_C8_Cancer_16April2012_v2.pdf. Accessed 11 May 2020.

Chang ET et al. (2014). A critical review of perfluorooctanoate and perfluorooctanesulfonate exposure and cancer risk in humans. Critical Reviews in Toxicology. 44:pp 1–81. doi: 10.3109/10408444.2014.905767.

Chen Q et al. (2018). Prenatal exposure to perfluoroalkyl and polyfluoroalkyl substances and childhood atopic dermatitis: A prospective birth cohort study. Environmental Health. 17: doi: 10.1186/s12940-018-0352-7.

Chengelis CP et al. (2009). A 90-day repeated dose oral (gavage) toxicity study of perfluorohexanoic acid (PFHxA) in rats (with functional observational battery and motor activity determinations). Reproductive Toxicology. 27:pp 342–351. doi: 10.1016/j.reprotox.2009.01.006.

Corsini E et al. (2012). *In vitro* characterization of the immunotoxic potential of several perfluorinated compounds (PFCs). Toxicology and Applied Pharmacology. 258:pp 248–255. doi: 10.1016/j.taap.2011.11.004.

Costa G, Sartori S and Consonni D. (2009). Thirty years of medical surveillance in perfluooctanoic acid production workers. Journal of Occupational and Environmental Medicine. 51:pp 364–372. doi: 10.1097/JOM.0b013e3181965d80.

Custer CM et al. (2012). Exposure and effects of perfluoroalkyl compounds on tree swallows nesting at Lake Johanna in east central Minnesota, USA. Reproductive Toxicology. 33:pp 556–562. doi: 10.1016/j.reprotox.2011.01.005.

Custer CM et al. (2014). Exposure and effects of perfluoroalkyl substances in tree swallows nesting in Minnesota and Wisconsin, USA. Archives of Environmental Contamination and Toxicology. 66:pp 120–138. doi: 10.1007/s00244-013-9934-0.

Danish Environmental Protection Agency. (2015). Short-chain polyfluoroalkyl substances (PFAS). A literature review of information on human health effects and environmental fate and effect aspects of short-chain PFAS. Project No. 1707, Copenhagen, Denmark.

Darrow LA, Stein CR and Steenland K. (2013). Serum perfluorooctanoic acid and perfluorooctane sulfonate concentrations in relation to birth outcomes in the Mid-Ohio Valley, 2005-2010. Environmental Health Perspectives. 121:pp 1207–1213. doi: 10.1289/ehp.1206372.

Das KP et al. (2008). Effects of perfluorobutyrate exposure during pregnancy in the mouse. Toxicological Sciences. 105:pp 173–181. doi: 10.1093/toxsci/kfn099.

Ding G et al. (2012a). Toxicity of polyfluorinated and perfluorinated compounds to lettuce (*Lactuca sativa*) and green algae (*Pseudokirchneriella subcapitata*). Archives of Environmental Contamination and Toxicology. 62:pp 49–55. doi: 10.1007/s00244-011-9684-9.

Ding G and Peijnenburg WJGM. (2013). Physicochemical properties and aquatic toxicity of poly- and perfluorinated compounds. Critical Reviews in Environmental Science and Technology. 43:pp 598–678. doi: 10.1080/10643389.2011.627016.

Ding GH et al. (2012b). Acute toxicity of poly- and perfluorinated compounds to two cladocerans, *Daphnia magna* and *Chydorus sphaericus*. Environmental Toxicology and Chemistry. 31:pp 605–610. doi: 10.1002/etc.1713.

Emmett EA et al. (2006). Community exposure to perfluorooctanoate: Relationships between serum levels and certain health parameters. Journal of Occupational and Environmental Medicine. 48:pp 771–779. doi: 10.1097/01.jom.0000233380.13087.37.

Environment Canada. (2006). Ecological screening assessment report on perfluorooctane sulfonate, its salts and its precursors that contain the C8F17SO2 or C8F17SO3, or C8F17SO2N moiety. June, 2006. Available at: https://www.ec.gc.ca/lcpe-cepa/documents/substances/spfo-pfos/ecological_sar_pfos_eng.pdf.

Environment Canada. (2012). Ecological screening assessment report: Long-chain (C9–C20) perfluorocarboxylic acids, their salts and their precursors. August, 2012. Available at: https://www.ec.gc.ca/ese-ees/CA29B043-86E4-4308-BBC0-5CE163FEAC40/PFCAs_EN.pdf.

Eriksen KT et al. (2009). Perfluorooctanoate and perfluorooctanesulfonate plasma levels and risk of cancer in the general Danish population. Journal of the National Cancer Institute. 101:pp 605–609. doi: 10.1093/jnci/djp041.

Eriksen KT et al. (2010). Genotoxic potential of the perfluorinated chemicals PFOA, PFOS, PFBS, PFNA and PFHxA in human HepG2 cells. Mutation Research. 700:pp 39–43. doi: 10.1016/j.mrgentox.2010.04.024.

Fei C et al. (2009). Maternal levels of perfluorinated chemicals and subfecundity. Human Reproduction. 24:pp 1200–1205. doi: 10.1093/humrep/den490.

Fei C et al. (2007). Perfluorinated chemicals and fetal growth: A study within the Danish National Birth Cohort. Environmental Health Perspectives. 115:pp 1677–1682. doi: 10.1289/ehp.10506.

Fisher M et al. (2013). Do perfluoroalkyl substances affect metabolic function and plasma lipids?—Analysis of the 2007–2009, Canadian Health Measures Survey (CHMS) Cycle 1. Environmental Research. 121:pp 95–103. doi: 10.1016/j.envres.2012.11.006.

Foguth RM et al. (2019). Developmental exposure to perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) selectively decreases brain dopamine levels in Northern leopard frogs. Toxicology and Applied Pharmacology. 377:pp 114623. doi: 10.1016/j.taap.2019.114623.

Gallo V et al. (2012). Serum perfluorooctanoate (PFOA) and perfluorooctane sulfonate (PFOS) concentrations and liver function biomarkers in a population with elevated PFOA exposure. Environmental Health Perspectives. 120:pp 655–660. doi: 10.1289/ehp.1104436.

Giesy JP et al. (2010). Aquatic toxicology of perfluorinated chemicals. Reviews of Environmental Contamination and Toxicology. 202:pp 1–52. doi: 10.1007/978-1-4419-1157-5_1.

Gilliland FD and Mandel JS. (1993). Mortality among employees of a perfluorooctanoic acid production plant. Journal of Occupational & Environmental Medicine. 35:pp 950–954.

Gordon SC. (2011). Toxicological evaluation of ammonium 4,8-dioxa-3H-perfluorononanoate, a new emulsifier to replace ammonium perfluorooctanoate in fluoropolymer manufacturing. Regulatory Toxicology and Pharmacology. 59:pp 64–80. doi: 10.1016/j.yrtph.2010.09.008.

Gorrochategui E et al. (2014). Perfluorinated chemicals: Differential toxicity, inhibition of aromatase activity and alteration of cellular lipids in human placental cells. Toxicology and Applied Pharmacology. 277:pp 124–130. doi: 10.1016/j.taap.2014.03.012.

Grandjean P et al. (2012). Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA. 307: doi: 10.1001/jama.2011.2034.

Grandjean P et al. (2016). Serum vaccine antibody concentrations in adolescents exposed to perfluorinated compounds. Environmental Health Perspectives. 125:pp 077018. doi: 10.1289/EHP275.

Granum B et al. (2013). Pre-natal exposure to perfluoroalkyl substances may be associated with altered vaccine antibody levels and immune-related health outcomes in early childhood. Journal of Immunotoxicology. 10:pp 373–379. doi: 10.3109/1547691X.2012.755580.

Grice MM et al. (2007). Self-reported medical conditions in perfluorooctanesulfonyl fluoride manufacturing workers. Journal of Occupational and Environmental Medicine. 49:pp 722–729. doi: 10.1097/JOM.0b013e3180582043.

Hagenaars A et al. (2011). Structure–activity relationship assessment of four perfluorinated chemicals using a prolonged zebrafish early life stage test. Chemosphere. 82:pp 764–772. doi: 10.1016/j.chemosphere.2010.10.076.

Hardell E et al. (2014). Case-control study on perfluorinated alkyl acids (PFAAs) and the risk of prostate cancer. Environment International. 63:pp 35–39. doi: 10.1016/j.envint.2013.10.005.

Hoffman K et al. (2010). Exposure to polyfluoroalkyl chemicals and attention deficit/hyperactivity disorder in U.S. children 12-15 years of age. Environmental Health Perspectives. 118:pp 1762–1767. doi: 10.1289/ehp.1001898.

Hoke RA et al. (2012). Comparative acute freshwater hazard assessment and preliminary PNEC development for eight fluorinated acids. Chemosphere. 87:pp 725–733. doi: 10.1016/j.chemosphere.2011.12.066.

Hoover G et al. (2019). *In vitro* and *in silico* modeling of perfluoroalkyl substances mixture toxicity in an amphibian fibroblast cell line. Chemosphere. 233:pp 25–33. doi: 10.1016/j.chemosphere.2019.05.065.

Hoover GM et al. (2017). Uptake and depuration of four per/polyfluoroalkyl substances (PFASS) in northern leopard frog *Rana pipiens* tadpoles. Environmental Science & Technology Letters. 4:pp 399–403. doi: 10.1021/acs.estlett.7b00339.

IARC. (2016). International Agency for Research on Cancer (IARC): Perfluorooctanoic acid. IARC Monographs on the evaluation of carcinogenic risks to humans. Volume 110. Lyon, France.

Impinen A et al. (2018). Prenatal exposure to perfluoralkyl substances (PFASs) associated with respiratory tract infections but not allergy- and asthma-related health outcomes in childhood. Environmental Research. 160:pp 518–523. doi: 10.1016/j.envres.2017.10.012.

Ji K et al. (2008). Toxicity of perfluorooctane sulfonic acid and perfluorooctanoic acid on freshwater macroinvertebrates (*Daphnia magna* and *Moina macrocopa*) and fish (*Oryzias latipes*). Environmental Toxicology and Chemistry. 27:pp 2159. doi: 10.1897/07-523.1.

Johnson PI et al. (2014). The Navigation Guide—Evidence-based medicine meets environmental health: Systematic review of human evidence for PFOA effects on fetal growth. Environmental Health Perspectives. 122:pp 1028–1039. doi: 10.1289/ehp.1307893.

Klaunig JE et al. (2015). Evaluation of the chronic toxicity and carcinogenicity of perfluorohexanoic acid (PFHxA) in Sprague-Dawley rats. Toxicologic Pathology. 43:pp 209–220. doi: 10.1177/0192623314530532.

Latala A, Nędzi M and Stepnowski P. (2009). Acute toxicity assessment of perfluorinated carboxylic acids towards the Baltic microalgae. Environmental Toxicology and Pharmacology. 28:pp 167–171. doi: 10.1016/j.etap.2009.03.010.

Lau C et al. (2007). Perfluoroalkyl acids: a review of monitoring and toxicological findings. Toxicological Sciences. 99:pp 366–394. doi: 10.1093/toxsci/kfm128.

Lazor-Blanchet C et al. (2004). Acute pulmonary toxicity following occupational exposure to a floor stain protector in the building industry in Switzerland. International Archives of Occupational and Environmental Health. 77:pp 244–248. doi: 10.1007/s00420-004-0505-6.

Leonard RC et al. (2008). Retrospective cohort mortality study of workers in a polymer production plant including a reference population of regional workers. Annals of Epidemiology. 18:pp 15–22. doi: 10.1016/j.annepidem.2007.06.011.

Li Y et al. (2018). Half-lives of PFOS, PFHxS and PFOA after end of exposure to contaminated drinking water. Occupational and Environmental Medicine. 75:pp 46–51. doi: 10.1136/oemed-2017-104651.

Lieder PH et al. (2009). Toxicological evaluation of potassium perfluorobutanesulfonate in a 90-day oral gavage study with Sprague-Dawley rats. Toxicology. 255:pp 45–52. doi: 10.1016/j.tox.2008.10.002.

Lin CY et al. (2009). Association among serum perfluoroalkyl chemicals, glucose homeostasis, and metabolic syndrome in adolescents and adults. Diabetes Care. 32:pp 702–707. doi: 10.2337/dc08-1816.

Liu W et al. (2008). Toxic effect of serial perfluorosulfonic and perfluorocarboxylic acids on the membrane system of a freshwater alga measured by flow cytometry. Environmental Toxicology and Chemistry. 27:pp 1597. doi: 10.1897/07-459.1.

Looker C et al. (2014). Influenza vaccine response in adults exposed to perfluorooctanoate and perfluorooctanesulfonate. Toxicological Sciences. 138:pp 76–88. doi: 10.1093/toxsci/kft269.

Lopez-Espinosa MJ et al. (2012). Thyroid function and perfluoroalkyl acids in children living near a chemical plant. Environmental Health Perspectives. 120:pp 1036–1041. doi: 10.1289/ehp.1104370.

Loveless SE et al. (2009). Toxicological evaluation of sodium perfluorohexanoate. Toxicology. 264:pp 32–44. doi: 10.1016/j.tox.2009.07.011.

Lundin JI et al. (2009). Ammonium perfluorooctanoate production and occupational mortality. Epidemiology. 20:pp 921–928. doi: 10.1097/EDE.0b013e3181b5f395.

MacNeil J et al. (2009). A cross-sectional analysis of type II diabetes in a community with exposure to perfluorooctanoic acid (PFOA). Environmental Research. 109:pp 997–1003. doi: 10.1016/j.envres.2009.08.002.

Maisonet M et al. (2012). Maternal concentrations of polyfluoroalkyl compounds during pregnancy and fetal and postnatal growth in British girls. Environmental Health Perspectives. 120:pp 1432–1437. doi: 10.1289/ehp.1003096.

Mariussen E. (2012). Neurotoxic effects of perfluoroalkylated compounds: Mechanisms of action and environmental relevance. Archives of Toxicology. 86:pp 1349–1367. doi: 10.1007/s00204-012-0822-6.

Martin JW et al. (2009). Bioactivation of fluorotelomer alcohols in isolated rat hepatocytes. Chemico-Biological Interactions. 177:pp 196–203. doi: 10.1016/j.cbi.2008.11.001.

McCoy JA et al. (2017). Associations between perfluorinated alkyl acids in blood and ovarian follicular fluid and ovarian function in women undergoing assisted reproductive treatment. Science of The Total Environment. 605–606:pp 9–17. doi: 10.1016/j.scitotenv.2017.06.137.

Melzer D et al. (2010). Association between serum perfluorooctanoic acid (PFOA) and thyroid disease in the U.S. National Health and Nutrition Examination Survey. Environmental Health Perspectives. 118:pp 686–692. doi: 10.1289/ehp.0901584.

Mommaerts V et al. (2011). Impact of a perfluorinated organic compound PFOS on the terrestrial pollinator *Bombus terrestris* (*Insecta*, *Hymenoptera*). Ecotoxicology. 20:pp 447–456. doi: 10.1007/s10646-011-0596-2.

Mundt DJ et al. (2007). Clinical epidemiological study of employees exposed to surfactant blend containing perfluorononanoic acid. Occupational and Environmental Medicine. 64:pp 589–594. doi: 10.1136/oem.2007.032664.

Nelson JW, Hatch EE and Webster TF. (2010). Exposure to polyfluoroalkyl chemicals and cholesterol, body weight, and insulin resistance in the general U.S. population. Environmental Health Perspectives. 118:pp 197–202. doi: 10.1289/ehp.0901165.

Nolan LA et al. (2009). The relationship between birth weight, gestational age and perfluorooctanoic acid (PFOA)-contaminated public drinking water. Reproductive Toxicology. 27:pp 231–238. doi: 10.1016/j.reprotox.2008.11.001.

Norden M, Berger U and Engwall M. (2016). Developmental toxicity of PFOS and PFOA in great cormorant (*Phalacrocorax carbo sinensis*), herring gull (*Larus argentatus*) and chicken (*Gallus gallus domesticus*). Environmental Science and Pollution Research. 23:pp 10855–10862. doi: 10.1007/s11356-016-6285-1.

Nørgaard AW et al. (2010). Lung damage in mice after inhalation of nanofilm spray rroducts: the role of perfluorination and free hydroxyl groups. Toxicological Sciences. 116:pp 216–224. doi: 10.1093/toxsci/kfq094.

OEHHA. (2017). Office of Environmental Health Hazard Assessment (OEHHA): Chemicals Listed Effective November 10, 2017 as Known to the State of California to Cause Reproductive Toxicity: Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS). Available at: https://oehha.ca.gov/proposition-65/crnr/chemicals-listed-effective-november-10-2017-known-state-california-cause. Accessed 9 Jan 2018.

Olsen GW et al. (2000). Plasma cholecystokinin and hepatic enzymes, cholesterol and lipoproteins in ammonium perfluorooctanoate production workers. Drug and Chemical Toxicology. 23:pp 603–620. doi: 10.1081/DCT-100101973.

Olsen GW et al. (1998). An epidemiologic investigation of reproductive hormones in men with occupational exposure to perfluorooctanoic acid. Journal of Occupational & Environmental Medicine. 40:pp 614–622. doi: 10.1097/00043764-199807000-00006.

Olsen GW et al. (2003). Epidemiologic assessment of worker serum perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA) concentrations and medical surveillance examinations. Journal of Occupational and Environmental Medicine. 45:pp 260–270. doi: 10.1097/01.jom.0000052958.59271.10.

Olsen GW and Zobel LR. (2007). Assessment of lipid, hepatic, and thyroid parameters with serum perfluorooctanoate (PFOA) concentrations in fluorochemical production workers. International Archives of Occupational and Environmental Health. 81:pp 231–246. doi: 10.1007/s00420-007-0213-0.

Rae CJM et al. (2015). Evaluation of chronic toxicity and carcinogenicity of ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate in Sprague–Dawley rats. Toxicology Reports. 2:pp 939–949. doi: 10.1016/j.toxrep.2015.06.001.

Raleigh KK et al. (2014). Mortality and cancer incidence in ammonium perfluorooctanoate production workers. Occupational and Environmental Medicine. 71:pp 500–506. doi: 10.1136/oemed-2014-102109.

Ramhøj L et al. (2018). Perfluorohexane sulfonate (PFHxS) and a mixture of endocrine disrupters reduce thyroxine levels and cause anti-androgenic effects in rats. Toxicological Sciences. 163:pp 579–591. doi: 10.1093/toxsci/kfy055.

RIVM. (2016). Dutch National Institute for Public Health and the Environment (RIVM): Evaluation of substances used in the GenX technology by Chemours. Dordrecht. Available at: https://www.rivm.nl/bibliotheek/rapporten/2016-0174.pdf.

Rosenmai AK et al. (2016). Fluorinated alkyl substances and technical mixtures used in food paper-packaging exhibit endocrine-related activity in vitro. Andrology. 4:pp 662–672. doi: 10.1111/andr.12190.

Rosenmai AK et al. (2018). Relationship between peroxisome proliferator-activated receptor alpha activity and cellular concentration of 14 perfluoroalkyl substances in HepG2 cells. Journal of Applied Toxicology. 38:pp 219–226. doi: 10.1002/jat.3515.

Rushing BR et al. (2017). Evaluation of the immunomodulatory effects of 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate in C57BL/6 mice. Toxicol Sci. doi: 10.1093/toxsci/kfw251.

Sakr CJ et al. (2009). Ischaemic heart disease mortality study among workers with occupational exposure to ammonium perfluorooctanoate. Occupational and Environmental Medicine. 66:pp 699–703. doi: 10.1136/oem.2008.041582.

Sakr CJ et al. (2007a). Longitudinal study of serum lipids and liver enzymes in workers with occupational exposure to ammonium perfluorooctanoate. Journal of Occupational and Environmental Medicine. 49:pp 872–879. doi: 10.1097/JOM.0b013e318124a93f.

Sakr CJ et al. (2007b). Cross-sectional study of lipids and liver enzymes related to a serum biomarker of exposure (ammonium perfluorooctanoate or APFO) as part of a general health survey in a cohort of occupationally exposed workers. Journal of Occupational and Environmental Medicine. 49:pp 1086–1096. doi: 10.1097/JOM.0b013e318156eca3.

Sammi SR et al. (2019). Perfluorooctane sulfonate (PFOS) produces dopaminergic neuropathology in *Caenorhabditis elegans*. Toxicological Sciences. 172:pp 417–434. doi: 10.1093/toxsci/kfz191.

Savitz DA et al. (2012a). Perfluorooctanoic acid exposure and pregnancy outcome in a highly exposed community. Epidemiology. 23:pp 386–392. doi: 10.1097/EDE.0b013e31824cb93b.

Savitz DA et al. (2012b). Relationship of perfluorooctanoic acid exposure to pregnancy outcome based on birth records in the mid-Ohio Valley. Environmental Health Perspectives. 120:pp 1201–1207. doi: 10.1289/ehp.1104752.

Seacat AM. (2002). Subchronic toxicity studies on perfluorooctanesulfonate potassium salt in cynomolgus monkeys. Toxicological Sciences. 68:pp 249–264. doi: 10.1093/toxsci/68.1.249.

Sheng N et al. (2017). Comparative hepatotoxicity of 6:2 fluorotelomer carboxylic acid and 6:2 fluorotelomer sulfonic acid, two fluorinated alternatives to long-chain perfluoroalkyl acids, on adult male mice. Archives of Toxicology. 91:pp 2909–2919. doi: 10.1007/s00204-016-1917-2.

Sheng N et al. (2018). Cytotoxicity of novel fluorinated alternatives to long-chain perfluoroalkyl substances to human liver cell line and their binding capacity to human liver fatty acid binding protein. Archives of Toxicology. 92:pp 359–369. doi: 10.1007/s00204-017-2055-1.

Shi G et al. (2017a). 6:2 fluorotelomer carboxylic acid (6:2 FTCA) exposure induces developmental toxicity and inhibits the formation of erythrocytes during zebrafish embryogenesis. Aquatic Toxicology. 190:pp 53–61. doi: 10.1016/j.aquatox.2017.06.023.

Shi G et al. (2017b). 6:2 Chlorinated polyfluorinated ether sulfonate, a PFOS alternative, induces embryotoxicity and disrupts cardiac development in zebrafish embryos. Aquatic Toxicology. 185:pp 67–75. doi: 10.1016/j.aquatox.2017.02.002.

Skuladottir M et al. (2015). Examining confounding by diet in the association between perfluoroalkyl acids and serum cholesterol in pregnancy. Environmental Research. 143:pp 33–38. doi: 10.1016/j.envres.2015.09.001.

Slotkin TA et al. (2008). Developmental neurotoxicity of perfluorinated chemicals modeled *in vitro*. Environmental Health Perspectives. 116:pp 716–722. doi: 10.1289/ehp.11253.

Song W et al. (2018). Polyfluorinated iodine alkanes regulated distinct breast cancer cell progression through binding with estrogen receptor alpha or beta isoforms. Environmental Pollution. 239:pp 300–307. doi: 10.1016/j.envpol.2018.04.037.

Steenland K, Fletcher T and Savitz DA. (2010). Epidemiologic evidence on the health effects of perfluorooctanoic acid (PFOA). Environmental Health Perspectives. 118:pp 1100–1108. doi: 10.1289/ehp.0901827.

Steenland K and Woskie S. (2012). Cohort mortality study of workers exposed to perfluorooctanoic acid. American Journal of Epidemiology. 176:pp 909–917. doi: 10.1093/aje/kws171.

Steenland K, Zhao L and Winquist A. (2015). A cohort incidence study of workers exposed to perfluorooctanoic acid (PFOA). Occupational and Environmental Medicine. 72:pp 373–380. doi: 10.1136/oemed-2014-102364.

Stein CR, Savitz DA and Bellinger DC. (2013). Perfluorooctanoate and neuropsychological outcomes in children. Epidemiology. 24:pp 590–599. doi: 10.1097/EDE.0b013e3182944432.

Stein CR, Savitz DA and Dougan M. (2009). Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome. American Journal of Epidemiology. 170:pp 837–846. doi: 10.1093/aje/kwp212.

UNEP/POPS/POPRC. (2016a). United Nations Environment Programme, Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee (UNEP/POPS/POPRC): Draft risk profile: Pentadecafluorooctanoic acid (CAS No: 335-67-1, PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds (UNEP/POPS/POPRC.12/3, 14 June 2016). Twelfth meeting, 19–23 September 2016, Rome, Italy.

UNEP/POPS/POPRC. (2016b). United Nations Environment Programme, Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee (UNEP/POPS/POPRC): Additional information document (UNEP/POPS/POPRC.12/INF/5, 18 July 2016). Twelfth meeting, 19–23 September 2016, Rome, Italy.

UNEP/POPS/POPRC. (2016c). United Nations Environment Programme, Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee (UNEP/POPS/POPRC): Comments and responses relating to the draft profile (UNEP/POPS/POPRC.12/INF/6*, 20 July 2016). Twelfth meeting, 19–23 September 2016, Rome, Italy.

UNEP/POPS/POPRC. (2016d). United Nations Environment Programme, Stockholm Convention on Persistent Organic Pollutants, Persistent Organic Pollutants Review Committee (UNEP/POPS/POPRC): Additional comments and responses relating to the third draft (UNEP/POPS/POPRC.12/INF/6/Add1, 14 September 2016). Twelfth meeting, 19–23 September 2016, Rome, Italy.

U.S. EPA. (2020). U.S. Environmental Protection Agency (U.S. EPA): Drinking water health advisories for PFOA and PFOS. Available at: https://www.epa.gov/ground-water-and-drinking-water/drinking-water-health-advisories-pfoa-and-pfos. Accessed 11 May 2020.

U.S. EPA. (2016). U.S. Environmental Protection Agency (U.S. EPA): Health effects support document for perfluorooctanoic acid (PFOA). Office of Water (4304T), Health and Ecological Criteria Division, Washington, DC, EPA Document Number 822-R-16-003, May 2016. Available at: https://www.epa.gov/sites/production/files/2016-05/documents/pfoa_hesd_final_508.pdf. Accessed 28 Nov 2017.

Velez MP, Arbuckle TE and Fraser WD. (2015). Maternal exposure to perfluorinated chemicals and reduced fecundity: the MIREC study. Human Reproduction. 30:pp 701–709. doi: 10.1093/humrep/deu350.

Viberg H, Lee I and Eriksson P. (2013). Adult dose-dependent behavioral and cognitive disturbances after a single neonatal PFHxS dose. Toxicology. 304:pp 185–191. doi: 10.1016/j.tox.2012.12.013.

Vieira VM et al. (2013). Perfluorooctanoic acid exposure and cancer outcomes in a contaminated community: A geographic analysis. Environmental Health Perspectives. 121:pp 318–323. doi: 10.1289/ehp.1205829.

Wang J et al. (2012). Association of perfluorooctanoic acid with HDL cholesterol and circulating miR-26b and miR-199-3p in workers of a fluorochemical plant and nearby residents. Environmental Science & Technology. 46:pp 9274–9281. doi: 10.1021/es300906q.

Weiss JM et al. (2009). Competitive binding of poly- and perfluorinated compounds to the thyroid hormone transport protein transthyretin. Toxicological Sciences. 109:pp 206–216. doi: 10.1093/toxsci/kfp055.

Winquist A and Steenland K. (2014a). Modeled PFOA exposure and coronary artery disease, hypertension, and high cholesterol in community and worker cohorts. Environmental Health Perspectives. 122:pp 1299–1305. doi: 10.1289/ehp.1307943.

Winquist A and Steenland K. (2014b). Perfluorooctanoic acid exposure and thyroid disease in community and worker cohorts. Epidemiology. 25:pp 255–264. doi: 10.1097/EDE.0000000000000040.

Wolf CJ et al. (2008). Activation of mouse and human peroxisome proliferator-activated receptor alpha by perfluoroalkyl acids of different functional groups and chain lengths. Toxicological Sciences. 106:pp 162–171. doi: 10.1093/toxsci/kfn166.

Zhang C et al. (2015). A prospective study of prepregnancy serum concentrations of perfluorochemicals and the risk of gestational diabetes. Fertility and Sterility. 103:pp 184–189. doi: 10.1016/j.fertnstert.2014.10.001.

Zhao B et al. (2010). The inhibition of human and rat 11β-hydroxysteroid dehydrogenase 2 by perfluoroalkylated substances. The Journal of Steroid Biochemistry and Molecular Biology. 125:pp 143–147. doi: 10.1016/j.jsbmb.2010.12.017.