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FINAL

Maximum Contaminant Level Goals (MCLGs) for Three Individual Per- and Polyfluoroalkyl Substances (PFAS) and a Mixture of Four PFAS

Individual MCLGs for Three Per- and Polyfluoroalkyl Substances (PFAS):

- HFPO-DA
- PFNA
- PFHxS

Mixture MCLG for Mixtures of Four PFAS:

- HFPO-DA
- PFNA
- PFHxS
- PFBS

Maximum Contaminant Level Goals (MCLGs) for Three Individual Per- and Polyfluoroalkyl Substances (PFAS) and a Mixture of Four PFAS

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Notices

This document has been reviewed in accordance with EPA policy and approved for publication.

This document provides a summary of information used to develop the final individual MCLGs for HFPO-DA[1], PFNA, and PFHxS and a final MCLG for mixtures of HFPO-DA, PFNA, PFHxS, and/or PFBS. Molecular formulas and Chemical Abstract Service registry numbers (CASRNs) for these four PFAS are as follows:

- HFPO-DA ($C_6F_{11}O_3^-$; CASRN 122499-17-6)
- PFNA ($C_9F_{17}CO_2^-$; CASRN 72007-68-2)
- PFHxS ($C_6F_{13}SO_3^-$; CASRN 108427-53-8)
- PFBS ($C_4F_9SO_3^-$; CASRN 45187-15-3)

These PFAS may exist in multiple forms, such as isomers or associated salts, and each form may have a separate CASRN or no CASRN at all. Additionally, these compounds have various names under different classification systems. However, at environmentally relevant pHs, these PFAS are expected to dissociate in water to their anionic (negatively charged) forms. For instance, HFPO-DA is an anionic molecule which has an ammonium salt (CASRN 62037-80-3), a conjugate acid (CASRN 13252-13-6), a potassium salt (CASRN 67118-55-2), and an acyl fluoride precursor (CASRN 2062-98-8), among other variations. At environmentally relevant pHs these all dissociate into the propanoate/anion form (CASRN 122499-17-6). Each PFAS listed has multiple variants with differing chemical connectivity, but the same molecular composition (known as isomers). Commonly, the isomeric composition of PFAS is categorized as 'linear,' consisting of an unbranched alkyl chain, or 'branched,' encompassing a potentially diverse group of molecules including at least one, but potentially more, offshoots from the linear molecule. While broadly similar, isomeric molecules may have differences in chemical properties. The final National Primary Drinking Water Regulation for PFAS covers all salts, isomers, precursors, and derivatives of the chemicals listed, including derivatives other than the anionic form which might be created or identified.

Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

[1]The EPA notes that HFPO-DA is used in a processing aid technology developed by DuPont to make fluoropolymers without using perfluorooctanoic acid (PFOA). The chemicals associated with this process are commonly known as GenX Chemicals and the term is often used interchangeably for HFPO-DA.

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Abbreviations and Acronyms

AFFF	aqueous film-forming foam
AMAP	Arctic Monitoring and Assessment Programme
AOP	adverse outcome pathway
ATSDR	Agency for Toxic Substances and Disease Registry
BAF	bioaccumulation factor
BDL	below the detection limit
BMD	benchmark dose
CASRN	Chemical Abstract Service registry number
CDC	Centers for Disease Control and Prevention
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CTEPP	Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants
DA	dose addition
DF	detection frequency
DWI-BW	body weight-adjusted drinking water intake
DWTP	drinking water treatment plant
DWR	durable water repellent
E	duration-relevant exposure
EEE	electrical and electronic equipment
EFSA	European Food Safety Authority
EPA	U.S. Environmental Protection Agency
FAO	Food and Agriculture Organization Area
FCM	food contact materials
FDA	U.S. Food and Drug Administration
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
GCA	groundwater contamination area
GenX chemicals	hexafluoropropylene oxide dimer acid (HFPO-DA) and HFPO-DA ammonium salt
GD	gestational day
g/L	grams per liter
HA	health advisory
HBWC	health-based water concentration
HED	human equivalent dose
HDPE	high-density polyethylene
HFPO	hexafluoropropylene oxide
HI	hazard index
HQ	hazard quotient
IA	integrated addition
IRIS	Integrated Risk Information System
K+PFBS	potassium perfluorobutane sulfonate
L/kg/day	liters per kilogram body weight per day
LAS	land application site
LB	lower bound
LOAEL	lowest-observed-adverse-effect level
LOD	limit of detection
LOQ	limit of quantitation
MAMA	Methods Advancement for Milk Analysis
MCLG	Maximum Contaminant Level Goal
MDL	method detection limit

MF	modifying factor
mg/kg/day	milligrams per kilogram body weight per day
mg/L	milligrams per liter
MOA	mode of action
MRL	minimal risk level
mRNA	messenger ribonucleic acid
MSW	municipal solid waste
MQL	method quantification limit
NAS	National Academy of Sciences
NCDHHS	North Carolina Department of Health and Human Services
NCDEQ	North Carolina Department of Environmental Quality
ng/g dw	naanograms per gram dry weight
ng/kg	nanograms per kilogram
ng/L	nanograms per liter
NHANES	National Health and Nutrition Examination Survey
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NPDWR	National Primary Drinking Water Regulation
NRSA	National Rivers and Streams Assessment
NTP	National Toxicology Program
OECD	Organisation for Economic Co-operation and Development
ORD	Office of Research and Development
PECO	Population, Exposure, Comparator, and Outcome
PFAA	perfluoroalkyl acids
PFAS	per- and polyfluoroalkyl substances
PFBS	perfluorobutanesulfonic acid
PFC	perfluorochemicals
PFCA	perfluoroalkyl carboxylic acids
PFHpA	perfluoroheptanoic acid
PFHxA	perfluorohexanoic acid
PFHxS	perfluorohexanesulfonic acid
PFNA	perfluorononanoic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctanesulfonic acid
pg/g	picograms per gram
pg/L	picograms per liter
pg/m^3	picograms per cubic meter
PHG	provisional health goal
PND	postnatal day
POD	point of departure
POP	persistent organic pollutants
PPARα	peroxisome proliferator-activated receptor alpha
ppt	parts per trillion
PSA	prostate-specific antigen
PWS	public water system
RA	response addition
RfD	reference dose
RfV	reference value
RPF	relative potency factor
RSC	relative source contribution
SAB	Science Advisory Board
SDWA	Safe Drinking Water Act

SLEA	Screening Level Exposure Assessment
SPM	suspended particulate matter
TOFMS	time-of-flight mass spectrometry
TOSHI	target-organ-specific hazard indices
TRI	Toxics Release Inventory
TSCATS	Toxic Substances Control Act Test Submissions
TTD	target organ toxicity dose
UB	upper bound
UCMR	Unregulated Contaminant Monitoring Rule
UCMR 3	third Unregulated Contaminant Monitoring Rule
UF	uncertainty factor
UF_A	interspecies uncertainty factor
UF_D	database uncertainty factor
UF_H	human interindividual variability uncertainty factor
UF_S	extrapolation from subchronic-to-chronic exposure duration uncertainty factor
µg/kg	micrograms per kilogram
µg/L	micrograms per liter
$\mu g/m^2$	micrograms per square meter
WOS	Web of Science
WWTP	wastewater treatment plant

1 Introduction and Background

1.1 Purpose

Section 1412(a)(3) of the Safe Drinking Water Act (SDWA) requires the Administrator of the U.S. Environmental Protection Agency (EPA) to propose a Maximum Contaminant Level Goal (MCLG) simultaneously with the National Primary Drinking Water Regulation (NPDWR). The MCLG is set, as defined in Section 1412(b)(4)(A), at "the level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety." The MCLG incorporates a margin of safety to reflect scientific uncertainty and, in some cases, the particular susceptibility of some groups (e.g., children) within the general population. Consistent with SDWA 1412(b)(3)(C)(i)(V), in developing the MCLG, the EPA considers "the effects of the contaminant on the general population and on groups within the general population such as infants, children, pregnant women, the elderly, individuals with a history of serious illness, or other subpopulations that are identified as likely to be at greater risk of adverse health effects due to exposure to contaminants in drinking water than the general population." Other factors considered in determining MCLGs include health effects data for drinking water contaminants and potential sources of exposure other than drinking water. MCLGs are not regulatory levels and are not enforceable.

The purpose of this document is to provide a summary of the health effects and exposure information and analyses and to describe the derivation of the EPA's final MCLGs for the following per- and polyfluoroalkyl substances (PFAS), for which the EPA is finalizing a NPDWR: hexafluoropropylene oxide dimer acid (HFPO-DA) (also known as GenX chemicals)[2], perfluorononanoic acid (PFNA), perfluorohexanesulfonic acid (PFHxS), and perfluorobutane sulfonic acid (PFBS).[3] The EPA is finalizing individual MCLGs for HFPO-DA, PFNA, and PFHxS. The EPA is also finalizing a PFAS mixture MCLG for mixtures of two or more of four PFAS—HFPO-DA, PFNA, PFHxS, and PFBS—that accounts for dose-additive health effects when these PFAS co-occur in drinking water. The PFAS mixture MCLG is based on a hazard index (HI) approach, a commonly used component-based mixtures risk assessment method (see Section 1.4 and USEPA, 2024a). This document summarizes key elements (e.g., reference doses (RfDs)) from recently published, peer-reviewed, publicly available human health toxicity assessments for HFPO-DA (USEPA, 2021c), PFBS (USEPA, 2021d), PFNA (ATSDR, 2021), and PFHxS (ATSDR, 2021) that the EPA used to develop MCLGs for HFPO-DA, PFNA, and PFHxS and an MCLG for mixtures of two or more of these PFAS plus PFBS. The MCLG represents the level below which adverse health effects over a lifetime of exposure are not expected to occur, including for sensitive populations and life stages, and with an adequate margin of safety. This document is not intended to be an exhaustive description of all health effects or modeled endpoints (i.e., human health toxicity assessment) nor is it a drinking water health advisory (HA).

[2]The EPA notes that HFPO-DA is used in a processing aid technology developed by DuPont to make fluoropolymers without using PFOA. The chemicals associated with this process are commonly known as GenX chemicals and the term is often used interchangeably for HFPO-DA along with its ammonium salt.

[3]Note: The EPA is also finalizing individual MCLGs for two other PFAS: perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) (see USEPA, 2024c)).

1.2 Occurrence and Co-Occurrence of PFAS in Drinking Water

Improved analytical monitoring and detection methods have enabled detection of the occurrence and co-occurrence of multiple PFAS in drinking water, ambient surface waters, aquatic organisms, and other environmental media (see Appendices A through D and USEPA, 2024d). The two PFAS perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) have historically been target analytes, and the focus of many environmental monitoring studies. More recent monitoring studies, however, have focused on additional PFAS via advanced analytical instruments/methods and nontargeted analysis (De Silva et al., 2021; McCord et al., 2020; McCord and Strynar, 2019).

The EPA uses the Unregulated Contaminant Monitoring Rule (UCMR) to collect occurrence data nationwide for contaminants that are suspected to be present in drinking water. Between 2013 and 2015, the EPA's third UCMR (UCMR 3) required all large public water systems (PWSs) (each serving more than 10,000 people) and a statistically selected, nationally representative sample of 800 small PWSs (each serving 10,000 people or fewer) to monitor for 30 unregulated contaminants in drinking water, including PFNA, PFHxS, and PFBS. In addition to the UCMR 3 data collection, many states have undertaken more recent efforts to monitor for PFAS in both source and finished drinking water using newer analytical methods and reflecting lower reporting limits than those in UCMR 3. These results and other peer-reviewed studies show continued PFAS occurrence and co-occurrence in multiple geographic locations (USEPA, 2024b; Cadwallader et al., 2022). These data also show certain PFAS (including PFNA, PFHxS, and PFBS) measured at lower concentrations and significantly greater frequencies than were measured under UCMR 3. Additionally, these state monitoring data include results for HFPO-DA (which was not included in the suite of PFAS analyzed in UCMR 3) and demonstrate HFPO-DA occurrence (and co-occurrence with other PFAS) in drinking water. From 2023–2025, monitoring data for 29 PFAS including HFPO-DA, PFBS, PFNA, and PFHxS are being collected under UCMR 5. These drinking water occurrence and co-occurrence data for HFPO-DA, PFNA, PFHxS, PFBS, as well as additional PFAS are detailed in the EPA's *PFAS Occurrence and Contaminant Background Support Document for the Final PFAS NPDWR* (USEPA, 2024b).

1.3 Dose Additivity for PFAS Mixtures

1.3.1 Overview of Scientific Support

Dose additivity means that when two or more chemicals (in this case, PFHxS, PFNA, HFPO-DA, and/or PFBS) exist in one mixture, the risk of adverse health effects following exposure to the mixture is equal to the sum of the individual doses or concentrations scaled for potency (USEPA, 2000a). Studies with PFAS and other classes of chemicals support the health-protective conclusion that toxicologically similar chemicals (i.e., those that elicit similar observed adverse effects following individual exposure, even if at different exposure levels) should be assumed to act in a dose-additive manner when present in a mixture unless data demonstrate otherwise. Experimental data demonstrate that PFAS elicit similar adverse health effects on several of the same biological systems and functions including thyroid hormone signaling, lipid synthesis and metabolism, development, and immune and liver function. Thus, exposure to these PFAS, at doses that individually would not likely result in adverse health effects, when combined in a mixture may pose health risks.

Numerous published studies across multiple chemical classes, biological effects, and study designs support a dose-additive mixture assessment approach for PFAS because they demonstrate that experimentally observed responses to exposure to PFAS mixtures and other chemical mixtures are consistent with modeled predictions of dose additivity (see the EPA's *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances* (PFAS) (hereafter "PFAS Mixtures Framework;" (USEPA, 2024a)). Since the EPA's draft PFAS Mixtures Framework underwent SAB review in 2021−2022, new studies from the EPA and others have provided robust evidence of combined toxicity of PFAS in mixtures, corroborating and confirming earlier findings (USEPA, 2024a; e.g., Conley et al., 2023; Conley et al., 2022). Additionally, the National Academies of Sciences, Engineering, and Medicine (NASEM, 2022) recently recommended that clinicians apply an additive approach for evaluating patient levels of PFAS currently measured in the National Health and Nutrition Examination Survey (NHANES) in order to protect human health from additive effects from PFAS co-exposure.

Data from *in vivo* studies that rigorously tested accuracy of Dose Additivity (DA), Integrated Addition (IA), and Response Additivity (RA) model predictions of mixtures with components that disrupted the same pathways (i.e., were toxicologically similar) demonstrated that DA models provided predictions that were better than or equal to IA and RA predictions of the observed mixture effects (Section 3.2 in USEPA, 2024a). In some circumstances the different additivity models provide highly similar predictions of mixture effects and thus are essentially equally effective. In situations where the models provide very different predictions, experimental data have demonstrated that DA-based models consistently provide more accurate predictions of observed mixture effects than RA or IA. This strongly supports the use of dose additivity as the default method for estimating mixture effects of compounds that are toxicologically similar. The National Academy of Sciences (NAS) conclusions on phthalates (and related chemicals) (NRC, 2008) and systematic reviews of the published literature (USEPA, 2024a; Martin et al., 2021; Boobis et al., 2011) support dose additivity as the default model for estimating mixture effects in some circumstances, even when the mixtures included chemicals with diverse MOAs (but the same target organs/effects). Systematic reviews of mixture studies with chemical classes other than PFAS also indicate that departures from dose additivity are uncommon and rarely exceed minor deviations (~2-fold) from predictions based on dose additivity (Martin et al., 2021; Boobis et al., 2011). Boobis et al. (2011) examined literature from 1990 to 2008 that discussed synergy in mammalian test systems, with an emphasis on "low dose" studies. They found that of the 11 available studies with synergy data that reported the magnitude of the difference between the dose-additive estimates of toxicity and observed toxicity, six studies reported magnitudes of synergy that were generally small, and the authors concluded that deviations from dose additivity at low doses were not common. Additionally, Martin et al. (2021) reviewed more than 1,200 mixture studies and concluded that there was little evidence for synergy (greater than additive effects) or antagonism (less than additive effects) among chemicals in mixtures, and that dose additivity should be considered as the default model. This supports the health-protective conclusion that a mixture of PFHxS, PFNA, HFPO-DA, and/or PFBS should be assumed to act in a dose-additive manner unless data demonstrate otherwise.

Although some available *in vitro* studies do not provide conclusive evidence of dose additivity for PFAS mixtures, their results also do not justify drawing a conclusion other than dose additivity. For example, a study on PFAS cytotoxicity in a human liver cell line (Ojo et al., 2020)

reported synergistic effects of mixtures of perfluoroalkyl acids (PFAAs; a type of PFAS) compared to a dose addition model, but also reported evidence of antagonistic effects. Other *in vitro* studies that have assessed PFAS mixture-based effects do not report these results; that is, they do not offer strong evidence for synergistic or antagonistic effects, particularly at environmentally relevant concentrations. For example, Wolf et al. (2014) evaluated *in vitro* PPARα activation and reported that effects seen following exposure to combinations of different PFAS were consistent with dose additivity in the lower tested concentration ranges. Wolf et al. (2014) also reported slightly greater than additive effects at higher test concentrations (approximately 500 parts per billion to over 800 parts per million); however, in environmental media such as drinking water, PFAS are not likely to occur at these higher concentrations (e.g., see USEPA, 2024b). Carr et al. (2013) reported slightly less than additive effects for *in vitro* PPARα activation of binary mixtures of PFAAs including PFOA, PFNA, PFOS, and PFHxS. Addicks et al. (2023) evaluated mRNA transcription in primary human liver spheroids exposed to seven different PFAS mixtures and found that all tested mixtures produced effects that were consistent with effects predicted using dose addition. To summarize, the available *in vitro* data do not support a conclusion other than dose additivity for PFAS mixtures.

Available *in vivo* data on this subject similarly support dose additivity. Two studies with PFAS mixtures in zebrafish reported no indications of synergy (Menger et al., 2020; Ding et al., 2013). Additionally, recent EPA Office of Research and Development (ORD) studies provide robust evidence that PFAS behave in a dose-additive manner (Gray et al., 2024; Conley et al., 2023; Conley et al., 2022). For example, results of a developmental toxicity study of exposure to PFOA and PFOS mixtures in rats showed that the observed results for almost all tested endpoints were consistent with dose additivity (Conley et al., 2022). Likewise, a rat developmental study of a PFAS mixture of PFOS, HFPO-DA, and Nafion byproduct 2 (an emerging polyfluoroethersulfonic acid compound recently detected in human serum (Kotlarz et al., 2020)) found that multiple tested endpoints in both parental females and offspring conformed to dose additivity and no endpoints demonstrated synergy (Conley et al., 2023).

Additionally, as described in the final PFAS Mixtures Framework (USEPA, 2024a), over the past two decades, many *in vivo* experimental animal studies have been published in which toxicity of chemical mixtures has been systematically evaluated (e.g., Conley et al., 2023; Conley et al., 2022; Martin et al., 2021; Hass et al., 2017; Howdeshell et al., 2015; Moser et al., 2012; Rider et al., 2010; Kortenkamp and Haas, 2009; Rider et al., 2009; Rider et al., 2008; Crofton et al., 2005; Moser et al., 2005; Walker et al., 2005; Gennings et al., 2004; Altenburger et al., 2000). These studies span different chemical classes, proposed MOAs, and health outcomes, but they generally show that chemicals in mixtures typically act dose additively. Even when mixture components with different MOAs/adverse outcome pathways (AOPs) are combined, they induce toxic effects consistent with dose additivity (Rider et al., 2009). This concept was further articulated in the National Research Council's 2008 report *Phthalates and cumulative risk assessment: The tasks ahead* (NRC, 2008), wherein that expert panel provided significant evidence that mixture components that elicit similar adverse health effects individually will demonstrate dose additivity when combined in a mixture, regardless of similarity in MOA.

This evidence base supports the longstanding recommendation in EPA chemical mixtures guidance for dose additivity as a default approach for evaluation of mixture toxicity (USEPA,

2000a, 1986). This position is further supported and articulated in the newly published EPA Risk Assessment Forum's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (USEPA, 2023b).

1.3.1.1 Science Advisory Board Support

The EPA's conclusions regarding dose additivity of PFAS were supported by the SAB during its 2021–2022 review of the EPA's draft *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances*. The EPA directly asked the SAB for feedback on PFAS dose additivity as part of its review of technical materials supporting development of the PFAS MCLG and NPDWR. Specifically, the EPA asked the SAB to, "[p]lease comment on the appropriateness of this approach for a component-based mixture evaluation of PFAS under an assumption of dose additivity" (USEPA SAB, 2022). The SAB strongly supported the scientific soundness of this approach when evaluating PFAS and concurred that it was a health protective conclusion. For example, the SAB said:

> "The SAB supports dose additivity based on a common outcome, instead of a common mode of action as a health protective default assumption and does not propose another default approach." (USEPA SAB, 2022)
>
> "…The information included in the draft framework supports the conclusion that toxicological interactions of chemical mixtures are frequently additive or close to additive. It also supports the conclusion that dose additivity is a public health protective assumption that typically does not underestimate the toxicity of a mixture..." (USEPA SAB, 2022)
>
> "The SAB Panel agrees with use of the default assumption of dose additivity when evaluating PFAS mixtures that have similar effects and concludes that this assumption is health protective." (USEPA SAB, 2022)
>
> "…dose additivity can provide an estimate of composite effects." (USEPA SAB, 2022)

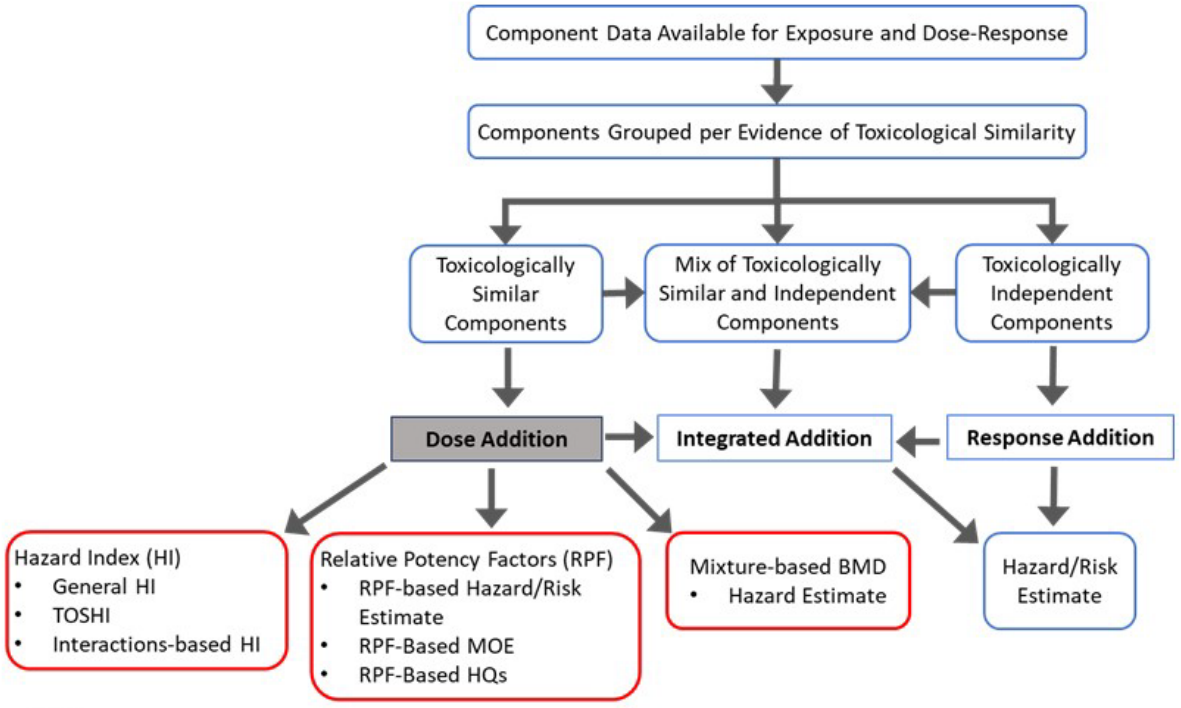
While the SAB also noted that there remain some questions about PFAS interaction in mixtures (USEPA SAB, 2022), the available data justify an approach that accounts for PFAS dose additivity. As described above, studies that have assessed PFAS mixture-based effects do not provide support for a conclusion other than dose additivity (i.e., they do not offer strong evidence for synergistic/antagonistic effects) (USEPA, 2024a).

1.3.2 Toxicological Similarity of PFHxS, PFNA, HFPO-DA, and PFBS

1.3.2.1 Background on Concept of "Toxicological Similarity"

This concept and application of dose additivity for "toxicologically similar components" in mixtures assessment is consistent with EPA mixtures guidance (USEPA, 2000a, 1986) and the EPA Risk Assessment Forum's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (USEPA, 2023b). Specifically, the EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a) notes that although the shared MOA metric for application of dose addition is optimal, MOA data are not always available and that toxicological similarity in the context of mixtures risk assessment can be based on adverse

effects observed at the organ or system level (USEPA, 2000a). This concept is further described in the EPA Risk Assessment Forum's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (USEPA, 2023b): "The primary criterion for choosing between dose addition and response addition methods is toxicological similarity among the chemicals in the mixture [(USEPA, 2000a)]. "Toxicological similarity" is used here as an overarching concept with a wide range of specificity across levels of biological organization, allowing similarity judgments to be tailored to both the specific goals of the mixture risk assessment and the availability of hazard and dose-response information across components." Unless there are available data that suggest deviation(s) from dose additivity, mixture chemicals that are "toxicologically similar" (e.g., same/similar effect or profile of effect[s], regardless of differences in potencies) prototypically behave dose additively. This concept is depicted in Figure 1-1 below, which shows that dose additivity is the logical default approach for "toxicologically similar" components and that component-based mixture assessment approaches including hazard index (HI), relative potency factor (RPF), and mixture-benchmark dose (Mixture-BMD) are options for mixture assessment in such cases (see Section 1.5).



Notes:
Modification of Figure 4-3b in USEPA (2007). BMD = benchmark dose; HI = hazard index; HQ = hazard quotient; MOE = margin of exposure; RPF = relative potency factor; TOSHI = target organ-specific hazard index.
Component-based methods selection is based on the relevant evidence supporting toxicological similarity (dose addition) or toxicological independence (response addition or effect summation). Integrated addition methods are reserved for mixtures of component chemicals that demonstrate a profile of both toxicological similarity and independence.

Figure 1-1. Flow chart for evaluating chemical mixtures using component-based additive methods. (Reproduction of Figure 2-1 from USEPA, 2024a).

1.3.2.2 Overview of Scientific Support

The EPA's approach is to evaluate risks from exposure to mixtures of PFAS based on similar adverse health effects (but with differing potencies for effect(s)) of the individual PFAS mixture components, rather than similar MOA. MOA describes key changes in cellular or molecular

events that may cause functional or structural changes that lead to adverse health effects and can be a useful metric by which risk can be assessed. It is considered a key determinant of chemical toxicity, and chemicals can often be classified by their type of toxicity pathway(s) or MOA(s). PFAS are an emerging chemical class, and MOA data are limited or entirely lacking for many PFAS. Although similarities among some PFAS have been shown at the level of molecular and cellular perturbations, no conserved MOAs have been identified across PFAS for noncancer health effects assessed thus far. Therefore, the EPA's approach for assessing risks of PFAS mixtures is based on the conclusion that PFAS that are "toxicologically similar"—that is, elicit the same or similar adverse health effects (but with differing potencies for effect(s))—will produce dose-additive effects from co-exposures (see USEPA, 2024a).

Available epidemiological and animal toxicological data demonstrate that exposure to each of these four PFAS (PFHxS, PFNA, HFPO-DA, and PFBS) is associated with many of the same or similar adverse health endpoints and outcomes, and thus they are "toxicologically similar" (see Table 1-1, Table 1-2, and Table 1-3 below). Further, these four PFAS are well-studied PFAS for which the EPA or ATSDR has developed human health assessments and toxicity reference values (i.e., reference doses (RfDs), minimal risk levels (MRLs)). Available animal toxicological and/or epidemiological studies demonstrate that PFHxS, PFNA, HFPO-DA, and PFBS are documented to affect at least five (5) of the same major health outcomes: lipids, developmental, immune, endocrine, and hematologic (Table 1-1). Similarly, according to the 2023 Interagency PFAS Report to Congress (US OSTP, 2023), available animal toxicological data show that PFHxS, PFNA, HFPO-DA, and PFBS significantly affect at least eight (8) of the same major health effect domains: body weight, respiratory, hepatic, renal, endocrine, immunological, reproductive, and developmental (Table 1-2). Furthermore, numerous *in vivo* and *in vitro* studies demonstrate that these four PFAS share many common health effects across diverse health outcome categories (e.g., developmental, immunological, and endocrine), and that they induce some of the same effects at the molecular level along biological pathways. Table 1-3 below shows specific endpoints shared across these four PFAS, including toxicologically relevant molecular perturbations (*in vitro*), and health effects (*in vivo*) from oral repeated-dose studies in rats and/or mice (note that this table is a summary of select studies for illustrative purposes and should not be construed to represent a systematic review or MOA analysis).

Table 1-1. Affected health outcomes in animal toxicological and/or epidemiological studies for the four PFAS included in the HI MCLG (adapted from Table 6-7 in USEPA, 2024d).

Health Outcome	HFPO-DA	PFNA	PFHxS	PFBS
Lipids	X	X	X	X
Developmental	X	X	X	X
Hepatic	X	X	X	-
Immune	X	X	X	X
Endocrine	X	X	X	X
Renal	X	-	-	X
Hematologic	X	X	X	X

Notes: (X) Health outcome examined, evidence of association; (-) health outcome examined, no evidence of association.

Table 1-2. Affected health endpoints in animal toxicity studies for the four PFAS included in the HI MCLG (adapted from Table 4 in US OSTP, 2023).

Health Endpoint	HFPO-DA	PFNA	PFHxS	PFBS
Body weight	X	X	X	X
Respiratory	X	X	X	X
Cardiovascular			X	X
Gastrointestinal		X	X	X
Hematological	X		X	X
Musculoskeletal			X	X
Hepatic	X	X	X	X
Renal	X	X	X	X
Dermal	X			
Ocular	X			X
Endocrine	X	X	X	X
Immunological	X	X	X	X
Neurological	X		X	X
Reproductive	X	X	X	X
Developmental	X	X	X	X
Other noncancer	X	X		

Notes: (X) Health outcome examined, evidence of association.

Table 1-3. Specific Endpoints Affected by One or More of the Four PFAS Included in the HI MCLG.

Endpoint	HFPO-DA	PFNA	PFHxS	PFBS
Molecular/Cellular Perturbations				
PPAR alpha binding/activation	Evans et al. (2022); Nielsen et al. (2021)	Evans et al. (2022); Nielsen et al. (2021); Rosenmai et al. (2018); Wolf et al. (2012)	Evans et al. (2022); Nielsen et al. (2021); Rosenmai et al. (2018); Wolf et al. (2012)	Evans et al. (2022); Rosenmai et al. (2018); Wolf et al. (2012)
PPAR gamma binding/activation	Evans et al. (2022); Houck et al. (2021)	Evans et al. (2022); Houck et al. (2021)	Evans et al. (2022); Houck et al. (2021)	Evans et al. (2022)
Liver gene induction (PPAR signaling pathway)	Conley et al. (2019); Blake et al. (2022)	NTP (2019b); Rosen et al. (2017); Rosen et al. (2013)	NTP (2019c); Rosen et al. (2017); Rosen et al. (2013); Chang et al. (2018)	NTP (2019c); Rosen et al. (2013)
Liver gene induction (CAR signaling pathway)	-	NTP (2019b)	NTP (2019c)	NTP (2019c)

Endpoint	HFPO-DA	PFNA	PFHxS	PFBS
Serum bile salts/acids (increased)	DuPont (2010c)	NTP (2019b)	-	NTP (2019c)
Serum globulin (reduced)	DuPont (2009); DuPont (2008b); DuPont (2008a)	NTP (2019b)	NTP (2019c)	NTP (2019c)
Serum albumin:globulin (increased)	DuPont (2009); DuPont (2008b); DuPont (2008a)	NTP (2019b)	NTP (2019c); Butenhoff et al. (2009)	NTP (2019c)
Health Effects				
Serum lipids (reduced cholesterol and/or triglycerides)	DuPont (2008b); DuPont (2009); DuPont (2008a)	NTP (2019b)	NTP (2019c); Chang et al. (2018); Butenhoff et al. (2009)	NTP (2019c)
Serum liver enzymes (increased ALT, AST, and/or ALKP)	DuPont (2008b); DuPont (2010c)	NTP (2019b)	-	NTP (2019c)
Serum thyroid hormones (reduced T4, T3)	Conley et al. (2019)	NTP (2019b)	NTP (2019c); Gilbert et al. (2021)	NTP (2019c)
Liver weight (increased)	DuPont (2008b); Blake et al. (2020); Conley et al. (2021); Conley et al. (2019); DuPont (2009); Rushing et al. (2017); DuPont (2008a)	NTP (2019b); Das et al. (2015)	NTP (2019c); Chang et al. (2018)	NTP (2019c); Lieder et al. (2009b)
Liver histopathology (nonneoplastic effects)	DuPont (2008b); DuPont (2010c); DuPont	NTP (2019b)	Chang et al. (2018); NTP (2019c)	NTP (2019c)

Endpoint	HFPO-DA	PFNA	PFHxS	PFBS
	(2008a); NTP (2019a)			
Thymus weight (reduced)	DuPont (2009)	NTP (2019b)	-	NTP (2019c)
Spleen weight (reduced)	-	NTP (2019b)	-	NTP (2019c); Lieder et al. (2009a)
Kidney weight (increased)	DuPont (2009); DuPont (2008b)	NTP (2019b)	NTP (2019c)	NTP (2019c)
Reduced fetal/pup bodyweight	Conley et al. (2021); DuPont (2010a); DuPont (2010b)	Das et al. (2015)	-	Feng et al. (2017)
Reduced fetal/pup survival	Conley et al. (2021)	Das et al. (2015)	Chang et al. (2018)	-
Reduced adult bodyweight	DuPont (2013)	NTP (2019b)	-	NTP (2019c); Lieder et al. (2009a)
Overt toxicity (lethality)	DuPont (2009)	NTP (2019b)	-	NTP (2019c)

Note: (-) indicates no statistically significant effect reported by study authors of cited studies at dose levels and dose interval used and/or effect not measured in cited studies.

1.3.2.3 Science Advisory Board Support

The SAB strongly supported the EPA's decision to focus on similarity of adverse health effects rather than similarity of MOA to assess risks of exposure to PFAS mixtures during its 2021–2022 review of the EPA's draft PFAS Mixtures Framework. Specifically, the EPA asked the SAB, "If common toxicity endpoint/health effect is not considered an optimal similarity domain for those PFAS with limited or no available MOA-type data, please provide specific alternative methodologies for integrating such chemicals into a component-based mixture evaluation(s)" (USEPA SAB, 2022). The SAB strongly supported the EPA's approach of using a similar toxicity endpoint/health effect instead of a common MOA as a default approach for evaluating mixtures of PFAS using dose additivity and did not recommend an alternative methodology. The SAB panel stated that:

> "The Panel agreed with use of a similar toxicity endpoint/health effect instead of a common MOA as a default approach for evaluating mixtures of PFAS. This approach makes sense because multiple physiological systems and multiple MOAs can contribute

> to a common health outcome. Human function is based on an integrated system of systems and not on single molecular changes as the sole drivers of any health outcome. The Panel concluded that rather than the common MOA, as presented in the EPA draft mixtures document, common physiological outcomes should be the defining position" (USEPA SAB, 2022).
>
> "Furthermore, many PFAS, including the four used in the examples in the draft EPA mixtures document and others, elicit effects on multiple biological pathways that have common adverse outcomes in several biological systems (e.g., hepatic, thyroid, lipid synthesis and metabolism, developmental and immune toxicities)" (USEPA SAB, 2022).

1.3.3 Summary

The available scientific evidence supports the conclusion that PFAS that elicit similar adverse health effects following individual exposure (even if with differing potencies for effect(s)) should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise. This means that individual PFAS, each at doses that are not anticipated to result in adverse health effects, when combined in a mixture may result in adverse health effects. (For a more complete discussion of the evidence supporting dose additivity as the default approach for assessing mixtures of PFAS, see the final PFAS Mixtures Framework (USEPA, 2024a)). The EPA's conclusions regarding PFAS dose additivity were supported by the SAB during its review of the EPA's draft PFAS Mixtures Framework (USEPA SAB, 2022) and are consistent with longstanding agency chemical mixtures guidance (USEPA, 2000a, 1986) and a recent EPA white paper (USEPA, 2023b). The SAB also strongly supported the EPA's default assumption of dose additivity in the absence of other information and the EPA's approach of using similar toxicity endpoints/health effects instead of a common MOA for evaluating mixtures of PFAS (USEPA SAB, 2022). This approach of basing the concept of toxicological similarity on same/similar adverse effects in the absence of adequate MOA information is also consistent with the EPA's chemical mixtures guidance (USEPA, 2000a, 1986) and the EPA Risk Assessment Forum's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (USEPA, 2023b).

1.4 General Hazard Index (HI) Approach for PFAS Mixtures

1.4.1 Background/Overview

The EPA's final determination that mixtures of the four PFAS "may have an adverse effect on the health of persons" is based on the health-protective conclusion that chemicals that are toxicologically similar (i.e., have similar observed adverse health effects, regardless of potency differences) following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise (see Section 1.3 and USEPA, 2024a). This means that where drinking water contains any combination of two or more of the four PFAS that are the subject of the NPDWR—PFHxS, PFNA, HFPO-DA, and PFBS—the hazard associated with each PFAS in the mixture must be added together to determine whether the mixture exceeds a level of public health concern.

The SDWA requires the agency to establish a health-based MCLG set at "a level at which no known or anticipated adverse effects on the health of persons occur and which allows for an adequate margin of safety." The MCLG "incorporates a margin of safety to reflect scientific

uncertainty and, in some cases, the particular susceptibility of some groups (e.g., children) within the general population" (see S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3). In the context of this NPDWR, the general HI is the approach used to determine if a mixture of two or more of four PFAS in drinking water—PFHxS, PFNA, HFPO-DA, and PFBS—exceeds the level of health concern with a margin of safety. A general HI equal to 1 is the MCLG for any mixture of these four PFAS.

Based on the scientific record, each of these four PFAS has a health-based water concentration (HBWC), which is set at the level below which adverse effects are not anticipated to occur and allows for an adequate margin of safety (see Section 2). The general HI approach accounts for the measured drinking water concentration of each of the four PFAS in the mixture, and the toxicity (represented by the HBWC) of each of the four PFAS. The general HI is derived by first calculating the ratio of the measured concentration of each of the four PFAS to its toxicity (the HBWC) to yield a "hazard quotient" (HQ) for each of the four PFAS. HQs are then added together to account for the dose-additive health concerns that these PFAS present. Adding the four HQs together yields the general HI. If the general HI exceeds 1, then the hazard from the combined amounts of the four PFAS present together in drinking water exceeds a level of public health concern.

The EPA has determined that in the context of SDWA, the general HI is an appropriate methodology for determining the level at and below which there are no known or anticipated adverse human health effects with an adequate margin of safety with respect to certain PFAS mixtures in drinking water. The general HI approach is the most practical approach for establishing an MCLG for PFAS mixtures that meets the statutory requirements outlined in Section 1412(b)(1)(A) of SDWA. As noted above, the general HI assesses the exposure level of each component PFAS relative to its HBWC, which is based on the most sensitive known adverse health effect (based on the weight of evidence) and considers sensitive population(s) and life stage(s) as well as potential exposure sources beyond drinking water. The general HI also accounts for dose-additive health concerns by summing the hazard contributions from each mixture component. In this way, the general HI approach ensures that mixtures of two or more of these four PFAS are not exceeding the level below which there are no known or anticipated adverse health effects and allows for an adequate margin of safety.

1.4.2 Consideration of Mixtures Assessment Approaches and Selection of General HI Approach

In selecting an approach to develop the MCLG for mixtures of two or more of four PFAS—PFHxS, PFNA, HFPO-DA, and PFBS—the EPA followed its *Guidelines for the Health Risk Assessment of Chemical Mixtures* (USEPA, 1986), *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a), and *Risk Assessment Guidance for Superfund* (e.g., USEPA, 1991b). As described below, the EPA first considered whether data were available for the whole mixture or a "sufficiently similar" mixture, per agency guidance (USEPA, 2000a, 1986), and then considered several mixture component-based assessment methods (USEPA, 2024a), ultimately selecting the general HI approach for PFAS mixture MCLG derivation.

The EPA's guidance documents (USEPA, 2000a, 1991b, 1986) propose a hierarchy of mixtures assessment approaches, with the preferred approach being evaluation of health risk using hazard and dose-response data for a specific whole mixture of concern, or alternatively, a "sufficiently similar" mixture. Whole-mixture data are rare; there are often many chemical combinations and proportions in the environment (e.g., parent chemicals, metabolites, and/or abiotic degradants), introducing a level of complexity that complicates evaluation and characterization. The exponential diversity of PFAS co-occurring in different combinations and proportions makes whole-mixture evaluations complex and unfeasible. Due to differing fate and transport properties, biotic (metabolism) and abiotic (degradation) processes, pH, ultraviolet radiation, media temperature, and so on, chemicals commonly co-occur in the environment in an array of parent species, metabolites, and/or degradants, making characterization and evaluation of any given mixture complicated. In controlled experimental study designs, whole mixtures can be assembled with defined component membership and proportions, but the relevance of toxicity associated with exposure to a defined mixture in a laboratory setting may not be translatable to environmental mixtures of different component combinations and proportions across time and space in environmental media. The complexities associated with the diversity of PFAS co-occurring in different component proportions (see USEPA, 2024b) make evaluating each unique whole mixture of PFAS intractable. This is why component-based mixture assessment approaches are considered particularly useful and appropriate for addressing human exposure(s) to mixtures of PFAS (see Sections 5–7 in USEPA, 2024a). For a more detailed discussion on whole-mixture and component-based approaches for PFAS risk assessment, please see the final *PFAS Mixtures Framework* (USEPA, 2024a).

The EPA considered several component-based assessment approaches to develop an MCLG for mixtures of PFHxS, PFNA, HFPO-DA, and/or PFBS under an assumption of dose additivity, including the general HI, the target organ-specific HI (TOSHI), the Relative Potency Factor (RPF) approach, and the mixture-benchmark dose (mixture-BMD) approach (USEPA, 2024a). As part of the technical support materials for the PFAS NPDWR, the EPA's draft PFAS Mixtures Framework (USEPA, 2021f) was submitted to the SAB for expert review. The SAB supported the EPA's proposed component-based approaches under the assumption of dose additivity in the absence of information to support a conclusion other than dose additivity (see Section 1.3). Following the SAB review, the EPA addressed the SAB's recommendations. Then, the EPA solicited public comment on the draft PFAS Mixtures Framework as part of the proposed NPDWR (88 FR 18638; USEPA (2023c)). The EPA evaluated potential component-based mixture assessment options, and ultimately proposed using the general HI approach as the most appropriate option based on available data and consistent with the statutory definition of MCLG.

The EPA first considered a "whole mixture" approach. Although use of data from whole mixtures or "sufficiently similar mixtures" is ideal in a theoretical sense, it is not practical, possible, or necessary for evaluating mixtures of PFAS in drinking water. Instead, the EPA is using the general HI approach, a longstanding component-based mixtures assessment approach which was endorsed by the SAB in the context of assessing risk associated with exposure to PFAS mixtures in drinking water (USEPA SAB, 2022), as discussed below. The goal of this component-based mixtures assessment approach is to approximate what the whole-mixture toxicity would be if the whole mixture could be tested and relies on toxicity information for each individual component in a mixture (USEPA, 2000a). A whole-mixture approach for regulating

mixtures of these four PFAS in drinking water is not possible because it would entail developing a single toxicity reference value (e.g., RfD) for one specific mixture of PFHxS, PFNA, HFPO-DA, and PFBS with defined proportions of each PFAS. Toxicity studies are typically conducted with only one test substance to isolate that particular substance's effects on test organisms, and whole-mixture data are exceedingly rare. There are no known whole-mixture studies for PFHxS, PFNA, HFPO-DA, and PFBS, and even if they were available, a toxicity reference value derived from such a study (i.e., a single RfD for a specific mixture of these four PFAS) would only be directly applicable to that specific mixture. Thus, a more flexible approach is necessary—one that considers the potential for the four PFAS to co-occur in different combinations and at different concentrations across time and space. The general HI approach affords this flexibility; the general HI indicates risk from exposure to a mixture and is useful to ensure a health-protective MCLG for PFAS mixtures that can be spatially and/or temporally variable. Given the variability of PFAS occurrence in drinking water across the nation (USEPA, 2024b), the general HI allows the EPA to regulate mixtures of these PFAS in drinking water by taking into account site-specific data at each PWS. HQs for the four different PFAS are expected to differ depending on the actual measured concentrations of each of the four PFAS at each PWS. The general HI approach thus allows for flexibility beyond a one-size-fits-all approach and is tailored to address risk at each PWS. Furthermore, the EPA's application of the general HI approach accounts for the dose additivity that was the basis for the EPA's final determination to regulate mixtures of two or more of these PFAS.

The EPA considered the two main types of HI approaches: 1) the general HI, which allows for component chemicals in the mixture to have different health effects or endpoints as the basis for their toxicity reference values (e.g., RfDs, minimal risk levels), and 2) the TOSHI, which relies on toxicity reference values based on the same specific target organ or system effects (e.g., effects on the liver or thyroid; effects on developmental or reproductive systems) (USEPA, 2000a). The general HI approach uses the most health-protective RfD (or minimal risk level) available for each mixture component, irrespective of whether the RfDs for all mixture components are based on effects in the same target organs or systems. These "overall" RfDs (as they are sometimes called) are protective of all other adverse health effects because they are based on the most sensitive known endpoints as supported by the weight of the evidence. As a result, this approach is protective of all types of toxicity/adverse effects, and thus ensures that the MCLG is the level at and below which there are no known or anticipated adverse human health effects with an adequate margin of safety with respect to certain PFAS mixtures in drinking water.

The TOSHI produces a less health protective indicator of risk than the general HI because the basis for the mixture component toxicity reference values has been limited to a specific target organ or system effect, which may occur at higher exposure levels than other effects (i.e., be a less sensitive endpoint). In other words, a TOSHI may not be health protective compared to the general HI if available data for a mixture component show effects in other organs at lower exposure levels compared to the critical effect observed in the target organ used for the TOSHI. Additionally, since a TOSHI relies on toxicity reference values aggregated for the same specific target organ or system endpoint/effect, an absence or lack of data on the specific target organ or system endpoint/effect for a mixture component may result in that component not being adequately accounted for in this approach (thus, underestimating health risk of the mixture). A TOSHI can only be derived for those PFAS for which the same target organ or system

endpoint/effect-specific RfDs have been calculated. For example, a TOSHI based on changes in thyroid effects illustrates why the target organ-specific approach underestimates risk in the context of these four PFAS in drinking water. To develop a thyroid effects-based TOSHI for mixtures of these four PFAS, only those PFAS with chronic toxicity reference values based on thyroid effects—PFHxS (MRL) and PFBS (RfD)—would be included in the TOSHI calculation; HFPO-DA and PFNA have chronic toxicity reference values based on other effects (i.e., liver and developmental effects, respectively) and thus would not be included in a thyroid effects-based TOSHI. Although thyroid effects are not the basis for the RfDs for HFPO-DA and PFNA, studies have shown that these two PFAS significantly affect the thyroid; for example, both have been shown to significantly affect serum thyroid hormone levels (reduced T4, T3) (Conley et al., 2019; NTP, 2019b). According to the Interagency Report to Congress on PFAS, "Multiple studies on diverse species (developing rodents and fish) suggest that some PFAS (e.g., PFOS, PFOA, ***PFNA, GenX chemicals*****,** PFHxS, PFDA, PFBA, PFBS, PFHxA) interfere with thyroid hormone signaling pathways and thyroid homeostasis through various mechanisms, including regulation of hepatic glucuronidation enzymes and deiodinases in the thyroid gland" (emphasis added, US OSTP, 2023). Therefore, a thyroid-specific HI that excluded HFPO-DA and PFNA would underestimate the dose additivity concerns for thyroid effects from the total mixture.

Many PFAS have data gaps in epidemiological or animal toxicological dose-response information for multiple types of health effects, thus limiting derivation of target organ-specific toxicity reference values; target organ-specific toxicity reference values for the same target for all four PFAS are not currently available for PFHxS, PFNA, HFPO-DA, and PFBS. The EPA's guidance recognizes the potential for organ- or system-specific data gaps and supports use of overall RfDs in a general HI approach, stating, "The target organ toxicity dose (TTD) is not a commonly evaluated measure and currently there is no official EPA activity deriving these values, as there is for the RfD and RfC" … "Because of their much wider availability than TTDs, standardized development process including peer review, and official stature, the RfD and RfC are recommended for use in the default procedure for the HI" (USEPA, 2000a). Even if target organ-specific toxicity reference values (TTDs) were available for PFHxS, PFNA, HFPO-DA, and PFBS, the general HI approach would still be more appropriate for this specific application because it is protective of all adverse health effects rather than just those associated with a specific organ or system, consistent with the statutory definition of MCLG.

Although these four PFAS elicit many of the same adverse health effects, the most sensitive known endpoint for each of the four PFAS is different, and thus the toxicity reference values used to calculate the HBWCs in the general HI approach are different. Epidemiological and/or experimental animal studies have demonstrated that exposure to PFHxS, PFNA, HFPO-DA, and PFBS individually is associated with many of the same observed adverse health effects (e.g., effects on lipids, as well as developmental, immune, endocrine, and hematologic endpoints; see Section 1.3), but with differing potencies for effect(s). In other words, two or more PFAS may elicit the same adverse effects, but at different exposure levels; for example, liver effects are associated with all four PFAS (PFHxS, PFNA, HFPO-DA, and PFBS) but HFPO-DA is the only one of the four for which liver effects represent the most sensitive known endpoint and serve as the basis for its toxicity reference value (i.e., RfD). The fact that the toxicity reference values (i.e., RfDs or MRLs) for the four PFAS are based on different health endpoints does not mean that the four PFAS are not toxicologically similar; rather, it means that based on the available data, the most sensitive endpoint currently known is different for each of these PFAS. The

general HI approach uses the most health-protective toxicity reference value available for each of the four PFAS to derive HBWCs, irrespective of whether they are based on effects in the same target organs or systems. Since each RfD (or MRL) is based on the most sensitive known endpoint based on the weight of evidence (i.e., toxicity reference value selection is not limited to a specific organ or system), this approach is protective of all other adverse health effects. This approach of allowing for component chemicals in the mixture to have different health effects or endpoints as the basis for their toxicity reference values is consistent with EPA guidance (see examples in USEPA, 2000a; USEPA, 1991b) and was supported by SAB (see Section 1.4.2.1).

The general HI is a well-established methodology that has been used for several decades in at least one other regulatory context to account for dose additivity in mixtures assessments. The EPA routinely uses the HI approach to consider the risks from multiple contaminants of concern in the Remedial Investigations and Feasibility Studies for cleanup sites on the Superfund National Priorities List under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Noncarcinogenic effects are summed to provide an HI that is compared to an acceptable index, generally 1. This approach assumes dose additivity in the absence of information on a specific mixture. These assessments of hazards from multiple chemical exposures are important factors to help inform the selection of remedies that are ultimately captured in the Superfund Records of Decision.

1.4.2.1 Science Advisory Board Support

The EPA directly asked the SAB about the utility and scientific defensibility of the general HI approach (in addition to other methods, including TOSHI) during the SAB's 2021–2022 review of the EPA's draft *Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS*. Specifically, the EPA asked the SAB to "Please provide specific feedback on whether the HI approach is a reasonable methodology for indicating potential risk associated with mixtures of PFAS. If not, please provide an alternative;" and "Please provide specific feedback on whether the proposed HI methodologies in the framework are scientifically supported for PFAS mixture risk assessment" (USEPA SAB, 2022). In its report (USEPA SAB, 2022), the SAB stated its support for the general HI approach:

> "The HI methodology is a reasonable approach for estimating the potential aggregate health hazards associated with the occurrence of chemical mixtures in environmental media. The HI is an approach based on dose additivity (DA) that has been validated and used by the EPA…. This approach is mathematically straightforward and may readily identify mixtures of potential toxicological concern, as well as identify chemicals that drive the toxicity within a given mixture." (USEPA SAB, 2022).
>
> "In general, the screening level Hazard Index (HI) approach, in which Reference Values (RfVs) for the mixture components are used regardless of the effect on which the RfVs are based, is appropriate for initial screening of whether exposure to a mixture of PFAS poses a potential risk that should be further evaluated. Toxicological studies to inform human health risk assessment are lacking for most members of the large class of PFAS, and mixtures of PFAS that commonly occur in environmental media, overall. For these reasons, the HI methodology is a reasonable approach for estimating the potential aggregate health hazards associated with the occurrence of chemical mixtures in environmental media. The HI is an approach based on dose additivity (DA) that has been

> validated and used by the EPA. The HI does not provide quantitative risk estimates (i.e., probabilities) for mixtures, nor does it provide an estimate of the magnitude of a specific toxicity. This approach is mathematically straightforward and may readily identify mixtures of potential toxicological concern, as well as identify chemicals that drive the toxicity within a given mixture." (USEPA SAB, 2022).

The SAB recognized the need for regulatory agencies to make decisions in the face of uncertainty to reduce exposures to PFAS. The SAB stated,

> "Given the agency's desire to support fit-for-purpose approaches, not every PFAS mixture scenario will be one that warrants a tiered or hierarchical approach. In some instances, an HI or target-organ-specific hazard indices (TOSHI) might provide enough information for decision-making about PFAS (or other chemicals) contamination in drinking water (or other media). Tiered approaches that require increasingly complex information before reaching a final decision point can be extremely challenging for data-poor chemicals such as PFAS. Data gaps identified in a such tiered methodologies could result in a bottleneck through which these chemicals may never emerge…" (USEPA SAB, 2022).

1.5 Establishment of Individual MCLGs for PFHxS, HFPO-DA, PFNA, and/or PFBS

The EPA has determined that sufficient information is available to satisfy the statutory requirements for individual regulation of PFHxS, HFPO-DA, and PFNA (in addition to PFOA and PFOS). To support this determination, the EPA carefully examined the health effects information from available peer-reviewed final human health assessments as well as published studies, reviewed PFAS drinking water occurrence data collected as part of the UCMR 3 and state-led monitoring efforts, and considered public comments received. The EPA finds that oral exposure to PFHxS, HFPO-DA, or PFNA individually may lead to adverse health effects in humans; that each of these three PFAS have a substantial likelihood of occurring in finished drinking water with a frequency and at levels of public health concern; and that, in the sole judgment of the Administrator, regulation of PFHxS, HFPO-DA, and PFNA individually presents a meaningful opportunity for health risk reductions for persons served by PWSs.

The agency is deferring the final individual regulatory determination for PFBS to further consider whether occurrence information supports a finding that there is substantial likelihood that PFBS will individually occur in PWSs and at a level of public health concern. Therefore, no individual MCLG for PFBS is being established at this time. However, when evaluating PFBS in mixture combinations with PFHxS, PFNA, and/or HFPO-DA, the EPA has determined that based on the best available information, PFBS does meet all three statutory criteria for regulation when a part of these mixtures, including that it is anticipated to have dose-additive adverse health effects; there is a substantial likelihood of its co-occurrence in combinations with PFHxS, PFNA, and/or HFPO-DA with a frequency and at levels of public health concern; and that there is a meaningful opportunity for health risk reduction by regulating mixture combinations of these four PFAS (USEPA, 2023c). Therefore, although the agency is deferring the individual final regulatory determination for PFBS, PFBS is included in the final determination to regulate mixture combinations containing two or more of PFHxS, PFNA, HFPO-DA, and PFBS. The

establishment of individual MCLGs for PFHxS, PFNA, and HFPO-DA as well as an HI MCLG for mixtures of two or more of PFHxS, PFNA, HFPO-DA, and PFBS addresses potential health risks related to individual PFAS exposure as well as dose additive adverse health effects from exposure to mixtures of two or more of these four PFAS.

1.6 Overview of Individual MCLG and Mixture Hazard Index (HI) MCLG Approaches

To establish an MCLG for an individual contaminant, the EPA assesses the peer-reviewed science examining cancer and noncancer health effects associated with oral exposure to the contaminant. For contaminants determined to be known or likely human carcinogens (USEPA, 2005) with a linear carcinogenic MOA (i.e., where there is a proportional relationship between dose and carcinogenicity at low concentrations) or for which there is insufficient information to determine that a carcinogen has a threshold dose below which no carcinogenic effects have been observed, the EPA has a longstanding practice of establishing the MCLG at zero (see USEPA (1998); USEPA (2000c); USEPA (2001); see S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3). For contaminants determined to be known or likely human carcinogens but with a nonlinear carcinogenic MOA,[4] contaminants that are designated as having suggestive evidence of carcinogenic potential in humans (USEPA, 2005), and noncarcinogenic contaminants, the EPA typically establishes the MCLG based on a noncancer toxicity reference value (RfV) that represents the best available science (e.g., EPA RfD or ATSDR MRL).

An MCLG that is based on noncancer effects is designed to be protective of noncancer effects over a lifetime of exposure with an adequate margin of safety, including for sensitive populations and life stages, consistent with SDWA 1412(b)(3)(C)(i)(V) and 1412(b)(4)(A). The inputs for a noncancer MCLG include an oral noncancer toxicity RfV (e.g., RfD or MRL), body weight-adjusted drinking water intake (DWI-BW), and a relative source contribution (RSC), as presented in Equation 1:

$$\boldsymbol{MCLG} = \left(\frac{Oral\ RfV}{DWI\text{-}BW}\right) * RSC \qquad \text{(Eqn. 1)}$$

Where:

RfV = Chronic toxicity reference value (EPA RfD or ATSDR MRL). An RfD is an estimate of a daily exposure to the human population (including sensitive populations) that is likely to be without an appreciable risk of deleterious effects during a lifetime (USEPA, 2002). An MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure (ATSDR, 2021).

DWI-BW = Body weight-adjusted drinking water intake – an exposure factor for the 90th percentile body weight-adjusted drinking water intake for the identified population or life stage, in units of liters of water consumed per kilogram body weight per day

[4] A carcinogen with a nonlinear MOA is a chemical agent for which the associated cancer response does not increase in direct proportion to the exposure level and for which there is scientific evidence demonstrating a threshold level of exposure below which there is no appreciable cancer risk (USEPA, 2005).

(L/kg/day). The DWI-BW considers both direct and indirect consumption of drinking water (indirect water consumption encompasses water added in the preparation of foods or beverages, such as tea or coffee). Chapter 3 of the EPA's *Exposure Factors Handbook* (USEPA, 2019b) provides the most up-to-date DWI-BWs for various populations or life stages within the U.S. general population based on publicly available, peer-reviewed data such as from the National Health and Nutrition Examination Survey (NHANES).

RSC = Relative source contribution – the percentage of the total exposure attributed to drinking water sources (USEPA, 2000b), with the remainder of the exposure allocated to all other routes or sources. The purpose of the RSC is to ensure that the level of a contaminant allowed by one criterion (e.g., MCLG), when combined with other identified sources of exposure common to the population and contaminant of concern, will not result in exposures that exceed the RfD. The RSC is derived by applying the Exposure Decision Tree approach (USEPA, 2000b).

The EPA's approach to DWI-BW selection includes a step to identify the sensitive population(s) or life stage(s) (i.e., those that may be more susceptible or sensitive to a chemical exposure) by considering the available data for the contaminant, including the adverse health effects observed in the toxicity study on which the RfD/minimal risk level was based (known as the critical effect within the critical or principal study). Although data gaps can complicate identification of the most sensitive population (e.g., not all windows or life stages of exposure and/or health outcomes may have been assessed in available studies), the critical effect and point of departure (POD) that form the basis for the RfD (or minimal risk level) can provide some information about sensitive populations because the critical effect is typically observed at the lowest tested dose among the available data. Evaluation of the critical study, including the exposure window, may identify a sensitive population or life stage (e.g., pregnant women, formula-fed infants, lactating women). In such cases, the EPA can select the corresponding DWI-BW for that sensitive population or life stage from the Exposure Factors Handbook (USEPA, 2019b). DWI-BWs in the Exposure Factors Handbook are based on information from publicly available, peer-reviewed studies, and were updated in 2019. In the absence of information indicating a sensitive population or life stage, the DWI-BW corresponding to the general population may be selected. Following this approach, the EPA selected appropriate DWI-BWs for each of the four PFAS included in the HI MCLG (see Section 2). The EPA did consider infants as a sensitive life stage for all four PFAS; however, the agency did not select the infant DWI-BW because the exposure intervals of the critical studies supporting the chronic toxicity reference values did not correspond to infants. Instead, the exposure intervals were relevant to other sensitive target populations (i.e., lactating women or women of childbearing age) or the general population.

The EPA applies an RSC to account for potential aggregate risk from exposure routes and exposure pathways other than oral ingestion of drinking water to ensure that an individual's total exposure to a contaminant does not exceed the daily exposure associated with toxicity (i.e., threshold level or RfD). Application of the RSC in this context is consistent with EPA methods (USEPA, 2000b) and long-standing EPA practice for establishing drinking water MCLGs and NPDWRs (e.g., see USEPA, 2010, 2004; USEPA, 1989). The RSC represents the proportion of an individual's total exposure to a contaminant that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee, tea, or soup, as well as from dietary items prepared with drinking water) relative to other exposure pathways. The remainder of the exposure equal to the RfD (or MRL) is allocated to other potential exposure sources (USEPA,

2000b). The purpose of the RSC is to ensure that the level of a contaminant (e.g., MCLG) in drinking water, when combined with other identified potential sources of exposure for the population of concern, will not result in total exposures that exceed the RfD (or MRL) (USEPA, 2000b). This ensures that the MCLG under SDWA meets the statutory requirement that it is a level of a contaminant in drinking water at or below which no known or anticipated adverse effects on human health occur and allowing an adequate margin of safety.

To determine the RSCs for the four PFAS, the agency assessed the available scientific literature on potential sources of human exposure other than drinking water (see Section 2). The EPA conducted literature searches and reviews for each of the four PFAS to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media (see Appendices A–D). Considering this exposure information, the EPA followed its longstanding, peer-reviewed Exposure Decision Tree Approach in EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000b) to determine the RSC for each PFAS. The EPA carefully evaluated studies that included information on potential exposure to these four PFAS (PFHxS, PFNA, HFPO-DA, and PFBS) via sources other than drinking water, such as food, soil, sediment, and air. For each of the four PFAS, the findings indicated that there are significant known or potential uses/sources of exposure beyond drinking water ingestion (e.g., food, indoor dust) (USEPA, 2000b), but that data are insufficient to allow for quantitative characterization of the different exposure sources (Box 8A in USEPA, 2000b). The EPA's Exposure Decision Tree approach states that when there are insufficient environmental and/or exposure data to permit quantitative derivation of the RSC, the recommended RSC for the general population is 20 percent (Box 8B in USEPA, 2000b). This means that 20 percent of the exposure equal to the RfD is allocated to drinking water, and the remaining 80 percent is attributed to all other potential exposure sources.

In the general HI approach, a HQ is calculated as the ratio of human exposure (E) to a health-based RfV for each mixture component chemical (i) (USEPA, 1986). The HI involves the use of RfVs for each PFAS mixture component (in this case, PFHxS, HFPO-DA, PFNA, and/or PFBS) which are based on the most sensitive known health outcomes and which are expected to be protective of all other adverse health effects observed after exposure to the individual PFAS. This approach, which protects against all adverse effects and not just a single adverse outcome/effect, is a conservative risk indicator and appropriate for MCLG derivation. The HI is dimensionless, so in the HI formula, E and the RfV must be in the same units (Equation 2). For example, if E is the oral intake rate (milligrams per kilogram per day (mg/kg/day)), then the RfV could be the RfD or MRL, which have the same units. Alternatively, the exposure metric can be a media-specific metric such as a measured water concentration (e.g., nanograms per liter or ng/L) and the RfV can be an HBWC (e.g., ng/L). The component chemical HQs are then summed across the mixture to yield the HI (Equation 2). A mixture HI exceeding 1 indicates potential risk for a given environmental medium or site. The HI provides an indication of: (1) concern for the overall mixture and (2) potential driver PFAS (i.e., those PFAS mixture components with high(er) HQs). For a detailed discussion of PFAS dose additivity and the HI approach, see the PFAS Mixtures Framework (USEPA, 2024a).

$$HI = \sum_{i=1}^{n} HQ_i = \sum_{i=1}^{n} \frac{E_i}{RfV_i} \qquad \text{(Eqn. 2)}$$

Where:

HI = Hazard index

HQ_i = Hazard quotient for chemical i

E_i = Exposure, i.e., dose (mg/kg/day) or occurrence concentration, such as in drinking water (in milligrams per liter or mg/L), for chemical i

RfV_i = Reference value (e.g., oral RfD or MRL) (mg/kg/day), or corresponding HBWC; e.g., an MCLG for chemical i (in mg/L)

The HBWCs/MCLGs are based on the best available science and data collected by accepted methods (see Section III in the USEPA, 2023c). Specifically, peer-reviewed, publicly available toxicity assessments are available for HFPO-DA (USEPA, 2021c), PFBS (USEPA, 2021d), PFNA (ATSDR, 2021), and PFHxS (ATSDR, 2021) that provide the oral toxicity values (i.e., RfD or MRL) used to calculate the HBWCs; the EPA selected the corresponding DWI-BW for the relevant sensitive population or life stage from the Exposure Factors Handbook (USEPA, 2019b) based on the best available, peer-reviewed science from publicly available, peer-reviewed studies taking into account the relevant sensitive population(s) or life stage(s); and the RSCs are based on the best available, peer-reviewed science or best available methods taking into account the relevant sensitive population(s) or life stage(s) (USEPA, 2000b).

2 Calculating the Health-Based Water Concentrations for HFPO-DA, PFBS, PFNA, and PFHxS for the HI MCLG

2.1 HFPO-DA

HFPO-DA is a shorter-chain PFAS that was intended to be a replacement for the longer-chained PFOA. In water, HFPO-DA and its various salts dissociate to form the HFPO-DA anion (HFPO−) as a common analyte.

The HBWC for HFPO-DA that the agency is using in the HI MCLG was derived in part from information from the agency's 2021 human health toxicity assessment of HFPO-DA (specifically the chronic RfD of 3E-06 mg/kg/day) (USEPA, 2021c). Summaries of key information from the HFPO-DA toxicity assessment of (i.e., information about the RfD), as well as information about the DWI-BW and RSC that were used to derive the HBWC for HFPO-DA, are presented in the following sections. Based on this information, an HBWC of 10 ng/L for HFPO-DA is used in the HI MCLG for mixtures of two or more of HFPO-DA, PFBS, PFNA, and PFHxS (see Section 3.2).

2.1.1 Toxicity

The HBWC for HFPO-DA is derived from a chronic RfD that is based on liver effects (specifically, a constellation of liver lesions including cytoplasmic alteration, single-cell and focal necrosis, and apoptosis) observed in parental female mice following oral exposure to HFPO-DA from pre-mating through lactation (53–64 days) (USEPA, 2021c).

As described in the EPA's human health toxicity assessment of HFPO-DA, oral toxicity studies in rodents exposed to HFPO-DA report a range of adverse effects. Repeated-dose oral exposure of rats and mice resulted in liver toxicity (e.g., increased relative liver weight, hepatocellular hypertrophy, apoptosis, and single-cell/focal necrosis), kidney toxicity (e.g., increased relative kidney weight), immune system effects (e.g., antibody suppression), hematological effects (e.g., decreased red blood cell count, hemoglobin, and hematocrit), reproductive/developmental effects (e.g., increased number of early deliveries, placental lesions, changes in maternal gestational weight gain, and delays in genital development in offspring), and cancer (e.g., liver and pancreatic tumors) (USEPA, 2021c).

The most sensitive noncancer effects observed among the available data were the adverse effects on liver, which were observed in both male and female mice and rats across a range of exposure durations and dose levels, including the lowest tested dose levels and shortest exposure durations (USEPA, 2021c). Noncancer liver effects formed the basis for the chronic RfD of 3E-06 mg/kg/day, which the EPA used to derive the HBWC for HFPO-DA. As described in the HFPO-DA toxicity assessment, to develop the chronic RfD for HFPO-DA, the EPA derived a human equivalent dose (HED) of 0.01 mg/kg/day from a no-observed-adverse-effect level (NOAEL) of 0.1 mg/kg/day for liver effects observed in the identified critical study (an oral

reproductive/developmental toxicity study in mice (DuPont, 2010b)). The EPA then applied a composite uncertainty factor (UF) of 3,000 (i.e., 10× for intraspecies variability (UF_H), 3× for interspecies differences (UF_A), 10× for extrapolation from a subchronic-to-chronic dosing duration (UF_S), and 10× for database deficiencies (UF_D)) to yield the chronic RfD (USEPA, 2021c).

The EPA determined that there is suggestive evidence of carcinogenic potential following oral exposure to HFPO-DA in humans, but the available data are insufficient to derive a cancer risk concentration in water for HFPO-DA (USEPA, 2021c).

2.1.2 Exposure Factor

To select an appropriate DWI-BW for use in derivation of the HBWC for HFPO-DA, the EPA considered the HFPO-DA exposure interval used in the oral reproductive/developmental toxicity study in mice that was the basis for chronic RfD derivation (the critical study). In this study, parental female mice were dosed from pre-mating through lactation, corresponding to three potentially sensitive human adult life stages that may represent critical windows of exposure for HFPO-DA: women of childbearing age (13 to < 50 years), pregnant women, and lactating women (Table 3-63 in USEPA, 2019b). Of these three, the DWI-BW for lactating women (0.0469 L/kg/day) is the highest (see Table 2-1), and therefore anticipated to be protective of the other two sensitive life stages. Therefore, the EPA used the DWI-BW for lactating women to calculate the HBWC for HFPO-DA in the HI MCLG.

Table 2-1. EPA Exposure Factors for Drinking Water Intake for Different Candidate Sensitive Populations or Life Stages, Based on the Critical Effect and Study for HFPO-DA.

Population	DWI-BW (L/kg bw-day)	Description of Exposure Metric	Source
Women of childbearing age	0.0354	90th percentile direct and indirect consumption of community water, consumer-only two-day average, 13 to < 50 years.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)
Pregnant women	0.0333	90th percentile direct and indirect consumption of community water, consumer-only two-day average.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)
Lactating women	**0.0469**	90th percentile direct and indirect consumption of community water, consumer-only two-day average.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010[a] (USEPA, 2019b)

Notes: L/kg bw-day = liters of water consumed per kilogram body weight per day; DWI-BW = body weight-adjusted drinking water intake; NHANES = National Health and Nutrition Examination Survey. The DWI-BW used to calculate the HFPO-DA HBWC is in bold.

[a] Estimates are less statistically reliable based on guidance published in the Joint Policy on Variance Estimation and Statistical Reporting Standards on NHANES III and CSFII Reports: HNIS/NCHS Analytical Working Group Recommendations (NCHS, 1993).

2.1.3 Relative Source Contribution

The EPA conducted literature searches and reviews for HFPO-DA to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media. Based on the physical properties, detected levels, and limited available exposure information for HFPO-DA, multiple non-drinking water sources (e.g., foods, indoor dust, air, soil, and sediment) are potential exposure sources (see Appendix A). Following the EPA's Exposure Decision Tree approach (USEPA, 2000b), potential sources other than drinking water ingestion were identified, but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources. Thus, the EPA used an RSC of 0.20 for HFPO-DA. This means that 20% of the RfD is attributed to drinking water, and the remaining 80% is attributed to all other potential exposure sources. As explained above (Section 1.6), applying the RSC ensures that an individual's total exposure to HFPO-DA from all sources does not exceed the daily exposure associated with the RfD, consistent with agency methodology (USEPA, 2000b) and longstanding practice for establishing drinking water MCLGs and NPDWRs.

2.1.4 Derivation of HFPO-DA HBWC

The HBWC for HFPO-DA is calculated as follows and summarized in Table 2-2:

$$\textbf{HFPO–DA HBWC} = \left(\frac{\text{RfD}}{\text{DWI-BW}}\right) * \text{RSC}$$

$$= \left(\frac{0.000003 \ \frac{\text{mg}}{\text{kg/day}}}{0.0469 \frac{\text{L}}{\text{kg/day}}}\right) * 0.2$$

$$= 0.00001 \ \frac{\text{mg}}{\text{L}}$$

$$= 0.01 \ \frac{\mu\text{g}}{\text{L}}$$

$$= 10 \ \frac{\text{ng}}{\text{L}} \text{ or parts per trillion (ppt)}$$

Table 2-2. HFPO-DA HBWC – Input Parameters and Value

Parameter	Value	Units	Source
Chronic oral RfD	3E-06	mg/kg/day	Final RfD based on liver effects (constellation of liver lesions as defined by the NTP Pathology Working Group) in parental female mice exposed to HFPO-DA by gavage from pre-mating through lactation (53–64 days) (USEPA, 2021c; DuPont, 2010b).
DWI-BW	0.0469	L/kg/day	90th percentile 2-day average, consumer-only estimate of combined direct and indirect community water ingestion for lactating women (age 13 to <50 years) based on 2005–2010 NHANES data (USEPA, 2019b).
RSC	0.2	N/A	Based on a review of the available scientific literature on HFPO-DA, potential exposure routes and sources exist but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources (see Appendix A).
HFPO-DA HBWC = 0.00001 mg/L or 10 ppt			

Notes: RfD = reference dose; DWI-BW = body weight-adjusted drinking water intake; HBWC = health-based water concentration; HFPO-DA = hexafluoropropylene oxide dimer acid; N/A = not applicable; NHANES = National Health and Nutrition Examination Survey; NTP = National Toxicology Program; RSC = relative source contribution.

2.2 PFBS

PFBS and its potassium salt (K^+PFBS) are shorter-chain PFAS that were developed as "safer" replacements for the longer-chained PFOS (USEPA, 2021d). In water, K^+PFBS dissociates to the deprotonated anionic form of PFBS (PFBS-) and the K^+ cation at environmental pH levels (pH 4–9). These three PFBS chemical forms are referred to collectively as PFBS.

The HBWC that the agency is using for the HI MCLG was derived from information in the agency's 2021 human health toxicity assessment for PFBS, specifically the chronic RfD of 3E-04 mg/kg/day based on thyroid effects observed in newborn mice born to mothers that had been orally exposed to PFBS throughout gestation (USEPA, 2021d). Summaries of key information from the PFBS toxicity assessment (i.e., information about the RfD), as well as information about the DWI-BW and RSC that were used to derive the HBWC for PFBS, are presented in the following sections. Based on this information, an HBWC of 2,000 ng/L for PFBS is used in the HI MCLG for mixtures of two or more of HFPO-DA, PFBS, PFNA, and PFHxS (see Section 3.2).

2.2.1 Toxicity

The HBWC for PFBS was derived using a chronic oral RfD based on thyroid effects seen in an oral toxicity study in mice (USEPA, 2021d).

The EPA's human health toxicity assessment for PFBS (USEPA, 2021d) considered all publicly available human, animal, and mechanistic studies of PFBS exposure and effects. The assessment

identified associations between PFBS exposure and thyroid, developmental, and kidney effects based on studies in animals. The limited evidence for thyroid or kidney effects in human studies was equivocal, and no studies evaluating developmental effects of PFBS in humans were available. Human and animal studies evaluated other health effects following PFBS exposure including effects on the reproductive system, liver, and lipid and lipoprotein homeostasis, but the evidence did not support clear associations between exposure and effect (USEPA, 2021d).

The most sensitive noncancer effect observed was an adverse effect on the thyroid (i.e., decreased serum total thyroxine) seen in newborn mice (postnatal day (PND) 1) born to mothers that had been orally exposed to K^+PFBS throughout gestation (USEPA, 2021d; Feng et al., 2017). This critical effect was the basis for the chronic RfD of 3E-04 mg/kg/day which the EPA used to derive the HBWC for PFBS (USEPA, 2021d). As described in the PFBS toxicity assessment, to develop the chronic RfD for PFBS,[5] the EPA derived an HED of 0.095 mg/kg/day from BMD modeling of the critical effect in mice. The EPA then applied a composite UF of 300 (i.e., 10× for UF_H, 3× for UF_A, and 10× for UF_D) to yield the chronic RfD (USEPA, 2021d). The EPA did not apply an additional UF to adjust for subchronic-to-chronic duration (i.e., UF_S) because the critical effects were observed during a developmental life stage[6] (USEPA, 2002).

There were no human or animal studies identified that evaluated the potential carcinogenicity of PFBS (USEPA, 2021d).

2.2.2 Exposure Factor

To select an appropriate DWI-BW for use in deriving the HBWC, the EPA considered the PFBS exposure interval used in the developmental toxicity study in mice that was the basis for chronic RfD derivation. In this study, pregnant mice were exposed throughout gestation, which is relevant to two human adult life stages: women of childbearing age (13 to < 50 years) who may be or become pregnant, and pregnant women and their developing embryo or fetus (Table 3-63 in USEPA, 2019b). Of these two, the EPA selected the DWI-BW for women of childbearing age (0.0354 L/kg/day) to derive the HBWC for PFBS because it is higher and therefore more health protective (see Table 2-3).

[5] Data for K^+PFBS were used to derive the chronic RfD for the free acid (PFBS), resulting in the same value (3E-04 mg/kg/day), after adjusting for differences in molecular weight between K+ PFBS (338.19) and PFBS (300.10) (USEPA, 2021d).

[6] As stated in USEPA (2002), "…This is because it is assumed that most endpoints of developmental toxicity can be caused by a single exposure. If, however, developmental effects are more sensitive than those seen after longer-term exposures, then even the chronic RfD/RfC should be based on such effects to reduce the risk of potential greater sensitivity in children. Because the standard studies currently conducted for developmental toxicity involve repeated exposures, data are not often available on which endpoints may be induced by acute, subacute, subchronic, or chronic dosing regimens and, therefore, on which should be used in setting various duration reference values."

Table 2-3. EPA Exposure Factors for Drinking Water Intake for Different Candidate Sensitive Populations or Life Stages, Based on the Critical Effect and Study for PFBS.

Population	DWI-BW (L/kg bw-day)	Description of Exposure Metric	Source
Women of childbearing age	**0.0354**	90th percentile direct and indirect consumption of community water, consumer-only two-day average, 13 to < 50 years.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)
Pregnant women	0.0333	90th percentile direct and indirect consumption of community water, consumer-only two-day average.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)

Notes: L/kg bw-day = liters of water consumed per kilogram body weight per day; DWI-BW = body weight-adjusted drinking water intake; NHANES = National Health and Nutrition Examination Survey. The DWI-BW used to calculate the PFBS HBWC is in bold.

2.2.3 Relative Source Contribution

The EPA conducted literature searches and reviews for PFBS to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media. Based on the physical properties, detected levels, and available exposure information for PFBS, multiple non-drinking water sources (seafood including fish and shellfish, other foods, indoor air, and some consumer products) are potentially significant exposure sources (see Appendix B). Following the EPA's Exposure Decision Tree approach (USEPA, 2000b), potential sources other than drinking water ingestion were identified, but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources. Thus, the EPA used an RSC of 0.20 for PFBS. This means that 20% of the RfD is attributed to drinking water, and the remaining 80% is attributed to all other potential exposure sources. As explained above (Section 1.6), applying the RSC ensures that an individual's total exposure to PFBS from all sources does not exceed the daily exposure associated with the RfD, consistent with agency methodology (USEPA, 2000b) and longstanding practice for establishing drinking water MCLGs and NPDWRs.

2.2.4 Derivation of PFBS HBWC

The HBWC for PFBS is calculated as follows and summarized in Table 2-4:

$$\textbf{PFBS HBWC} = \left(\frac{\text{RfD}}{\text{DWI-BW}}\right) * \text{RSC}$$

$$= \left(\frac{0.0003 \ \frac{\text{mg}}{\text{kg/day}}}{0.0354 \frac{\text{L}}{\text{kg/day}}}\right) * 0.2$$

$$= 0.0017 \frac{\text{mg}}{\text{L}} \left(\text{rounded to } 0.002 \frac{\text{mg}}{\text{L}}\right)$$

$$= 2 \frac{\mu\text{g}}{\text{L}}$$

$$= 2{,}000 \frac{\text{ng}}{\text{L}} \text{ or ppt}$$

Table 2-4. PFBS HBWC – Input Parameters and Value

Parameter	Value	Units	Source
Chronic RfD	3E-04	mg/kg/day	Final RfD based on critical effect of decreased serum total thyroxine in newborn mice after gestational exposure (USEPA, 2021d; Feng et al., 2017).
DWI-BW	0.0354	L/kg/day	90th percentile 2-day average, consumer-only estimate of combined direct and indirect community water ingestion for women of childbearing age (13 to <50 years) based on 2005−2010 NHANES data (USEPA, 2019b).
RSC	0.2	N/A	Based on a review of the available scientific literature on PFBS, potential exposure routes and sources exist but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources (see Appendix B).
PFBS HBWC = 0.002 mg/L or 2,000 ppt			

Note: RfD = reference dose; DWI-BW = body weight-adjusted drinking water intake; N/A = not applicable; NHANES = National Health and Nutrition Examination Survey; HBWC = health-based water concentration; PFBS = perfluorobutanesulfonic acid; RSC = relative source contribution.

2.3 PFNA

PFNA has been used as a processing aid in the production of fluoropolymers, primarily polyvinylidene fluoride, which is a plastic designed to be temperature resistant and chemically nonreactive (USEPA, 2020b; NJSWQI, 2017; Prevedouros et al., 2006). PFNA has been used since the 1950's in a wide variety of industrial and consumer products. It has also been used in aqueous film-forming foam (AFFF) for fire suppression (USEPA, 2020b; Laitinen et al., 2014).

ATSDR has published a toxicological profile for a group of PFAS including PFNA and has developed an intermediate-duration oral MRL for PFNA (ATSDR, 2021). The EPA's derived HBWC for PFNA (described below) is based on the ATSDR MRL (ATSDR, 2021), a DWI-BW (selected by the EPA) that corresponds to this MRL, and an RSC determined by the EPA. There is no published EPA human health toxicity assessment for PFNA at the time of this writing; however, the EPA's Integrated Risk Information System (IRIS) program is developing a human health toxicity assessment for PFNA, which is expected to be finalized in 2024 (USEPA, 2023a, 2021e). The EPA's IRIS assessment will use systematic review methods to evaluate the

epidemiological and toxicological literature for PFNA, including consideration of relevant mechanistic evidence (USEPA, 2021e).

2.3.1 Toxicity

The HBWC for PFNA is based on an ATSDR intermediate-duration oral MRL that was based on developmental effects seen in mice after oral PFNA exposure (ATSDR, 2021).

Studies of oral PFNA exposure in rodents have reported adverse effects on the liver, development, and reproductive and immune systems (ATSDR, 2021). The most sensitive noncancer effects and basis for the ATSDR intermediate-duration oral MRL were decreased body weight gain and impaired development (i.e., delayed eye opening, preputial separation, and vaginal opening) in mice born to mothers that were treated with PFNA from gestational days (GDs) 1–17 (with presumed continued indirect exposure of offspring via lactation), and monitoring until PND 287 (ATSDR, 2021). The study reporting these effects (Das et al., 2015) was selected by ATSDR as the principal study for MRL derivation. To derive the MRL, an HED of 0.001 mg/kg/day was calculated from the NOAEL of 1 mg/kg/day identified in the study. ATSDR applied a total UF of 30 (i.e., 10× for UF_H and 3× for UF_A) and a modifying factor (MF) of 10× for database deficiencies to account for the small number/limited scope of studies examining PFNA toxicity following intermediate-duration exposure. The resulting intermediate-duration oral MRL was 3E-06 mg/kg/day (ATSDR, 2021). The EPA did not apply an additional UF_S to calculate the HBWC because the critical effects were observed during a developmental life stage[6] (USEPA, 2002). Toxicological assessments from other sources (e.g., states) report toxicity reference values based on animal studies for PFNA that are in the same range, providing additional support (USEPA, 2021e).

The carcinogenic potential of PFNA has been examined in three epidemiological studies. No consistent associations between serum PFNA levels and breast cancer or prostate cancer were found (ATSDR, 2021). The EPA has not yet completed a final evaluation and classification of the carcinogenicity of PFNA.

2.3.2 Exposure Factor

Based on the life stages of exposure in the principal study from which the intermediate-duration MRL was derived (i.e., directly to maternal animals during gestation, and indirectly to offspring during gestation and lactation), the EPA identified three potentially sensitive life stages that may represent critical windows of exposure for PFNA: women of childbearing age (13 to <50 years), pregnant women, and lactating women (Table 3-63 in USEPA, 2019b). The DWI-BW for lactating women (0.0469 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only 2-day average) was selected to calculate the HBWC for PFNA because it is the highest of the three DWI-BWs and is anticipated to be protective of the other two sensitive life stages (see Table 2-5).

Table 2-5. EPA Exposure Factors for Drinking Water Intake for Different Candidate Sensitive Populations and Life Stages, Based on the Critical Effect and Study for PFNA.

Population	DWI-BW (L/kg bw-day)	Description of Exposure Metric	Source
Women of childbearing age	0.0354	90th percentile direct and indirect consumption of community water, consumer-only two-day average, 13 to < 50 years.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)
Pregnant women	0.0333	90th percentile direct and indirect consumption of community water, consumer-only two-day average.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010 (USEPA, 2019b)
Lactating women	**0.0469**	90th percentile direct and indirect consumption of community water, consumer-only two-day average.	2019 Exposure Factors Handbook Chapter 3, Table 3-63, NHANES 2005–2010[a] (USEPA, 2019b)

Notes: L/kg bw-day = liters of water consumed per kilogram body weight per day; DWI-BW = body weight-adjusted drinking water intake; NHANES = National Health and Nutrition Examination Survey. The DWI-BW used to calculate the PFNA HBWC is in bold.

2.3.3 Relative Source Contribution

The EPA conducted literature searches and reviews for PFNA to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media. Based on the physical properties, detected levels, and available exposure information for PFNA, multiple non-drinking water sources (fish and shellfish, non-fish food, some consumer products, indoor dust, and air) are potentially significant exposure sources (see Appendix C). Following the Exposure Decision Tree approach (USEPA, 2000b), potential sources other than drinking water ingestion were identified but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources. Thus, the EPA used an RSC of 0.20 for PFNA. This means that 20% of the RfV is attributed to drinking water, and the remaining 80% is attributed to all other potential exposure sources. As explained above (Section 1.6), applying the RSC ensures that an individual's total exposure to PFNA from all sources does not exceed the daily exposure associated with the RfV, consistent with agency methodology (USEPA, 2000b) and longstanding practice for establishing drinking water MCLGs and NPDWRs.

2.3.4 Derivation of PFNA HBWC

The HBWC for PFNA is calculated as follows and summarized in Table 2-6:

$$\textbf{PFNA HBWC} = \left(\frac{\text{RfV}}{\text{DWI-BW}}\right) * \text{RSC}$$

$$= \left(\frac{0.000003 \ \frac{\text{mg}}{\text{kg/day}}}{0.0469 \frac{\text{L}}{\text{kg/day}}}\right) * 0.2$$

$$= 0.000014 \ \frac{\text{mg}}{\text{L}} \left(\text{rounded to } 0.00001 \ \frac{\text{mg}}{\text{L}}\right)$$

$$= 0.01 \ \frac{\mu\text{g}}{\text{L}}$$

$$= 10 \ \frac{\text{ng}}{\text{L}} \text{ or ppt}$$

Table 2-6. PFNA HBWC – Input Parameters and Value

Parameter	Value	Units	Source
RfV	3E-06[a]	mg/kg/day	Based on decreased body weight gain and delayed eye opening, preputial separation, and vaginal opening in mouse offspring after gestational and presumed lactational exposure (ATSDR, 2021; Das et al., 2015).
DWI-BW	0.0469	L/kg/day	90th percentile 2-day average, consumer-only estimate of combined direct and indirect community water ingestion for lactating women (13 to <50 years) based on 2005−2010 NHANES data (USEPA, 2019b).
RSC	0.2	N/A	Based on a review of the current scientific literature on PFNA, potential exposure routes and sources exist but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources (see Appendix C).
PFNA HBWC = 0.00001 mg/L or 10 ppt			

Notes: RfV = chronic toxicity reference value; DWI-BW = body weight-adjusted drinking water intake; N/A = not applicable; NHANES = National Health and Nutrition Examination Survey; PFNA = perfluorononanoic acid; RSC = relative source contribution.

[a] Note that ATSDR MRLs and EPA RfDs are not identical (e.g., intermediate-duration MRL vs. chronic RfD; the EPA and ATSDR may apply different uncertainty/modifying factors) and are developed for different purposes. In this case, the EPA did not apply an additional UF_S to calculate the HBWC for PFNA because the critical effect is identified in a developmental population (USEPA, 2002).

2.4 PFHxS

PFHxS has been used in laboratory applications and as a raw material or a precursor for the manufacture of PFAS/perfluoroalkyl sulfonate-based products, though production of PFHxS in

the United States was phased out by its major manufacturer in 2002 (Sigma-Aldrich, 2014 as cited in NCBI, 2022; Backe et al., 2013; Buck et al., 2011; OECD, 2011). PFHxS has also been used in firefighting foam and carpet treatment solutions, and it has been used as a stain and water repellant (Garcia and Harbison, 2015 as cited in NCBI, 2022).

ATSDR has published a toxicological profile for a group of PFAS including PFHxS and has calculated an intermediate-duration oral MRL for PFHxS (ATSDR, 2021). The EPA's derived HBWC for PFHxS (described below) is based on the ATSDR MRL (ATSDR, 2021), a DWI-BW (selected by the EPA) that corresponds to the MRL, and an RSC determined by the EPA. There is no published EPA human health toxicity assessment for PFHxS at the time of this writing; however, the EPA's IRIS program is developing a human health toxicity assessment for PFHxS, which is expected to be finalized in 2024 (USEPA, 2023a, 2021e). The EPA's IRIS assessment will use systematic review methods to evaluate the epidemiological and toxicological literature for PFHxS, including consideration of relevant mechanistic evidence (USEPA, 2021e).

2.4.1 Toxicity

The HBWC for PFHxS is derived using an ATSDR intermediate-duration oral MRL based on thyroid effects seen in male rats after oral PFHxS exposure (ATSDR, 2021).

Toxicity studies of oral PFHxS exposure to animals have reported health effects on the liver, thyroid, and development (ATSDR, 2021). The most sensitive noncancer effect observed was thyroid follicular epithelial hypertrophy/hyperplasia in parental male rats that had been exposed for 42–44 days, identified in the principal developmental toxicity study selected by ATSDR (NOAEL of 1 mg/kg/day for this effect) (ATSDR, 2021; Butenhoff et al., 2009). This critical effect was the basis for the ATSDR intermediate-duration oral MRL which the EPA used to derive the HBWC for PFHxS. An HED of 0.0047 mg/kg/day was calculated from the NOAEL of 1 mg/kg/day identified in the principal study. ATSDR applied a total UF of 30 (i.e., 10× for UF_H and 3× for UF_A) and an MF of 10× for database deficiencies to yield an intermediate-duration oral MRL of 2E-05 mg/kg/day (ATSDR, 2021). To calculate the HBWC, the EPA applied an additional UF_S of 10, per agency guidelines (USEPA, 2002), because the effect is not in a developmental population (i.e., thyroid follicular epithelial hypertrophy/hyperplasia in parental male rats). The resulting adjusted chronic reference value is 2E-06 mg/kg/day. Toxicological assessments from other sources (e.g., states) report toxicity reference values based on animal studies for PFHxS that are in the same range, providing additional support (USEPA, 2021e).

The carcinogenic potential of PFHxS has been examined in four epidemiological studies (ATSDR, 2021). Bonefeld-Jørgensen et al. (2014) reported a significant negative correlation between serum PFHxS levels (mean concentration 1.2 ng/mL) and breast cancer risk among Danish women. However, a study in Greenland found a significant, positive association between high serum levels of PFHxS and breast cancer risk (Wielsøe et al., 2017). The median serum PFHxS concentration among cases in that study was 2.52 ng/mL and serum levels ranged from 0.19 ng/mL to 23.40 ng/mL (Wielsøe et al., 2017). Hardell et al. (2014) found a statistically significant interaction between above-median PFHxS concentrations and increased risk for prostate cancer among men with genetics as a risk factor (first-degree relative). Prostate-specific antigen (PSA) levels were not associated with serum PFHxS levels (mean concentration 3.38 ng/mL) in men 20–49 or 50–69 years of age (Ducatman et al., 2015). The EPA has not yet completed a final evaluation and classification of the carcinogenicity of PFHxS.

2.4.2 Exposure Factor

No sensitive population or life stage was identified for DWI-BW selection for PFHxS because the critical effect on which the ATSDR MRL was based (thyroid alterations) was observed in adult male rats. Since this exposure life stage does not correspond to a sensitive population or life stage, a DWI-BW for adults within the general population (0.034 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only 2-day average, adults 21 years and older) was selected for HBWC derivation (USEPA, 2019b).

2.4.3 Relative Source Contribution

The EPA conducted literature searches and reviews for PFHxS to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media. Based on the physical properties, detected levels, and available exposure information for PFHxS, multiple non-drinking water sources (fish and shellfish, non-fish food, some consumer products, indoor dust, and soil) are potentially significant exposure sources (see Appendix D). Following the Exposure Decision Tree approach (USEPA, 2000b), potential sources other than drinking water ingestion were identified but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources. Thus, the EPA used an RSC of 0.20 for PFHxS. This means that 20% of the RfV is attributed to drinking water, and the remaining 80% is attributed to all other potential exposure sources. As explained above (Section 1.6), applying the RSC ensures that an individual's total exposure to PFHxS from all sources does not exceed the daily exposure associated with the RfV, consistent with agency methodology (USEPA, 2000b) and longstanding practice for establishing drinking water MCLGs and NPDWRs.

2.4.4 Derivation of PFHxS HBWC

The HBWC for PFHxS is calculated as follows and summarized in Table 2-7:

$$\textbf{PFHxS HBWC} = \left(\frac{\text{RfV}}{\text{DWI-BW}}\right) * \text{RSC}$$

$$= \left(\frac{0.000002 \ \frac{\text{mg}}{\text{kg/day}}}{0.034 \frac{\text{L}}{\text{kg/day}}}\right) * 0.2$$

$$= 0.000012 \ \frac{\text{mg}}{\text{L}} \left(\text{rounded to } 0.00001 \ \frac{\text{mg}}{\text{L}}\right)$$

$$= 0.01 \ \frac{\mu\text{g}}{\text{L}}$$

$$= 10 \ \frac{\text{ng}}{\text{L}} \text{ or ppt}$$

Table 2-7. PFHxS HBWC – Input Parameters and Value

Parameter	Value	Units	Source
RfV	2E-06[a]	mg/kg/day	Based on thyroid follicular epithelial hypertrophy/hyperplasia in parental male rats (exposed 42–44 days) (ATSDR, 2021; Butenhoff et al., 2009).
DWI-BW	0.034	L/kg/day	90th percentile 2-day average, consumer-only estimate of combined direct and indirect community water ingestion for adults 21 years and older based on 2005−2010 NHANES data (USEPA, 2019b).
RSC	0.2	N/A	Based on a review of the current scientific literature on PFHxS, potential exposure routes and sources exist but the available information is limited and does not allow for the quantitative characterization of the relative levels of exposure among these different sources (see Appendix D).
PFHxS HBWC = 0.00001 mg/L or 10 ppt			

Notes: RfV = chronic toxicity reference value; DWI-BW = body weight-adjusted drinking water intake; N/A = not applicable; NHANES = National Health and Nutrition Examination Survey; RSC = relative source contribution; PFHxS = perfluorohexanesulfonic acid.

[a] Note that ATSDR MRLs and EPA RfDs are not identical (e.g., intermediate-duration MRL vs. chronic RfD; the EPA and ATSDR may apply different uncertainty/modifying factors) and are developed for different purposes. The EPA applied an additional UF of 10 to the ATSDR MRL for PFHxS to account for subchronic-to-chronic duration (i.e., UF_S) yielding a chronic reference value of 2E-06 mg/kg/day, which was used to calculate the HBWC for PFHxS (USEPA, 2002).

3 Derivation of MCLGs

3.1 Individual MCLGs for HFPO-DA, PFNA, and PFHxS

The EPA is setting the individual MCLGs for HFPO-DA, PFHxS, and PFNA at 10 ng/L based on their respective RfVs, DWI-BWs, and RSCs (see Section 2, specifically Sections 2.1, 2.3, and 2.4). The MCLG for each of these PFAS is set, as defined in Section 1412(b)(4)(A), at "the level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety." Each of these MCLGs (shown in Table 3-1 below) is set at the same level as the HBWCs derived in Section 2.

Table 3-1. Individual MCLGs

PFAS	MCLG (ng/L or ppt)
HFPO-DA	10
PFHxS	10
PFNA	10

Notes: HFPO-DA = hexafluoropropylene oxide dimer acid; MCLG = maximum contaminant level goal; PFAS = per- and polyfluoroalkyl substances; PFHxS = perfluorohexanesulfonic acid; PFNA = perfluorononanoic acid; ng/L = nanograms per liter; ppt = parts per trillion.

3.2 PFAS Mixtures Hazard Index MCLG

In consideration of the known toxic effects, potential dose additivity, and occurrence and likely co-occurrence of these PFAS in drinking water, the EPA is finalizing an HI of 1 (unitless) as the MCLG for any mixture of two or more of PFHxS, PFNA, HFPO-DA, and PFBS. As described in Section 1.6, a mixture HI can be calculated when HBWCs for a set of PFAS are available or can be calculated. HQs are calculated for each of the mixture components by dividing the measured component PFAS concentration in water (e.g., expressed as ng/L) by the relevant HBWC (e.g., expressed as ng/L), as shown in the equation below. Component HQs are summed across the PFAS mixture to yield the HI MCLG. A PFAS mixture HI MCLG greater than 1 (i.e., rounded to one significant digit) indicates an exceedance of the health-protective level and indicates potential human health risk for noncancer effects from oral exposure to the PFAS mixture in water. For more details, please see USEPA (2024a). The final PFAS mixture HI MCLG for mixtures of two or more of HFPO-DA, PFBS, PFNA, and PFHxS is calculated as follows:

$$HI\ MCLG = \left(\frac{[HFPO\text{-}DA_{ng/L}]}{[HFPO\text{-}DA_{HBWC}]}\right) + \left(\frac{[PFBS_{ng/L}]}{[PFBS_{HBWC}]}\right) + \left(\frac{[PFNA_{ng/L}]}{[PFNA_{HBWC}]}\right) + \left(\frac{[PFHxS_{ng/L}]}{[PFHxS_{HBWC}]}\right) = 1$$

$$HI\ MCLG = \left(\frac{[HFPO\text{-}DA_{ng/L}]}{[10\ ng/L]}\right) + \left(\frac{[PFBS_{ng/L}]}{[2000\ ng/L]}\right) + \left(\frac{[PFNA_{ng/L}]}{[10\ ng/L]}\right) + \left(\frac{[PFHxS_{ng/L}]}{[10\ ng/L]}\right) = 1$$

Where:

$[PFAS_{ng/L}]$ = the measured component PFAS concentration in water (in ng/L) and

$[PFAS_{HBWC}]$ = the HBWC of a component PFAS.

In summary, although current weight of evidence suggests that PFAS vary in their precise structure and function, exposure to different PFAS can result in the same or similar adverse health effects; as a result, PFAS co-exposures are likely to result in dose-additive effects and therefore the conclusion of dose additivity is appropriate (see Section 1.3 and also USEPA, 2024a). While individual PFAS can pose a potential risk to human health if the exposure level exceeds the chemical-specific toxicity reference value (RfD or MRL) (i.e., individual PFAS HQ >1.00), mixtures of PFAS at lower individual PFAS concentrations can result in dose-additive adverse health effects. For example, if the individual HQs for PFHxS, HFPO-DA, PFNA, and PFBS were each 0.90, that would indicate that the measured concentration of each individual PFAS in drinking water is below the level of appreciable risk for each individual PFAS (recall that an RfV, such as an oral RfD, represents an estimate at which no appreciable risk of deleterious effects exists). However, the overall HI for that mixture would be 4 (i.e., 3.6, sum of four HQs of 0.90, rounded to 4), indicating risk. Thus, setting an MCLG based on the concentration of an individual PFAS without considering the potential dose-additive effects from other PFAS in a mixture would not provide a sufficiently protective MCLG with an adequate margin of safety. To account for dose-additive noncancer effects associated with co-occurring PFAS to protect against health impacts from multi-chemical exposures to mixtures of two or more of PFHxS, HFPO-DA, PFNA, and PFBS, the agency is finalizing use of the HI approach for the MCLG for mixtures of two or more of these four PFAS. Consistent with the statutory requirement under 1412(b)(4)(A) of SDWA, establishing the MCLG for mixtures of two or more of PFHxS, HFPO-DA, PFNA, and PFBS at an HI equal to 1 ensures that the MCLG is set at a level at which there are no known or anticipated adverse effects on the health of persons and which ensures an adequate margin of safety.

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Appendix A. HFPO-DA: Summary of Occurrence in Water and Detailed Relative Source Contribution

A.1. Occurrence in Water

HFPO-DA can enter the aquatic environment through industrial discharges, runoff into surface water, and leaching into groundwater from soil and landfills (USEPA, 2021c). HFPO-DA is water-soluble, with solubilities of greater than 751 grams per liter (g/L) and greater than 739 g/L for HFPO-DA and its ammonium salt, respectively, at 20°C (USEPA, 2021c). Volatilization from water surfaces is expected to be an important fate process for HFPO-DA and its ammonium salt (USEPA, 2021c).

A.1.1. Ground Water

Pétré et al. (2021) quantified the mass transfer of PFAS, including HFPO-DA, from contaminated groundwater to five tributaries of the Cape Fear River. All sampling sites were located within 5 km of a manufacturing plant known known to be a major source of PFAS contamination. HFPO-DA and another fluoroether (perfluoro-2-[perfluoromethoxy] propanoic acid) together accounted for 61% of the total quantified PFAS. The study authors calculated that approximately 32 kg/year of PFAS is discharged from contaminated groundwater to the five tributaries. These data indicate that the discharge of contaminated groundwater has led to long-term contamination from HFPO-DA in surface water and could lead to subsequent impacts on downstream drinking water (Pétré et al., 2021).

A.1.2. Surface Water

Chemours has reported that HFPO-DA has been discharged into the Cape Fear River for several decades as a byproduct of other manufacturing processes (NCDEQ, 2017). An exposure assessment of the Chemours Fayetteville Works Facility located in Bladen County, North Carolina evaluated HFPO-DA in surface water data collected between July 2017 and October 2019 at four locations in the Cape Fear River (one upstream, one facility-adjacent location, and two downstream), one pond located in the facility site, and one offsite pond (Geosyntec, 2019). HFPO-DA was detected in surface water at the six sampled locations where mean (range) concentrations were 5 ng/L (n=1), 23.37 ng/L (2.1-160 ng/L; n=26), 133.6 ng/L (8.6-580 ng/L; n= 17), 16.38 ng/L (2.14-76 ng/L; n=79), 800 ng/L (730-940 ng/L; n=4), and 303.3 ng/L (290-310 ng/L; n=3), respectively (Geosyntec, 2019). Additionally, several other studies evaluated the occurrence of HFPO-DA in surface waters in North America (see Table A-1). As noted in the EPA's human health toxicity assessment for HFPO-DA (USEPA, 2021c), HFPO-DA was first detected in North Carolina's Cape Fear River and its tributaries in the summer of 2012 (Pritchett et al., 2019; Strynar et al., 2015). Since that finding, U.S. studies of surface waters, some of which are source waters for PWSs, have reported results of sampling efforts from contaminated areas near the Cape Fear River (McCord et al., 2018; Sun et al., 2016) and in Ohio and West Virginia (Galloway et al., 2020).

In studies of the Cape Fear River basin by McCord et al. (2018) and Sun et al. (2016), surface water concentrations of HFPO-DA ranged from below the North Carolina Department of Health and Human Services (NCDHHS) provisional health goal (PHG) of 140 ng/L to a maximum level of 4,560 ng/L. Sun et al. (2016) analyzed surface water from two sites upstream of a drinking water treatment plant (DWTP) and one site downstream. They reported a median HFPO-DA concentration of 304 ng/L with a maximum of 4,560 ng/L in the source water of the plant. HFPO-DA levels did not exceed the quantitation limit (10 ng/L) at the two upstream locations. In source water samples collected from the Cape Fear River near a DWTP downstream of a fluorochemical manufacturer, McCord et al. (2018) reported initial HFPO-DA concentrations of approximately 700 ng/L. After the manufacturer diverted waste stream emissions from one of its manufacturing lines, the measured concentrations decreased to levels below the NCDHHS PHG (140 ng/L).

In Ohio and West Virginia, Galloway et al. (2020) sampled rivers and streams located upstream, downstream, and downwind to the north and northeast of the Chemours Washington Works facility outside Parkersburg, West Virginia. The downwind sampling was intended to explore potential airborne deposition. Some of the downstream sampling sites were in the vicinity of landfills. Reported levels of HFPO-DA in these waters ranged from non-detectable levels to a maximum of 227 ng/L. The highest HFPO-DA concentrations were measured downwind of the facility (i.e., to the northeast). The study observed an exponentially declining trend of HFPO-DA concentrations in surface water with distance from the facility in this direction and attributed its occurrence in surface water to air dispersion of emissions from the facility. The most distant site where HFPO-DA was detected was 24 km north of the facility.

In one study of sites located in highly industrialized commercial waterways (authors did not indicate whether sampling sites were in the vicinity of known PFAS point sources), Pan et al. (2018) detected HFPO-DA in 100% of samples from sites in the Delaware River (n=12), reporting median and maximum concentrations of 2.02 ng/L and 8.75 ng/L, respectively, in surface waters.

Globally, HFPO-DA occurrence has been reported in surface waters from Germany (Pan et al., 2018; Heydebreck et al., 2015), China (Li et al., 2020a; Pan et al., 2018; Song et al., 2018; Pan et al., 2017; Heydebreck et al., 2015), the Netherlands (Pan et al., 2018; Gebbink et al., 2017; Heydebreck et al., 2015), the United Kingdom (Pan et al., 2018), South Korea (Pan et al., 2018), and Sweden (Pan et al., 2018). HFPO-DA was also detected with a mean concentration of 30 picograms per liter (pg/L; 0.030 ng/L) in Artic seawater samples, suggesting long-range transport (Joerss et al., 2020).

In one study of surface water collected from industrialized areas in Europe (authors did not indicate whether sampling sites were in the vicinity of known PFAS point sources), Pan et al. (2018) reported HFPO-DA detections in 100% of samples from the Thames River in the United Kingdom (n=6 sites), the Rhine River in Germany and the Netherlands (n=20 sites), and the Malaren Lake in Sweden (n=10 sites). Across these three river systems, median HFPO-DA concentrations ranged from 0.90 to 1.38 ng/L and the highest concentration detected was 2.68 ng/L. In another study, Heydebreck et al. (2015) detected HFPO-DA at 17% of sampling locations on the industrialized non-estuarine reaches of the Rhine River, with a maximum concentration of 86.08 ng/L; however, HFPO-DA was not detected at locations on the Elbe River.

Gebbink et al. (2017) evaluated surface water samples upstream and downstream of a fluorochemical production plant in the Netherlands and reported only one of three samples upstream of the plant with detectable HFPO-DA concentrations (22 ng/L; method quantification limit [MQL] = 0.2 ng/L). Downstream of the fluorochemical plant, HFPO-DA was detected in 100% of samples, with a mean concentration of 178 ng/L and a range of 1.7 to 812 ng/L. Vughs et al. (2019) analyzed surface water from 11 water suppliers in the Netherlands and Belgium, some of which were in the vicinity of a fluoropolymer manufacturing plant. The authors reported HFPO-DA detections in 77% of surface water samples (n=13) with a mean concentration of 2.2 ng/L and a maximum of 10.2 ng/L; however, only three samples in the study had HFPO-DA concentrations exceeding 1 ng/L.

Of the five studies conducted in China, one study evaluated surface water samples from an industrialized region (authors did not indicate whether sampling sites were in the vicinity of known PFAS point sources) (Pan et al., 2018), one study evaluated surface water river and reservoir samples in an industrialized river basin with potential PFAS point sources (Li et al., 2020a), and three studies examined samples from sites along the Xiaoqing river at locations upstream, downstream, or in the vicinity of known PFAS sources (Song et al., 2018; Pan et al., 2017; Heydebreck et al., 2015). HFPO-DA was detected in freshwater systems sampled in all five studies, though HFPO-DA concentrations appeared to be positively correlated with proximity to known PFAS point sources. Song et al. (2018), Pan et al. (2017), and Heydebreck et al. (2015) sampled sites in the Xiaoqing River system, including one of its tributaries, nearby a known fluoropolymer production facility. These three studies reported maximum HFPO-DA concentrations of 9,350, 2,060, and 3,060 ng/L, respectively. HFPO-DA concentrations in samples collected upstream of the facility did not exceed 3.64 ng/L. Other Chinese freshwater systems evaluated in the other two studies (Li et al., 2020a; Pan et al., 2018) generally reported maximum concentrations like those from the upstream Xiaoqing River system sites (≤ 10.3 ng/L), except for one site in Tai Lake which was reported to have a maximum HFPO-DA concentration of 143 ng/L. Similarly, in a study that sampled an industrialized river in South Korea (authors did not report whether sampling sites were in the vicinity of known PFAS point sources), HFPO-DA was found in 100% of samples and the maximum concentration found was 2.49 ng/L (Pan et al., 2018).

Table A-1. Compilation of Studies Describing HFPO-DA Occurrence in Surface Water

Study	Location	Site Details	Results
North America			
Sun et al. (2016)	United States (North Carolina, Cape Fear River Basin)	Source waters of three community drinking water treatment plants, two upstream and one downstream of a PFAS manufacturing plant (LOQ = 10 ng/L)	Community A (upstream): DF 0% Community B (upstream): DF NR, median (range) = ND (ND-10 ng/L) Community C (downstream): DF NR, mean = 631 ng/L, median (range) = 304 (55–4,560) ng/L
McCord et al. (2018)	United States (North Carolina, Cape Fear River Basin)	Source water of a drinking water treatment plant near the industrial waste outfall of a fluorochemical manufacturer, before and after the manufacturer diverted a waste stream (exact values NR, estimated values from Figure 3)	Before waste diversion (estimated): DF NR, measured concentration = ~ >700 ng/L After waste diversion (estimated): DR NR, measured concentration = < 140 ng/L
Galloway et al. (2020)	United States (Ohio and West Virginia, Ohio River Basin)	Rivers and tributaries located upstream, downstream, and downwind of a fluoropolymer production facility; some sample locations potentially impacted by local landfills	DF = 21/24 unique sites with detections > LOQ, median[a] (range) = 46.7 (ND–227) ng/L
Europe			
Gebbink et al. (2017)	The Netherlands	Upstream and downstream of the Dordrecht fluorochemical production plant; two control sites	Control sites: DF 0% Upstream of plant (n=3): DF[a] 33%, point = 22 ng/L Downstream of plant (n=13): DF 100%, mean[a] (range) = 178 (1.7–812) ng/L (MQL = 0.2)

Study	Location	Site Details	Results
Vughs et al. (2019)	The Netherlands and Belgium	Thirteen surface water samples collected from eleven water suppliers, some near a fluoropolymer manufacturing plant. The study did not map the distribution of reported concentrations by geographic location or with respect to distance from the fluoropolymer manufacturing plant.	DF 77%, mean (range) = 2.2 (ND–10.2) ng/L (LOQ = 0.2 ng/L)
Asia			
Pan et al. (2017)	China (Xiaoqing River and tributary)	Upstream and downstream of a fluoropolymer production plant in an industrialized region	Upstream of plant in the Xiaoqing River (n=6): DF[a] 100%, median[a] (range) = 2.10 (1.61–3.64) ng/L Tributary directly receiving plant effluent (n=4): DF[a] 100%, median[a] (range) = 1,855 (2.34-2,060) ng/L Downstream of plant in the Xiaoqing River receiving tributary waters (n=8): DF[a] 100%, median[a] (range) = 311 (118–960) ng/L
Song et al. (2018)	China (Xiaoqing River)	Near the Dongyue group industrial park, including a fluoropolymer production plant	DF NR, mean, median (range) = 519, 36.7 (<LOQ–9,350) ng/L (n=25 sites; LOQ=0.24 ng/L)
Li et al. (2020)	China (Hai River Basin)	40 surface water samples from 8 rivers and 3 reservoirs – many of the rivers flowed through industrialized areas, some with potential PFAS point sources	DF[b] 80%, mean (range) = 0.316 (<MDL–2.6) ng/L (MDL = 0.0132 ng/L)
Multiple Continents			

Study	Location	Site Details	Results
Heydebreck et al. (2015)	Germany (Elbe and Rhine Rivers), the Netherlands (Rhine-Meuse delta)	All sampling locations in industrialized areas	Rhine River (n=23): DF[a] 17%, range = ND–86.08 ng/L Elbe River (n=22): DF 0%
	China (Xiaoqing River)	Some sampling locations were downstream of PFAS point sources	Xiaoqing River (n=20): DF[a] 65%, range = ND–3,060 ng/L
Pan et al. (2018)	United States (Delaware River)	Sampling sites along industrialized river systems that were not proximate to known point sources of PFAS from fluorochemical facilities	Delaware River (n=12): DF 100%, mean, median (range) = 3.32, 2.02 (0.78–8.75) ng/L
	United Kingdom (Thames River), Germany and the Netherlands (Rhine River), Sweden (Malaren Lake)	Sampling sites along industrialized river systems that were not proximate to known point sources of PFAS from fluorochemical facilities	Thames River (n=6): DF 100%, mean, median (range) = 1.12, 1.10 (0.70–1.58) ng/L Rhine River (n=20): DF 100%, mean, median (range) = 0.99, 0.90 (0.59–1.98) ng/L Malaren Lake (n=10): DF 100%, mean, median (range) = 1.47, 1.38 (0.88–2.68) ng/L

Study	Location	Site Details	Results
	South Korea (Han River), China (Liao, Huai, Yellow, Yangtze, and Pearl Rivers; Chao and Tai Lakes)	Sampling sites along industrialized river systems that were not proximate to known point sources of PFAS from fluorochemical facilities	Han River (n=6): DF 100%, mean, median (range) = 1.38, 1.16 (0.78–2.49) ng/L Liao River (n=6): DF 100%, mean, median (range) = 1.44, 0.88 (0.62–4.51) ng/L Huai River (n=9): DF 100%, mean, median (range) = 1.66, 1.40 (0.83–3.62) ng/L Yellow River (n=15): DF 67%, mean, median (range) = 1.01, 1.30 (< LOQ–1.74) ng/L Yangtze River (n=35): DF 94%, mean, median (range) = 0.73, 0.67 (< LOQ–1.54) ng/L Pearl River (n=13): DF 100%, mean, median (range) = 1.51, 0.70 (0.21–10.3) ng/L Chao Lake (n=13): DF 100%, mean, median (range) = 1.92, 1.81 (0.93–3.32) ng/L Tai Lake (n=15): DF 100%, mean, median (range) = 14.0, 0.77 (0.38–143.7) ng/L (LOQ = 0.05 ng/L; MDL = 0.38 ng/L)
	All locations	Sampling sites were not proximate to known point sources of any fluorochemical facilities	All locations (n=160): DF 96%, mean, median (range) = 2.55, 0.95 (0.18–144) ng/L (LOQ = 0.05 ng/L; MDL = 0.38 ng/L)

Notes:
DF = detection frequency; LOQ = limit of quantification; ND = not detected.; ng/L = nanograms per liter; NR = not reported; MQL = method quantification limit; MDL = method detection limit.
[a] The DF, median, and/or mean was not reported in the study and was calculated in this synthesis. Mean values were only calculated if DF = 100%.

[b] The DF in Li et al. (2020a) was reported as 82.5% in the main article. The DF of 80% shown in this table is based on the supporting information data, which show only 32/40 samples with data > MDL.
[c] The Xiaoqing River results reported in Heydebreck et al. (2015) included samples from Laizhou Bay. The EPA considered freshwater samples only.

A.2. RSC for HFPO-DA, Literature Search and Screening Methodology

The EPA applies an RSC to the RfD when calculating an MCLG based on noncancer effects or for carcinogens that are known to act through a nonlinear mode of action to account for the fraction of an individual's total exposure allocated to drinking water (USEPA, 2000b). The EPA emphasizes that the purpose of the RSC is to ensure that the level of a chemical allowed by a criterion (e.g., the MCLG for drinking water) or multiple criteria, when combined with other identified sources of exposure (e.g., diet, ambient and indoor air) common to the population of concern, will not result in exposures that exceed the RfD. In other words, the RSC is the portion of total daily exposure equal to the RfD that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee tea or soup, as well as from transfer to dietary items prepared with drinking water) relative to other exposure sources; the remainder of the exposure equal to the RfD is allocated to other potential exposure sources. For example, if for a particular chemical, drinking water were to represent half of total exposure and diet were to represent the other half, then the drinking water contribution (or RSC) would be 50%. The EPA considers any potentially significant exposure source when deriving the RSC.

The RSC is derived by applying the Exposure Decision Tree approach published in the EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000b). The Exposure Decision Tree approach allows flexibility in the RfD apportionment among sources of exposure and considers several characteristics of the contaminant of interest, including the adequacy of available exposure data, levels of the contaminant in relevant sources or media of exposure, and regulatory agendas (i.e., whether there are multiple health-based criteria or regulatory standards for the contaminant). The RSC is developed to reflect the exposure to the U.S. general population or a sensitive population within the U.S. general population and may be derived qualitatively or quantitatively, depending on the available data.

A quantitative RSC determination first requires "data for the chemical in question… representative of each source/medium of exposure and… relevant to the identified population(s)" (USEPA, 2000b). The term "data" in this context is defined as ambient sampling measurements in the media of exposure, not internal human biomonitoring metrics. More specifically, the data must adequately characterize exposure distributions including the central tendency and high-end exposure levels for each source and 95% confidence intervals for these terms (USEPA, 2000b). Frequently, an adequate level of detail is not available to support a quantitative RSC derivation. When adequate quantitative data are not available, the agency relies on the qualitative alternatives of the Exposure Decision Tree approach. A qualitatively-derived RSC is an estimate that incorporates data and policy considerations and thus, is sometimes referred to as a "default" RSC (USEPA, 2000b). Both the quantitative and qualitative approaches recommend a "ceiling" RSC of 80% and a "floor" RSC of 20% to account for uncertainties including unknown sources

of exposure, changes to exposure characteristics over time, and data inadequacies (USEPA, 2000b).

In cases in which there is a lack of sufficient data describing environmental monitoring results and/or exposure intake, the Exposure Decision Tree approach results in a recommended RSC of 20%. In the case of MCLG development, this means that 20% of the exposure equal to the RfD is allocated to drinking water and the remaining 80% is reserved for other potential sources, such as diet, air, consumer products, etc. This 20% RSC value can be replaced if sufficient data are available to develop a scientifically defensible alternative value. If scientific data demonstrating that sources and routes of exposure other than drinking water are not anticipated for a specific pollutant, the RSC can be raised as high as 80% based on the available data, allowing the remaining 20% for other potential sources (USEPA, 2000b). Applying a lower RSC (e.g., 20%) is a more conservative approach to public health and results in a lower MCLG.

A.2.1. Literature Search and Screening

In support of the EPA's human health toxicity assessment for HFPO-DA (USEPA, 2021c), literature searches were conducted of four databases (PubMed, Toxline, Web of Science (WOS), and Toxic Substances Control Act Test Submissions (TSCATS)) to identify publicly available literature using CASRN, synonyms, and additional relevant search strings (see USEPA (2021c) for details). Due to the limited search results, additional databases were searched for information on physicochemical properties, health effects, toxicokinetics, and mechanism of action. The initial date-unlimited database searches were conducted in July 2017 and January/February 2018, with updates completed in February 2019, October 2019, and March 2020. In addition, information on toxicokinetics; acute, short-term, subchronic, and chronic toxicity; developmental and reproductive toxicity; neurotoxicity; immunotoxicity; genotoxicity; and cancer in animals was submitted with premanufacture notices to the EPA by DuPont/Chemours, the manufacturer of HFPO-DA, as required under the Toxic Substances Control Act pursuant to a consent order (USEPA, 2009b) or reporting requirements (15 U.S.C. § 2607.8(e)). The results of the literature searches of publicly available sources and submitted studies from DuPont/Chemours are available through the EPA's Health & Environmental Resource Online website at https://hero.epa.gov/hero/index.cfm/project/page/project_id/2627.

The HFPO-DA literature search results and all studies submitted by DuPont/Chemours were imported into SWIFT-Review (Sciome, LLC, Research Triangle Park, NC) and filtered through the Evidence Stream tags to identify human studies and non-human (i.e., those not identified as being in humans) studies. Studies identified as human studies were further categorized into seven major PFAS pathway categories (Cleaning Products, Clothing, Environmental Media, Food Packaging, Home Products/Articles/Materials, Personal Care Products, and Specialty Products) as well as an additional category for Human Exposure Measures. Non-human studies were grouped into the same seven major PFAS pathway categories, except that the Environmental Media category did not include soil, wastewater, or landfill.

Application of the SWIFT-Review tags identified 52 studies for title and abstract screening. An additional three references were identified through gray literature sources that were included to supplement the search results. Title and abstract screening to determine relevancy followed the populations, exposures, comparators, and outcomes (PECO) criteria in Table A-2:

Table A-2. Populations, Exposures, Comparators, and Outcomes (PECO) Criteria

PECO Element	Inclusion Criteria
Population	Adults (including women of childbearing age) and/or children in the general populations from any country
Exposure	Primary data from peer-reviewed studies collected in any of the following media: ambient air, consumer products, drinking water, dust, food, food packaging, groundwater, human blood/serum/urine, indoor air, landfill, sediment, soil, surface water (freshwater), wastewater/biosolids/sludge
Comparator	Not applicable
Outcome	Measured concentrations of HFPO-DA (or measured emissions from food packaging and consumer products only)

The title and abstract of each study were independently screened for relevance by two screeners using *litstream*^{TM}. A study was included as relevant if it was unclear from the title and abstract whether it met the inclusion criteria. When two screeners did not agree if a study should be included or excluded, a third reviewer was consulted to make a final decision. The title and abstract screening resulted in 24 studies tagged as relevant (i.e., data on occurrence of HFPO-DA in one of the media of interest were presented in the study) that were further screened with full-text review using the same inclusion criteria. Of these 24 studies, 4 contain only human biomonitoring data and are not discussed further here. Based on full-text review, 15 studies were identified as relevant and are summarized below. At the full-text review stage, two additional studies were identified as only containing biomonitoring data.

A.2.2. Additional Screening

To supplement the primary literature database, the EPA also searched the following publicly available gray literature sources in February 2022 for information related to relative exposure of HFPO-DA for all potentially relevant routes of exposure (oral, inhalation, dermal) and exposure pathways relevant to humans:

- USEPA (2021c) *Human Health Toxicity Values for Hexafluoropropylene Oxide (HFPO) Dimer Acid and Its Ammonium Salt (CASRN 13252-13-6 and CASRN 62037-80-3) Also Known as "GenX Chemicals"*;
- ATSDR's *Toxicological Profiles*;
- Centers for Disease Control and Prevention's (CDC's) national reports on human exposures to environmental chemicals;
- EPA's CompTox Chemicals Dashboard;
- EPA's fish tissue studies;
- EPA's Toxics Release Inventory;
- Relevant documents submitted under the Toxic Substances Control Act and relevant reports from the EPA's Office of Chemical Safety and Pollution Prevention;

- U.S. Food and Drug Administration's (FDA's) *Total Diet Studies* and other similar publications from FDA, U.S. Department of Agriculture, and Health Canada;
- National Oceanic and Atmospheric Administration's (NOAA's) National Centers for Coastal Ocean Science data collections;
- National Science Foundation direct and indirect food and/or certified drinking water additives;
- PubChem compound summaries;
- Relevant sources identified in the relative source contribution discussions (Section 5) of the EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA)/Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water*; and
- Additional sources, as needed.

The EPA has included available information from these gray literature sources for HFPO-DA relevant to its uses, chemical and physical properties, and for occurrence in ambient or indoor air, foods (including fish and shellfish), soil, dust, and consumer products. The EPA has also included available information specific to HFPO-DA below on any regulations that may restrict HFPO-DA levels in media (e.g., water quality standards, air quality standards, food tolerance levels).

The EPA incorporated six references (Li et al., 2022; Burkhard, 2021; Feng et al., 2021; Straková et al., 2021; Semerád et al., 2020; Geosyntec, 2019) that were not identified in the EPA's RSC literature search strategy; these references were provided by Chemours as part of their outreach to the EPA on uses and sources for HFPO-DA in April 2022 and/or through the public comment period for the proposed PFAS NPDWR in 2023.

A.3. Summary of Potential Exposure Sources of HFPO-DA Other than Water

A.3.1. Dietary Sources

HFPO-DA was included in a suite of individual PFAS selected as part of PFAS-targeted reexaminations of samples collected for the U.S. Food and Drug Administration's (FDA's) Total Diet Study (US FDA, 2022b, 2022a, 2021b, 2021a, 2020b, 2020a); however, it was not detected in any of the food samples tested. It should be noted that FDA indicated that the sample sizes were limited and that the results should not be used to draw definitive conclusions about PFAS levels or presence in the general food supply (US FDA, 2022c). HFPO-DA was not detected in cow milk samples collected from a farm with groundwater known to be contaminated with PFAS; however, it was detected in produce (collard greens, cabbage) collected from an area near a PFAS production plant in FDA studies of the potential exposure to the U.S. population to PFAS (US FDA, 2021c, 2018). HFPO-DA was detected at low levels in 14% of vegetable garden crops (endive, beets, celery, lettuce, and tomatoes) grown near a PFAS manufacturing facility in the Netherlands (NCDEQ, 2018c; Mengelers et al., 2018). An exposure assessment of the Chemours Fayetteville Works Facility located in Bladen County, North Carolina evaluated HFPO-DA in fish fillet tissue samples collected between July and September 2019 at five locations within the Cape Fear River (Geosyntec, 2019). HFPO-DA was detected in three

samples of largemouth bass from the Bladen Bluffs location which is approximately eight miles downstream of the Fayetteville Works Facility at concentrations of 68,000 ng/kg, 54,000 ng/kg, and 24,000 ng/kg (Geosyntec, 2019). HFPO-DA was not detected in fish (largemouth bass, catfish, bluegill, or redbreast sunfish) collected from the other sites located upstream, adjacent to the site, 4 miles downstream, or in an on-site pond.

Feng et al. (2021) measured HFPO-DA in food samples collected from up to ten home gardens or farms in villages within 15 km of a large fluoropolymer facility located on the Dongzhulong River in Shandong Province, China. The authors detected HFPO-DA in wheat (mean concentration: 5.53 nanograms per gram dry weight [ng/g dw]; range: 2.27–9.19 ng/g dw; detection frequency [DF] 100%), maize (mean concentration: 1.17 ng/g dw; range: not detected (ND)–1.94 ng/g dw; DF 80%), and vegetable samples (mean concentration: 20.1 ng/g dw; range: ND–67.2 ng/g dw; DF 82%). In fish collected at two sites along the Dongzhulong River, HFPO-DA was detected at concentrations of 43.9 and 3.23 ng/g dw at sites approximately 3 km and 15 km downstream of the fluoropolymer facility, respectively. HFPO-DA was not found in eggs (home-produced and store-bought), store-bought meat or seafood, or milk from domestic goats (Feng et al., 2021). Except for the fish sampled at two sites, the study did not report HFPO-DA concentrations in food according to sampling location or proximity to the fluoropolymer facility.

HFPO-DA was not a target chemical in the EPA's National Lake Fish Tissue Study or the EPA's 2015 Great Lakes Human Health Fish Fillet Tissue Study, nor in the EPA's 2008–2009 or 2013–2014 National Rivers and Streams Assessment studies (USEPA, 2021e, 2020a; Stahl et al., 2014; USEPA, 2009a). HFPO-DA was detected in a redear sunfish fillet composite sample collected from a privately-owned lake near a PFAS manufacturing facility in North Carolina at a concentration of 270 nanograms per kilogram (ng/kg) (wet weight tissue) (USEPA, 2021c; NCDEQ, 2018c). HFPO-DA was not included in the NOAA's National Centers for Coastal Ocean Science, National Status and Trends Data (NOAA, 2022). Burkhard (2021) identified a single bioaccumulation factor (BAF) in muscle tissue/fillet reported in the literature of 4.07 L/kg wet weight (reported as a logBAF of 0.61 L/kg).

A.3.2. Food Contact Materials

In an analysis performed at the Department of Food Analysis and Nutrition of the University of Chemistry and Technology in Prague, Czech Republic, HFPO-DA was not detected in 42 samples of disposable food packaging and tableware purchased from six different European countries between May and December 2020 (LOQ = 1.7 mg/kg) (Straková et al., 2021).

A.3.3. Consumer Products

Although no specific studies on the occurrence of HFPO-DA in consumer products were identified, DuPont began transitioning to GenX processing aid technology in 2009 to work toward eliminating long-chain PFAS as part of the company's commitment under the 2010/2015 PFOA Stewardship Program (USEPA, 2021c). It is unknown if HFPO-DA in consumer products have increased as a result of this transition.

A.3.4. Indoor Dust

Feng et al. (2021) detected HFPO-DA in indoor dust samples taken from homes from 10 villages within 15 km of a large fluoropolymer facility in Shandong Province, China, at concentrations

ranging from ND to 841 ng/g (mean concentration 159 ng/g; DF 72%). Contaminated dust was found in homes as far as 15 km from the fluoropolymer facility and HFPO-DA concentrations were highest in homes nearest to the facility. Although only one study on the occurrence of HFPO-DA in indoor dust was identified, other PFAS have been detected in indoor dust and on window films (ATSDR, 2021).

A.3.5. Air

PFAS have been released to air from wastewater treatment plants, waste incinerators, and landfills (USEPA, 2016a). HFPO-DA could be transported in the vapor phase or with particulates (USEPA, 2021c). When released to air or volatilized from water, HFPO-DA is stable and short- and long-range transport has occurred (D'Ambro et al., 2021; Galloway et al., 2020). Galloway et al. (2020) analyzed HFPO-DA concentrations in soil samples downwind of and surface water samples upstream of the Chemours Washington Works facility outside of Parkersburg, West Virginia, and results suggest atmospheric transport of HFPO-DA emissions. Additionally, a study that modeled the atmospheric transport of a PFAS mixture containing HFPO-DA from a fluoropolymer manufacturing facility in North Carolina (D'Ambro et al., 2021) predicted that only 2.5% of total HFPO-DA (consisting of HFPO-DA and HFPO-DA fluoride) would be deposited within 150 km of the facility (USEPA, 2021c).

HFPO-DA is persistent in air (half-lives longer than 6 months), and not readily broken down by biodegradation, direct photolysis, or hydrolysis (USEPA, 2021c). In the vapor phase, HFPO-DA is expected to undergo hydroxyl radical-catalyzed indirect photolysis slowly, with a predicted average hydroxylation rate of 8.50 x 10-13 cubic centimeters (cm^3)/molecule - second (USEPA, 2022f, 2022e, 2021c). Based on a measured vapor pressure of 2.7 mm Hg at 20°C for HFPO-DA, volatilization is expected to be an important fate process for this chemical (USEPA, 2021c). EPA's Toxics Release Inventory reported release data for HFPO-DA in 2020 (USEPA, 2022c). HFPO-DA is not listed as a hazardous air pollutant (USEPA, 2022d).

HFPO-DA has been identified in air emissions. North Carolina Department of Environmental Quality (NCDEQ) estimates for the Chemours Fayetteville Works plant, located in the North Carolina Cape Fear watershed, indicate that annual emissions of HFPO-DA could have exceeded 2,700 pounds per year during the reporting period (2017–2018) (NCDEQ, 2018a). Rainwater samples collected within a seven-mile radius of this facility were reported to have detectable levels of HFPO-DA (NCDEQ, 2018b), with the highest concentration of 810 ng/L found in a rainwater sample collected five miles from the facility. The three samples collected seven miles from the plant had HFPO-DA concentrations ranging from 45.3 to 60.3 ng/L (NCDEQ, 2018b).

A.3.6. Soil

When HFPO-DA is deposited on or applied to soil, it is expected to run off into surface waters or rapidly leach to groundwater (USEPA, 2021c). PFAS can also be taken up from contaminated soil by plants (ATSDR, 2021). No specific studies on the occurrence of HFPO-DA in biosolids were identified. An exposure assessment of the Chemours Fayetteville Works Facility located in Bladen County, North Carolina evaluated HFPO-DA in soil samples collected between July and September 2019 at twelve offsite locations (Geosyntec, 2019). HFPO-DA was detected in two of four surface soil samples (0 to 0.5 ft depth) collected within a 2.5 km radius of the facility at concentrations of 2,600 ng/kg and 360 ng/kg. HFPO-DA was also detected in two of four

subsurface soil samples (4 to 4.5 ft depth) collected within a 2.5 km radius of the facility at concentrations of 430 ng/kg and 590 ng/kg, and in one of four subsurface samples (excluding duplicates) collected within a 5 km radius to the facility at a concentration of 400 ng/kg (Geosyntec, 2019). HFPO-DA was not detected in surface or subsurface soil samples collected in a 10 km radius to the facility.

Two peer-reviewed studies reported HFPO-DA concentrations in soil. In the United States, Galloway et al. (2020) analyzed 13 soil samples for HFPO-DA at locations in Ohio and West Virginia that were upstream and downwind of the Chemours Washington Works facility in order to evaluate HFPO-DA contamination due to atmospheric deposition. HFPO-DA was detected in 5 out of 13 samples, with a maximum concentration of 8.14 ng/g dw. In China, Li et al. (2020b) collected and analyzed residential soil samples throughout the country from 31 provincial-level administrative regions (consisting of 26 provinces, 4 municipalities, and 1 special administrative region). HFPO-DA was detected in 40.5% of soil samples at concentrations up to 967 picograms per gram (pg/g) dw and a mean level of 19.1 pg/g dw. PFOA was detected in these soils more frequently (96.6%) and at higher mean levels (354 pg/g dw), leading the authors to conclude that HFPO-DA consumption was still limited at the national scale of China, despite its use as a PFOA replacement.

One study measured concentrations of HFPO-DA in and/or on grass and leaves collected from sites various distances from a fluoropolymer manufacturing plant in the Netherlands (Brandsma et al., 2019). HFPO-DA concentrations ranged from 86 ng/g in leaves from a site closest to the plant to ND furthest from the plant. A similar pattern was observed for grass samples, except the maximum HFPO-DA concentration was lower (27 ng/g). The study authors note that it hadn't rained for five days prior to sample collection.

Semerád et al. (2020) investigated occurrence of HFPO-DA in sewage sludge from 43 facilities in the Czech Republic. HFPO-DA was detected in 7 of 43 samples at concentrations ranging from 0.3 to 1.2 ng/g dw. The authors raised concerns about the agricultural use of sludge containing PFAS for growing crops.

A.3.7. Sediment

HFPO-DA is expected to remain in water and exhibit low partitioning to sediment (USEPA, 2021c). One study evaluated the occurrence of HFPO-DA in sediments from the North and Baltic Seas in Europe, and reported that HFPO-DA was not detected in any of the 24 sediment samples taken in the North and Baltic Seas in the vicinity of Germany (Joerss et al., 2019). An additional four studies analyzed sediments in China (Li et al., 2022; Li et al., 2020b; Wang et al., 2019a; Song et al., 2018). Of the four studies, Wang et al. (2019a) analyzed sediment from the South China Sea coastal region in the area of the highly industrialized Pearl River Delta and reported that HFPO-DA was below the LOQ in all 53 samples. Li et al. (2020) analyzed 20 sediment samples from eight rivers and three reservoirs in the Hai River Basin in the vicinity of several industrialized areas. HFPO-DA was reportedly detected at minimal levels, but the authors did not report actual concentrations. Song et al. (2018) analyzed concentrations of HFPO-DA in 24 sediment samples from the Xiaoqing River in the vicinity of a fluoropolymer production facility. The study reported a maximum HFPO-DA concentration in sediment of 22.3 ng/g dw, with median and mean levels below the LOQ. Li et al. (2022) also analyzed sediment samples

from five sites of the Xiaoqing River estuary and reported a mean HFPO-DA concentration of 0.23 ng/g dw.

A.4. Recommended RSC

The EPA followed the Exposure Decision Tree approach to determine the RSC for HFPO-DA (USEPA, 2000b). The EPA first identified three potential populations of concern (Box 1): pregnant women, lactating women, and women of childbearing age (see Section 2.1.2). However, limited information was available regarding specific exposure of these populations to HFPO-DA in different environmental media. The EPA considered exposures in the general U.S. population as likely being applicable to these two populations. Second, the EPA identified several relevant HFPO-DA exposures and pathways (Box 2), including dietary consumption, incidental oral consumption via dust and soil or dermal exposure via soil and dust, and inhalation exposure via ambient air. Several of these may be potentially significant exposure sources. Third, the EPA determined that there was inadequate quantitative data to describe the central tendencies and high-end estimates for all of the potentially significant sources (Box 3). For example, a study from China indicates that exposure via dust may be a significant pathway for HFPO-DA. At the time of the literature search, the EPA was unable to identify studies assessing HFPO-DA concentrations in dust samples from the U.S. and therefore, the agency does not have adequate quantitative data to describe the central tendency and high-end estimate of exposure for this potentially significant source in the U.S. population. However, the agency determined there were sufficient data, physical/chemical property information, fate and transport information, and/or generalized information available to characterize the likelihood of exposure to relevant sources (Box 4). Notably, based on the studies summarized in the sections above, there are significant known or potential uses/sources of HFPO-DA other than drinking water (Box 6), though there is not information available on each source to make a characterization of exposure (Box 8A). For example, the EPA identified physico-chemical properties of HFPO-DA indicating that volatilization may be an important fate process for this chemical. There is evidence in the literature of atmospheric transport of HFPO-DA and occurrence in rainwater. However, monitoring data describing the occurrence of HFPO-DA in ambient air is limited to a single report from an area located nearby a known point source of this chemical. The levels of HFPO-DA in ambient air at this source may not be representative of the U.S. as a whole. Therefore, it is not possible to determine whether ambient air can be considered a major or minor contributor to total HFPO-DA exposure in the U.S. general population. Similarly, it is not possible to determine whether the other potentially significant exposure sources such as vegetables and soil should be considered major or minor contributors to total HFPO-DA exposure. Given these considerations, following recommendations of the Exposure Decision Tree (USEPA, 2000b), the EPA recommends an RSC of 20% (0.20) for HFPO-DA.

Appendix B. PFBS: Summary of Occurrence in Water and Detailed Relative Source Contribution

B.1. Occurrence in Water

PFBS can enter the aquatic environment through releases from manufacturing sites, industrial uses, fire/crash training areas, and wastewater treatment facilities, as well as from the use of contaminated biosolids (USEPA, 2021d; ATSDR, 2021). PFBS has a high solubility in water (52.6 g/L at 22.5–24 °C for the potassium salt) and high mobility in the environment (log Koc 1.2 to 2.7) (ECHA, 2019). A literature review of physicochemical properties and environmental monitoring data for PFBS by the Norwegian Environment Agency determined that volatilization from water is negligible, but that the presence of PFBS in ambient air can result from direct emissions or transport of droplets in contaminated water (Arp and Slinde, 2018). PFBS has been found in rain as well as in snow/ice in the Arctic and Antarctic (Arp and Slinde, 2018).

The EPA collected information about PFBS occurrence in water (described below). To better understand PFBS sources and occurrence patterns in water, this section includes studies conducted within and outside the United States. Overall, studies that analyzed water from sites receiving inputs from or in proximity to known sources of PFAS (as reported by study authors) did not provide a consistent pattern of detection; increased PFBS detection frequencies (DFs) or concentrations were not only observed in studies of sites with known sources of PFAS contamination from point sources. Specifically, DFs of 0% were reported at some sites with known, suspected, or historic PFAS contamination, and DFs of 100% were reported at some sites with no known point sources of PFAS contamination. However, the maximum reported PFBS concentrations in groundwater and surface water were measured at sites with known PFAS contamination from AFFF usage (Anderson et al., 2016).

B.1.1. Ground Water

Several studies evaluated the occurrence of PFBS in raw groundwater in the United States or Europe (see Table B-1). Most of the available studies sampled from groundwaters known or suspected to be contaminated with PFAS through various sources, as reported by the study authors. Importantly, some of these groundwaters are known to be used as input sources for PWSs.

The EPA identified four U.S. studies assessing PFBS concentrations in groundwater at sites known to be contaminated with PFAS from the use of AFFF (Steele et al., 2018; Eberle et al., 2017; Anderson et al., 2016; Moody et al., 2003). Of the three studies that reported PFBS detections, two reported DFs of 78.26% and 100% (Eberle et al., 2017; Anderson et al., 2016); the third study did not report a PFBS DF across sample sites but indicated a range of PFBS concentrations (ND–48 ng/L) (Steele et al., 2018). The fourth study, which analyzed groundwater from the decommissioned Wurtsmith Air Force Base, did not detect PFBS at any of the ten sites sampled, though other PFAS were detected (Moody et al., 2003). However, a case study published by the Association of State and Territorial Solid Waste Management Officials reported quantifiable levels of PFBS in four of seven samples tested from the Wurtsmith Air

Force Base; one site sampled directly below the fire training area was reported to have a PFBS concentration of 4,100 ng/L (ASTSWMO, 2015).

Additionally, PFBS has been detected at concentrations ranging from 0.00211 ng/L to 0.0261 ng/L in groundwater wells (100% well DF) at a site near the 3M Cottage Grove perfluorochemical manufacturing facility in Minnesota (ATSDR, 2021; 3M, 2007). Lee et al. (2015) evaluated urban shallow groundwater contaminated by wastewater effluent discharge and reported a DF of 20% (1 of 5 shallow sites) and a maximum PFBS level of 36.3 ng/L. In contrast, Procopio et al. (2017) collected groundwater from 17 sampling sites (53 total across all water types sampled), some of which were located downstream of an industrial facility that used materials containing PFOA. PFBS was not detected in groundwater collected from any of the sampling locations. Post et al. (2013) assessed raw water from PWS intakes in New Jersey; these intake locations were selected to represent New Jersey geographically and they were not necessarily associated with any known PFAS release. PFBS was detected pre-treatment in 1 of 18 systems at a concentration of 6 ng/L (minimum reporting level = 5 ng/L). Lindstrom et al. (2011) analyzed water from 13 wells intended for uses other than drinking water (e.g., livestock, watering gardens) in areas impacted by up to 12 years of field applications of biosolids contaminated by a fluoropolymer manufacturer. PFBS was detected in three of the wells (mean concentration 10.3 ng/L; range: ND–76.6 ng/L).

Of the 10 identified studies conducted in Europe, seven studies evaluated groundwater samples from sites with known or suspected PFAS releases associated with AFFF use, fluorochemical manufacturing, or other potential emission sources including landfill/waste disposal sites, skiing areas, or areas of unspecific industries that use PFAS in manufacturing (e.g., metal plating) (Dauchy et al., 2019; Høisæter et al., 2019; Gobelius et al., 2018; Dauchy et al., 2017; Gyllenhammar et al., 2015; Wagner et al., 2013; Dauchy et al., 2012). All of these studies reported PFBS detections in at least one sample or site, though only two studies (both conducted in the vicinity of areas with known AFFF usage) reported PFBS concentrations ≥ 100 ng/L (Dauchy et al., 2019; Gyllenhammar et al., 2015). The remaining three studies of the 10 identified did not provide information on whether there were potential sources of PFAS at the sampling locations or were designed to be regionally, nationally, or internationally representative (Barreca et al., 2020; Boiteux et al., 2012; Loos et al., 2010). At these sites, PFBS was detected infrequently (DFs 4 to 18%) with a maximum concentration of 25 ng/L across the three studies.

Table B-1. Compilation of Studies Describing PFBS Occurrence in Groundwater

Study	Location	Site Details	Results
North America			
Lee et al. (2015)	United States (California)	Samples from 5 urban shallow groundwater wells with wastewater contamination	DF[a] 20%, range = ND–36.3 ng/L

Study	Location	Site Details	Results
Appleman et al. (2014)	United States (New Jersey)	Samples from 5 New Jersey groundwater source waters for PWSs impacted by upstream wastewater effluent discharge	DF[a] 100%, mean[a] (range) = 2.4 (0.43–3.7) ng/L
Post et al. (2013)	United States (New Jersey)	Raw water from 18 public drinking water system groundwater intakes	DF 6%, range = ND–6 ng/L
Steele et al. (2018)	United States (Alaska)	Military base contaminated with PFAS from AFFF use (4 wells sampled once per month for 8 months)	DF[a] NR, range = ND–48 ng/L
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, VA)	Former fire training site, site characterization and pretreatment groundwater samples	Site characterization: DF 100%, mean[a] (range) = 3,700 (1,100–13,000) ng/L (10 wells) Pretreatment: DF 100%, mean[a] (range) = 3,400 (1,200–5,000) ng/L (5 wells, 2 laboratory samples/well)
Anderson et al. (2016)	United States (national)	Ten active U.S. Air Force installations with historic AFFF release	DF 78.26%, median of detects (range) = 200 (ND–110,000) ng/L
Moody et al. (2003)	United States (Oscoda, MI)	Groundwater plume at former Wurtsmith Air Force Base; firefighting training area active from 1952 to 1993	DF 0%
Procopio et al. (2017)	United States (New Jersey)	Samples collected from temporary wells in a small area of an industrial/business park located within the Metedeconk River Watershed	DF 0%

Study	Location	Site Details	Results
Lindstrom et al. (2011)	United States (Alabama)	Samples from 13 wells used for purposes aside from drinking water (e.g., livestock, watering gardens, washing), located in areas with historical land application of fluorochemical industry-impacted biosolids	DF[a] 23%, mean (range) = 10.3 (ND–76.6) ng/L
Europe			
Barreca et al. (2020)	Italy (Lombardia region)	Groundwater sampling stations representative of region	DF 18%[a], concentrations NR
Boiteux et al. (2012)	France (national)	Raw water from 2 sampling campaigns of DWTPs, some sites possibly affected by industrial or commercial releases	DF 4%, range = ND–9 ng/L
Loos et al. (2010)	23 European countries	Monitoring stations were not necessarily representative of surrounding area or contaminated	DF 15.2%, range = ND–25 ng/L
Gobelius et al. (2018)	Sweden (national)	Sampling locations selected based on potential vicinity of PFAS hot spots and importance as a drinking water source area	DF 26%[a] (triplicate samples removed), range = ND–22 ng/L
Dauchy et al. (2012)	France (unspecified)	Raw water from 2 DWTPs supplied by alluvial wells; DWTPs located 15 km downstream of fluorochemical manufacturing facility	DF[a] 40%, range = ND–4 ng/L

Study	Location	Site Details	Results
Høisæter et al. (2019)	Norway (unspecified)	Samples from 19 sampling campaigns of 5 pumping wells placed to intercept a groundwater contamination plume originating from a firefighting training facility that ceased usage of PFAS- and fluorotelomer-based AFFF 15 years prior	Detections reported but DF and concentrations not provided
Dauchy et al. (2019)	France (unspecified)	Samples collected over 2 campaigns from 6 areas (13 monitoring wells) of a firefighter training site	DF[a] 77%, range = ND–750 ng/L
Dauchy et al. (2017)	France (unspecified)	Samples collected near 3 sites (A, C, D) impacted by the use of AFFF. Site A results describe 1 sampling location with 2 sampling events. Site C results describe a single sampling location and event. Site D results describe 5 sampling locations, each with a single sampling event	Site A: DF[a] 100% mean[a] = 8 ng/L Site C: point = 6 ng/L Site D: DF[a] 20%, range = ND–59 ng/L

Study	Location	Site Details	Results
Gyllenhammar et al. (2015)	Sweden (Uppsala)	Samples from local aquifers extracted by 21 production wells, 6 observation wells or 1 private well located in the vicinity of a potential AFFF point source (military airport). Results for all well sites were not provided.	Site 1 (production well): DF 0% (n = NR) Site 3 (observation wells): DF 100%, median = 100 ng/L (n = 3) Site 5 (observation well): DF 0% (n = NR) Site 6 (production well): DF 0% (n = NR) Site 7 (observation well): DF 100%, median = 35 ng/L (n = 3) Site 8 (production well): DF[a] 91%, median = 13 ng/L (n = 103) Site 10 (production well): DF[a] 2%, median = ND (n = 50)
Wagner et al. (2013)	Germany (unspecified)	Samples (n = 3) taken downstream from a site contaminated by AFFF from firefighting activities	DF[a] 100%, concentrations NR

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; DWTP = drinking water treatment plant; km = kilometer; ND = not detected; ng/L = nanogram per liter; PFAA = perfluoroalkyl acid; PFAS = per- and polyfluoroalkyl substances; NR = not reported; WWTP = wastewater treatment plant.
[a] The DF and/or mean was calculated using point data. Means were calculated only when DF = 100%.

B.1.2. Surface Water

Studies evaluating the occurrence of PFBS in surface water in North America or Europe are summarized in Table B-2. Broadly, studies either targeted surface waters used as drinking water sources, surface waters known to be contaminated with PFAS (as reported by the study authors), or surface waters over a relatively large geographic area (i.e., statewide) with some or no known point sources of PFAS.

Zhang et al. (2016) identified major sources of surface water PFAS contamination by collecting samples from 37 rivers and estuaries in the northeastern United States (metropolitan New York area and Rhode Island). PFBS was detected at 82% of sites and the range of PFBS concentrations was ND to 6.2 ng/L. Appleman et al. (2014) collected samples of surface water that were impacted by wastewater effluent discharge in several states. PFBS was detected in 64% of samples from 11 sites with a range of PFBS concentrations from ND to 47 ng/L. Several other studies from North America (four from the United States and two from Canada) evaluated

surface waters from sites for which authors did not indicate whether sites were associated with any specific, known PFAS releases (Pan et al., 2018; Yeung et al., 2017; Subedi et al., 2015; Veillette et al., 2012; Nakayama et al., 2010). Nakayama et al. (2010) also collected samples across several states, but no specific source of PFAS was identified. The DF in the Nakayama et al. (2010) study was 43% with median and maximum PFBS levels of 0.71 and 84.1 ng/L, respectively. Pan et al. (2018) sampled surface water sites in the Delaware River and reported a 100% DF, though PFBS levels were relatively low (0.52 to 4.20 ng/L); Yeung et al. (2017) reported results for a creek (PFBS concentration of 0.02 ng/L) and a river (no PFBS detected) in Canada. Veillette et al. (2012) analyzed surface water from an Arctic lake and detected PFBS at concentrations ranging from 0.011 to 0.024 ng/L. Subedi et al. (2015) evaluated lake water potentially impacted by septic effluent from adjacent residential properties, and detected PFBS in only one sample at a concentration of 0.26 ng/L.

Additional available studies assessed surface water samples at U.S. sites contaminated with PFAS from nearby PFAS manufacturing facilities (ATSDR, 2021; Galloway et al., 2020; Newsted et al., 2017; Newton et al., 2017) or facilities that manufacture products containing PFAS (Procopio et al., 2017; Zhang et al., 2016; Lasier et al., 2011). A few of these studies identified potential point sources of PFAS contamination, including industrial facilities (e.g., textile mills, metal plating/coating facilities), airports, landfills, and wastewater treatment plants (WWTPs) (Galloway et al., 2020; Zhang et al., 2016). Among these sites, DFs (0 to 100%) and PFBS levels (ND to 336 ng/L) varied. In general, DFs that ranged from 0 to 3% were associated with samples collected upstream of PFAS point sources, and higher DFs (up to 100%) and PFBS concentrations were associated with samples collected downstream of point sources. An additional study (Lindstrom et al., 2011) sampled pond and stream surface water in areas impacted by up to 12 years of field applications of biosolids contaminated by a fluoropolymer manufacturer, and the maximum and mean PFBS concentrations were 208 and 26.3 ng/L, respectively.

Another group of studies from the United States evaluated sites known to be contaminated from military installations with known or presumed AFFF use (Anderson et al., 2016; Post et al., 2013; Nakayama et al., 2007). The highest PFBS levels reported among these available studies were from Anderson et al. (2016) who performed a national study of 40 AFFF-impacted sites across 10 military installations and reported a maximum PFBS concentration of 317,000 ng/L. Lescord et al. (2015) examined PFAS levels in Meretta Lake, a Canadian lake contaminated with runoff from an airport and military base, which are likely sources of PFAS from AFFF use. The authors reported a 70-fold higher mean PFBS concentration for the contaminated lake versus a control lake. In addition to AFFF, Nakayama et al. (2007) identified industrial sources, including metal-plating facilities and textile and paper production, as contributing to the total PFAS contamination in North Carolina's Cape Fear River Basin. Nakayama et al. (2007) reported a PFBS DF of 17% and PFBS concentrations ranging from ND to 9.41 ng/L at these sites.

The EPA identified additional studies evaluating surface water samples from sites in Europe with known or suspected PFAS releases associated with AFFF use (Mussabek et al., 2019; Gobelius et al., 2018; Dauchy et al., 2017) or fluorochemical manufacturing (Bach et al., 2017; Boiteux et al., 2017; Gebbink et al., 2017; Valsecchi et al., 2015). PFBS levels were comparable at the AFFF-impacted sites (< 300 ng/L overall). Of the four study sites potentially contaminated based on proximity to fluorochemical manufacturing sites, two (from studies conducted in France) did not have PFBS detections (Bach et al., 2017; Boiteux et al., 2017). PFBS levels were low at most

sampling locations of the remaining two studies (up to approximately 30 ng/L) except for the site in River Brenta in Italy (maximum PFBS concentration of 1,666 ng/L) which is also impacted by nearby textile and tannery manufacturers (Valsecchi et al., 2015).

Eight studies in Europe evaluated areas close to urban areas, commercial activities, or industrial activities (e.g., textile manufacturing) (Lorenzo et al., 2015; Zhao et al., 2015; Boiteux et al., 2012; Eschauzier et al., 2012; Rostkowski et al., 2009) and/or wastewater effluent discharges (Wilkinson et al., 2017; Lorenzo et al., 2015; Labadie and Chevreuil, 2011; Möller et al., 2010). Among these sites, DFs varied (0 to 100%) and PFBS levels were < 250 ng/L overall.

Ten studies conducted in Europe evaluated sites with no known fluorochemical point source of contamination (Barreca et al., 2020; Pan et al., 2018; Loos et al., 2017; Shafique et al., 2017; Munoz et al., 2016; Eriksson et al., 2013; Wagner et al., 2013; Ahrens et al., 2009b; Ahrens et al., 2009a; Ericson et al., 2008b). Pan et al. (2018) analyzed surface water from sites in the United Kingdom (Thames River), Germany and the Netherlands (Rhine River), and Sweden (Mälaren Lake). None of the sites sampled were proximate to known sources of PFAS, but PFBS was detected in all three water bodies. Concentrations of PFBS ranged from 0.46 to 146 ng/L; the highest level (146 ng/L) was detected in the Rhine River and was more than 20 times greater than any maximum level found in the other water bodies. In the remaining nine studies, reported PFBS levels ranged from ND to 26 ng/L, except for one study in Italy that reported a PFBS DF of 39% and levels in the µg/L range at three out of 52 locations within the same river basin: Legnano (16,000 ng/L), Rho (15,000 ng/L), and Pero (3,400 ng/L) (Barreca et al., 2020).

Table B-2. Compilation of Studies Describing PFBS Occurrence in Surface Water

Study	Location	Site Details	PFBS Results
North America			
Yeung et al. (2017)	Canada (Ontario; Mimico Creek, Rouge River)	Two water samples at each of the sites	Mimico Creek: point = 0.020 ng/L Rouge River: DF 0%
Subedi et al. (2015)	United States (New York; Skaneateles Lake)	Lake water along the shoreline of residences that use an enhanced treatment unit for onsite wastewater treatment	DF[a] 4% (n=28); single detection value = 0.26 ng/L
Appleman et al. (2014)	United States (Wisconsin, Oklahoma, Alaska, California, Alabama, Colorado, Ohio, Nevada, Minnesota, New Jersey)	Raw surface waters from 11 sites, some impacted by upstream wastewater effluent discharge	DF[a] 64% (n=25); range = ND–47 ng/L (MRL = 0.3)
Veillette et al. (2012)	Canada (Ellesmere Island, Nunavut)	A lake near the northwest coast with no known sources of PFAS	DF[a] 100%, mean (range) = 0.016 (0.011–0.024) ng/L

Study	Location	Site Details	PFBS Results
Nakayama et al. (2010)	United States (Illinois, Iowa, Minnesota, Missouri, Wisconsin; Upper Mississippi River Basin and Missouri River Basin)	88 sampling sites from tributaries and streams	DF 43%, median (range) = 0.71 (ND–84.1) ng/L
Galloway et al. (2020)	United States (Ohio and West Virginia; Ohio River Basin)	Rivers and tributaries 58 km upstream to 130 km downwind of a fluoropolymer production facility, some sample locations potentially impacted by local landfills	DF NR, range[a] = ND–28.0 ng/L
Newsted et al. (2017)	United States (Minnesota; Upper Mississippi River Pool 2)	Upstream and downstream of 3M Cottage Grove facility outfall, which is a source of PFAS	Upstream: DF[a] 3%, point = 4.2 ng/L Downstream: DF[a] 67%, range = ND–336.0 ng/L
Procopio et al. (2017)	United States (New Jersey; Metedeconk River Watershed)	Downstream of suspected illicit discharge to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs	DF[a] 5%, range = ND–100 ng/L
Newton et al. (2017)	United States (Decatur, Alabama; Tennessee River)	6 sites upstream and 3 sites downstream of fluorochemical manufacturing facilities	Upstream: DF 0% Downstream: DF[a] 100%, mean[a] (range) = 69 (10–160) ng/L
Zhang et al. (2016)	United States (Rhode Island, New York Metropolitan Region)	Rivers and creeks, some sampling locations downstream from industrial activities, airport, textile mills, and WWTP. PFAS are used for water resistant coating in textiles.	DF[a] 85%, range = ND–6.181 ng/L

Study	Location	Site Details	PFBS Results
Lescord et al. (2015)	Canada (Resolute Bay, Nunavut)	One lake (Meretta) contaminated with runoff from an airport, which is a known source of PFAS; one control lake (9 Mile)	Meretta: DF NR, mean = 4.9 ng/L 9 Mile: DF NR, mean = 0.07 ng/L
Lasier et al. (2011)	United States (Georgia; Coosa River watershed)	Upstream (sites 1 and 2) and downstream (sites 3–8) of a land-application site where effluents from carpet manufacturers (suspected of producing wastewaters containing perfluorinated chemicals) are processed at a WWTP and the treated WWTP effluent is sprayed onto the site. Site 4 was downstream of a manufacturing facility for latex and polyurethane backing material.	Upstream Sites 1 and 2: DF 0% Downstream Site 3: DF NR, mean = 205 ng/L Site 4: DF NR, mean = 260 ng/L Site 5: DF NR, mean = 125 ng/L Site 6: DF NR, mean = 134 ng/L Site 7: DF NR, mean = 122 ng/L Site 8: DF NR, mean = 105 ng/L
Anderson et al. (2016)	United States (national)	Ten U.S. Air Force installations with historic AFFF release	DF 80.00%, median (range) = 106 (ND–317,000) ng/L
Post et al. (2013)	United States (New Jersey)	6 rivers and 6 reservoirs from public drinking water system intakes, some sites may include nearby small industrial park and civil-military airport	DF 17%, range = ND–6 ng/L
Nakayama et al. (2007)	United States (North Carolina; Cape Fear River Basin)	80 sampling sites in river basin; some sites near industrial areas and Fort Bragg and Pope Air Force Base with suspected use of AFFF at the Air Force Base	DF 62%, mean (range) = 2.58 (ND–9.41) ng/L

Study	Location	Site Details	PFBS Results
Lindstrom et al. (2011)	United States (Alabama)	32 surface water samples (ponds and streams) from areas with historical land application of fluorochemical industry-impacted biosolids	DF[a] 63%, range = ND–208 ng/L
Bradley et al. (2020)	United States (Lake Michigan)	Untreated Lake Michigan water from treatment plant intake (4 sites)	DF 29%, range = ND–0.5 ng/L
Europe			
Barreca et al. (2020)	Italy (Lombardia Region)	Rivers and streams with no known fluorochemical sources	DF[a] 39%, range = ND–16,000 ng/L
Loos et al. (2017)	Austria, Bulgaria, Croatia, Moldova, Romania, Serbia, Slovakia (Danube River and tributaries)	Some sampling locations downstream of major cities	DF 94%, mean (range) = 1.6 (ND–3.7) ng/L
Wilkinson et al. (2017)	England (Greater London and southern England; Hogsmill River, Chertsey Bourne River, Blackwater River)	50 m upstream and 250 m and 1,000 m downstream from WWTP effluent outfalls	Upstream: DF NR, mean = 20.4 ng/L Downstream 250 m: DF NR, mean = 40.3 ng/L Downstream 1,000 m: DF NR, mean = 41.1 ng/L
Shafique et al. (2017)	Germany (Leipzig, Pleiße-Elster River, Saale River, and Elbe River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	Pleiße-Elster: DF NR, mean = 1.2 ng/L Saale: DF NR, mean = 7.5 ng/L Elbe: DF NR, mean = 4.3 ng/L
Munoz et al. (2016)	France (Seine River)	Two sites downstream of Greater Paris and one site unaffected by the Greater Paris region	DF 70%, range = ND–3.1 ng/L

Study	Location	Site Details	PFBS Results
Lorenzo et al. (2015)	Spain (Guadalquivir River Basin, Ebro River Basin)	Guadalquivir sampling locations included downstream of WWTPs, near industrial areas, near a military camp, or through major cities; Ebro sampling locations included nearby ski resorts and downstream of WWTP and industrial areas	Guadalquivir: DF 8%, mean (range) = 10.1 (ND–228.3) ng/L Ebro: DF 0%
Zhao et al. (2015)	Germany (Elbe River and lower Weser River)	Some sampling sites near Hamburg city and industrial plants	Elbe: DF 100%, mean (range) = 7.4 (0.24–238) ng/L Weser: DF 100%, mean (range) = 1.41 (0.75–1.85) ng/L
Eriksson et al. (2013)	Denmark (Faroe Islands)	Lakes Leitisvatn, Havnardal, Kornvatn, and Á Mýranar with no known point sources of any fluorochemical facilities	Leitisvatn: DF 0% Havnardal Lake: DF 0% Kornvatn Lake: DF 0% Á Mýranar: DF 0%
Wagner et al. (2013)	Germany (Rhine River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean[b] (range[b]) = 18 (9–26) ng/L
Boiteux et al. (2012)	France (national)	Rivers; some locations may have upstream industrial sources	DF 1%, range = ND–5 ng/L
Eschauzier et al. (2012)	The Netherlands (Amsterdam; Lek Canal, tributary of Rhine River)	Downstream of an industrial point source in the German part of the Lower Rhine	DF[a] 100%, mean (range) = 35 (31–42) ng/L
Labadie and Chevreuil (2011)	France (Paris; River Seine)	Urban stretch of the River Seine during a flood cycle, sampling location under the influence of two urban WWTPs and two major combined sewer overflow outfalls	DF 100%, mean (range) = 1.3 (0.6–2.6) ng/L

Study	Location	Site Details	PFBS Results
Möller et al. (2010)	Germany (Rhine River watershed)	Upstream and downstream of Leverkusen, where effluent of a WWTP treating industrial wastewater was discharged; other major rivers and tributaries	Rhine upstream Leverkusen: DF 100%, mean (range) = 3.19 (0.59–6.58) ng/L Rhine downstream Leverkusen: DF 100%, mean (range) = 45.4 (15.0–118) ng/L River Ruhr: DF 100%, mean (range) = 7.08 (2.87–11.4) ng/L River Moehne: point = 31.1 ng/L Other tributaries: DF 100%, mean (range) = 2.84 (0.22–6.82) ng/L
Ahrens et al. (2009b)	Germany (Elbe River)	Sampling sites in Hamburg city (sites 16–18) and from Laurenburg to Hamburg (sites 19–24)	Hamburg: Dissolved: DF[a] 100%, mean (range) = 1.6 (1.1–2.5) ng/L Laurenburg to Hamburg: Dissolved: DF[a] 100%, mean (range) = 1.1 (0.53–1.5) ng/L
Ahrens et al. (2009a)	Germany (Elbe River)	Sampling locations 53 to 122 km (sites 1 to 9)[c] upstream of estuary mouth of Elbe River	DF NR; range of mean (for different locations) = 1.8–3.4 ng/L
Rostkowski et al. (2009)	Poland (national)	Rivers, lakes, and streams in northern and southern Poland, some southern locations near chemical industrial activities	North: DF[a] 60%, range = ND–10 ng/L South: DF[a] 73%, range = ND–16.0 ng/L
Ericson et al. (2008b)	Spain (Tarragona Province; Ebro River, Francolí River, Cortiella River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	Ebro site 1: DF 0% Ebro site 2: DF 0% Francolí: DF 0% Cortiella: DF 0%
Bach et al. (2017)	France (southern)	Upstream and downstream from discharge point that receives wastewater from an industrial site with two fluoropolymer manufacturing facilities	Upstream: DF 0% Downstream: DF 0%

Study	Location	Site Details	PFBS Results
Boiteux et al. (2017)	France (northern)	River samples from upstream and downstream of an industrial WWTP that processes raw sewage from fluorochemical manufacturing facility	Upstream: DF 0% Downstream: DF 0%
Gebbink et al. (2017)	The Netherlands (Dordrecht)	Upstream and downstream of Dordrecht fluorochemical production plant; two control sites	Control sites: DF[a] 100%, mean[a] (range) = 17 (12–22) ng/L Upstream: DF[a] 100%, mean[a] (range) = 19.7 (18–21) ng/L Downstream: DF[a] 100%, mean[a] (range) = 21 (16–27) ng/L
Valsecchi et al. (2015)	Italy (Po River Basin, Brenta River Basin, Adige River Basin, Tevere River Basin, and Arno River Basin)	Two river basins (Po and Brenta) which receive discharges from two chemical plants that produce fluorinated polymers and intermediates; three river basins (Adige, Tevere, Arno) with no known point sources of any fluorochemical facilities	Po: DF[a] 56%, range = ND–30.4 ng/L Brenta: DF[a] 100%, mean[a] (range) = 707 (23.1–1,666) ng/L Adige: DF[a] 20%, range = ND–4.3 ng/L Tevere: DF 0% Arno: DF[a] 58%, range = ND–31.4 ng/L
Mussabek et al. (2019)	Sweden (Luleå)	Samples from lake and pond near a firefighting training facility at the Norrbotten Air Force Wing known to use PFAS-containing AFFF	Lake: DF NR, mean = 200 ng/L Pond: DF NR, mean = 150 ng/L
Gobelius et al. (2018)	Sweden (national)	Sampling locations selected based on potential vicinity of PFAS hot spots and importance as a drinking water source area, some sites include firefighting training sites at airfields and military areas	DF[a] 29%, range = ND–299 ng/L

Study	Location	Site Details	PFBS Results
Dauchy et al. (2017)	France (unspecified)	Samples collected near 3 sites (B, C, D) impacted by the use of firefighting foams	Site B: DF 0% Site C: DF 0% Site D: DF[a] 30%, range = ND–138 ng/L
Multiple Continents			
Pan et al. (2018)	United States (Delaware River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 2.19 (0.52–4.20) ng/L
	United Kingdom (Thames River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 5.06 (3.26–6.75) ng/L
	Germany and the Netherlands (Rhine River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 21.9 (0.46–146) ng/L
	Sweden (Mälaren Lake)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 1.43 (0.75–1.92) ng/L

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; km = kilometer; m = meter; ND = not detected; ng/L = nanogram per liter; NR = not reported; PFAA = perfluoroalkyl acid; PFAS = per- and polyfluoroalkyl substances; WWTP = wastewater treatment plant; µg/L = microgram per liter.

[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

[b] For Wagner et al. (2013), PFBS concentrations were calculated using the fluorine concentrations reported in Table 4 from the study.

[c] Freshwater locations determined as sites with conductivity < 1.5 milliSiemens/cm.

B.2. RSC for PFBS, Literature Search and Screening Methodology

The EPA applies an RSC to the RfD when calculating an MCLG based on noncancer effects or for carcinogens that are known to act through a nonlinear mode of action to account for the fraction of an individual's total exposure allocated to drinking water (USEPA, 2000b). The EPA emphasizes that the purpose of the RSC is to ensure that the level of a chemical allowed by a criterion (e.g., the MCLG for drinking water) or multiple criteria, when combined with other identified sources of exposure (e.g., diet, ambient and indoor air) common to the population of concern, will not result in exposures that exceed the RfD. In other words, the RSC is the portion of total daily exposure equal to the RfD that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee tea or soup, as well as from transfer to dietary items prepared

with drinking water) relative to other exposure sources; the remainder of the exposure equal to the RfD is allocated to other potential exposure sources. For example, if for a particular chemical, drinking water were to represent half of total exposure and diet were to represent the other half, then the drinking water contribution (or RSC) would be 50%. The EPA considers any potentially significant exposure source when deriving the RSC.

The RSC is derived by applying the Exposure Decision Tree approach published in the EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000b). The Exposure Decision Tree approach allows flexibility in the RfD apportionment among sources of exposure and considers several characteristics of the contaminant of interest, including the adequacy of available exposure data, levels of the contaminant in relevant sources or media of exposure, and regulatory agendas (i.e., whether there are multiple health-based criteria or regulatory standards for the contaminant). The RSC is developed to reflect the exposure to the U.S. general population or a sensitive population within the U.S. general population and may be derived qualitatively or quantitatively, depending on the available data.

A quantitative RSC determination first requires "data for the chemical in question… representative of each source/medium of exposure and… relevant to the identified population(s)" (USEPA, 2000b). The term "data" in this context is defined as ambient sampling measurements in the media of exposure, not internal human biomonitoring metrics. More specifically, the data must adequately characterize exposure distributions including the central tendency and high-end exposure levels for each source and 95% confidence intervals for these terms (USEPA, 2000b). Frequently, an adequate level of detail is not available to support a quantitative RSC derivation. When adequate quantitative data are not available, the agency relies on the qualitative alternatives of the Exposure Decision Tree approach. A qualitatively-derived RSC is an estimate that incorporates data and policy considerations and thus, is sometimes referred to as a "default" RSC (USEPA, 2000b). Both the quantitative and qualitative approaches recommend a "ceiling" RSC of 80% and a "floor" RSC of 20% to account for uncertainties including unknown sources of exposure, changes to exposure characteristics over time, and data inadequacies (USEPA, 2000b).

In cases in which there is a lack of sufficient data describing environmental monitoring results and/or exposure intake, the Exposure Decision Tree approach results in a recommended RSC of 20%. In the case of MCLG development, this means that 20% of the exposure equal to the RfD is allocated to drinking water and the remaining 80% is reserved for other potential sources, such as diet, air, consumer products, etc. This 20% RSC value can be replaced if sufficient data are available to develop a scientifically defensible alternative value. If scientific data demonstrating that sources and routes of exposure other than drinking water are not anticipated for a specific pollutant, the RSC can be raised as high as 80% based on the available data, allowing the remaining 20% for other potential sources (USEPA, 2000b). Applying a lower RSC (e.g., 20%) is a more conservative approach to public health and results in a lower MCLG.

B.2.1. Literature Search and Screening

In 2020, the EPA conducted a literature search to evaluate evidence for pathways of human exposure to eight PFAS (PFOA, PFOS, PFBA, PFBS, PFDA, perfluorohexanoic acid (PFHxA), PFHxS, and PFNA) (Holder et al., 2023). This search was not date limited and spanned the

information collected across the Web of Science (WOS), PubMed, and ToxNet/ToxLine (now ProQuest) databases. The results of the PFBS literature search of publicly available sources are available through the EPA's Health & Environmental Resource Online website at https://hero.epa.gov/hero/index.cfm/project/page/project_id/2610.

The 654 literature search results for PFBS were imported into SWIFT-Review (Sciome, LLC, Research Triangle Park, NC) and filtered through the Evidence Stream tags to identify human studies and nonhuman (i.e., those not identified as human) studies. Studies identified as human studies were further categorized into seven major PFAS pathway categories (Cleaning Products, Clothing, Environmental Media, Food Packaging, Home Products/Articles/Materials, Personal Care Products, and Specialty Products) as well as an additional category for Human Exposure Measures. Nonhuman studies were grouped into the same seven major PFAS pathway categories, except that the Environmental Media category did not include soil, wastewater, or landfill. Only studies published between 2003 and 2020 were considered. Application of the SWIFT-Review tags identified 343 peer-reviewed papers matching these criteria for PFBS.

The 343 articles were screened to identify studies reporting measured occurrence of PFBS in human matrices and media commonly related to human exposure (human blood/serum/urine, drinking water, food, food contact materials, consumer products, indoor dust, indoor and ambient air, and soil). For this synthesis, additional screening was conducted to identify studies relevant to surface water (freshwater only) and groundwater using a keyword[7] search for water terms.

Following the Populations, Exposures, Comparators, and Outcomes (PECO) criteria outlined in Table B-3, the title and abstract of each study were independently screened for relevance by two screeners using *litstream*^{TM}. A study was included as relevant if it was unclear from the title and abstract whether it met the inclusion criteria. When two screeners did not agree whether a study should be included or excluded, a third reviewer was consulted to make a final decision. The title and abstract screening resulted in 191 unique studies being tagged as relevant (i.e., having data on occurrence of PFBS in exposure media of interest) that were further screened with full-text review using the same inclusion criteria. After additional review of the evidence collected by Holder et al. (2023), 87 studies originally identified for other PFAS also contained information relevant to PFBS. Based on full-text review, 147 studies were identified as having relevant, extractable data for PFBS from the United States, Canada, or Europe for environmental media, not including studies with only human biomonitoring data. Of these 147 studies, 130 were identified from Holder et al. (2023), where primary data were extracted into a comprehensive evidence database. Parameters of interest included sampling dates and locations, numbers of collection sites and participants, analytical methods, limits of detection and detection frequencies, and occurrence statistics. Seventeen of the 147 studies were identified in this synthesis as containing primary data on only surface water and/or groundwater.

The evidence database of Holder et al. (2023) additionally identified 18 studies for which the main article was not available for review. As part of this synthesis, 17 of the 18 studies could be retrieved. An additional three peer-reviewed references were identified through gray literature sources that were included to supplement the search results. The combined 20 studies underwent

[7] Keyword list: water, aquifer, direct water, freshwater, fresh water, groundwater, ground water, indirect water, lake, meltwater, melt water, natural water, overland flow, recreation water, recreational water, river, riverine water, riverwater, river water, springwater, spring water, stream, surface water, total water, water supply

full-text screening using the inclusion criteria in Table B-3. Based on full-text review, four studies were identified as relevant.

Table B-3. Populations, Exposures, Comparators, and Outcomes (PECO) Criteria

PECO Element	Inclusion Criteria
Population	Adults and/or children in the general population and populations in the vicinity of PFAS point sources from the United States, Canada, or Europe
Exposure	Primary data from peer-reviewed studies collected in any of the following media: ambient air, consumer products, drinking water, dust, food, food packaging, groundwater[a], human blood/serum/urine, indoor air, landfill, sediment, soil, surface water[a] (freshwater), wastewater/biosolids/sludge
Comparator	Not applicable
Outcome	Measured concentrations of PFBS (or measured emissions from food packaging and consumer products only)

Notes: PFBS = perfluorobutane sulfonic acid
[a] Surface water and groundwater were not included as relevant media in Holder et al. (2023). Studies were re-screened for these two media in this synthesis.

Using the screening results from the evidence database and this synthesis, a total of 151 peer-reviewed studies were identified as relevant.

B.2.2. Additional Screening

The EPA also searched the following publicly available gray literature sources for information related to relative exposure of PFBS for all potentially relevant routes of exposure (oral, inhalation, dermal) and exposure pathways relevant to humans:

- USEPA (2021d). *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)*;
- ATSDR's *Toxicological Profiles;*
- CDC's national reports on human exposures to environmental chemicals;
- The EPA's CompTox Chemicals Dashboard;
- The EPA's fish tissue studies;
- The EPA's Toxics Release Inventory;
- Relevant documents submitted under the Toxic Substances Control Act and relevant reports from the EPA's Office of Chemical Safety and Pollution Prevention;
- U.S. Food and Drug Administration's (FDA's) *Total Diet Studies* and other similar publications from FDA, U.S. Department of Agriculture, and Health Canada;
- National Oceanic and Atmospheric Administration's (NOAA's) National Centers for Coastal Ocean Science data collections;
- National Science Foundation direct and indirect food and/or certified drinking water additives;
- *Throwaway Packaging, Forever Chemicals: European wide survey of PFAS in disposable food packaging and tableware* (Straková et al., 2021);

- PubChem compound summaries;
- Relevant sources identified in the relative source contribution discussions (Section 5) of the EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA)/Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water*; and
- Additional sources, as needed.

The EPA has included available information from these gray literature sources for PFBS relevant to its uses, chemical and physical properties, and for occurrence in ambient or indoor air, foods (including fish and shellfish), soil, dust, and consumer products. The EPA has also included available information specific to PFBS below on any regulations that may restrict PFBS levels in media (e.g., water quality standards, air quality standards, food tolerance levels).

B.3. Summary of Potential Exposure Sources of PFBS Other than Water

B.3.1. Food

PFBS was included in a suite of individual PFAS selected as part of PFAS-targeted reexaminations of samples collected for the U.S. Food and Drug Administration's (FDA's) Total Diet Study (US FDA, 2022b, 2022a, 2021b, 2021a, 2020b, 2020a); however, it was not detected in any of the food samples tested. It should be noted that FDA indicated that the sample sizes were limited and that the results should not be used to draw definitive conclusions about PFAS levels or presence in the general food supply (US FDA, 2022c). PFBS was detected in cow milk samples collected from a farm with groundwater known to be contaminated with PFAS, as well as in produce (collard greens) collected from an area near a PFAS production plant, in FDA studies of the potential exposure of the U.S. population to PFAS (US FDA, 2021c, 2018). Maximum residue levels for PFBS were not found in the Global MRL Database (Bryant Christie Inc, 2022).

In addition to efforts by FDA, peer-reviewed studies conducted in North America, Europe, and across multiple continents analyzed PFBS in food items obtained from home, recreational, or commercial sources (see Table B-4). Food types evaluated include fruits and vegetables, grains, meat, seafood, dairy, and fats/other (e.g., eggs, spices, and oils), with seafood showing the highest levels of PFBS detected. PFBS was not detected in any of the eight studies that analyzed human milk for PFAS (not shown in Table B-4)—one in the United States (von Ehrenstein et al., 2009) and seven in Europe (Abdallah et al., 2020; Nyberg et al., 2018; Cariou et al., 2015; Lankova et al., 2013; Beser et al., 2011; Kärrman et al., 2010; Kärrman et al., 2007).

Of the studies conducted in North America, four U.S. studies (Scher et al., 2018; Byrne et al., 2017; Blaine et al., 2014; Schecter et al., 2010) found PFBS in at least one food item. Locations and food sources varied in these studies. In Schecter et al. (2010), PFBS was detected in cod samples but not in any of the other foods collected from Texas grocery stores. Scher et al. (2018) detected PFBS in plant parts (leaf and stem samples) analyzed from garden produce collected at homes in Minnesota within a GCA impacted by a former 3M PFAS production facility (PFBS concentrations ranged from ND to 0.065 nanograms per gram [ng/g]). The authors suggested that

the PFBS detections in plant parts were likely associated with PFAS present in irrigation water that had accumulated in produce. Blaine et al. (2014) found PFBS in radish, celery, tomato, and peas that were grown in soil amended with industrially impacted biosolids. They also found PFBS in these crops grown in soil that had received municipal biosolid applications over 20 years. In unamended control soil samples, PFBS was only detected in radish root with an average value of 22.36 ng/g (Blaine et al., 2014). In a similar study conducted by Blaine et al. (2013), PFBS was found in lettuce, tomato, and corn grown in industrially impacted biosolids-amended soils in greenhouses. Young et al. (2012) analyzed 61 raw and retail milk samples from 17 states for PFAS, but PFBS was not detected.

Based on the available data collected to date, seafood (including fish and shellfish) has been found to contain the highest concentrations of PFBS out of all food types examined. Burkhard (2021) identified 16 studies reporting BAFs for PFBS and calculated a median (standard deviation) bioaccumulation factor (BAF) in muscle tissue/fillet of 22.39 ± 6.92 L/kg wet weight (reported as a logBAF of 1.35 ± 0.84 L/kg). Several large-scale sampling efforts have been conducted by the EPA and other agencies to determine PFAS levels in fish. In the EPA's 2013–2014 National Rivers and Streams Assessment (NRSA), PFBS was detected at concentrations between the quantitation limit (1 ng/g) and the method detection limit (0.1 ng/g) at 0.571 ng/g in a largemouth bass fish fillet sample collected from Big Black River, Mississippi; 0.475 ng/g in a smallmouth bass fillet composite collected from Connecticut River, New Hampshire; and 0.148 ng/g in a walleye fillet composite collected from Chenango River, New York (USEPA, 2020a). Notably, PFBS was not detected in any fish species sampled in the 2008–2009 NRSA (Stahl et al., 2014). PFBS was also detected at a concentration of 0.36 ng/g in a smallmouth bass fillet composite collected from Lake Erie, New York in the EPA's 2015 Great Lakes Human Health Fish Fillet Tissue Study (USEPA, 2021g). PFBS has been detected in Irish pompano, silver porgy, grey snapper, and eastern oyster from the St. Lucie Estuary in the National Oceanic and Atmospheric Administration's (NOAA's) National Centers for Coastal Ocean Science, National Status and Trends Data (NOAA, 2022). PFBS was not a target chemical in the EPA's National Lake Fish Tissue Study (USEPA, 2009a).

Several peer-reviewed publications that examined PFBS concentrations in fish and shellfish are also available. As mentioned previously, Schecter et al. (2010) detected PFBS in cod samples. Mean PFBS levels in cod from this study (0.12 ng/g wet weight [ww]) were much lower than maximum levels detected in Alaska blackfish obtained from the Suqi River, Alaska in remote locations upstream and downstream of a former (unnamed) defense site (59.2 ng/g ww) (Byrne et al., 2017). In this study, blackfish were considered sentinel species but are not among the traditional fish consumed in the area. The authors noted that the presence of PFAS in fish from remote sites is suggestive of atmospheric deposition. In two additional studies from North America, PFBS was not detected in samples of farmed and wild-caught seafood (Chiesa et al., 2019; Young et al., 2013).

The European Food Safety Authority (EFSA) reported the presence of PFBS in various food and drink items, including fruits, vegetables, cheese, and bottled water (EFSA, 2012). For average adult consumers, the estimated exposure ranges for PFBS were 0.03−1.89 nanograms per kilogram body weight per day (ng/kg bw-day) (minimum) to 0.10−3.72 ng/kg bw-day (maximum) (EFSA, 2012). Of the studies conducted in Europe, 12 found PFBS in at least one food type (Table B-4). Eight of the 12 studies included food samples obtained solely from

markets (Scordo et al., 2020; Sznajder-Katarzyńska et al., 2019; Surma et al., 2017; D'Hollander et al., 2015; Pérez et al., 2014; Eschauzier et al., 2013; Hlouskova et al., 2013; Domingo et al., 2012). Across studies, PFBS detections were found in seafood; other animal products such as meat, dairy, and eggs; fruits and vegetables; tap water-based beverages such as coffee; sweets; and spices.

Papadopoulou et al. (2017) analyzed duplicate diet samples with PFBS detected in only one solid food sample (ND−0.001 ng/g; DF 2%; food category unspecified). Eriksson et al. (2013) evaluated foods that were farmed or freshly caught in the Faroe Islands, and only detected PFBS in cow milk (0.019 ng/g ww) and packaged dairy milk (0.017 ng/g ww) samples among the products analyzed. In eight of the European studies where PFBS was not detected, foods were primarily obtained from commercial sources, but wild-caught seafood was also included.

Two of the 12 European studies examined both market-bought and fresh-caught fish, and PFBS was detected in seafood from both sources (Vassiliadou et al., 2015; Yamada et al., 2014). Yamada et al. (2014) found higher PFBS in fresh-caught river fish samples (0.16 ng/g ww maximum) versus fresh or frozen market samples (0.03 ng/g ww maximum) in France. Vassiliadou et al. (2015) detected PFBS in raw shrimp (from Greek markets) but did not detect PFBS in either fried shrimp, raw hake (from Greek fishing sites), or fried hake.

In summary, in Europe and North America, PFBS has been detected in multiple food types, including fruits, vegetables, meats, seafoods, and other fats. Several large-scale fish tissue sampling efforts conducted by the EPA and others indicate that fish consumption may be an important PFBS exposure source. Future large-scale sampling efforts by FDA and others may help to similarly elucidate PFBS concentrations in other food types. Although several U.S. studies have evaluated PFBS in meats, fats/oils, fruits, vegetables, and other non-seafood food types, many of these sampling efforts were localized to specific cities or markets and/or used relatively small sample sizes. Broader-scale sampling efforts will be helpful in determining the general levels of PFBS contamination in these food types, as well as the impact of known PFAS contamination sources on PFBS concentrations in foods.

Table B-4. Compilation of Studies Describing PFBS Occurrence in Food

Study	Location and Source	Food Types	Results
North America			
Schecter et al. (2010)	United States (Texas) Grocery stores	Dairy, fruits and vegetables, grain, meat, seafood, fats/other	**Cod: DF NR, mean = 0.12 ng/g ww** ND in salmon, canned sardines, canned tuna, fresh catfish fillet, frozen fish sticks, tilapia, cheeses (American, mozzarella, Colby, cheddar, Swiss, provolone, and Monterey jack), butter, cream cheese, frozen yogurt, ice cream, whole milk, whole milk yogurt, potatoes, apples, cereals, bacon, canned chili, ham, hamburger, roast beef, sausages, sliced chicken breast, sliced turkey, canola oil, margarine, olive oil, peanut butter, eggs
Byrne et al. (2017)	United States (Alaska) Upstream/downstream of former defense site (Suqi River)	Seafood	**Blackfish: DF 48%, range = ND–59.2 ng/g ww** Highest concentration was upstream
Scher et al. (2018)	United States (Minnesota) Home gardens Near former 3M PFAS production facility, homes within and outside a GCA	Fruits and vegetables	Within GCA: **Leaf: DF 6%, max = 0.061 ng/g** **Stem: DF 4%, max = 0.065 ng/g** ND in floret, fruit, root, seed Outside GCA: ND
Blaine et al. (2014)	United States (Midwestern) Greenhouse study, unamended controls	Fruits and vegetables	**Radish root: DF NR, mean = 22.36 ng/g** ND in celery shoot, pea fruit
Blaine et al. (2013)	United States (Midwestern) Greenhouse and field studies, unamended controls	Fruits and vegetables, grain	ND in corn, lettuce, tomato in unamended soil.
Young et al. (2013)	United States (Maryland, Mississippi, Tennessee, Florida, New York, Texas, Washington, D.C.) Retail markets	Seafood	ND in crab, shrimp, striped bass, farm raised catfish, farm raised salmon
Young et al. (2012)	United States (17 states) Retail markets	Dairy	ND in retail cow's milk

Study	Location and Source	Food Types	Results
Europe			
Domingo et al. (2012)	Spain (Catalonia) Local markets, small stores, supermarkets, big grocery stores	12 food categories	**Vegetables: DF NR, mean = 0.013 ng/g fw** **Fish and seafood: DF NR, mean = 0.054 ng/g fw** ND in meat and meat products, tubers, fruits, eggs, milk, dairy products, cereals, pulses, industrial bakery, oils
Pérez et al. (2014)	Serbia (Belgrade and Novi Sad), Spain (Barcelona, Girona, and Madrid) Various supermarkets and retail stores	8 food categories	Categories included cereals, pulses and starchy roots, tree-nuts, oil crops and vegetable oils, vegetables and fruits, meat and meat products, milk, animal fats, dairy products, and eggs, fish and seafood, and others such as candies or coffee **Spain: DF 3.2%, range = ND–13 ng/g (primarily fish, oils)** **Serbia: DF 5.2%, range = ND–0.460 ng/g (primarily meat and meat products, cereals)**
D'Hollander et al. (2015)	Belgium, Czech Republic, Italy, Norway PERFOOD study; items from 3 national retail stores of different brands and countries of origin	Fruit, cereals, sweets, salt	**Sweets: DF[a] 25%, range = ND–0.0016 ng/g** **Fruit: DF[a] 19%, range = ND–0.067 ng/g** ND in cereals, salt
Hlouskova et al. (2013)	Belgium, Czech Republic, Italy, Norway Several national supermarkets	Pooled milk/dairy products, meat, fish, hen eggs	**DF 5%, mean (range) = 0.00975 (0.006–0.012) ng/g**
Eriksson et al. (2013)	Denmark Farm, dairy farm, fish from Faroe Shelf area	Dairy, fruits and vegetables, seafood	**Milk:** **Farmer (Havnardal): point = 0.019 ng/g ww** **Diary (Faroe Island): point = 0.017 ng/g ww;** ND or NQ in 4 samples ND in yogurt, creme fraiche, potatoes, farmed salmon, wild-caught cod, wild-caught saithe

Study	Location and Source	Food Types	Results
Sznajder-Katarzyńska et al. (2019)	Poland Markets	Dairy	**All dairy: sum PFBS = 0.04 ng/g** **Butter: range = 0.01–0.02 ng/g** ND in camembert-type cheese, cottage cheese, milk, natural yogurt, sour cream, kefir (bonny clabber)
Yamada et al. (2014)	France Freshwater fish from 6 major French rivers; fresh and frozen fish from markets	Seafood	**Freshwater fish: DF NR, range = 0.06–0.16 ng/g ww** **Fresh or frozen fish: DF NR, range = 0.02–0.03 ng/g ww**
Vassiliadou et al. (2015)	Greece Local fish markets, mariculture farm, fishing sites	Seafood	**Hake: raw mean = 0.45 ng/g ww, fried mean = 0.83 ng/g ww** **Shrimp: raw mean = 1.37 ng/g ww** ND in raw, fried, and grilled anchovy, bogue, picarel, sand smelt, sardine, squid, striped mullet, raw and fried mussel, fried shrimp, and grilled hake
Eschauzier et al. (2013)	The Netherlands (Amsterdam) Cafés, universities, supermarkets	Fats/other	**Brewed coffee (manual): mean (range) = 1.6 (1.3–2.0) ng/L** **Brewed coffee (machine): mean (range) = 2.9 (ND–9.8) ng/L** **Cola: mean (range) = 7.9 (ND–12) ng/L**
Surma et al. (2017)	Spain, Slovakia Source NR	Fats/other	**Spices: ND–1.01 ng/g** Spain: Detected in anise, star anise, fennel, coriander, cinnamon, peppermint, parsley, thyme, laurel, cumin, and oregano ND in white pepper, cardamon, clove, nutmeg, allspice, vanilla, ginger, garlic, black paper, and hot pepper (mild and hot) Slovakia: ND in anise, star anise, white pepper, fennel, cardamom, clove, coriander, nutmeg, allspice, cinnamon, vanilla, and ginger

Study	Location and Source	Food Types	Results
Papadopoulou et al. (2017)	Norway A-TEAM project: food and drinks collected by participants as duplicate diet samples	Solid foods (11 food categories), liquid foods (5 drinks)	**Solid foods (unspecific food category): DF 2%, range = ND–0.001 ng/g** ND in liquid foods (coffee, tea and cocoa, milk, water, alcoholic beverages and soft drinks)
Scordo et al. (2020)	Italy Supermarkets	Fruits	**Olives: DF[a] 100%, mean[a] (range) = 0.294 (0.185–0.403) ng/g dw** ND in strawberries
Ericson et al. (2008a)	Spain Local markets, large supermarkets, grocery stores	18 food categories	ND in all categories: veal, pork, chicken, lamb, white fish, seafood, tinned fish, blue fish, whole milk, semi-skimmed milk, dairy products, vegetables, pulses, cereals, fruits, oil, margarine, and eggs
Noorlander et al. (2011)	The Netherlands Several Dutch retail store chains with nationwide coverage	15 food categories	ND in all categories: flour, fatty fish, lean fish, pork, eggs, crustaceans, bakery products, vegetables/fruit, cheese, beef, chicken/poultry, butter, milk, vegetable oil, and industrial oil
Jogsten et al. (2009)	Spain (Catalonia) Local markets, large supermarkets, grocery stores	Fruits and vegetables, meat, seafood, fats/other	ND in lettuce, raw, cooked, and fried meat (veal, pork, and chicken), fried chicken nuggets, black pudding, lamb liver, pate of pork liver, foie gras of duck, “Frankfurt” sausages, home-made marinated salmon, and common salt
Sznajder-Katarzyńska et al. (2018)	Poland Markets	Fruits and vegetables	ND in apples, bananas, cherries, lemons, oranges, strawberries, beetroots, carrots, tomatoes, potatoes, and white cabbage
Falandysz et al. (2006)	Poland Gulf of Gdañsk, Baltic Sea south coast	Meat, seafood	ND in eider duck, cod
Barbosa et al. (2018)	Belgium, France, the Netherlands, Portugal Various markets	Seafood	ND in raw and steamed fish (*P. platessa, M. australis, M. capenis, K. pelamis*, and *M. edulis*)

Study	Location and Source	Food Types	Results
Hölzer et al. (2011)	Germany Fish from Lake Möhne and river Möhne, contaminated with PFCs from use of polluted soil conditioner on agricultural lands; retail trade, wholesale trade, supermarkets, and producers	Seafood	Lake Möhne /River Möhne: ND in cisco, eel, perch, pike, and roach Trade/markets: ND in eel, pike/perch, and trout
Jörundsdóttir et al. (2014)	Iceland Collected during biannual scientific surveys, commercially produced	Seafood	ND in anglerfish, Atlantic cod, blue whiting, lemon sole, ling, lumpfish, plaice, and pollock
Rivière et al. (2019)	France Based on results of national consumption survey	Seafood, fats/other	ND in infant food, vegetables, non-alcoholic beverages, dairy-based desserts, milk, mixed dishes, fish, ultra-fresh dairy products, meat, poultry and game
Lankova et al. (2013)	Czech Republic Retail market	Fats/other	ND in infant formula
Zafeiraki et al. (2016a)	Greece, the Netherlands Home and commercially produced	Fats/other	ND in chicken eggs
Gebbink et al. (2015)	Sweden Major grocery chain stores, market basket samples	12 food categories	ND in all categories: dairy products, meat products, fats, pastries, fish products, egg, cereal products, vegetables, fruit, potatoes, sugar and sweets, soft drinks
Herzke et al. (2013)	Belgium, Czech Republic, Italy, Norway PERFOOD study: items from 3 national retail stores of different brands per location	Vegetables	ND for all vegetables
Zafeiraki et al. (2016b)	The Netherlands Local markets and slaughterhouses	Meat	ND for horse, sheep, cow, pig, and chicken liver

Study	Location and Source	Food Types	Results
Multiple Continents			
Chiesa et al. (2019)	United States (Pacific Ocean) Wholesale fish market	Seafood	ND in wild-caught salmon
	Canada Wholesale fish market	Seafood	ND in wild-caught salmon
	Norway Wholesale fish market	Seafood	ND in farm salmon
	Scotland Wholesale fish market	Seafood	ND in wild-caught and farm salmon

Notes: DF = detection frequency; dw = dry weight; fw = fresh weight; GCA = groundwater contamination area; ND = not detected; ng/g = nanogram per gram; ng/L = nanogram per liter; NR = not reported; PFAS = per- and polyfluoroalkyl substances; NQ = not quantified; µg/L = microgram per liter; ww = wet weight.
Bold indicates detected levels of PFBS in food.
[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

B.3.2. Food Contact Materials

PFBS is not authorized for use in food packaging in the United States; however, PFBS has been detected in food packaging materials in the few available studies that investigate this potential route of exposure (USEPA, 2021d; ATSDR, 2021). In one report from the United States, PFBS was detected in fast-food packaging (7/20 samples) although the concentrations detected were not reported (Schaider et al., 2017). Additionally, in an analysis performed at the Department of Food Analysis and Nutrition of the University of Chemistry and Technology in Prague, Czech Republic, PFBS was not detected in 42 samples of disposable food packaging and tableware purchased from six different European countries between May and December 2020 (LOQ = 1.7 mg/kg) (Straková et al., 2021).

The EPA identified five peer-reviewed studies in Europe (conducted in Poland, Norway, Greece, Czech Republic, and Germany) analyzed the occurrence of PFBS in food packaging or food contact materials (FCMs), such as baking papers and fast-food boxes and wrappers. Surma et al. (2015) measured levels of 10 perfluorinated compounds in three different brands of common FCMs commercially available in Poland, including wrapping papers (n = 3), breakfast bags (n = 3), baking papers (n = 3), and roasting bags (n = 3). PFBS was detected in one brand of baking paper at 0.02 picograms per square centimeter (pg/cm^2), but PFBS was not detected at or below the LOQ in all other FCMs. Vestergren et al. (2015) analyzed paper plates (n = 2), paper cups (n = 1), baking covers (n = 1), and baking molds (n = 1) purchased from retail stores in Tromsø and Trondheim, Norway. PFBS was detected in one paper plate at 6.9 pg/cm^2.

The remaining three studies did not detect PFBS in FCMs. Zafeiraki et al. (2014) analyzed FCMs made of paper, paperboard, or aluminum foil collected from a Greek market. PFBS was

not detected in any of the samples of beverage cups (n = 8), ice cream cups (n = 1), fast-food paper boxes (n = 8), fast-food wrappers (n = 6), paper materials for baking (n = 2), microwave bags (n = 3), and aluminum foil bags/wrappers (n = 14). The study concluded that the use of perfluorinated compound alternatives such as fluorophosphates and fluorinated polyethers in the local manufacturing process potentially explains the low levels of other PFAS (i.e., perfluorobutanoic acid [PFBA], perfluorohexanoic acid [PFHxA], perfluoroheptanoic acid [PFHpA], perfluorononanoic acid [PFNA], perfluorodecanoic acid [PFDA], and perfluorododecanoic acid [PFDoDA]) detected in the sampled FCMs. Vavrouš et al. (2016) analyzed 15 samples of paper FCMs acquired from a market in the Czech Republic. FCMs included paper packages of wheat flour (n = 2), paper bags for bakery products (n = 2), sheets of paper for food packaging in food stores (n = 2), cardboard boxes for packaging of various foodstuffs (n = 3), coated bakery release papers for oven baking at temperatures up to 220°C (n = 3), and paper filters for coffee preparation (n = 3). PFBS was not detected in any samples. Kotthoff et al. (2015) analyzed 82 samples for perfluoroalkane sulfonate (PFSA) and perfluoroalkyl carboxylic acid (PFCA) compounds in 10 consumer products including individual paper-based FCMs (n = 33) from local retailers in Germany in 2010. PFBS was not detected in paper-based FCMs.

Overall, the single available studies conducted in the United States indicate that PFBS may be present in food packaging materials; however, further research is needed to understand which packaging materials generally contain PFBS at the highest concentrations and with the greatest frequency. There are also uncertainties related to data gaps on topics that may influence whether food packaging is a significant PFBS exposure source in humans, including differences in transfer efficiency from different packaging types directly to humans or indirectly through foodstuffs.

B.3.3. Consumer Products

Several studies examined a range of consumer products and found multiple PFAS, including PFBS, at various levels (van der Veen et al., 2020; Zheng et al., 2020; Schultes et al., 2018; Bečanová et al., 2016; Favreau et al., 2016; Gremmel et al., 2016; Kotthoff et al., 2015; Vestergren et al., 2015; Liu et al., 2014). Two of the studies collected consumer products in the United States, five purchased consumer products in Europe, and two studies did not report the purchase location(s) of the consumer products that were tested.

Zheng et al. (2020) determined the occurrence of ionic and neutral PFAS in items collected from childcare environments in the United States. Nap mats (n = 26; 20 polyurethane foam, 6 vinyl cover samples) were collected from seven Seattle childcare centers. PFBS was detected in 5% of nap mat samples at a maximum concentration of 0.04 ng/g. Liu et al. (2014) analyzed the occurrence of PFAS in commonly used consumer products (carpet, commercial carpet-care liquids, household carpet/fabric-care liquids, treated apparel, treated home textiles, treated non-woven medical garments, floor waxes, membranes for apparel, and thread-sealant tapes) purchased from retail outlets in the United States. PFBS was detected in 100% of commercial carpet/fabric-care liquids samples (n = 2) at concentrations of 45.8 and 89.6 ng/g, in 75% of household carpet/fabric-care liquids and foams samples (n = 4) at concentrations up to 911 ng/g, in one treated apparel samples (n = 2) at a concentration of 2 ng/g, in the single treated floor wax and stone/wood sealant sample (143 ng/g, n = 2), and in the single apparel membrane sample

(30.7 ng/g, n = 2). PFBS was not detected in treated home textile and upholstery (n = 2) or thread-sealant tapes and pastes (n = 2).

van der Veen et al. (2020) examined the effects of weathering on PFAS content in durable water-repellent clothing collected from six suppliers in Sweden (1 pair of outdoor trousers, 7 jackets, 4 fabrics for outdoor clothes, 1 pair of outdoor overalls). Two pieces of each of the 13 fabrics were cut. One piece of each fabric was exposed to elevated ultraviolet radiation, humidity, and temperature in an aging device for 300 hours (assumed lifespan of outdoor clothing); the other was not aged. Both pieces of each fabric were analyzed for ionic PFAS (including PFBS) and volatile PFAS. In general, aging of outdoor clothing resulted in increased perfluoroalkylated acid (PFAA) levels of 5-fold or more. For 8 of 13 fabrics, PFBS was not detected before or after aging. For three fabrics, PFBS was detected before and after aging, increasing approximately 3- to 14-fold in the aged fabric (i.e., from 43 to 140 micrograms per square meter [µg/m^2], 45 to 350 µg/m^2, and 9.6 to 130 µg/m^2 respectively for the 3 fabrics). For the remaining two fabrics, PFBS was not detected prior to aging but was detected afterward at concentrations of 0.57 and 1.7 µg/m^2, respectively. The authors noted that possible explanations for this could be weathering of precursor compounds (e.g., fluorotelomer alcohols) to PFAAs such as PFBS or increased extractability due to weathering.

Kotthoff et al. (2015) analyzed 82 samples for PFSA and PFCA compounds in outdoor textiles (n = 3), gloves (n = 3), carpets (n = 6), cleaning agents (n = 6), impregnating sprays (n = 3), leather (n = 13), wood glue (n = 1), ski wax (n = 13), and awning cloth (n = 1). Individual samples were bought from local retailers or collected by coworkers of the involved institutes or local clubs in Germany. The age of the samples ranged from a few years to decades. PFBS was detected in outdoor textiles (level not provided), carpet samples (up to 26.8 µg/m^2), ski wax samples (up to 3.1 micrograms per kilogram [µg/kg]), leather samples (up to 120 µg/kg), and gloves (up to 2 µg/kg). Favreau et al. (2016) analyzed the occurrence of 41 PFAS in a wide variety of liquid products (n = 132 consumer products, 194 total products), including impregnating agents, lubricants, cleansers, polishes, AFFFs, and other industrial products purchased from stores and supermarkets in Switzerland. PFBS was not detected in impregnation products (n = 60), cleansers (n = 24), or polishes (n = 18). PFBS was detected in 13% of a miscellaneous category of products (n = 23) that included foam-suppressing agents for the chromium industry, paints, ski wax, inks, and tanning substances, with mean and maximum concentrations of 998 and 2,992 parts per million (ppm), respectively (median = ND).

The remaining two European studies from Norway (Vestergren et al., 2015) and Sweden (Schultes et al., 2018) did not detect PFBS in the consumer products analyzed. Vestergren et al. (2015) analyzed furniture textile, carpet, and clothing samples (n = 40) purchased from retail stores in Tromsø and Trondheim, Norway, while Schultes et al. (2018) determined levels of 39 PFAS in 31 cosmetic products collected in Sweden. Both studies found measurable concentrations of at least one PFAS; however, PFBS was not detected in any of the samples.

Of the two studies for which purchase location(s) were not specified, Gremmel et al. (2016) determined levels of 23 PFAS in 16 new outdoor jackets since it has been shown that outdoor jackets emit PFAS to the air as well as into water during washing. The jackets were selected based on factors such as fabric and origin of production (primarily Asia, with some origins not specified). PFBS (concentration of 0.51 µg/m^2) was only detected in one large hardshell jacket made of 100% polyester that was polyurethane-coated and finished with Teflon® (production

origin unknown). Bečanová et al. (2016) analyzed 126 samples of (1) household equipment (textiles, floor coverings, electrical and electronic equipment (EEE), and plastics); (2) building materials (oriented strand board, other composite wood and wood, insulation materials, mounting and sealant foam, facade materials, polystyrene, air conditioner components); (3) car interior materials; and (4) wastes of electrical and electronic equipment (WEEE) for 15 target PFAS, including PFBS. The condition (new versus used) and production year of the samples varied; the production year ranged from 1981 to 2010. The origin(s) of production were not specified. PFBS was detected in 31/55, 9/54, 7/10, and 6/7 household equipment, building materials, car interior, and WEEE samples, respectively. The highest level was 11.4 µg/kg found in a used 1999 screen associated with WEEE.

In summary, in the few studies available from North America and Europe, PFBS was detected in a wide range of consumer products including clothing, household textiles and products, children's products, and commercial/industrial products. However, there is some uncertainty in these results as the number and types of products tested in each study were often limited in terms of sample size. While there is evidence indicating PFBS exposure may occur through the use of or contact with consumer products, more research is needed to understand the DF and concentrations of PFBS that occur in specific products, as well as how the concentrations of PFBS change in these products with age or weathering.

B.3.4. Indoor Dust

Dust ingestion may be an important exposure source of PFAS including PFBS (ATSDR, 2021), though it should be noted that dust exposure may also occur via inhalation and dermal routes. The EPA identified several studies conducted in the United States, Canada, various countries in Europe, and across multiple continents that analyzed PFBS in dust of indoor environments (primarily in homes, but also schools, childcare facilities, offices, and vehicles; see Table B-5). Most of the studies sampled dust from areas not associated with any known PFAS activity or release. PFBS concentrations in dust measured in these studies ranged from ND to 170 ng/g with three exceptions: two studies (Kato et al., 2009; Strynar and Lindstrom, 2008) reported maximum PFBS concentrations > 1,000 ng/g in dust from homes and daycare centers, and a third study (Huber et al., 2011) reported a PFBS concentration of 1,089 ng/g in dust from a storage room that had been used to store "highly contaminated PFC [polyfluorinated compounds] samples and technical mixtures for several years."

Of the two available studies that measured PFBS in dust from vehicles, one (in the United States) detected no PFBS (Fraser et al., 2013) and the other (in Ireland) reported a DF of 75% and PFBS concentrations ranging from ND to 170 ng/g (Harrad et al., 2019).

One U.S. study, Scher et al. (2019) evaluated indoor dust from 19 homes in Minnesota within a GCA impacted by the former 3M PFAS production facility. House dust samples were collected from both interior living rooms and entryways to the yard. The DFs for PFBS were 16% and 11% for living rooms and entryways, respectively, and a maximum PFBS concentration of 58 ng/g was reported for both locations.

Haug et al. (2011) indicated that house dust concentrations are likely influenced by a number of factors related to the building (e.g., size, age, floor space, flooring type, ventilation); the residents or occupants (e.g., number of people, housekeeping practices, consumer habits such as

buying new or used products); and the presence and use of certain products (e.g., carpeting, carpet or furniture stain-protective coatings, waterproofing sprays, cleaning agents, kitchen utensils, clothing, shoes, cosmetics, insecticides, electronic devices). In addition, the extent and use of the products affects the distribution patterns of PFAS in dust of these buildings.

At this time, there is uncertainty regarding the extent of human exposure to PFBS through indoor dust compared with other exposure pathways.

Table B-5. Compilation of Studies Describing PFBS Occurrence in Indoor Dust

Study	Location	Site Details	Results
North America			
Zheng et al. (2020)	United States (Seattle, Washington and West Lafayette, Indiana)	Childcare facilities (20 samples from 7 facilities in Seattle and 1 in West Lafayette)	DF 90%, mean (range) = 0.34 (ND–0.86) ng/g
Byrne et al. (2017)	United States (St. Lawrence Island, Alaska)	Homes (49)	DF 16%, median = ND; 95th percentile = 1.76 ng/g
Fraser et al. (2013)	United States (Boston, Massachusetts)	Homes (30); offices (31); vehicles (13)	Homes: DF 3% (single detection), range = ND–4.98 ng/g Offices: DF 10%, range = ND–12.0 ng/g Vehicles: DF 0%
Knobeloch et al. (2012)	United States (Great Lakes Basin, Wisconsin)	Homes (39)	DF 59%, median (range) = 1.8 (ND–31) ng/g
Strynar and Lindstrom (2008)	United States (Cities in North Carolina and Ohio)	Homes (102) and daycare centers (10); samples had been collected in 2000–2001 during EPA's Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) study	DF 33%, mean (range) = 41.7 (ND–1,150) ng/g

Study	Location	Site Details	Results
Scher et al. (2019)	United States (Twin Cities metropolitan region, Minnesota)	Near former 3M PFAS production facility; 19 homes within the GCA	Entryway: DF 11%, median (range) = ND (ND–58 ng/g) Living room: DF 16%, median (range) = ND (ND–58 ng/g)
Kubwabo et al. (2005)	Canada (Ottawa)	Homes (67)	DF 0%
Europe			
de la Torre et al. (2019)	Spain (unspecified), Belgium (unspecified), Italy (unspecified)	Homes (65)	Spain: DF 52%, median (range) = 0.70 (ND–12.0) ng/g Belgium: DF 27%, median (range) = 0.40 (ND–56.7) ng/g Italy: DF 18%, median (range) = 0.40 (ND–11.6) ng/g
Harrad et al. (2019)	Ireland (Dublin, Galway, and Limerick counties)	Homes (32); offices (33); cars (31); classrooms (32)	Homes: DF 81%, mean (range) = 17 (ND–110) ng/g Offices: DF 88%, mean (range) = 19 (ND–98) ng/g Cars: DF 75%, mean (range) = 12 (ND–170) ng/g Classrooms: DF 97%, mean (range) = 17 (ND–49) ng/g
Giovanoulis et al. (2019)	Sweden (Stockholm)	Preschools (20)	DF 0%
Winkens et al. (2018)	Finland (Kuopio)	Homes (63 children's bedrooms)	DF 12.7%, median (range) = ND (ND–13.5) ng/g
Padilla-Sánchez and Haug (2016)	Norway (Oslo)	Homes (7)	DF 14% (single detection), range = ND–3 ng/g
Jogsten et al. (2012)	Spain (Catalonia)	Homes (10)	DF 60%, range = ND–6.5 ng/g
Haug et al. (2011)	Norway (Oslo)	Homes (41)	DF 22%, mean (range) = 1.3 (0.17–9.8) ng/g

Study	Location	Site Details	Results
Huber et al. (2011)	Norway (Tromsø)	Homes (7; carpet, bedroom, sofa); one office; one storage room that had been used for storage of "highly contaminated PFC [polyfluorinated compounds] samples and technical mixtures for several years"	All homes: DF NR, median = 1.1 ng/g Living room: DF[a] 57%, range = ND–10.6 ng/g Carpet, bedroom, sofa: DF 0% Office: point = 3.8 ng/g Storage room: point = 1,089 ng/g
D'Hollander et al. (2010)	Belgium (Flanders)	Homes (45); offices (10)	Homes: DF 47%, median = 0 ng/g dw Offices: DF NR, median = 0.2 ng/g dw
Multiple Continents			
Kato et al. (2009)	United States (Atlanta, Georgia), Germany (unspecified), United Kingdom (unspecified), Australia (unspecified)	Homes (39)	DF 92.3%, median (range) = 359 (ND–7,718) ng/g
Karásková et al. (2016)	United States (unspecified)	Homes (14)	DF 60%, mean (range) = 1.4 (ND–2.6) ng/g
	Canada (unspecified)	Homes (15)	DF 55%, mean (range) = 1.6 (ND–5.8) ng/g
	Czech Republic (unspecified)	Homes (12)	DF 37.5%, mean (range) = 3.6 (ND–14.4) ng/g

Notes: DF = detection frequency; GCA = groundwater contamination area; ND = not detected; ng/g = nanogram per gram; NR = not reported; dw = dry weight

[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

B.3.5. Air

PFAS have been released to air from WWTPs, waste incinerators, and landfills (USEPA, 2016a). ATSDR (2021) noted that PFAS have been detected in particulates and in the vapor phase in air and can be transported long distances via the atmosphere; they have been detected at low concentrations in areas as remote as the Arctic and ocean waters. However, EPA's Toxic Release Inventory did not report release data for PFBS in 2020 (USEPA, 2022c). In addition, PFBS is not listed as a hazardous air pollutant (USEPA, 2022d).

B.3.5.1. Indoor Air

No studies from the U.S. reporting levels of PFBS in indoor air were identified from the peerr-reviewed or gray literature. However, the EPA identified studies from Europe that are summarized below. These three studies were conducted in Norway (Barber et al., 2007), Spain (Jogsten et al., 2012), and Ireland (Harrad et al., 2019).

In Norway, neutral and ionic PFAS were analyzed in four indoor air samples collected from homes in Tromsø (Barber et al., 2007). PFBS levels were below the limit of quantitation. The authors noted that measurable amounts of other ionic PFAS were found in indoor air samples, but levels were not significantly elevated above levels in outdoor air. In Spain, Jogsten et al. (2012) collected indoor air samples (n = 10) from selected homes in Catalonia and evaluated levels of 27 perfluorinated chemicals (PFCs). PFBS was not detected.

In Ireland, Harrad et al. (2019) measured eight target PFAS in air from cars (n = 31), home living rooms (n = 34), offices (n = 34), and school classrooms (n = 28). PFBS was detected in all four indoor microenvironments, at DFs of 53%, 90%, 41%, and 54% in samples from homes, cars, offices, and classrooms, respectively. The mean (maximum) concentrations were 22 (270) picograms per cubic meter (pg/m^3) in homes, 54 (264) pg/m^3 in cars, 37 (313) pg/m^3 in offices, and 36 (202) pg/m^3 in classrooms.

There is some evidence from European studies indicating PFBS exposure via indoor air. However, further research is needed to understand the DF and concentrations of PFBS that occur in indoor environments in the United States.

B.3.5.2. Ambient Air

Similar to studies on indoor PFBS air concentrations, no studies from the U.S. reporting levels of PFBS in ambient air were identified from the peer-reviewed or gray literature. Four studies conducted across Europe (Harrad et al., 2020; Jogsten et al., 2012; Beser et al., 2011; Barber et al., 2007) and one study conducted in Canada (Ahrens et al., 2011) analyzed ambient air samples for PFBS. Two of the studies (Harrad et al., 2020; Barber et al., 2007) found detectable levels of PFBS in outdoor air. Barber et al. (2007) collected air samples from four field sites in Europe (one semirural site [Hazelrigg] and one urban site [Manchester] in the United Kingdom, one rural site from Ireland, and one rural site from Norway) for analysis of neutral and ionic PFAS. The study authors did not indicate whether any of the sites had a history of local PFAS-related activities (e.g., AFFF usage, PFAS manufacturing or use). PFBS was detected in the particle phase of outdoor air samples during one of the two sampling events in Manchester at 2.2 pg/m^3 and one of the two sampling events in Hazelrigg at 2.6 pg/m^3. PFBS was not detected above the method quantification limit at the Ireland and Norway sites. Harrad et al. (2020) measured PFBS

in air near 10 Irish municipal solid waste landfills located in non-industrial areas. Air samples were collected upwind and downwind of each landfill. PFBS was detected in more than 20% of the samples, with mean concentrations (ranges) at downwind and upwind locations of 0.50 (< 0.15–1.4) pg/m^3 and 0.34 (< 0.15–1.2) pg/m^3, respectively. Beser et al. (2011) and Jogsten et al. (2012) did not detect PFBS in ambient air samples in Spain. Beser et al. (2011) analyzed fine airborne particulate matter ($PM_{2.5}$) in air samples collected from five stations located in Alicante province, Spain (3 residential, 1 rural, 1 industrial) to determine levels of 12 ionic PFAS. PFBS was below the method quantification limit at all five locations. Jogsten et al. (2012) did not detect PFBS in ambient air samples collected outside homes in Catalonia, Spain.

In the one study identified from North America, Ahrens et al. (2011) determined levels of PFAS in air around a WWTP and two landfill sites in Canada. PFBS was not detected in any sample above the method detection limit.

PFBS has been detected in Artic air in one study, with a DF of 66% and mean concentration of 0.1 pg/m^3 (Arp and Slinde, 2018; Wong et al., 2018).

As with exposure to PFBS via indoor air, there is some evidence from European studies indicating PFBS is present in some ambient air samples. Further research is needed to understand the DF and concentrations of PFBS that occur in ambient environments in the United States.

B.3.6. Soil

PFBS can be released into soil from manufacturing facilities, industrial uses, fire/crash training sites, and biosolids containing PFBS (USEPA, 2021d; ATSDR, 2021). The EPA identified 16 studies that evaluated the occurrence of PFBS and other PFAS in soil, conducted in the United States, Canada, or Europe (see Table B-6). Two U.S. studies and two Canadian studies (Cabrerizo et al., 2018; Venkatesan and Halden, 2014; Blaine et al., 2013; Dreyer et al., 2012) were conducted in areas not reported to be associated with any known PFAS release or were experimental studies conducted at research facilities. At these sites, PFBS levels were low (≤ 0.10 ng/g) or below detection limits in non-amended or control soils. Two U.S. studies by Scher et al. (2019; 2018) evaluated soils at homes in Minnesota within and outside of a GCA impacted by a former 3M PFAS production facility; for sites within the GCA, one of the studies reported a DF of 10% and a 90th percentile PFBS concentration of 0.02 ng/g, and the other reported a DF of 9% and a maximum PFBS concentration of 0.017 ng/g. For sites outside of the GCA, the DF was 17% and the maximum PFBS concentration was 0.031 ng/g. Three U.S. studies and one Canadian study analyzed soils potentially impacted by AFFF used to fight fires—one at U.S. Air Force installations with historic AFFF use (Anderson et al., 2016), two at former fire training sites (Nickerson et al., 2020; Eberle et al., 2017), and another at the site of a train derailment and fire in Canada (Mejia-Avendaño et al., 2017). In these four studies, DFs ranged from 35 to 100%. PFBS concentrations in the study of the U.S. Air Force installations ranged from ND–79 ng/g, and PFBS concentrations ranged from ND–58.44 ng/g at one fire training site (Nickerson et al., 2020). The study of the other fire training site measured PFBS pre-treatment (0.61–0.6.4 ng/g) and post-treatment (0.07–0.83 ng/g) (Eberle et al., 2017). The DFs and range of PFBS concentrations measured in soils at the site of the train derailment were 75% DF and ND–3.15 ng/g, respectively, for the AFFF run-off area (measured in 2013, the year of accident) and 36% DF and ND–1.25 ng/g, respectively, at the burn site and adjacent area (measured in 2015) (Mejia-Avendaño et al., 2017).

Of the six European studies, one study (Harrad et al., 2020) analyzed soil samples collected upwind and downwind of 10 municipal solid waste landfills in Ireland and found PFBS levels to be higher in soils from downwind locations. Based on the overall study findings, however, the authors concluded there was no discernible impact of the landfills on concentrations of PFAS in soil surrounding these facilities. Grønnestad et al. (2019) investigated soils from a skiing area in Norway to elucidate exposure routes of PFAS into the environment from ski products, such as ski waxes. The authors found no significant difference in mean total PFAS in soil samples from the Granåsen skiing area and the Jonsvatnet reference area but noted that the skiing area samples were dominated by long-chain PFAS (C8–C14; ≥ 70%) and the reference area samples were dominated by short-chain PFAS (> 60%), which included PFBS. A study in Belgium (Groffen et al., 2019) evaluated soils collected at a 3M fluorochemical plant in Antwerp and at four sites located at increasing distances from the plant. PFBS levels were elevated at the plant site and decreased with increasing distance from the plant. The other three studies analyzed soil samples from areas near firefighting training sites in Norway and France, and reported PFBS concentrations varying from ND to 101 ng/g dry weight (Dauchy et al., 2019; Skaar et al., 2019; Hale et al., 2017).

A U.S. study of biosolid samples from 94 WWTPs across 32 states and the District of Columbia detected PFBS in 60% of samples at a mean concentration (range) of 3.4 (2.5–4.8) ng/g (Venkatesan and Halden, 2013). As mentioned, PFBS has been detected in drinking water wells, food types, and plant samples from soils or fields that have received biosolids applications that were industrially impacted (Blaine et al., 2014; Blaine et al., 2013; Lindstrom et al., 2011).

In summary, results of some available studies suggest that proximity to a PFAS production facility or a site with historical AFFF use or firefighting is correlated with increased PFBS soil concentrations compared to soil from sites not known to be impacted by PFAS. However, few available studies examined PFBS concentrations in soils not known to have nearby sources of PFBS. Additional research is needed that quantifies ambient levels of PFBS in soils in the United States.

Table B-6. Compilation of Studies Describing PFBS Occurrence in Soil

Study	Location	Site Details	Results
North America			
Venkatesan and Halden (2014)	United States (Baltimore, Maryland)	Control (nonamended) soil from Beltsville Agricultural Research Center	DF 0%
Blaine et al. (2013)	United States (Midwestern)	Urban and rural full-scale field study control (nonamended) soil	Urban control: DF NR, mean = 0.10 ng/g Rural control: DF NR, mean = ND
Scher et al. (2019)	United States (Twin Cities metropolitan region, Minnesota)	Near former 3M PFAS production facility, homes within a GCA	DF 10%, median (p90) = ND (0.02) ng/g

Study	Location	Site Details	Results
Scher et al. (2018)	United States (Twin Cities metropolitan region, Minnesota)	Near former 3M PFAS production facility, homes within and outside a GCA	Within GCA: DF 9%, median (range) = ND (ND–0.17 ng/g) Outside GCA: DF 17%, median (range) = ND (ND–0.031 ng/g)
Anderson et al. (2016)	United States (unspecified)	Ten U.S. Air Force installations with historic AFFF release, surface and subsurface soils	Surface soil: DF 35%, median (range) = 0.775 (ND–52.0) ng/g Subsurface soil: DF 35%, median (range) = 1.30 (ND–79.0) ng/g
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, Virginia)	Firefighting training site, pre- and posttreatment	Pretreatment: DF 60%, range = 0.61–6.4 ng/g Posttreatment: DF 100%, range = 0.07–0.83 ng/g
Nickerson et al. (2020)	United States (unspecified)	Two AFFF-impacted soil cores from former fire-training areas	Core E: DF[a] 91%, range = ND–27.37 ng/g dw Core F: DF 100%, range = 0.13–58.44 ng/g dw
Cabrerizo et al. (2018)	Canada (Melville and Cornwallis Islands)	Catchment areas of lakes	DF 100%, mean[a] (range) = 0.0024 (0.0004–0.0083) ng/g dw
Dreyer et al. (2012)	Canada (Ottawa, Ontario)	Mer Bleue Bog Peat samples (core samples)	Detected once at 0.071 ng/g in 1973 sample and not considered for further evaluation
Mejia-Avendaño et al. (2017)	Canada (Lac-Mégantic, Quebec)	Site of 2013 Lac-Mégantic train accident (oil and AFFF runoff area [sampled 2013], burn site and adjacent area [sampled 2015])	Background: DF NR, mean = 0.035 ng/g dw 2013: DF 75%, mean range = ND–3.15 ng/g dw 2015: DF 36%, mean range = ND–1.25 ng/g dw
Europe			
Harrad et al. (2020)	Ireland (multiple cities)	10 landfills, samples collected upwind and downwind	Downwind: DF NR, mean (range) = 0.0059 (ND–0.044) ng/g dw Upwind: DF NR, mean (range) = 0.0011 (ND–0.0029) ng/g dw
Grønnestad et al. (2019)	Norway (Granåsen, Jonsvatnet)	Granåsen (skiing area); Jonsvatnet (reference site)	Skiing area: DF 0%[b] Reference area: DF 70%, mean (range) = 0.0093 (ND–0.0385 ng/g dw)

Study	Location	Site Details	Results
Groffen et al. (2019)	Belgium (Antwerp)	3M perfluorochemical plant and 4 sites with increasing distance from plant	Plant: DF 92%, mean (range) = 7.84 (ND–33) ng/g dw Vlietbos (1 km from plant): DF 90%, mean (range) = 2.79 (ND–7.04) ng/g dw 2.3 km, 3 km, 11 km from plant: DF 0%
Dauchy et al. (2019)	France (unspecified)	Firefighting training site, samples collected in 6 areas collected up to 15-m depth; in areas 2 and 6, foams used more intensely and/or before concrete slab was built	Areas 1, 3, 4, and 5 combined: DF[a] 0–10%, range = ND–7 ng/g dw, across all depths Area 2: DF[a] 35%, range = ND–82 ng/g dw, across all depths Area 6: DF[a] 55%, range = ND–101 ng/g dw, across all depths
Skaar et al. (2019)	Norway (Ny-Ålesund)	Research facility near firefighting training site	Background: DF 0% Contaminated: DF 100%, mean[a] (range) = 4.9 (2.64–7.13) ng/g dw
Hale et al. (2017)	Norway (Gardermoen)	Firefighting training site	DF 0%

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; dw = dry weight; GCA = groundwater contamination area; km = kilometer; ND = not detected; ng/g = nanogram per gram; NR = not reported; PFAS = per- and polyfluoroalkyl substances; p90 = 90th percentile

[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

[b] Grønnestad et al. (2019) reported a DF = 10% but a range, mean, and standard deviation of < LOQ.

B.4. Recommended RSC

The EPA followed the Exposure Decision Tree approach to determine the RSC for PFBS (USEPA, 2000b). The EPA first identified two potential populations of concern (Box 1): pregnant women and their developing fetuses and women of childbearing age (see Section 2.2.2). However, limited information was available regarding specific exposure of these populations to PFBS in different environmental media. The EPA considered exposures in the general U.S. population as likely being applicable to these two populations. Second, the EPA identified several relevant PFBS exposures and pathways (Box 2), including dietary consumption, incidental oral consumption via dust, consumer products, and soil or dermal exposure via soil, consumer products, and dust, and inhalation exposure via indoor or ambient air. Several of these may be potentially significant exposure sources. Third, the EPA determined that there was inadequate quantitative data to describe the central tendencies and high-end estimates for all of the potentially significant sources (Box 3). For example, studies from Canada and Europe indicate that indoor and ambient air may be a significant source of exposure to PFBS. At the time of the literature search, the EPA was unable to identify studies assessing PFBS concentrations in

indoor or ambient air samples from the U.S. and therefore, the agency does not have adequate quantitative data to describe the central tendency and high-end estimate of exposure for this potentially significant source in the U.S. population. However, the agency determined there were sufficient data, physical/chemical property information, fate and transport information, and/or generalized information available to characterize the likelihood of exposure to relevant sources (Box 4). Notably, based on the studies summarized in the sections above, there are significant known or potential uses/sources of PFBS other than drinking water (Box 6), though there is not information available on each source to make a characterization of exposure (Box 8A). For example, there are several studies from the U.S. indicating that PFBS may occur in dust sampled from various microenvironments (e.g., homes, offices, daycare centers, vehicles). However, the majority of studies sampled in only one location and few studies examined dust samples outside of the home (e.g., one study from the U.S. assessed PFBS occurrence in dust sampled from vehicles). Additionally, though several studies from around the U.S. measured PFBS concentrations in dust from houses, the detection frequencies in these studies varied widely (from 3% to 59%) and may be a result of uncertainties including home characteristics, behaviors of the residents, and the presence or absence of PFBS-containing materials or products (Haug et al., 2011). Therefore, it is not possible to determine whether dust can be considered a major or minor contributor to total PFBS exposure. Similarly, it is not possible to determine whether the other potentially significant exposure sources such as seafood and consumer products should be considered major or minor contributors to total PFBS exposure. Given these considerations, following recommendations of the Exposure Decision Tree (USEPA, 2000b), the EPA recommends an RSC of 20% (0.20) for PFBS.

Appendix C. PFNA: Summary of Occurrence in Water and Detailed Relative Source Contribution

C.1. Occurrence in Water

The use and production of PFNA could result in its release to the aquatic environment through various waste streams (NCBI, 2022b). PFNA has an estimated water solubility of 62.5 µg/L (6.25×10^{-2} mg/L) at 25°C and when released to surface water, it is expected to adsorb to suspended solids and sediment (NCBI, 2022b). Volatilization from water surfaces is not expected to be an important fate process for PFNA (NCBI, 2022b).

C.1.1. Groundwater

Several peer-reviewed studies were identified that examined PFNA occurrence in groundwater sources from the U.S. In three studies, sampling locations included sites known or suspected to contain PFAS but not related to AFFF use (Procopio et al., 2017; Post et al., 2013; Lindstrom et al., 2011). Procopio et al. (2017) evaluated groundwater samples from a small area of an industrial/business park located within the South Branch Metedeconk River watershed. Two sampling events were conducted as part of a source trackdown study to identify potential sources of PFAS contamination after elevated PFOA levels were discovered at a raw surface water intake of the Brick Township Municipal Utilities Authority. Samples were collected following the installation of 16 temporary monitoring wells by the New Jersey Geological and Water Survey or a contract driller during August 2013 and June and July 2014. PFNA was detected in 32% of samples (n = 19), with concentrations ranging from < 5 to 63 ng/L; a mean concentration was not reported. The maximum PFNA level found (63 ng/L) occurred at a well located in the middle of the industrial/business park where the highest PFAAs were detected in the study. Based on the results of all PFAAs analyzed, the authors concluded a strong likelihood of a groundwater plume of PFAS contamination resulting from the suspected illicit discharge of liquid waste to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs. Post et al. (2013) evaluated raw groundwater samples from public drinking water system intakes in two sampling campaigns. Between August 2009 and February 2010, groundwater samples from 18 drinking water systems were obtained from 1 confined well (sunk into an aquifer located between two impermeable strata) and 17 unconfined wells in the upper unconfined aquifer that were chosen to represent New Jersey geographically. The sampled locations included one site with a nearby industrial facility that previously used large quantities of PFNA. PFNA was found in 5 of 18 samples with concentrations ranging from not detected (ND) to 96 ng/L. The maximum PFNA concentration was at the site with the nearby industrial facility. Sampling was also conducted in 2010–2013 from unconfined wells of two additional public drinking water systems with groundwater known to be contaminated by PFOA. Four wells were sampled at the first system and one well at the second system. PFNA was detected in all five wells. Sample detection frequencies were not reported but concentrations ranged from ND to 16 ng/L across the four wells in the first system and from 24 to 72 ng/L in the second system. Lindstrom et al. (2011) analyzed well water samples from 13 wells used for livestock, watering gardens, and washing in Decatur, Alabama. The samples were collected in

February 2009 from farms that had applied PFC-contaminated biosolids to local agricultural fields as a soil amendment for at least 12 years. PFNA was below the LOQ (10 ng/L) in all 13 wells.

Three studies from the U.S. evaluated groundwater potentially impacted by wastewater (Boone et al., 2019; Appleman et al., 2014; Quiñones and Snyder, 2009). In the first study, Boone et al. (2019), evaluated 17 PFAS in source and treated waters collected in 2010–2012. Of the three groundwater sources evaluated, PFNA was detected in two out of three samples at levels of 1.25 and 0.156 ng/L. In Appleman et al. (2014), authors assessed source water from five utilities in New Jersey from November 2011 to September 2012, the majority of which were selected because they were either known from previous monitoring or expected to contain detectable PFAS because they were impacted by upstream wastewater effluent discharge. PFNA was detected in six of seven groundwater samples. The study did not report an average value for PFNA, but concentrations ranged from below the method RL (0.5 ng/L) to 47 ng/L. In the third study, Quiñones and Snyder (2009) examined levels of eight PFAS at two sites in Nevada that were highly impacted from treated wastewater. Mean PFNA levels at the two sites were 6.9 and 5.7 ng/L at sites 1 (n = 7) and 2 (n = 8), respectively.

The remaining three U.S. studies identified addressed sites of current and/or historic use of AFFFs (Steele et al., 2018; Eberle et al., 2017; Anderson et al., 2016). Anderson et al. (2016) assessed 40 sites across 10 active Air Force installations throughout the continental United States and Alaska between March and September 2014. Installations were included if there was known historic AFFF release in the period 1970–1990. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. The selected sites were not related to former fire training areas and were characterized according to volume of AFFF release—low (n = 24), medium (n = 100), and high (n = 25). Across all sites, the PFNA detection frequency was 46.38% and the median concentration at sites with detectable levels was 105 ng/L. PFNA was detected only at low- and medium-volume release sites with detection frequencies of 37.5% and 40.6%, respectively, and mean concentrations of 300 and 900 ng/L, respectively. Authors noted that given PFNA is not present in 3M AFFF formulations, there may be some degree of telomer-based AFFF contamination. Steele et al. (2018) investigated a contaminated military base in Alaska and former Pease Air Force Base (the latter being historical, secondary data). Authors reported the primary source of contamination for the Alaska military base to be from prior legacy AFFF use and wells were selected for sampling based on historical data that indicated PFOS and PFOA contamination. Well samples at the Alaska base were collected monthly from July 2016 to March 2017 to determine if monthly variations in PFAS concentrations existed. For four wells, PFNA was detected one to three times during the monthly sampling at concentrations ranging from 0.91 to 6.6 ng/L. PFNA was not detected in any of the eight monthly timepoints in two other wells. A seventh well was only sampled in July 2016 and reported a PFNA concentration of 1.3 ng/L. The authors found that PFAS concentrations did not vary significantly on the scale of weeks or months. Eberle et al. (2017) collected groundwater samples first in April and December 2012 as part of a screening/site characterization analysis. Additional samples were collected in 2013–2014 before and after a pilot scale field test at a former fire training site at Joint Base Langley-Eustis, Virginia. Monthly fire training activities were conducted at the site from 1968 to 1980 and irregular fire training activities continued until 1990. Of the data reported, samples collected for site characterization showed PFNA was detected in all wells (seven deep, three shallow)

sampled. PFNA was also detected in all pre-treatment samples (n = 5), ranging from approximately 100 to 1,500 ng/L.

The EPA also identified studies from Canada and Europe reporting the occurrence of PFNA in groundwater, which are briefly summarized below. Detailed results from each study are presented in Table C-1. PFNA was detected in groundwater samples collected in 2010 from the Highland Creek watershed in Canada at concentrations ranging from 0.071 ng/L to 0.54 ng/L (Meyer et al., 2011, as cited in NCBI, 2022). In this study, the authors reported that none of the sampling sites receive water that is impacted by known PFAS point sources (Meyer et al., 2011). In Europe, PFNA investigations in groundwater were conducted in France (Dauchy et al., 2019; Bach et al., 2017; Boiteux et al., 2017; Dauchy et al., 2017; Gellrich et al., 2013; Boiteux et al., 2012; Dauchy et al., 2012), Germany (Gellrich et al., 2013), Ireland (Harrad et al., 2020), Italy (Barreca et al., 2020; Ciofi et al., 2018; Gellrich et al., 2013), Malta (Sammut et al., 2019), Norway (Høisæter et al., 2019), Spain (Jurado-Sánchez et al., 2013; Llorca et al., 2012) and Sweden (Gobelius et al., 2018; Gyllenhammar et al., 2015). Loos et al. (2010) also collected groundwater from 164 groundwater monitoring stations of participating European Union laboratories, within 23 countries, in the fall of 2008. They detected PFNA in 15.2% of these collected samples with concentrations up to 10 ng/L. Gellrich et al. (2013) also collected samples in multiple countries along the Rhine River, collecting both river filtrate and combined groundwater and percolated water from the Rhine riverbed in Germany, France, and Italy. In this campaign, they did not detect PFNA from the Rhine River or riverbed groundwater.

In the studies conducted in countries along the Mediterranean Sea, there was little-to-no detected PFNA in groundwater sources. In Spain, no PFNA was detected in well water samples collected by Jurado-Sánchez et al. (2013) in the southeast of the country nor by Llorca et al. (2012) in Barcelona. In the Lombardia region of Italy, Barreca et al. (2020) detected PFNA in 3% of 130 collected groundwater samples across 57 sampling stations in 2018. Ciofi et al. (2018) collected groundwater samples at 12 locations across Tuscany, including Siena, Florence, and Prato. One grab sample was collected at each of these 12 locations, detecting PFNA at each with concentrations between <0.26–4.8 ng/L (Ciofi et al., 2018). Sammut et al. (2019) collected groundwater from ten boreholes across the island country in 2015-2016, which were sites used by the Malta Water Services Corporation for both water extraction and quality analysis sampling. Across these ten boreholes, they detected PFNA in one borehole at a concentration of 0.90 ng/L (Sammut et al., 2019).

The EPA identified a number of studies reporting PFNA measurements within France (Dauchy et al., 2019; Bach et al., 2017; Boiteux et al., 2017; Dauchy et al., 2017; Gellrich et al., 2013; Boiteux et al., 2012; Dauchy et al., 2012). Boiteux et al. (2012) analyzed raw water from drinking water treatment plants distributed across 100 French departments, representing approximately 20% of the national water supply flow in 2009 and 2010. In their first sampling campaign in 2009, they detected PFNA in 6% of collected samples, with a maximum concentration of 14 ng/L. In their second sampling campaign in 2010, they did not detect PFNA at a limit of detection of 1.3 ng/L (Boiteux et al., 2012). In 2013, Bach et al. (2017) and Boiteux et al. (2017) evaluated the PFNA concentration in alluvial wells that are influent groundwater to drinking water treatment plants in southern and northern France, respectively. Both Bach et al. (2017) and Boiteux et al. (2017) sampled groundwater downstream from industrial sites which produce fluoropolymers and fluorotelomer-based products. In southern France, Bach et al. (2017) detected FPNA in 86–100% of collected samples, with concentrations from <4 pg/m^3 (the

limit of quantification) to 37 pg/m^3. Alternatively, in northern France, Boiteux et al. (2017) did not detect PFNA in any of the sampled alluvial wells.

A number of studies focused specifically on investigating sites where PFAS-containing material had been heavily used. In Norway, Høisæter et al. (2019) analyzed groundwater at a firefighting training site that extensively used PFAS-containing materials until their ban in 2011. Five monitoring wells were sampled in 2016, totaling 19 sampling campaigns, which detected a mean concentration of approximately 950 ng/L in the monitoring wells at the site. Dauchy et al. (2019) investigated a similar firefighting training facility with heavy PFAS use in France in 2015–2016 but detected no groundwater contamination. Neighboring Norway, in Sweden, Gyllenhammar et al. (2015) sampled monitoring, private, and production wells for four drinking water treatment plants downstream of a military airport with firefighting training activities. Unlike the findings of Høisæter et al. (2019) in Norway, they found no detectable PFNA across the wells they sampled in 2012–2014 (Gyllenhammar et al., 2015). Gobelius et al. (2018) also sampled at "PFAS hot spots" (e.g., firefighter training sites, sewage treatment plants, landfills) across Sweden, detecting PFNA in 27% of samples with concentrations between <0.08–66 ng/L. In Ireland, Harrad et al. (2020) collected groundwater samples from boreholes downgradient from ten municipal solid waste landfills across the country, which accepted municipal waste, non-hazardous industrial waste, construction and demolition waste, and biomedical waste. Across these sites, they detected PFNA in 10% of groundwater samples, with concentrations ranging from <0.1–0.22 ng/L.

Dauchy et al. (2012) sampled raw water from monitoring groundwater wells at a fluoropolymer manufacturing plant and at two drinking water treatment plants downstream of the manufacturing plant and many other domestic and industrial activities. At the fluoropolymer manufacturing plant, three of the four sampled wells reported detectable levels of PFNA from 21–724 ng/L. Downstream, at the drinking water treatment plants, five of five sampled monitoring wells contained detectable PFNA ranging from 13–35 ng/L (Dauchy et al., 2012). Dauchy et al. (2017) also conducted sampling campaigns in late 2014 through early 2015, investigating a previously operated oil refinery, a military airport, and a training center for firefighters. These sites were selected due to their heavy use of fluorosurfactant-based foams, and samples were collected from groundwater monitoring wells. PFNA was detected in the oil storage depot, between 11–12 ng/L in October 2014 and March 2015. During this period, no PFNA was detected in the military airport or firefighter training center groundwater (limit of quantification=4 ng/L) (Dauchy et al., 2017).

Table C-1. Summary of Studies Reporting the Occurrence of PFNA in Groundwater

Study	Location	Site Details	Results
United States			
Procopio et al. (2017)	United States (New Jersey)	Groundwater from an industrial/business park located within the South Branch Metedeconk River watershed, where there was suspected illicit discharge to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs. Samples were collected following the installation of 16 temporary monitoring wells by the NJ Geological and Water Survey or a contract driller during August 2013 (sampling event #7) and June–July 2014 (sampling event #8). Samples were taken from the upper 1.5 m (5 ft) of the water table from each well, except for one "profile well" in which samples were collected at three different depths (3.7–4.6, 6.7–7.6, and 10.7–11.6 m below grade; 12–15, 22–25, and 35–38 ft below grade, respectively).	n = 19, DF[a] 32%, range = <5–63 ng/L (minimum reporting level = 5 ng/L)
Post et al. (2013)	United States (New Jersey)	Raw water collected from public drinking water system intakes in two sampling campaigns. In the first sampling campaign, samples from 18 drinking water systems were collected between August 2009 and February 2010 from 1 confined well (sunk into an aquifer located between two impermeable strata) and 17 unconfined wells in the upper unconfined aquifer; sites were chosen to represent NJ geographically and included 1 site with a nearby industrial facility that previously used large quantities of PFNA (site 5). In the second sampling campaign, samples from two drinking water systems (PWS-A and PWS-B) were collected in 2010–2013 from five unconfined wells. Groundwater at these two systems were known to be contaminated by PFOA.	1st sampling campaign: n = 18, DF 28%, range = ND–96 ng/L 2nd sampling campaign: PWS-A, WF1A: n = 5, DF NR, range = ND–6 ng/L PWS-A, WF1B: n = 4, DF NR, range = ND–12 ng/L PWS-A, WF2A: n = 9, DF NR, range = ND–16 ng/L PWS-A, WF2B: n = 9, DF NR, range = ND–7 ng/L PWS-B: n = 8, DF NR, range = 24–72 ng/L (minimum reporting level = 5 ng/L)
Lindstrom et al. (2011)	United States (Decatur, Alabama)	Thirteen samples collected in February 2009 from 13 wells located on farms with historical land application of PFC-contaminated biosolids to local agricultural fields between 1995 and 2008. Biosolids obtained from local municipal WWTP where sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers,	n = 13, DF (frequency of quantification) 0% (LOQ = 10 ng/L)

Study	Location	Site Details	Results
		fluorocarbon fibers, polymers, polymer films, and resins.	
Boone et al. (2019)	United States (unspecified)	Three groundwater sites used as source waters for three DWTPs, collected in 2010−2012; some locations with known or suspected sources of wastewater in the source water, but study did not differentiate which locations had known or suspected sources.	n = 3, DF[a] 67%, range = ND–1.25 ng/L (LCMRL = 0.094 ng/L)
Appleman et al. (2014)	United States (New Jersey)	Groundwater source water for five DWTPs, sampled November 2011 to September 2012. Majority of the utilities were selected because they were either known from previous monitoring or expected based on their source waters to contain detectable PFAS (i.e., impacted by upstream wastewater effluent discharge). Two sites were sampled twice and three sites were sampled only once.	n = 7, DF[a] 86%, range = <Method RL–47 ng/L (method reporting limit = 0.5 ng/L)
Quiñones and Snyder (2009)	United States (Nevada)	Samples collected in 2008 from two groundwater sites in Las Vegas Wash, Nevada that were highly impacted from treated wastewater.	Site 1: n = 7, DF NR, mean (maximum) = 6.9 (8.8) ng/L Site 2: n = 8, DF NR, mean (maximum) = 5.7 (8.9) ng/L (method reporting limit = 1.0 ng/L)
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between 1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were not specifically controlled for in the site selection process; active remedies had not been applied at any of the sites selected. Approximately ten samples were collected between March and September 2014 at each site; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases. Groundwater	Overall: n = 149, DF 46.38%, median (maximum) = 105 (3,000) ng/L *Breakdown by site group:* Emergency Response (low-volume release): n = 24, DF 37.5%, mean (range) = 300 (57–450) ng/L Hangars and Buildings (medium-volume release): n = 100, DF 40.6%, mean (range) = 900 (22–10,000) ng/L Testing and Maintenance (high-volume release): n = 25, DF 0% (median reporting limit = 18 ng/L) *Minimum of detected values reported *Median calculated using quantified detections

Study	Location	Site Details	Results
		samples were collected from existing monitoring wells and temporary monitoring wells installed with direct push technology.	*Non-detects were substituted with ½ the reporting limit
Steele et al. (2018)	United States (Alaska)	Monthly samples collected from a military installation during July 2016–March 2017; six wells from around the installation were sampled each month, along with a seventh well that was only sampled in July 2016. PFAS contamination predominately from prior legacy AFFF use. Wells selected based on historical sample data indicating PFAS contamination.	Data for July, August, September, October, November, December, January, and February, respectively: Well A: 1, ND, ND, ND, ND, ND, ND, ND ng/L Well B: 1.2, ND, ND, ND, ND, ND, ND, ND ng/L Well D: 1.3 ng/L (no values provided for other months) Well E: ND, ND, ND, ND, ND, ND, ND, ND ng/L Well F: ND, ND, ND, ND, ND, ND, ND, ND ng/L DK: 4.8, ND, ND, ND, 6.4, 6.6, ND, ND ng/L FG: 0.91, 0.91, ND, ND, ND, ND, ND, ND ng/L (method detection limit not reported)
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, Virginia)	Pilot testing area in former fire training area (Training Site 15) at Joint Base Langley-Eustis where monthly fire training activities were conducted from 1968 to 1980 in a zigzag pattern burn pit. Facility was abandoned in 1980 but irregular fire training activities using an above-ground germed burn pit continued until 1990. Groundwater samples collected for screening/site characterization (April and December 2012), and for pre- (April 2013) and post- (October 2013 and February 2014) in situ chemical oxidation treatment using a peroxone activated persulfate (OxyZone) technology. Treatment was conducted in Test Cell 1 over 113 days (April through August 2013). Pre-treatment samples were collected from 14 wells screened in the deep zone, and 3 wells screened in the shallow zone. Post-treatment samples were collected from the same wells as the pre-treatment samples with an additional three wells (two shallow, one deep) sampled. Wells EC-1, EC-2, EC-3, EC-4, I-	Screening/site characterization: EC-1 (deep, sentry): 100 ng/L EC-2 (deep, sentry): 600 ng/L I-1 (deep): 700 ng/L I-2 (deep, sentry): 400 ng/L I-4 (deep): 900 ng/L I-5 (shallow): 100 ng/L I-6 (shallow): 200 ng/L MW-2904 (deep): 100 ng/L U-16D (deep): 1,700 ng/L U-16S (shallow): 200 ng/L Pre-treatment (values reported for two different laboratories): EC-2: 900; 500 ng/L EC-3: 1,500; 700 ng/L I-1: 200; 300 ng/L I-2: 200; 100 ng/L I-4: 1,700; 800 ng/L

Study	Location	Site Details	Results
		2, and I-3 were sentry wells to monitor the possible migration of oxidants and contaminants outside Test Cell 1. No PFNA data reported for post-treatment samples.	(LOQ not reported)
Europe			
Bach et al. (2017)	France (southern)	Samples were collected from alluvial wells that provide source water for two DWTPs. The two DWTPs are located on both sides of a river, ~15 km downstream from an industrial site where two facilities produce fluoropolymers; the industrial site discharges its effluents at three points along a river. The alluvial wells are located along the river, with wells for the first DWTP (DWTP A) located on the left shore and alluvial wells for the second DWTP (DWTP B) located on the right shore, on an island formed by a backwater. Sample collection occurred in April, July, October, and December 2013.	Alluvial wells for DWTP A: April 2013: n = 7, DF[a] 86%, range = <4–25 ng/L July 2013: n = 7, DF[a] 86%, range = <4–25 ng/L October 2013: n = 7, DF[a] 86%, range = <4–37 ng/L December 2013: n = 7, DF[a] 86%, range = <4–30 ng/L Alluvial wells for DWTP B: April 2013: n = 8, DF[a] 100%, mean[a] (range) = 8.13 (4–17) ng/L July 2013: n = 8, DF[a] 88%, range = <4–15 ng/L October 2013: n = 7, DF[a] 100%, mean[a] (range) = 9.43 (5–15) ng/L December 2013: n = 8, DF[a] 100%, mean[a] (range) = 7.38 (4–10) ng/L (LOQ = 4 ng/L) *DF represents frequency of quantification
Boiteux et al. (2017)	France (northern)	Samples were collected in four sampling campaigns (May, July, October, and December 2013) from alluvial wells that provide source water for two DWTPs. The two DWTPs (A and B) are located downstream of an industrial WWTP that processes raw sewage from a facility that manufactures fluorotelomer-based products and side-chain-fluorinated polymers used in firefighting foams and stain repellents. DWTP A is located 15 km downstream from the WWTP and is supplied by five alluvial wells. DWTP B is located 20 km downstream of the WWTP and is supplied by four alluvial wells.	DWTP A: May 2013: n = 5, DF 0% July 2013: n = 5, DF 0% October 2013: n = 5, DF 0% December 2013: n = 5, DF 0% DWTP B: May 2013: n = 4, DF 0% July 2013: n = 4, DF 0% October 2013: n = 4, DF 0% December 2013: n = 4, DF 0% (LOQ = 4 ng/L) *DF represents frequency of quantification
Dauchy et al. (2012)	France (unspecified)	Raw water sampled in June 2010 from four monitoring wells at a fluoropolymer manufacturing	Fluoropolymer manufacturing plant:

Study	Location	Site Details	Results
		plant (P13, P14, P15, P01). Groundwater flowed from well P14 to P01 and well P15 is nearest to the polyvinylidene fluoride production area. Raw water resources also collected from two DWTPs (five sampling sites – DWA-1, DWA-2, DWA-3, DWA-4, DWB-1); the first DWTP (DWA) is supplied by four alluvial wells, and the second DWTP (DWB) is supplied by one alluvial well. The two DWTPs are located on both sides of a river, 15 km downstream of fluorochemical manufacturing facility. The river receives wastewater from many domestic and industrial activities.	P13: not quantifiable due to dilution or matrix effects P14: n = NR, DF NR, 21 ng/L P15: n = NR, DF NR, 342 ng/L P01: n = NR, DF NR, 724 ng/L DWA: DWA-1: n = NR, DF NR, 13 ng/L DWA-2: n = NR, DF NR, 35 ng/L DWA-3: n = NR, DF NR, 30 ng/L DWA-4: n = NR, DF NR, 33 ng/L DWB: DWB-1: n = NR, DF NR, 21 ng/L (LOQ = 4 ng/L) *Study did not indicate whether concentrations reported were point values or means
Harrad et al. (2020)	Ireland (multiple cities)	Groundwater samples collected between November 2018 and January 2019 from ten municipal solid waste landfills at two sampling points down-gradient from the main body of each landfill. Each sampling point consisted of a borehole leading down to water reservoirs at a minimum depth of 5 m below ground level. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction and demolition, and biomedical waste.	n = 10, DF[a] 10%, range = <0.1–0.22 ng/L (LOD = <0.1 ng/L) *Non-detects replaced by ½ LOD
Gobelius et al. (2018)	Sweden (national)	Sampling conducted between May and August 2015, with the majority in July and a few samples in September and November 2015. Samples were collected in 21 regional counties by the County Administration Boards. Sampling locations selected based on potential vicinity of PFAS hot spots (i.e., fire training sites, unspecific industry, sewage treatment plant effluent, landfill/waste disposal, skiing, and urban areas) and/or importance as a drinking water source. Sample numbers varied for each county and sampling sites were spread unevenly across Sweden.	n = 161, DF[a] 27%, range = <0.08–66 ng/L (method detection limit = 0.084 ng/L)

Study	Location	Site Details	Results
Boiteux et al. (2012)	France (national)	Raw water from DWTPs distributed across 100 French departments to represent ~20% of the national water supply flow; samples collected during two sampling campaigns in July–September 2009 (first campaign) and June 2010 (second campaign – focused on sites from first sampling campaign that had PFC levels >LOQ). Some sites possibly affected by industrial or commercial releases.	Overall: n = 196, DF (frequency of quantification) 3%, maximum = 14 ng/L *1st Sampling Campaign* n = 163, DF 6%, mean, median (maximum) = <1, <1 (14) ng/L *2nd Sampling Campaign* n = 33; results not reported (LOD = 1.3 ng/L, LOQ = 4 ng/L)
Loos et al. (2010)	23 European countries	Groundwater collected from 164 groundwater monitoring stations of participating European Union Member State laboratories during an 8-week window in Fall 2008. There were no strict selection criteria for the sampling sites such as “representative” or “contaminated”. Most monitoring stations were “official” monitoring stations also used for drinking water abstraction.	n = 164, DF 15.2%, mean, median (maximum) = 0, 0 (10) ng/L (LOD = 0.4 ng/L)
Dauchy et al. (2019)	France (unspecified)	Samples collected in two sampling campaigns in and around site where fluorosurfactant-based foams have been used extensively. From 1969 to 1984, the site was an oil refinery, with the exact location of the firefighting training area, frequency of training sessions, and history of firefighting training activities unknown. From 1987 to date, it has been a large training area for firefighters. First sampling campaign collected 13 samples from 9 monitoring wells and 4 springs in June 2015. Second sampling campaign collected from four monitoring wells in October 2016. Monitoring wells MW-1 to MW-5 were located upgradient from the firefighter training site around a landfill site. Monitoring well MW-11 and springs SW-A, SW-B, and SW-D located downgradient from the landfill or firefighter training site but not in the direction of groundwater flow. Monitoring wells MW-6 to MW-13 and spring SW-C were located downgradient from the firefighter training site in the direction of groundwater flow.	Upgradient: Monitoring wells: n = 5, DF 0% Downgradient but not in the direction of groundwater flow: Monitoring wells: n = 1, DF 0% Spring water: n = 3, DF 0% Downgradient in the direction of groundwater flow: Monitoring wells: n = 7, DF 0% Spring water: n = 1, DF 0% (LOQ = 4 ng/L)
Høisæter et al. (2019)	Norway (unspecified)	Firefighting training site with an airport that extensively used AFFF containing PFOS since the early 1990s until 2001 when it was replaced by fluorotelomer containing AFFF. All PFAS containing	n = 19, DF NR, mean* = 950 ng/L (LOD/LOQ not reported) *Mean estimated from Figure 4b in the paper

Study	Location	Site Details	Results
		firefighting foams was banned at the airport in 2011. Groundwater samples collected in 2016 at five pumping wells installed down gradient of the site to intercept and pump and treat the plume spreading from the firefighting training site. A total of 19 sampling campaigns were performed.	
Dauchy et al. (2017)	France (unspecified)	Samples collected in the vicinity of three sites (A, C, D) where fluorosurfactant-based foams are or were being heavily used. Site A is an oil storage depot located in a river port. In June 1987, a large explosion occurred in the depot and the fire was extinguished by applying a large amount of fluorosurfactant-based foams. Two groundwater samples were collected in October 2014 and March 2015 from a monitoring well located in the center of the depot. The water table lies 2.5 – 3.5 m below the ground. Site C is a military airport, with the exact location of the training area, frequency of the training sessions, and history of the firefighting training activities unknown. The well supplying the DWTP was sampled in March 2015. Site D is a training center for firefighters. From 1969 to 1984, the site was an oil refinery. Starting in 1987, the site became a training area for firefighters, with exercises carried out directly on the soil. From the 1990s, some exercise areas were covered with concrete. In November 2014, groundwater samples were collected from five springs.	Site A: October 2014: n = 1, point = 11 ng/L March 2015: n = 1, point = 12 ng/L Site C: n = 1, DF 0% Site D: n = 5, DF[a] 0% (LOQ = 4 ng/L)
Gyllenhammar et al. (2015)	Sweden (Uppsala)	Three observation well sites (Tuna backar: n = 3 wells; Svartbäcken: n = 1 well, Librobäck: n = 2 wells;) were sampled from September 2012 to January 2013. Four DWTP production well sites (Storvad: n = 9 wells; Galgbacken: n = 1 well; Stadsträdgården and Kronåsen: n = 6 wells; Sunnersta: n = 5 wells) were sampled from July 2012 to February 2014. One private well (Klastorp) was sampled in September 2012.	Observation wells: Tuna backar: n = 3, DF 0% Librobäck: n = 4, DF 0% Svartbäcken: n = 3, DF 0% Production wells: Storvad: n = 12, DF 0% Galgbacken: n = 7, DF 0% Stadsträdgården and Kronåsen: n = 103, DF 0% Sunnersta: n = 50, DF 0% Private well: Klastorp: n = 1, DF 0%

Study	Location	Site Details	Results
		All wells located downstream of a military airport with firefighting training activities up to the year 2003. It is not known when the usage of AFFF started.	(method detection limit = 10 ng/L)
Barreca et al. (2020)	Italy (Lombardia region)	Fifty-seven groundwater sampling stations throughout the region. Samples collected in 2018.	n = 130, DF[a] 3%, range NR (LOQ = 1 ng/L)
Sammut et al. (2019)	Malta	Groundwater collected from ten boreholes at different areas on the island during November and December 2015 and January 2016. Collection sites were the most commonly used extraction sites by the Malta Water Services Corporation for water extraction as well as for sampling for water quality analysis.	n = 10, DF 10%, range = ND–0.90 ng/L (LOD = 0.02 ng/L; LOQ = 0.04 ng/L)
Ciofi et al. (2018)	Italy (Tuscany)	Groundwater samples were collected at 12 locations. Sampling year was not reported. Each sample was collected from the phreatic layer with a mean depth between 10–75 m: GW-1: Siena, 10 m GW-2, GW-3, GW-4: Florence, 15 m GW-5: Prato, 75 m GW-6: Prato, 71 m GW-7: Prato, 70 m GW-8: Prato, 61 m GW-9: Florence, 17 m GW-10, GW-11, GW-12: Florence, 10 m	GW-1: n = 1, point = <0.28 ng/L GW-2: n = 1, point = <0.28 ng/L GW-3: n = 1, point = <0.28 ng/L GW-4: n = 1, point = <0.29 ng/L GW-5: n = 1, point = <0.33 ng/L GW-6: n = 1, point = <0.46 ng/L GW-7: n = 1, point = 3.2 ng/L GW-8: n = 1, point = 1.3 ng/L GW-9: n = 1, point = 4.8 ng/L GW-10: n = 1, point = <0.27 ng/L GW-11: n = 1, point = <0.26 ng/L GW-12: n = 1, point = <0.27 ng/L (MDL = 0.26–0.46 ng/L) *method detection limit varied by sample and was provided for 9 of 12 samples
Gellrich et al. (2013)	Germany (Hesse); France; Italy	Untreated water samples for preparation of mineral water included seven from Hesse, three from France, and four from Italy. The supplying waterworks obtain their untreated water either from Rhine river filtrate, a mixture of ground water and percolation water from the Rhine riverbed, drawn from wells 30–50 m deep or from wells in their closer vicinity. Sampling year not reported.	n = 14, DF (frequency of quantification) 0% (LOQ = 1 ng/L)

Study	Location	Site Details	Results
Jurado-Sánchez et al. (2013)	Spain (southeast)	Well water samples were collected and analyzed using a newly developed analytical method. Authors did not report sampling details or sampling year.	n = NR, DF 0% (LOD = 0.1 ng/L)
Llorca et al. (2012)	Spain (Barcelona)	Well water samples from two different sites were collected from the North of Barcelona metropolitan area in 2011.	n = 2, DF 0% (method LOD = 1.9; method LOQ = 6.3 ng/L)

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; DWTP = drinking water treatment plant; ft = feet; m = meter; ND = not detected; ng/L = nanogram per liter; PFAA = perfluoroalkyl acid; PFAS = per- and polyfluoroalkyl substances; NR = not reported; LOD = limit of detection; LOQ = limit of quantification; LCMRL = lowest concentration minimum reporting level; WWTP = wastewater treatment plant.

[a] The DF and/or mean was calculated using point data. Means were calculated only when DF = 100%.

C.1.2. Surface Water

The EPA identified many studies that reported on the occurrence of PFNA in surface water in both the U.S. and internationally. Overall, most U.S. studies reported PFNA detected in at least one surface water sample site in each study. Concentrations of PFNA in 12 remote and urban Minnesota surface water samples, including samples collected from Lake Michigan, ranged from <0.3 ng/L to 3.1 ng/L (<0.0003 µg/L to 0.0031 µg/L) (Simcik and Dorweiler, 2005, as cited in ATSDR, 2021). PFNA was detected in 38% of eight surface water samples from U.S. streams in the Great Lakes basin collected during 1994 to 2000 at concentrations between 0.03 ng/L and 0.4 ng/L (0.00003 µg/L and 0.0004 µg/L) (Klecka et al., 2010, as cited in NCBI, 2022). PFNA was detected in six locations in the Delaware River at concentrations ranging from 1.65 ng/L to 976 ng/L (0.00165 µg/L to 0.976 µg/L) in 2007 to 2009 (DRBC, 2013, as cited in ATSDR, 2021).

Three studies investigated surface water upstream and downstream of fluoropolymer facilities, with some sites also downstream of other potential PFAS sources (e.g., landfills, WWTPs) (Galloway et al., 2020; Newsted et al., 2017; Newton et al., 2017). Galloway et al. (2020) assessed several rivers and tributaries along the Ohio River in three sampling trips in 2016. The sampling sites ranged from upstream, downstream, and north/northeast of a fluoropolymer facility and known PFAS containing landfills. In June 2016, samples were collected on a 188 km stretch of the Ohio River, from 130 km downstream to 58 km upstream of the facility, and tributaries that pass near known PFAS-containing landfills. In July 2016, samples were collected from lakes, rivers, and creeks to the north and northeast of the facility as far as 16 km downwind. The December 2016 trip expanded the collection radius to more than 48 km downwind to the north and northeast of the facility. PFNA was detected in 92% of samples (n = 26) in June 2016, however all detects were below the LOQ (10 ng/L). From the second sampling trip, PFNA was not detected in any sample in July 2016 (n = 25). Finally, in December 2016, PFNA was detected at levels above the LOQ in one sample at 24.2 ng/L, detected but below the LOQ in 31 samples, and not detected in 8 samples. In Newsted et al. (2017), surface water samples were collected in August 2011 from a 3-mile section of the Upper Mississippi River: ten sampling reaches (three samples each) in an area between Ford Dam (between Minneapolis and St. Paul) and Hastings Dam (near Hastings) and which had been subject to 10–15 years of actions to reduce PFAS contamination from 3M Cottage Grove plant and other commercial/industrial entities. PFNA was detected in one sample from reach 10, immediately downstream the 3M Cottage Grove facility outfall, at a concentration of 2.0 ng/L. PFNA in all other samples was below the LOQ (2.0 ng/L). Newton et al. (2017) investigated surface water upstream and downstream of facilities that manufactured or used fluorinated materials along the Tennessee River near Decatur, Alabama. Six sampling sites were located upstream of the manufacturing facilities and three sites were downstream. Among the upstream sites, three were also upstream of a WWTP. All samples were collected in October 2015. PFNA was below the LOQ (10 ng/L) in all nine samples from the nine different sampling sites.

In four U.S. studies, sampling locations included surface waters potentially impacted by current and/or historic use of AFFFs (Anderson et al., 2016; Post et al., 2013; Nakayama et al., 2010; Nakayama et al., 2007). Anderson et al. (2016) assessed 40 sites across 10 active Air Force installations throughout the continental United States and Alaska between March and September 2014. Installations were included if there was known historic AFFF release in the period 1970–

1990. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. The selected sites were not related to former fire training areas and were characterized according to volume of AFFF release—low (n = 2), medium (n = 32), and high (n = 2). PFNA was detected only at medium-volume release sites (11.4% detection frequency; mean concentration of 15,400 ng/L). Across all 36 sites, the detection frequency was 36.00% with a median concentration of detects of 96 ng/L. Authors noted that given PFNA is not present in 3M AFFF formulations, there may be some degree of telomer-based AFFF contamination. Post et al. (2013) evaluated raw surface water samples from 12 public drinking water system intakes collected between August 2009 and February 2010. Six rivers and six reservoirs, including two reservoirs in Atlantic County near a civil-military airport with possible AFFF use, were selected to represent New Jersey geographically. PFNA was below the minimum RL (5 ng/L) in all six river samples. In reservoir samples, PFNA was detected in 67% of samples (n = 6) at a maximum level of 19 ng/L. At the two reservoir sites near the civil-military airport, PFNA was below the minimum RL (5 ng/L) at one, and the other found PFNA at 5 ng/L. Two studies from Nakayama et al. (2010; 2007) assessed surface water samples from the Upper Mississippi River, Missouri River, and Cape Fear River Basins. In Nakayama et al. (2010), a large-scale evaluation of the Upper Mississippi River Basin and portion of the Missouri River Basin was conducted to provide preliminary PFC data given the importance of the two basins in supplying drinking water. Between the two basins, 173 samples were collected across 88 sampling sites in March–August 2008 by several different agencies—Minnesota Pollution Agency, Wisconsin Department of Natural Resources, Illinois Environmental Protection Agency, and the EPA Region 7 Water Quality Monitoring Team. Overall, the detection frequency of PFNA was 87% with a median concentration of 0.71 ng/L. Authors reported higher PFC concentrations adjacent to chemical manufacturers, downstream of WWTPs receiving waste from those types of manufacturers, and near an airport with historic use of firefighting foams. In Nakayama et al. (2007), one hundred surface water samples were taken from 80 sites selected to reflect water quality throughout the basin. PFNA was detected in 89.9% of samples with mean and median concentrations of 33.6 and 5.70 ng/L, respectively. The highest concentrations were found in the middle reaches of the Cape Fear River and its two major tributaries. The authors noted possible sources of PFCs to the basin included firefighting foam from nearby air force bases and commercial/industrial facilities.

Three studies conducted in the U.S. examined surface water near or downstream of land application sites where PFC-contaminated WWTP effluent or biosolids were applied (Lasier et al., 2011; Lindstrom et al., 2011; Konwick et al., 2008). In Lindstrom et al. (2011), authors analyzed surface water samples from ponds and streams in Decatur, Alabama. The samples were collected in February 2009 from farms that had applied PFC-contaminated biosolids to local agricultural fields as a soil amendment for at least 12 years. The biosolids were obtained from a local municipal WWTP where authors noted that sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers, fluorocarbon fibers, polymers, polymer films, and resins, although specific sources could not be characterized. PFNA was detected in 28% of samples (n = 32), with levels ranging from below the LOQ (10 ng/L) to 286 ng/L. The remaining two studies (Lasier et al., 2011; Konwick et al., 2008) evaluated surface water upstream and downstream of a land application site (LAS) in Georgia, where treated WWTP effluent was sprayed. The WWTP processed effluents from multiple carpet manufacturers who were reported to use significant quantities of PFCs. Lasier et al. (2011) sampled along the Conasauga, Oostanaula, and Coosa Rivers during summer 2008; samples

included two sites upstream (sites 1 and 2) and six sites downstream (sites 3–8) of the LAS. Additionally, site 2 was downstream of a local airport, and site 4 was downstream of a manufacturing facility of latex and polyurethane back material—inputs for the carpet manufacturers. PFNA was below the MDL (0.005 ng/g) at both of the sites upstream. Mean concentrations for sites 3–8 were 35, 44, 17, 21, 20, and 21 ng/L, respectively. Authors reported highest concentrations downstream of the land application and backing-material sites and then decreased concentrations increasingly downstream as a result of dilution. Konwick et al. (2008) sampled from the Conasauga River near Dalton, Georgia in March 2006 at one location upstream, two downstream, and one at the LAS. Sampling was also conducted in January 2005 at one freshwater location in the Altamaha River, a river remote from the carpet industry, and from four different streams and ponds located in Dalton, Georgia. PFNA was detected at all sites. Mean PFNA concentrations for the four sites along the Conasauga River were 32.8, 201.6, 369, and 284 ng/L for the upstream, LAS site, and two downstream locations, respectively (n = 5 at each site), with a pattern of increasing concentration with distance downstream of the LAS before a decrease in concentration at the final site. Authors suggested sorption to sediments, particularly organic carbon, as a possible reason for the decrease in PFNA concentration at the final site. At the single freshwater site in the Altamaha River, PFNA was detected in one of three samples at a concentration of 0.6 ng/L. The range of PFNA concentrations in ponds and streams near Dalton were: 11.1–12.2, 40.6–41.0, 4.8–6.3, and 2.1–2.5 ng/L for sites 1, 2, 3, and 4, respectively (n = 2 at each site).

Three studies evaluated surface water potentially impacted by wastewater (Boone et al., 2019; Subedi et al., 2015; Appleman et al., 2014). Boone et al. (2019) evaluated 17 PFAS in source and treated waters collected in 2010–2012. Authors attempted to select locations with known or suspected sources of wastewater in the source water, but ultimately the site selection was dependent upon the willingness of DWTPs to participate. The study did not differentiate which locations had known or suspected sources. Of the 22 surface water sources evaluated (16 river and 6 lake/reservoir), PFNA was detected in all samples (n = 22), with a mean concentration of 2.93 ng/L. Subedi et al. (2015) collected 28 lake water samples from 3 sampling events in August–September 2012 and four sampling events in May–September 2013 from Skaneateles Lake. Sites were selected to be along the shoreline of homes that use an enhanced treatment unit for onsite wastewater treatment. Wastewater effluents were identified as a source of contamination to the lake. PFNA was detected in 57% of samples with mean and median concentrations of 0.36 and 0.26 ng/L, respectively. Appleman et al. (2014) assessed source water from 11 utilities in Alaska, Alabama, Colorado, Nevada, New Jersey, Ohio, Oklahoma, and Wisconsin from August 2011 to May 2012, the majority of which were selected because they were either known from previous monitoring or expected to contain detectable PFAS because they were impacted by upstream wastewater effluent discharge. Authors evaluated the utilities and their effectiveness for removing PFAS. The study did not report an average concentration for PFNA, but PFNA was detected in 14 of 25 samples (from 7 of 11 utilities) with a maximum concentration of 5.7 ng/L.

In two studies, surface water samples were collected from locations with potential sources of PFAS that were not related to AFFF use (Procopio et al., 2017; Zhang et al., 2016). Procopio et al. (2017) evaluated samples collected between September 2011 and July 2014 from the Metedeconk River. Eight sampling events were conducted as part of a source trackdown study to identify potential sources of PFAS contamination after elevated PFOA levels were discovered at

a raw surface water intake of the Brick Township Municipal Utilities Authority. In all 56 samples, PFNA was below the minimum laboratory RL (5 ng/L). Zhang et al. (2016) conducted analyses to determine major sources of surface water PFAS contamination. Freshwater sample collection sites included 22 sites in the state of Rhode Island (sampled June 2014) and 6 sites in the New York Metropolitan Area (sampled October 2014). Surface water sites were creeks and rivers in urban and rural locations. PFNA was detected at all sites with a mean of 1.914 ng/L (n = 28). Authors identified potential PFAS sources at these sites to be metal coating plating; paint, coating, adhesive manufacturing; paper manufacturing; petroleum coal products manufacturing; printing activity; printing ink manufacturing; semiconductor manufacturing; sewage treatment; textile mills; waste management including landfills, and airports.

Of the remaining four U.S.-based studies (Boone et al., 2014; Quiñones and Snyder, 2009)|(Pan et al., 2018; Kim and Kannan, 2007), Boone et al. (2014) analyzed PFCs in water samples collected from the surface of the Mississippi River at a low flow level (2.95 ft) in September 2010 and a high flow level (8.32 ft) in June 2009. PFNA levels were 1.800 and 3.30 ng/L at low and high flow levels, respectively. In Quiñones and Snyder (2009), surface water samples from the Boulder Basin, Hoover Dam, and the lower Colorado River were collected in 2008. Mean PFNA levels at all sites were below the method RL (1.0 ng/L). Kim and Kannan (2007) sampled two urban lakes in Albany, New York during five sampling trips from February–November 2006. The lakes, Washington Park Lake and Rensselaer Lake, are located in downtown Albany and receive surface runoff from nearby roadways and residential areas during stormwater runoff. PFNA was detected in Washington Park Lake (n = 6) at mean and median concentrations of 1.99 and 2.14 ng/L, respectively, and in Rensselaer Lake (n = 5) at mean and median concentrations of 1.35 and 1.45 ng/L, respectively. Overall, PFNA was detected in 81.8% of the 11 total samples. Finally, in a multicontinental study, Pan et al. (2018) assessed surface water samples from several countries including the United States (Delaware River), United Kingdom (Thames River), Germany and the Netherlands (Rhine River), and Sweden (Mälaren Lake). Twelve samples were collected in September–December 2016 along the Delaware River that spanned seven cities—Trenton, Bristol, Philadelphia, Chester, Delaware, Smyrna, and Frederica. Authors noted that all sampling sites were along the main stream of the rivers and not proximate to known point sources of any fluorochemical facilities. PFNA was detected in all samples from the Delaware River with a mean concentration of 2.51 ng/L and were similar to levels found in the Thames River.

The EPA also identified studies from Canada and Europe reporting the occurrence of PFNA in surface water, which are briefly summarized below. Detailed results from each study are presented in Table C-2. Most Canadian and European studies reported PFNA detected in at least one surface water sample site in each study. Concentrations of PFNA in creek and river samples measured throughout Canada ranged from <125 pg/L to 3,000 pg/L (D'Eon J et al., 2009 as cited in NCBI, 2022). Also, PFNA concentrations ranged from 0.80 ng/L to 2.4 ng/L in surface water samples collected from Highland Creek watershed, Canada in 2010 (Meyer et al., 2011 as cited in NCBI, 2022). Concentrations of PFNA in lake water samples collected from four lakes on Cornwallis Island, Canada from 2003 to 2005 ranged from not detected to 6.1 ng/L (Stock et al., 2007 as cited in NCBI, 2022).

Several studies in Europe sampled surface water from sites in proximity to fluoropolymer facilities or in locations with current or past AFFF usage. Four studies investigated surface water

in proximity to fluoropolymer facilities. One study in France (Boiteux et al., 2017) reported no detections, but the other three, from France, the Netherlands, and Italy, respectively reported at least one site or sampling event with detections from 0.49 ng/L to 2,637 ng/L (Bach et al., 2017; Gebbink et al., 2017; Valsecchi et al., 2015). In the study by Bach et al. (2017), the maximum level detected of 2,637 ng/L was from a sampling area in a river downstream from two facilities that produce fluoropolymers. Four additional studies included sampling locations potentially impacted by current or past AFFF usage (Mussabek et al., 2019; Skaar et al., 2019; Gobelius et al., 2018; Dauchy et al., 2017). The reported PFNA detection frequencies ranged from 0% to 89% with varying maximum levels from 1.81 ng/L to 26 ng/L.

Many European and Canadian studies sampled sites where there was no known PFAS manufacturing or use of PFAS-heavy materials (e.g., AFFF). In some cases, PFNA was not detected or concentrations were not reported (Barreca et al., 2020; Yeung et al., 2017; Lescord et al., 2015; Jurado-Sánchez et al., 2013; Villaverde-de-Sáa et al., 2012; Möller et al., 2010) , and in other cases there was 100% (or near 100%) detection frequency of PFNA (Zhao et al., 2015; Eschauzier et al., 2012; Kovarova et al., 2012; Labadie and Chevreuil, 2011; Ahrens et al., 2009a). Maximum PFNA concentrations reported among all PFNA detections in these studies ranged from low (e.g., 0.18 ng/L in Zhao et al. (2015)) to very high (e.g., 8,100 ng/L in Kovarova et al. (2012)). Several studies with high maximum detections of PFNA used sampling locations near or potentially near wastewater treatment plants or other industrial activity (Wilkinson et al., 2017; Lorenzo et al., 2015; Boiteux et al., 2012; Llorca et al., 2012). The remaining studies did not report details on the area surrounding sampling locations and how nearby activities may have impacted the results (Ciofi et al., 2018; Munoz et al., 2018; Loos et al., 2017; Shafique et al., 2017; Eriksson et al., 2013; Veillette et al., 2012; Ahrens et al., 2009b; Rostkowski et al., 2009; Ericson et al., 2008b); the detected concentrations in these studies ranged from 0.057 ng/L to 9.89 ng/L.

Table C-2. Summary of Studies Reporting the Occurrence of PFNA in Surface Water

Study	Location	Site Details	Results
United States			
Galloway et al. (2020)	United States (Ohio; West Virginia)	Rivers and tributaries near a fluoropolymer facility sampled throughout three trips on June, July, and December 2016. In June 2016, samples were collected on a 188 km stretch of the Ohio River, from 130 km downstream to 58 km upstream of the facility, and tributaries that pass near known PFAS-containing landfills. In July 2016, samples were collected from lakes, rivers, and creeks to the north and northeast of the facility as far as 16 km downwind. The December 2016 trip expanded the collection radius to more than 48 km downwind to the north and northeast of the facility.	June 2016: n = 26; DF[a] 92%* *PFNA was detected but below the LOQ in 24 samples, and ND in 2 samples July 2016: n = 25; DF[a] 0% December 2016: n = 40; DF[a] 80%*, range = <LOQ–24.2 ng/L *PFNA was detected above the LOQ in 1 sample, detected but below the LOQ in 31 samples, and ND in 8 samples (LOQ = 10 ng/L)
Newsted et al. (2017)	United States (Minnesota)	Ten sampling reaches (three samples each) spanned 3 miles of the Mississippi River within Pool 2, an area between Ford Dam (between Minneapolis and St. Paul) and Hastings Dam (near Hastings) and subject to ongoing PFAS reduction efforts by the Minnesota Pollution Control Agency for 10–15 years. Surface water samples were collected in August 2011.	Upstream of 3M Cottage Grove facility: n = 27, DF 0% Downstream of 3M Cottage Grove facility: DF[a] 33%, range = <LOQ–2.0 ng/L (LOQ = 2.0 ng/L)
Newton et al. (2017)	United States (Decatur, Alabama)	Samples collected in October 2015 from nine sites along the Tennessee River. Three sites were downstream of facilities that manufacture or use fluorinated materials and three sites were upstream. Among the upstream sites, three were also upstream of a WWTP.	Upstream: n = 6, DF 0% Downstream: n = 3, DF 0% (LOQ = 10 ng/L)
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between 1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were	Overall: n = 36, DF 36.00%, median (maximum) = 96 (10,000) ng/L *Breakdown by site:* Emergency Response (low-volume release): n = 2, DF 0% Hangars and Buildings (medium-volume release): n = 32, DF 11.4%, mean (range) = 15,400 (480–59,000) ng/L Testing and Maintenance (high-volume release):

Study	Location	Site Details	Results
		not specifically controlled for in the site selection process; active remedies had not been applied at any of the sites selected. Approximately ten samples were collected between March and September 2014 at each site; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases. Surface water sample locations included engineered storm water channels, engineered AFFF ponds, and natural streams.	n = 2, DF 0% (median reporting limit = 17 ng/L) *Median calculated using quantified detections *Non-detects were substituted with ½ the reporting limit
Post et al. (2013)	United States (New Jersey)	Raw water collected from 12 public drinking water system intakes between August 2009 and February 2010 from 6 rivers and 6 reservoirs. Sites were chosen to represent NJ geographically and included two reservoir sites near a civil-military airport with possible AFFF use.	Overall: n = 12, DF 33%, range = ND–19 ng/L Rivers: n = 6, DF 0% Reservoirs: n = 6, DF 67%, range = <5–19 ng/L (minimum reporting limit = 5 ng/L)
Nakayama et al. (2010)	United States (Illinois; Iowa; Minnesota; Missouri; Wisconsin)	Eighty-eight sampling sites collected between March and August 2008 from tributaries and streams in the Upper Mississippi River Basin and a portion of the Missouri River Basin. Samples were collected by the Minnesota Pollution Agency, Wisconsin Department of Natural Resources, Illinois Environmental Protection Agency, and U.S. EPA Region 7 Water Quality Monitoring Team. Each agency selected sampling sites with the intention of providing preliminary PFC data to the individual regions. Sampling sites included locations adjacent to chemical manufacturers, downstream of WWTPs receiving waste from those types of manufacturers, and near an airport with historic use of firefighting foams.	n = 173, DF 87%, median (range) = 0.71 (ND–72.9) ng/L (LOD = 0.02 ng/L) *ND data points were substituted with LOD/sqrt(2) = 0.014 ng/L
Nakayama et al. (2007)	United States (North Carolina)	Eighty sampling sites in river basin during spring 2006. The sites were selected to reflect water quality throughout the basin. Possible sources of PFCs include use of firefighting foam from Fort Bragg and Pope Air Force Base, metal-plating facilities, textile, and paper production, and other industries.	n = 100, DF 89.9%, mean, GM, median (range) = 33.6, 9.73, 5.70 (<LOQ–194) ng/L (LOQ = 1 ng/L, LOD = 0.05 ng/L) *Values below the LOQ were excluded from the calculation of the mean and GM
Lindstrom et al. (2011)	United States (Decatur, Alabama)	Samples collected in February 2009 from ponds and streams located on farms with historical land application of PFC-contaminated biosolids to local	n = 32, DF (frequency of quantification)[b] 28%, range = <LOQ–286 ng/L (LOQ = 10 ng/L)

Study	Location	Site Details	Results
		agricultural fields between 1995 and 2008. Biosolids obtained from local municipal WWTP where sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers, fluorocarbon fibers, polymers, polymer films, and resins.	
Lasier et al. (2011)	United States (Georgia)	Upstream (sites 1 and 2) and downstream (sites 3–8) of a land application site were sampled along the Conasauga, Oostanaula, and Coosa Rivers during summer 2008, where effluents from carpet manufacturers (suspected of producing wastewaters containing perfluorinated chemicals) are processed at a WWTP and the treated WWTP effluent is sprayed onto the site. Additionally, site 2 was downstream of a local airport and site 4 was downstream of a manufacturing facility for latex and polyurethane backing material.	Upstream: Sites 1 and 2: DF 0% Downstream: Site 3: DF NR, mean = 35 ng/L Site 4: DF NR, mean = 44 ng/L Site 5: DF NR, mean = 17 ng/L Site 6: DF NR, mean = 21 ng/L Site 7: DF NR, mean = 20 ng/L Site 8: DF NR, mean = 21 ng/L (MDL = 0.005 ng/g, LOQ = 0.010 ng/g) *Half of the MDL was used when measured concentrations were below the MDL *Concentrations measured in triplicate samples
Konwick et al. (2008)	United States (Georgia)	Samples collected in March 2006 from the Conasauga River near Dalton, a major carpet manufacturing city, where there is high use of PFAAs in the carpet industry. Four sites on the Conasauga River were sampled: one location upstream, two downstream, and one at the site of a land application system where treated wastewater (approximately 87% industrial source) is sprayed. One freshwater site on the Altamaha River, a reference site away from Dalton, was sampled in January 2005. Four ponds and streams in Dalton were also sampled in January 2005.	Conasauga River: Site 1 (upstream): n = 5, DF[a] 100%, mean (range) = 32.8 (12.3–75.4) mg/L Site 2 (at site): n = 5, DF[a] 100%, mean (range) = 201.6 (136–248) mg/L Site 3 (downstream): n = 5, DF[a] 100%, mean (range) = 369 (280–456) mg/L Site 4 (downstream): n = 5, DF[a] 100%, mean (range) = 284 (190–366) mg/L Altamaha River: Site 1 (freshwater): n = 3, DF[a] 33%, range = <0.6–0.6 ng/L Dalton streams/ponds: Site 1: n = 2, DF[a] 100%, range = 11.1–12.2 ng/L Site 2: n = 2, DF[a] 100%, range = 40.6–41.0 ng/L Site 3: n = 2, DF[a] 100%, range = 4.8–6.3 ng/L

Study	Location	Site Details	Results
			Site 4: n = 2, DF[a] 100%, range = 2.1–2.5 ng/L (LOD = 0.6 ng/L)
Boone et al. (2019)	United States (unspecified)	Twenty-two surface waters (16 rivers and 6 lakes/reservoirs) used as source waters for 22 DWTPs, collected in 2010−2012; some locations with known or suspected sources of wastewater in the source water, but study did not differentiate which locations had known or suspected sources.	n = 22, DF[a] 100%, mean[a] (range) = 2.93 (0.117–41.4) ng/L (LCMRL = 0.094 ng/L)
Subedi et al. (2015)	United States (New York)	Lake water along the shoreline of residences that use an enhanced treatment unit for onsite wastewater treatment; samples were collected ~40 ft from the lakeshore about 2 ft below surface. Sampling occurred August–September 2012 (three sampling events) and May–September 2013 (four sampling events). Wastewater effluents identified as source of contamination.	n = 28, DF[a] 57%, mean, median (range) = 0.36, 0.26 (ND–1.21) ng/L (LOQ = 0.2 ng/L) *Data points <LOQ were substituted with ½ LOQ and NDs were substituted with zero
Appleman et al. (2014)	United States (Wisconsin; Oklahoma; Alaska; Alabama; Colorado; Ohio; Nevada; New Jersey)	Surface water source water for 11 DWTPs, sampled August 2011 to May 2012; majority of the utilities selected because they were either known from previous monitoring or expected based on their source waters to contain detectable PFAS (i.e., impacted by upstream wastewater effluent discharge). Each site was sampled between one and four times.	n = 25, DF[a] 56%, range = <0.5–5.7 ng/L (Method reporting limit = 0.5 ng/L)
Procopio et al. (2017)	United States (New Jersey)	Surface water from the Metedeconk River, where there was suspected illicit discharge to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs. Majority of samples collected from a 4.0 km (2.5 mile) reach of the South Branch Metedeconk River, although samples from the North Branch were also collected in sampling event 1. Samples were collected in eight sampling events in September and December 2011; February, July, September, and December 2012; June and August 2013; and June and July 2014.	n = 56, DF 0% (Minimum reporting limit = 5 ng/L)
Zhang et al. (2016)	United States (Rhode Island; New York)	River and creek samples from 22 sites in Rhode Island collected in June 2014 and from six sites in the NY Metropolitan Area collected in October 2014. Both urban and rural locations were sampled.	Overall: n = 28, DF[a] 100%, mean[a] (range) = 1.914 (0.104–13.986) ng/L

Study	Location	Site Details	Results
			Rhode Island: n = 22, DF[a] 100%, mean[a] (range) = 1.868 (0.104–13.986) ng/L Urban: n = 10, DF[a] 100%, mean[a] (range) = 2.576 (0.308–13.986) ng/L Rural: n = 12, DF[a] 100%, mean[a] (range) = 1.278 (0.104–7.235) ng/L New York Metropolitan Area (all urban sites): n = 6, DF[a] 100%, mean[a] (range) = 2.084 (0.151–6.658) ng/L (LOD = 0.04 ng/L)
Boone et al. (2014)	United States (New Orleans, Louisiana)	Surface samples from the Mississippi River collected in June 2009 when the river was at a high flow level (8.32 ft) and in September 2010 when the river was at a low flow level (2.95 ft).	Low flow (2.95 ft): mean based on a primary and duplicate sample = 1.800 ng/L High flow (8.32 ft): mean based on four replicates = 3.30 ng/L (DL = 0.047 ng/L, LCMRL = 0.110 ng/L)
Quiñones and Snyder (2009)	United States (Arizona; Nevada)	Samples collected in 2008 from three sites in Boulder Basin, one site in Hoover Dam, and two sites from the lower Colorado River. PFC occurrence had not been previously determined or reported for these sites.	n = 40, DF NR (Method RL = 1.0 ng/L) *Mean values at all sites were <Method RL; Figure 3 in the paper shows the maximum value at one lower Colorado River site is >Method reporting limit
Kim and Kannan (2007)	United States (Albany, New York)	Samples collected from two urban lakes—Washington Park and Rensselaer Lake—during five sampling trips from February–November 2006. Both lakes are located in downtown Albany and receive surface runoff from nearby roadways and residential areas during stormwater runoff.	Total: n = 11, DF 81.8%, mean, median (range) = 1.70, 1.63 (ND–3.51) ng/L Washington Park Lake: n = 6, DF NR, mean, median (range) = 1.99, 2.14 (<LOQ–3.51) ng/L Rensselaer Lake: n = 5, DF NR, mean, median (range) = 1.35, 1.45 (ND–2.73) ng/L (LOQ = 0.25 ng/L) *Non-detects were set to zero; values below the LOQ were set to ½ LOQ
Canada			
Lescord et al. (2015)	Canada (Cornwallis Island, Nunavut)	Six lakes (Meretta, Resolute, Char, Small, North, and 9 Mile) located on Cornwallis Island and near the Inuit community of Resolute Bay were sampled weekly in July to August in 2010 and biweekly in July to August 2011. Two lakes (Meretta and Resolute) are	Meretta: n = 5, DF >0% Resolute: n = 5, DF >0% *Concentrations and summary statistics were not reported in tables or text; Figure 2 shows non-zero concentrations of PFNA in Meretta

Study	Location	Site Details	Results
		approximately 0.5 km downstream from a local airport, where wastewater from the airport and military base was discharged with little treatment into the Meretta catchment from 1949 to 1998.	and Resolute. Concentrations in Char, Small, North, and 9 Mile lakes are unclear (method detection limit = 0.0018 ng/L)
Yeung et al. (2017)	Canada (Ontario)	River water samples were collected from Mimico Creek and Rouge River in November 2014 and analyzed using two methods. Reference method used ultra performance liquid chromatograph and a newly developed method used ultra performance convergence chromatograph for separation.	Results presented as reference method, new method: Mimico: n = 1, point = <2, <5 ng/L Rouge: n = 1, point = <2, <5 ng/L (MLOQ for reference method not reported; MLOQ for new method reported in Table S3 as 200 ng/L but based on Table S7, this should likely be 2 ng/L) *Results in Table S7 are presented in ng/L but based on a comparison to Figure 4, Table S7 should be in ng/mL
Veillette et al. (2012)	Canada (Ellesmere Island, Nunavut)	Lake catchment area located on the northwest coast of the island. Surface water was collected from the center of the lake, the littoral zone (30 m from the delta), the delta, and lake inflow and outflow in July 2007, May 2008, and August 2008. Samples were collected at depths of 2 m (underneath the ice cover), 10 m (the bottom of the mixed layer), and 32 m (in the monimolimnion).	n = 11, DF[a] 100%, mean (range) = 0.118 (0.057–0.192) ng/L (method detection limit = 0.009 ng/L)
Europe			
Bach et al. (2017)	France (southern)	Grab water samples were collected from six locations along the shore of a river in April, July, October, and December 2013. The river selected for the study receives effluent at three points along the river from an industrial site where two facilities produce fluoropolymers. The first facility has been active since the 1960s, with production including PTFE synthesis from the beginning of the 1960s to 1985 with PFOA as a processing aid; more recently, PVDF has been synthesized since the early 1970s with fluorotelomer sulfonic acid (6:2 FTSA) or PFNA as a processing aid. The second facility, established in 2002, produced fluoropolymers with PFOA as a processing aid until 2008 when it was replaced with PFHxA. Samples were collected starting ~1.3 km upstream from the industrial site and covered ~15 km of the river.	Upstream: Sampling point #1: n = 1, point = <4 ng/L for April, July, October, and December 2013 Downstream: Sampling point #2: n = 1, point = 209, 2,637, 15, and 12 ng/L for April, July, October, and December 2013 Sampling point #3: n = 1, point = <4, <4, <4, and <4 ng/L for April, July, October, and December 2013 Sampling point #4: n = 1, point = 9, 42, <4, and <4 ng/L for April, July, October, and December 2013

Study	Location	Site Details	Results
		Samples (point #1 to #6) were collected from upstream to downstream.	Sampling point #5: n = 1, point = <4, <4, <4, and <4 ng/L for April, July, October, and December 2013 Sampling point #6: n = 1, point = <4, 87, <4, and <4 ng/L for April, July, October, and December 2013 (LOQ = 4 ng/L)
Boiteux et al. (2017)	France (northern)	Grab water samples were collected from seven locations along a river in May, July, October, and December 2013. The river selected for the study receives wastewater from an industrial WWTP that treats raw sewage coming from a facility that manufactures fluorotelomer-based products and side-chain-fluorinated polymers used in firefighting foams and stain repellents. Samples were collected starting ~1.2 km upstream of the WWTP discharge and encompassed ~65 km of the river. Samples were collected from upstream to downstream.	Upstream: Sampling point #1: n = 1, DF 0% for May, July, October, and December 2013 Downstream: Sampling points #3, 4, 5, 7, 9, 11: n = 1, DF 0% for May, July, October, and December 2013 (LOQ = 4 ng/L) *DF represents frequency of quantification
Gebbink et al. (2017)	The Netherlands (Dordecht)	River water samples collected in October 2016 at sites downstream (R1−R13) and upstream (R14−R16) of the Dordrecht fluorochemical production plant. Samples (R17−R18) were also collected from different waterbodies at control sites.	Control sites: n = 2, DF[a] (frequency of quantification) 100%, mean[a] (range) = 0.9 (0.8–1.0) ng/L Upstream: n = 3, DF[a] (frequency of quantification) 100%, mean[a] (range) = 0.75 (0.54–0.92) ng/L Downstream: n = 13, DF[a] (frequency of quantification) 100%, mean[a] (range) = 0.67 (0.49–1.0) ng/L (minimum quantification level = 0.03 ng/L)
Valsecchi et al. (2015)	Italy (River Basins Po, Brenta, Adige, Tevere, and Arno)	Five river basins were sampled between 2008 and 2013. Two river basins (Po and Brenta) receive discharges from two chemical plants that produce fluorinated polymers and intermediates; two river basins (Tevere and Adige) are not impacted by relevant industrial activities; and one river basin (Arno) has textile and tannery districts located along parts of the river. In total, 20 rivers were sampled at their basin closure stations. Rivers Arno, Tevere, and Po were also sampled along the course of the river.	Po: n = 105, DF[a] 64%, range = <LOD–70.3 ng/L Brenta: n = 5, DF[a] 40%, range = <LOD–1.4 ng/L Adige: n = 5, DF 0% Tevere: n = 7, DF 0% Arno: n = 19, DF[a] 95%, range = <LOD–30.1 ng/L (LOD = 0.5 ng/L)

Study	Location	Site Details	Results
Skaar et al. (2019)	Norway (Ny-Ålesund; Lake Linnévatnet area)	Freshwater samples were collected from Ny-Ålesund (research facility) in June 2016 and from the Lake Linnévatnet area (background site) in March 2014 and from April to June 2015. Surface water in Ny-Ålesund was contaminated by a local firefighting training site. Lake Linnévatnet receives water from meltwater of the adjacent glaciers and has few potential pollution sources.	Ny-Ålesund: Background: n = 7, DF 0% Contaminated: n = 3, DF[b] 67%, range = <0.02–1.81 ng/L Lake Linnévatnet: Background: n = 20, DF[b] 65%, range = <0.03–0.16 ng/L (LOD = 0.021 ng/L; LOQ = 0.085 ng/L)
Mussabek et al. (2019)	Sweden (Luleå)	Samples from a man-made lake and pond approximately 500 m southwest from a firefighting training facility at the Norrbotten Air Force Wing collected in October 2015. The training facility has been active since 1941 and has used PFAS-containing AFFFs in the last decades. The lake and pond lie above a groundwater reservoir with high permeable soil and were selected because they are isolated water bodies receiving PFAS contamination and can potentially impact groundwater.	Lake: n = 2, DF NR, mean = <0.5 ng/L Pond: n = 2, DF NR, mean = <0.5 ng/L (LOD = 0.5 ng/L)
Gobelius et al. (2018)	Sweden (national)	Sampling conducted between May and August 2015, with the majority in July and a few samples in September and November 2015. Samples were collected in 21 regional counties by the County Administration Boards. Sampling locations selected based on potential vicinity of PFAS hot spots (i.e., fire training sites, unspecific industry, sewage treatment plant effluent, landfill/waste disposal, skiing, and urban areas) and/or importance as a drinking water source. Sample numbers varied for each county and sampling sites were spread unevenly across Sweden. Surface water samples collected approximately 10 cm below the water surface.	n = 281, DF[a] 89%, range = <0.08–26 ng/L (MDL = 0.084 ng/L) *Two types of water (i.e., surface water and recipient water [surface water]) included
Dauchy et al. (2017)	France (unspecified)	Samples collected in the vicinity of three sites (B, C, D) where fluorosurfactant-based foams are or were being heavily used. Site B is an international civilian airport built in 1974. The exact location of the training area, frequency of training sessions, and history of firefighting training activities are unknown. In November 2014, surface water samples were collected in the only river running alongside the airport.	Site B: n = 5, DF 0% Site C: n = 9, DF 0% Site D: n = 2, DF 0% (LOQ = 4 ng/L)

Study	Location	Site Details	Results
		Downstream from the airport, this river joins two other rivers, which were also sampled. Site C is a military airport, with the exact location of the training area, frequency of the training sessions, and history of the firefighting training activities unknown. In April 2014, surface water samples were collected in several rivers surrounding the military base. Site D is a training center for firefighters. From 1969 to 1984, the site was an oil refinery. Starting in 1987, the site became a training area for firefighters, with exercises carried out directly on the soil. From the 1990s, some exercise areas were covered with concrete. In November 2014, two surface water samples were collected from the river receiving effluent from a WWTP at the site, one upstream and one downstream of the discharge pipe.	
Ciofi et al. (2018)	Italy (Tuscany)	Surface water samples were collected at 13 locations. Sampling year was not reported. SW-1: Arno river before the "Canale Maestro della Chiana" (Arezzo), receiving agricultural runoff and untreated urban wastewater SW-2: Arno river after the "Canale Maestro della Chiana" (Arezzo) SW-3: Arno river before entering in Florence SW-4: Arno river after the discharge of the Florence WWTP SW-5: Arno river after the confluence of the Bisenzio river SW-6: Arno river after the city of Empoli (Florence) SW-7: Arno river after receiving the WWTP effluent from the leather industrial district of Santa Croce (Pisa) SW-8: Arno river in the proximity of the mouth (Pisa) SW-9: Bisenzio river before the confluence with Arno river (Florence) SW-10: Serchio river in the proximity of the mouth (Lucca) SW-11: East area of the coastal lake "Massaciuccoli" (Lucca)	SW-1: n = 1, point = <0.23 ng/L SW-2: n = 1, point = <0.21 ng/L SW-3: n = 1, point = <0.25 ng/L SW-4: n = 1, point = <0.21 ng/L SW-5: n = 1, point = <0.22 ng/L SW-6: n = 1, point = <0.22 ng/L SW-7: n = 1, point = <0.20 ng/L SW-8: n = 1, point = <0.25 ng/L SW-9: n = 1, point = 2.7 ng/L SW-10: n = 1, point = <0.23 ng/L SW-11: n = 1, point = <0.19 ng/L SW-12: n = 1, point = <0.70 ng/L SW-13: n = 1, point = <0.27 ng/L (MDL = 0.19–0.27 ng/L; MQL = 0.70 ng/L) *MDL/MQL varied by sample. MDL provided for 11 of 13 samples; minimum quantitation level provided for 1 of 13 samples

Study	Location	Site Details	Results
		SW-12: West area of the coastal lake "Massaciuccoli" (Lucca) SW-13: Central area of the artificial lake "Bilancino" (Florence)	
Munoz et al. (2018)	France (Marnay-sur-Sein; Bougival; Triel-sur-Seine)	Surface water along the Seine River was collected during four sampling campaigns between September 2011 and December 2012, each conducted in a different season. For each campaign, two to four samples were collected over a one-month period. Three sampling sites were investigated: Marnay-sur-Sein, located 200 km upstream from Paris, was selected as a reference site, non-affected by the Greater Paris region; Bougival, situated 40 km downstream from Paris, was chosen to investigate the impact of Greater Paris on PFAS levels; Triel-sur-Seine, another 40 km further downstream, was selected to assess the global influence of the Paris urban area, including other inputs such as WWTPs.	n = 36, DF 97%, range = ND–2.0 ng/L (LOD = 0.02–0.3 ng/L for all PFAS) *PFNA concentrations were not reported in text or table by site but relative abundance by site is available in Figure 1
Wilkinson et al. (2017)	England (Greater London and southern England)	Three rivers selected due to their accessibility, receiving only STW effluent outfalls (i.e., no confluence with another major river in the study area) and accessibility of river headwater sampling for the Hogsmill and Blackwater Rivers. Each river sampled received inputs from at least one STW. Three STWs discharge into the Blackwater River, one STW discharges into the Hogsmill River, and one STW discharges to Chertsey Bourne River. Headwaters were evaluated for the Hogswill and Blackwater Rivers. Water samples were collected 50 m upstream and 250 m and 1,000 m downstream from STW effluent outfalls. In total, samples were collected on 3–4 separate occasions from 23 sites. Sampling dates were not reported.	Headwaters: n = 6, DF NR, mean = 2.75 ng/L Upstream: n = 19, DF NR, mean = 16.7 ng/L Downstream 250 m: n = 19, DF NR, mean = 32.5 ng/L Downstream 1,000 m: n = 19, DF NR, mean = 23.9 ng/L (LOD = 0.23 ng/L; LOQ = 0.75 ng/L)
Lorenzo et al. (2015)	Spain (Guadalquivir River Basin; Ebro River Basin)	Surface water was collected from the Guadalquivir River and its main tributaries and from the Ebro River and its main tributaries in October 2010. Guadalquivir sampling locations included downstream of WWTPs, near industrial areas, near a military camp, or through major cities; Ebro sampling locations included nearby ski resorts and downstream of WWTP and industrial areas.	Guadalquivir: n = 24, DF 8%, mean (range) = 5.1 (6.8–116.1) ng/L Ebro: n = 24, DF 8%, mean (range) = 0.5 (4.8–7.9) ng/L *Minimum reported is the lowest amount quantified

Study	Location	Site Details	Results
			*Mean was calculated with not detected concentrations as zeros (MQL = 0.4 ng/L)
Zhao et al. (2015)	Germany (Elbe River)	Four sampling campaigns conducted in February, April, August, and October 2011 to represent the four seasons. Freshwater samples (sites E619 to E689 with salinity <1 PSU) were collected at nine locations in the river Elbe. Some sampling sites were near Hamburg city and experienced occasional discharge of wastewater from industrial plants,	February: n = 8, DF[a] 100%, mean[a] (range) = 0.23 (0.16–0.43) ng/L April: n = 9, DF[a] 100%, mean[a] (range) = 0.18 (0.16–0.23) ng/L August: n = 9, DF[a] 100%, mean[a] (range) = 0.24 (0.19–0.36) ng/L October: n = 8, DF[a] 100%, mean[a] (range) = 0.15 (0.12–0.18) ng/L (MDL = 0.03 ng/L)
Boiteux et al. (2012)	France (national)	Raw water from rivers used as source water for DWTPs. Sites distributed across 100 French department to represent ~20% of the national water supply flow; samples collected during two sampling campaigns in July–September 2009 (first campaign) and June 2010 (second campaign – focused on sites from first sampling campaign that had PFC levels >LOQ). Some sites possibly affected by commercial/industrial releases.	Overall: n = 135, DF (frequency of quantitation) 2%, maximum = 52 ng/L *1st Sampling Campaign* n = 99, DF 5%, mean, median (maximum) = <1, <1 (4) ng/L *2nd Sampling Campaign* n = 36, results not reported (LOD = 1.3 ng/L, LOQ = 4 ng/L)
Eschauzier et al. (2012)	The Netherlands (Amsterdam)	Intake water from the Lek canal (n = 2) was collected in January and September 2010 to determine the behavior of PFAAs during the drinking water treatment processes. The Lek canal, a tributary of the river Rhine, is the source of drinking water for the city of Amsterdam and is downstream of an industrial point source in the German part of the Lower Rhine.	n = 2, DF[a] 100%, mean (range) = 0.6 (0.5–0.8) ng/L (LOQ = 0.24 ng/L)
Kovarova et al. (2012)	Czech Republic (Brno)	Seven locations in the Svitava and Svratka Rivers upstream and downstream of Brno, a city with highly developed chemical, engineering, textile, and food-processing industries. A sampler was installed at each site for 30 days twice a year (May and September 2008). Due to technical problems, samples were produced from only four of seven sites in May and from five of seven sites in September.	May: n = 4, DF[a] 100%, mean[a] (range) = 2,735 (240–5,700) ng/L September: n = 5, DF[a] 100%, mean[a] (range) = 2,690 (170–8,100) ng/L (LOD not reported)
Llorca et al. (2012)	Germany (Hesse), Spain (national)	Forty-eight surface river waters were sampled in 2010–2012 (24 from Spain and 24 from Germany).	Germany: n = 24, DF 0%

Study	Location	Site Details	Results
		Samples from Germany were collected from agriculturally or industrially influenced streams. Samples from Spain were collected from the Xúquer River Basin, Llobregat River Basin, and Ebro River Basin.	Spain: n = 24, DF 13%, mean, median (range) = 26, 20 (0.03–52) ng/L (MLOQ = 0.03 ng/L) *MLOQ reported above is from Table 3; Table 2 reports MLOD = 1.9 ng/L and MLOQ = 6.3 ng/L
Labadie and Chevreuil (2011)	France (Paris)	Samples collected weekly in January–May 2010 in an urban stretch of the River Seine at the Austerlitz Quay, downtown Paris during a flood cycle. The sampling station is under the influence of two major WWTPs and two major combined sewer overflow outfalls.	n = 16, DF 100%, mean, median (range) = 0.5, 0.5 (0.1–1.2) ng/L (LOQ = 0.17 ng/L)
Möller et al. (2010)	Germany (Rhine River watershed)	Raw freshwater samples collected in September–October 2008 along the River Rhine (stations 1–36) and major tributaries of the River Rhine (e.g., Rivers Neckar, Main, Rhur, stations 37–48). Along the River Rhine, samples were taken upstream and downstream of Leverkusen, where effluent of a WWTP treating industrial wastewater was discharged. All samples taken at a water depth ≤1 m.	n = 48, DF NR, authors noted that PFNA was quantified but results not provided (LOD = 0.014–1.60 ng/L for all PFAS)
Rostkowski et al. (2009)	Poland (national)	Inland surface water samples were collected at 12 locations in the southern part of Poland and 14 locations in the northern part of Poland in October and December 2004. Inland surface waters included rivers, lakes, and streams. The northern locations flowed through forested, agricultural, and rural areas; these areas are considered unpolluted with industrial chemicals. Some southern locations were near chemical industrial activities.	North: DF[a] 57%, range = <0.1–0.6 ng/L South: n = 11, DF[a] 36%, range = <0.1–0.6 ng/L (LOQ = 0.1–0.5 ng/L)
Barreca et al. (2020)	Italy (Lombardia Region)	Fifty-two surface water sampling stations (rivers and streams) throughout the region. Samples collected in 2018.	n = 286, DF[a] 6% (LOQ = 1 ng/L)
Loos et al. (2017)	Austria, Bulgaria, Croatia, Moldova, Romania, Serbia, Slovakia (Danube River and tributaries)	Samples were collected in August–September 2013 from 68 sites along a 2,581 km-stretch of the Danube River, with 14 of the sites in the mouths of tributaries or side arms. Three additional samples were also collected between the Iskar and Olt tributaries, the Olt River, and between the Siret and Prut tributaries. The investigated tributary rivers were the Morava (Austria/Slovakia), the Vah (Slovakia), the Drava	n = 71, DF 79%, mean, median (range) = 1.2, 1.1 (<LOQ–3.3) ng/L *Results <LOQ were replaced by zero (LOD = 0.29 ng/L, LOQ = 0.66 ng/L)

Study	Location	Site Details	Results
		(Croatia), the Tisa (Serbia), the Sava (Serbia), the Velika Morava (Serbia), the Timok (Serbia/Bulgaria), the Iskar (Bulgaria), the Olt (Romania), the Jantra (Bulgaria), the Russenski Lom (Bulgaria), the Arges (Romania), the Siret (Romania) and the Prut (Romania/Moldavia). Some sampling locations were downstream of major cities.	
Shafique et al. (2017)	Germany (River Elster; River Pleiße; River Saale; and River Elbe)	Surface water samples were collected from the River Elster, River Pleiße, River Saale, and River Elbe at the start of 2015.	Elster: n = 4, DF NR, mean = 0.96 ng/L Pleiße: n = 2, DF NR, mean = 1.65 ng/L Saale (Site A): n = 10, DF NR, mean = 6.57 ng/L Saale (Site B): n = 10, DF NR, mean = 9.89 ng/L Saale (Site C): n = 10, DF NR, mean = 0.83 ng/L Elbe: n = 2, DF NR, mean = 0.91 ng/L (MDL = 0.07 ng/L) *Values extracted from SI, which provides a more detailed breakdown of sites compared to that reported in the main text (where Elster and Pleiße sites were combined and Saale sites were combined)
Eriksson et al. (2013)	Denmark (Faroe Islands)	Grab samples collected in April–May 2012 from Lakes Leitisvatn, Havnardal, Kornvatn, and Á Mýranar.	Leitisvatn: n = 1, point = 0.16 ng/L Havnadal Lake: n = 1, point = 0.14 ng/L Kornvatn Lake: n = 1, point = 0.22 ng/L Á Mýranar: n = 1, point = 0.13 ng/L (LOD = 0.028 ng/L)
Jurado-Sánchez et al. (2013)	Spain (southeast)	Raw water samples were collected from a reservoir used as the source for tap water production and analyzed using a newly developed analytical method. Samples were collected in triplicate once a week in six different months at the intake of two different DWTPs. River water samples were also collected. Authors did not report sampling details or sampling year.	DWTP 1 intake: n = 3, DF[a] 0% DWTP 2 intake: n = 3, DF[a] 0% River water: n = NR, DF 0% (LOD = 0.1 ng/L)

Study	Location	Site Details	Results
Villaverde-de-Sáa et al. (2012)	Spain (Santiago de Compostela; Pontevedra)	Surface water samples were collected from Sar river in Santiago de Compostela on January 2011 and from Lérez river in Pontevedra on March 2011.	n = 3, DF 0% (LOD = 0.6 ng/L)
Ahrens et al. (2009a)	Germany (Hamburg; Laurenburg)	Nine samples collected from the river Elbe in Hamburg city (sites 16-18) and from Laurenburg to Hamburg (sites 19-24) in August 2006. Samples were collected at a water depth of 1 m. Dissolved and particulate phases were analyzed for each of the water samples.	Hamburg: Dissolved: n = 3, DF[a] 100%, mean (range) = 1.8 (1.7–2.0) ng/L Particulate: n = 3, DF[a] 67%, mean (range) = 0.040 (ND–0.088) ng/L Laurenburg to Hamburg: Dissolved: n = 6, DF[a] 100%, mean (range) = 1.7 (0.6–2.1) ng/L Particulate: n = 6, DF[a] 100%, mean (range) = 0.044 (0.003–0.074) ng/L (MDL = 0.045 ng/L for dissolved phase; 0.005 ng/L for particulate phase)
Ahrens et al. (2009b)	Germany (Elbe River)	Samples collected at 53 to 122 km (sites 1 to 9) upstream of estuary mouth of Elbe River in June 2007. *Only locations with conductivity <1.5 mS/cm were assumed to be freshwater and extracted	Site 1 (122 km): n = NR, DF NR, mean = 0.73 ng/L Site 2 (118 km): n = NR, DF NR, mean = 1.1 ng/L Site 3 (115 km): n = NR, DF NR, mean = 0.7 ng/L Site 4 (103 km): n = NR, DF NR, mean = 0.8 ng/L Site 5 (90 km): n = NR, DF NR, mean = 0.6 ng/L Site 6 (80 km): n = NR, DF NR, mean = 0.9 ng/L Site 7 (74 km): n = NR, DF NR, mean = 0.7 ng/L Site 8 (64 km): n = NR, DF NR, mean = 0.6 ng/L Site 9 (53 km): n = NR, DF NR, mean = 0.7 ng/L (MDL = 0.04 ng/L; MQL = 0.12 ng/L)
Ericson et al. (2008b)	Spain (Tarragona Province)	River water samples collected from the Ebro (at two points, Garcia and Mora), Francolí, and Cortiella Rivers in February 2007.	Ebro site 1: n = 1, point = 0.44 ng/L Ebro site 2: n = 1, point = 0.36 ng/L Francolí: n = 1, point = 0.64 ng/L

Study	Location	Site Details	Results
			Cortiella: n = 1, point = <0.42 ng/L (LOD = 0.42 ng/L)
Multiple Continents			
Pan et al. (2018)	United States (Delaware River)	Samples were collected from the Delaware River between September and December 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Trenton, Bristol, Philadelphia, Chester, Delaware, Smyrna, and Frederica.	n = 12, DF[a] 100%, mean, median (range) = 2.51, 2.36 (0.76–4.81) ng/L (MDL = 0.02 ng/L)
	United Kingdom (Thames River)	Samples were collected from the Thames River in October 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Oxford and London.	n = 6, DF[a] 100%, mean, median (range) = 1.18, 1.17 (0.77–1.71) ng/L (MDL = 0.02 ng/L)
	Germany and The Netherlands (Rhine River)	Samples were collected from the Rhine River in December 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities in Germany included Offenbach, Frankfurt, Goarshausen, Rheinbrohl, Bonn, Cologne, Leverkusen, Dormagen, Düsseldorf, Duisburg, Wesel, and Emmerich. Cities in The Netherlands included Arnhem, Lienden, Duurstede, Nijmegen, Wamel, and Zaltbommel.	n = 20, DF[a] 100%, mean, median (range) = 0.42, 0.39 (0.09–0.67) ng/L (MDL = 0.02 ng/L)
	Sweden (Mälaren Lake)	Samples were collected from Mälaren Lake in September 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Örebro and Stockholm.	n = 10, DF[a] 100%, mean, median (range) = 0.54, 0.54 (0.24–0.76) ng/L (MDL = 0.02 ng/L)

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; DWTP = drinking water treatment plant; ND = not detected; ng/L = nanogram per liter; PFAA = perfluoroalkyl acid; PFAS = per- and polyfluoroalkyl substances; NR = not reported; LOD = limit of detection; LOQ = limit of quantification; LCMRL = lowest concentration minimum reporting level; WWTP = wastewater treatment plant.

[a] The DF and/or mean was calculated using point data. Means were calculated only when DF = 100%.

C.2. RSC for PFNA, Literature Search and Screening Methodology

The EPA applies an RSC to the RfD when calculating an MCLG based on noncancer effects or for carcinogens that are known to act through a nonlinear mode of action to account for the fraction of an individual's total exposure allocated to drinking water (USEPA, 2000b). The EPA emphasizes that the purpose of the RSC is to ensure that the level of a chemical allowed by a criterion (e.g., the MCLG for drinking water) or multiple criteria, when combined with other identified sources of exposure (e.g., diet, ambient and indoor air) common to the population of concern, will not result in exposures that exceed the RfD. In other words, the RSC is the portion of total daily exposure equal to the RfD that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee tea or soup, as well as from transfer to dietary items prepared with drinking water) relative to other exposure sources; the remainder of the exposure equal to the RfD is allocated to other potential exposure sources. For example, if for a particular chemical, drinking water were to represent half of total exposure and diet were to represent the other half, then the drinking water contribution (or RSC) would be 50%. The EPA considers any potentially significant exposure source when deriving the RSC.

The RSC is derived by applying the Exposure Decision Tree approach published in the EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000b). The Exposure Decision Tree approach allows flexibility in the RfD apportionment among sources of exposure and considers several characteristics of the contaminant of interest, including the adequacy of available exposure data, levels of the contaminant in relevant sources or media of exposure, and regulatory agendas (i.e., whether there are multiple health-based criteria or regulatory standards for the contaminant). The RSC is developed to reflect the exposure to the U.S. general population or a sensitive population within the U.S. general population and may be derived qualitatively or quantitatively, depending on the available data.

A quantitative RSC determination first requires "data for the chemical in question… representative of each source/medium of exposure and… relevant to the identified population(s)" (USEPA, 2000b). The term "data" in this context is defined as ambient sampling measurements in the media of exposure, not internal human biomonitoring metrics. More specifically, the data must adequately characterize exposure distributions including the central tendency and high-end exposure levels for each source and 95% confidence intervals for these terms (USEPA, 2000b). Frequently, an adequate level of detail is not available to support a quantitative RSC derivation. When adequate quantitative data are not available, the agency relies on the qualitative alternatives of the Exposure Decision Tree approach. A qualitatively-derived RSC is an estimate that incorporates data and policy considerations and thus, is sometimes referred to as a "default" RSC (USEPA, 2000b). Both the quantitative and qualitative approaches recommend a "ceiling" RSC of 80% and a "floor" RSC of 20% to account for uncertainties including unknown sources of exposure, changes to exposure characteristics over time, and data inadequacies (USEPA, 2000b).

In cases in which there is a lack of sufficient data describing environmental monitoring results and/or exposure intake, the Exposure Decision Tree approach results in a recommended RSC of

20%. In the case of MCLG development, this means that 20% of the exposure equal to the RfD is allocated to drinking water and the remaining 80% is reserved for other potential sources, such as diet, air, consumer products, etc. This 20% RSC value can be replaced if sufficient data are available to develop a scientifically defensible alternative value. If scientific data demonstrating that sources and routes of exposure other than drinking water are not anticipated for a specific pollutant, the RSC can be raised as high as 80% based on the available data, allowing the remaining 20% for other potential sources (USEPA, 2000b). Applying a lower RSC (e.g., 20%) is a more conservative approach to public health and results in a lower MCLG.

C.2.1. Literature Search and Screening

In 2020, the EPA conducted a literature search to evaluate evidence for pathways of human exposure to eight PFAS chemicals (PFOA, PFOS, PFBA, PFBS, PFDA, perfluorohexanoic acid (PFHxA), PFHxS, and PFNA) (Holder et al., 2023). This search was not date limited and spanned the information collected across the Web of Science (WOS), PubMed, and ToxNet/ToxLine (now ProQuest) databases. The results of the PFNA literature search of publicly available sources are available through the EPA's Health & Environmental Resource Online website at https://hero.epa.gov/hero/index.cfm/project/page/project_id/2633.

The 2,408 literature search results for PFNA were imported into SWIFT-Review (Sciome, LLC, Research Triangle Park, NC) and filtered through the Evidence Stream tags to identify human studies and nonhuman (i.e., those not identified as human) studies (Holder et al., 2023). Studies identified as human studies were further categorized into seven major PFAS pathways (Cleaning Products, Clothing, Environmental Media, Food Packaging, Home Products/Articles/Materials, Personal Care Products, and Specialty Products) as well as an additional category for Human Exposure Measures. Nonhuman studies were grouped into the same seven major PFAS pathway categories, except that the Environmental Media category did not include soil, wastewater, or landfill. Only studies published between 2003 and 2020 were considered. Application of the SWIFT-Review tags identified 1,359 peer-reviewed papers matching these criteria for PFNA.

Holder et al. (2023) screened the 1,359 papers to identify studies reporting measured occurrence of PFNA in human matrices and media commonly related to human exposure (human blood/serum/urine, drinking water, food, food contact materials, consumer products, indoor dust, indoor and ambient air, and soil). For this synthesis, additional screening was conducted to identify studies relevant to surface water (freshwater only) and groundwater using a keyword[8] search for water terms.

Following the Population, Exposure, Comparator, and Outcome (PECO) criteria outlined in Table C-3, the title and abstract of each study were independently screened for relevance by two screeners using *litstream*^{TM}. A study was included as relevant if it was unclear from the title and abstract whether it met the inclusion criteria. When two screeners did not agree whether a study should be included or excluded, a third reviewer was consulted to make a final decision. The title and abstract screening of Holder et al. (2023) and of this synthesis resulted in 679 unique studies being tagged as relevant (i.e., having data on occurrence of PFNA in exposure media of interest)

[8] Keyword list: water, aquifer, direct water, freshwater, fresh water, groundwater, ground water, indirect water, lake, meltwater, melt water, natural water, overland flow, recreation water, recreational water, river, riverine water, riverwater, river water, springwater, spring water, stream, surface water, total water, water supply

that were further screened with full-text review using the same inclusion criteria. After additional review of the evidence collected by Holder et al. (2023), 98 studies originally identified for other PFAS also contained information relevant to PFNA. Based on full-text review, 171 studies were identified as having relevant, extractable data for PFNA from the United States, Canada, or Europe for environmental media, not including studies with only human biomonitoring data. Of these 171 studies, 156 were identified from (Holder et al., 2023), where primary data were extracted into a comprehensive evidence database. Parameters of interest included: sampling dates and locations, numbers of collection sites and participants, analytical methods, limits of detection and detection frequencies, and occurrence statistics. Fifteen of the 171 studies were identified in this synthesis as containing primary data on only surface water and/or groundwater.

The evidence database of Holder et al. (2023) additionally identified 18 studies for which the main article was not available for review. As part of this synthesis, 17 of the 18 studies could be retrieved. An additional three peer-reviewed references were identified through gray literature sources that were included to supplement the search results. The combined 20 studies underwent full-text screening using the inclusion criteria in Table C-3. Based on full-text review, five studies were identified as relevant.

Table C-3. Populations, Exposures, Comparators, and Outcomes (PECO) Criteria

PECO Element	Inclusion Criteria
Population	Adults and/or children in the general population and populations in the vicinity of PFAS point sources from the United States, Canada, or Europe
Exposure	Primary data from peer-reviewed studies collected in any of the following media: ambient air, consumer products, drinking water, dust, food, food packaging, groundwater[a], human blood/serum/urine, indoor air, landfill, sediment, soil, surface water[a] (freshwater), wastewater/biosolids/sludge
Comparator	Not applicable
Outcome	Measured concentrations of PFNA (or measured emissions from food packaging and consumer products only)

Notes: PFNA = perfluorononanoic acid.
[a] Surface water and groundwater were not included as relevant media in Holder et al. (2023). Studies were re-screened for these two media in this synthesis.

Using the screening results from the evidence database and this synthesis, a total of 176 studies were identified as relevant. Forty-seven of these contained information relevant to the United States and were summarized for this effort.

C.2.2. Additional Screening

The EPA also searched the following publicly available gray literature sources for information related to relative exposure of PFNA for all potentially relevant routes of exposure (oral, inhalation, dermal) and exposure pathways relevant to humans:

- ATSDR's *Toxicological Profiles;*
- CDC's national reports on human exposures to environmental chemicals;
- EPA's CompTox Chemicals Dashboard;

- EPA's fish tissue studies;
- EPA's Toxics Release Inventory;
- Relevant documents submitted under the Toxic Substances Control Act and relevant reports from the EPA's Office of Chemical Safety and Pollution Prevention;
- U.S. Food and Drug Administration's (FDA's) *Total Diet Studies* and other similar publications from FDA, U.S. Department of Agriculture, and Health Canada;
- National Oceanic and Atmospheric Administration's (NOAA's) National Centers for Coastal Ocean Science data collections;
- National Science Foundation direct and indirect food and/or certified drinking water additives;
- *Throwaway Packaging, Forever Chemicals: European wide survey of PFAS in disposable food packaging and tableware* (Straková et al., 2021);
- PubChem compound summaries;
- Relevant sources identified in the relative source contribution discussions (Section 5) of the EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA)/Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water; and*
- Additional sources, as needed.

The EPA has included available information from these gray literature sources for PFNA relevant to its uses, chemical and physical properties, and for occurrence in ambient or indoor air, foods (including fish and shellfish), soil, dust, and consumer products. The EPA has also included available information specific to PFNA below on any regulations that may restrict PFNA levels in media (e.g., water quality standards, air quality standards, food tolerance levels).

C.3. Summary of Potential Exposure Sources of PFNA Other than Water

C.3.1. Dietary Sources

C.3.1.1. Seafood

PFNA was detected in 108 of 157 fish tissue composite samples collected during the EPA's National Lake Fish Tissue Study, with a maximum concentration of 9.70 ng/g and a 50th percentile concentration of 0.32 ng/g (Stahl et al., 2014). It was detected in one of 162 fish tissue composite samples collected during the EPA's 2008–2009 National Rivers and Streams Assessment (NRSA) at a concentration of 2.48 ng/g (Stahl et al., 2014). More recently, PFNA was detected in 135 of 349 fish tissue composite samples at concentrations ranging from 0.100 ng/g to 1.910 ng/g in the EPA's 2013–2014 NRSA (USEPA, 2020a). PFNA was also detected in 119 of 152 fish tissue composite samples at concentrations ranging from 0.12 ng/g to 9.32 ng/g in the EPA's 2015 Great Lakes Human Health Fish Fillet Tissue Study (USEPA, 2021g). In 2001, PFNA was detected at mean concentrations of 1.0 ng/g, 0.57 ng/g, 2.8 ng/g, 2.9 ng/g, and 1.1 ng/g (wet weight) in whole body homogenates of lake trout collected from Lake Superior, Lake Michigan, Lake Huron, Lake Erie and Lake Ontario, respectively (Furdui et al., 2007 as cited in ATSDR, 2021 and NCBI, 2022). In addition, PFNA was detected in lake trout at concentrations of 0.70 ng/g for Lake Superior, 1.4 ng/g for Lake Huron, 2.6 ng/g for

eastern Lake Erie, and 0.90 ng/g for Lake Ontario; PFNA was also detected at a concentration of 1.2 ng/g in walleye collected from western Lake Erie (ATSDR, 2021; De Silva et al., 2011). PFNA was detected in mixtures of whole fish from the Missouri River, the Mississippi River, and the Ohio River at concentrations of 0.43 ng/g, 0.78 ng/g, and 1.03 ng/g, respectively (ATSDR, 2021; Ye et al., 2008). Concentrations of PFNA ranged from 0.01 ng/g to 0.73 ng/g in capelin whole body samples, <0.09 ng/g to 1.3 ng/g in cod muscle samples, and 0.05 ng/g to 8.0 ng/g in salmon muscle samples collected from the Hudson Bay region of northeast Canada in 1999 to 2003 (NCBI, 2022b; Kelly et al., 2009). PFNA was not included in NOAA's National Centers for Coastal Ocean Science, National Status and Trends Data (NOAA, 2022). Burkhard (2021) identified 79 studies reporting BAFs for PFNA and calculated a median (standard deviation) bioaccumulation factor (BAF) in muscle tissue/fillet of 144.54 ± 6.03 L/kg wet weight (reported as a logBAF of 2.16 ± 0.78 L/kg).

Among the peer-reviewed studies identified, there was considerable variability in sampling and analysis methodologies. Seafood products analyzed comprised a broad range of fish, crustaceans, and mollusks which were locally caught, farmed, obtained from local seafood markets, and/or obtained from large grocery chains. There was also considerable variability in sample preparation which might affect the interpretation and comparability of results. Some studies included tinned or prepared seafood while others focused only on unprepared or raw items; some studies composited many organisms into each analysis sample while others focused on single organism measurements; and some studies analyzed whole organisms while others measured only muscle tissue, potentially excluding fatty tissues likely to be higher in PFNA. Results from these studies are provided in detail in Table C-4.

Five U.S.-based studies were identified that evaluated PFNA levels in seafood (Young et al., 2022; Chiesa et al., 2019; Byrne et al., 2017; Young et al., 2013; Schecter et al., 2010). Four of these studies analyzed fish purchased from stores and fish markets. PFNA was detected infrequently in samples reported in Chiesa et al. (2019), Schecter et al. (2010) and Young et al. (2013): one of 10 samples of striped bass (1.4 ng/g) and in one of nine samples of shrimp (1.2 ng/g), but not in samples of crab meat, catfish, clams, cod, flounder, pangasius, pollock, tuna (including canned), salmon, scallops, tilapia, canned sardines, or frozen fish sticks. No other fish types were sampled in these three studies, and other than canned tuna and sardines, none were analyzed as prepared for eating. Seafood samples reported in Young et al. (2022) reported detectable PFNA in five out of the eight types of seafood evaluated. These included canned clams, canned tuna, cod, crab meat, and pollock (fish fillets and frozen fish sticks). No PFNA was detected in salmon, tilapia or shrimp. Seafood packaging was also evaluated for PFAS coatings, and it was determined the packaging did not contribute to any PFAS concentrations observed in the study.

One study evaluated fish samples collected directly from rivers and lakes (Byrne et al., 2017). As part of a study to assess exposure to PFNA and other PFAS among residents of two remote Alaska Native villages on St. Lawrence Island, Byrne et al. (2017) measured PFAS concentrations in stickleback and Alaska blackfish, resident fish used as sentinel species to detect accumulation of PFAS in the local environment. Stickleback were collected from three locations: Suqitughneq (Suqi) River watershed (n = 9 composite samples), Tapisaggak (Tapi) River (n = 2 composite samples), and Troutman Lake (n = 3 composite samples). Blackfish were collected from the Suqi River (n = 29) but were not found in the other water bodies. Authors reported that

the Suqi River watershed was upstream and downstream of a formerly used defense site and Tapi River was east of a military site, however at the start of the study none of the sites were known to be contaminated with PFAS. The sample dates were not reported. PFNA was not detected in the blackfish samples but was detected in 100%, 56%, and 50% of stickleback samples from Troutman Lake, Suqi River, and Tapi River, respectively, with authors noting that PFNA was the most frequently detected PFAS in stickleback. PFNA concentrations ranged between 2.72 ng/g and 4.13 ng/g ww at Troutman Lake, from below the limit of detection (LOD) to 1.52 ng/g ww at Suqi River, and <LOD–0.78 ng/g ww at Tapi River (LOD not reported; limit of quantitation (LOQ) = 0.5–1 ng/g ww). The authors reported that total PFAS levels were "exceptionally high" in Troutman Lake and hypothesized that stickleback were exposed to a local PFAS source and that contaminant may be leaching from village and military landfills.

The remaining four studies purchased seafood from stores and fish markets (Young et al., 2022; Chiesa et al., 2019; Young et al., 2013; Young et al., 2012; Schecter et al., 2010). Young et al. (2013) assessed fish and shellfish collected in 2010–2012 from retail markets across the continental United States. Retail markets in California, Florida, Illinois, Mississippi, New Jersey, New York, Tennessee, Texas, and Washington, D.C., were represented. Authors selected the 10 most consumed fish and shellfish in the United States that were farm raised, wild caught, or had unknown origin. Among the crab meat, shrimp, striped bass, catfish, clams, cod, flounder, pangasius, pollock, tuna, salmon, scallops, and tilapia, PFNA was only detected in one of nine samples of shrimp at a concentration of 1.2 ng/g and 1 of 10 samples of striped bass at a concentration of 1.4 ng/g. Young et al. (2022) evaluated fish and shellfish collected from retail markets in the Washington, D.C., metropolitan area, from March 2021 through May 2022. Some clam samples were also purchased online. Eight seafood products were selected that represented those in the top 10 types of seafood consumed in the United States. Seafood products were farm raised, wild caught, or of unknown origin. PFNA was detected in all clam (n = 10) and crab (n = 11) samples with concentrations ranging up to 796 ng/kg for clams and 350 ng/kg for crabs. Samples of cod (40%, n = 10), pollock (20%, n = 10) and canned/pouch tuna (30%, n = 10) also had detectable PFNA with concentrations of 45–103 ng/kg, 100–106 ng/kg and 44–77 ng/kg, respectively. Salmon, shrimp and tilapia did not have detectable levels of PFNA (MDL = 30–39 ng/kg). Schecter et al. (2010) evaluated PFNA and other PFAS in seafood collected from five Dallas, Texas, grocery stores in 2009. The origin or source of seafood was not described. Seafood included canned sardines in water, canned tuna, fresh catfish fillet, cod, frozen fish sticks, salmon, and tilapia (n = 1 composite sample for each seafood type). PFNA was not detected in any of the seafood samples. Finally, in a multicontinental study, Chiesa et al. (2019) collected salmon from a wholesale fish market in Milan, Italy; the sampling year was not reported. Wild-caught salmon samples originated from the United States (n = 7), Canada (n = 15), and Scotland (n = 2), while farmed salmon samples originated from Norway (n = 25) and Scotland (n = 17). Among the salmon that originated from the United States – Pacific Ocean (Food and Agriculture Organization Area (FAO) 67 and 77), two species – *Oncorhynchus kisutch* and *Oncorhynchus keta* – were analyzed, with PFNA not detected in either species (LOQ = 0.005 ng/g). PFNA was also not detected in wild-caught salmon from Canada and Scotland.

Results for all of the identified non-U.S. studies are presented in detail in Table C-4 and are summarized here. Among the non-U.S. studies, there were ten which provided PFNA measurements in seafood products identified as originating from the region of study. These

samples were wild caught, farmed, or purchased from local seafood markets and included seven species of fish caught off the coast of Iceland (Jörundsdóttir et al., 2014); three species of fish caught in lakes in Norway (Hansen et al., 2016); four fish species caught in Canadian Rivers (Bhavsar et al., 2014); rainbow trout collected from fish farms along the Swedish Baltic Sea coast (Johansson et al., 2014); shrimp, squid, mussels, and seven species of fish caught or farmed on the Greek coast (Vassiliadou et al., 2015); two species of fish originating from the West coast of Greenland (Carlsson et al., 2014); five species of fish collected from a river and lake in Germany (Hölzer et al., 2011); eight species of fish collected from commercially and recreationally important fishing areas across the Baltic Sea, a large freshwater Lake, and four fish farming facilities in Finland (Koponen et al., 2015); cod caught along the Polish Baltic Sea Coast (Falandysz et al., 2006); and two species of wild caught fish and one species of farmed fish from Faroe island area of Denmark (Eriksson et al., 2013). Two studies reported sampling in freshwater sources, one of which was likely to be contaminated with PFAS due to proximity to an airport with known AFFF usage (Hansen et al., 2016), and the other which had nearby industrial activities and previous monitoring results finding PFAS contamination (Bhavsar et al., 2014), leading the authors to expect elevated PFAS concentrations in fish captured from these sites. In both cases detectable levels of PFNA were measured in all fish sampled (Hansen et al., 2016; Bhavsar et al., 2014). The highest reported PFNA contents from Hansen et al. (2016) was 2.39 ng/g ww in brown trout muscle tissue, while the highest reported PFNA contents reported in Bhavsar et al. (2014) was 0.374 ng/g ww in fried Lake Trout. Among other European sites, there were often no organisms sampled with measurable concentrations of PFNA or it was present only in some organisms sampled.

Several studies from Europe provided PFNA measurements for seafood products purchased from supermarkets and other retailers. These samples were not identified as originating from the region of study and the product origin was often unknown or undisclosed. These included at least four fish species purchased in Germany (Hölzer et al., 2011); marine and freshwater fish purchased in Southern France (Yamada et al., 2014); fish purchased in the center region of France (Rivière et al., 2019); five species of fish purchased in Belgium, France, the Netherlands, and Portugal (Barbosa et al., 2018); fresh and processed fish samples purchased from major grocery store chains in Sweden (Gebbink et al., 2015; Vestergren et al., 2012); fish purchased in the Netherlands (Noorlander et al., 2011); and a variety of seafood products purchased in Spain (Domingo et al., 2012; Jogsten et al., 2009; Ericson et al., 2008a). Results from European market studies were similar to those of U.S. studies, with PFNA detected infrequently among the samples. However, several of the European seafood studies analyzed composites of all seafood products sampled rather than individual organisms, thus the results are less precise than in the U.S. studies.

Table C-4. Summary of PFNA Occurrence in Seafood

Study	Location and Source	Seafood Type	Results
United States			
Byrne et al. (2017)	United States (Alaska) Stickleback collected from three locations on St. Lawrence Island: Suqitughneq (Suqi) River watershed (upstream and downstream of a formerly used defense site), Tapisaggak (Tapi) River (located approximately 5 km east of military site), and Troutman Lake, a coastal lake situated adjacent to the village of Gambell. Alaska blackfish collected from the Suqi River but were absent from the other water bodies. Sampling year not reported. No sites were known to be contaminated with PFASs at the initiation of the study.	Stickleback and Alaska blackfish	**Stickleback:** **Troutman Lake: n = 3*, DF[a] 100%, mean[a] (range) = 3.43 (2.72–4.13) ng/g ww** **Suqi River: n = 9*, DF[a] 56%, range = <LOD–1.52 ng/g ww** **Tapi River: n = 2*, DF[a] 50%, range = <LOD–0.78 ng/g ww** Blackfish: n = 29, DF 0% (LOQ = 0.5–1 ng/g ww for all PFAS) *Number of composite samples, each composed of ~10 stickleback fish
Young et al. (2013)	United States (California; Illinois; Mississippi; Tennessee; Florida; New Jersey; New York; Texas; Washington, D.C.) Fish and shellfish collected from retail markets in 11 areas across the continental United States from 2010–2012. The fish and shellfish included farm raised, wild caught, and unknown origin, as well as freshwater fish, saltwater fish, and euryhaline fish. Crab meat, clams, cod, flounder, pangasius, salmon, scallops, and tilapia purchased from Washington, D.C. Shrimp purchased from Orlando, Florida; Memphis, Tennessee; and Nashville, Tennessee. Striped bass purchased from New York, New York and Cherry Hill, New Jersey. Catfish purchased from Indianola, Mississippi; Dallas, Texas; Tampa, Florida; and Orlando, Florida. Pollock purchased from Huntington Beach, California. Tuna purchased from Chicago, Illinois.	Crab, shrimp, striped bass, catfish, clams, cod, flounder, pangasius, pollock, tuna (can and pouch), salmon, scallops (bay and sea), tilapia	**Shrimp: n = 9, DF[a] 11%, range = ND–1.2* ng/g** **Striped bass: n = 10, DF[a] 10%, range = ND–1.4* ng/g** Crab meat: n = 1, DF 0% Catfish: n = 13, DF 0% Clams: n = 1, DF 0% Cod: n = 1, DF 0% Flounder: n = 1, DF 0% Pangasius: n = 1, DF 0% Pollock: n = 1, DF 0% Tuna: n = 3, DF 0% Salmon: n = 2, DF 0% Scallops: n = 2, DF 0% Tilapia: n = 1, DF 0% (MDL = 0.60 ng/g for all seafood) *This value was above the MDL but below the LOQ; LOQ is estimated as 3x the MDL
Schecter et al. (2010)	United States (Texas) Seafood samples from five different grocery stores in Dallas, Texas were collected in 2009. Ten individual samples were collected for each food type and combined to form composite samples. The origin/source of the food samples were not reported.	Salmon, canned tuna, fresh catfish fillet, tilapia, cod, canned sardines, frozen fish sticks	Salmon: n = 1, DF 0% Canned tuna: n = 1, DF 0% Fresh catfish fillet: n = 1, DF 0% Tilapia: n = 1, DF 0% Cod: n = 1, DF 0% Canned sardines: n = 1, DF 0% Frozen fish sticks: n = 1, DF 0% (LOD not reported for any seafood type)

Study	Location and Source	Seafood Type	Results
			*n reflects number of composite samples, each composed of ~10 individual samples
Canada			
Bhavsar et al. (2014)	Canada (Ontario) Recreationally caught fish from four rivers – Credit River, Thames River, Niagara River, Welland River – in summer and fall of 2010 and 2011. Chinook salmon were caught from Credit River, common carp from Thames River, lake trout from Niagara River, and walleye from Welland River. Elevated PFASs concentrations were expected in the fish based on nearby industrial activities or previous monitoring work conducted by the Ontario Ministry of Environment. Raw fish were analyzed, as well as cooked fish using three different cooking methods (baking, broiling, and frying).	Raw and cooked fish (chinook salmon, common carp, lake trout, walleye)	**Chinook salmon:** **Raw: n = 5, DF NR, mean = 0.067 ng/g ww** **Baked: n = 5, DF NR, mean = 0.086 ng/g ww** **Broiled: n = 5, DF NR, mean = 0.083 ng/g ww** **Fried: n = 5, mean = 0.078 ng/g ww** **Common carp:** **Raw: n = 5, DF NR, mean = 0.092 ng/g ww** **Baked: n = 5, DF NR, mean = 0.099 ng/g ww** **Broiled: n = 5, mean = 0.105 ng/g ww** **Fried: n = 5, mean = 0.101 ng/g ww** **Lake trout:** **Raw: n = 4, DF NR, mean = 0.298 ng/g ww** **Baked: n = 4, DF NR, mean = 0.370 ng/g ww** **Broiled: n = 4, mean = 0.358 ng/g ww** **Fried: n = 4, mean = 0.374 ng/g ww** **Walleye:** **Raw: n = 5, DF NR, mean = 0.063 ng/g ww** **Baked: n = 5, DF NR, mean = 0.079 ng/g ww** **Broiled: n = 5, mean = 0.074 ng/g ww** **Fried: n = 5, mean = 0.067 ng/g ww** (LOQ not reported)
Europe			
Hansen et al. (2016)	Norway (Evenes; Skånland) Fish were sampled from Lake Langvatnet, Lake Lavangsvatnet, River Tårstadelva, and the reference Lake Strandvatnet. A civilian airport (location also shared with the Air Station of the Royal Norwegian Air Force) is situated on a ridge between Lake Langvatnet and Lake Lavangsvatnet. These waters are affected by PFAS due to AFFF emissions from the airport.Lake Lavangsvatnet drains into the river Tårstadelva and Lake Strandvatnet is ~15 km away from the airport with no connection to the airport runoff. Samples of the dorsolateral muscle were taken from 10 salmon, 10 anadromous brown trout, 12 stationary brown trout, and 3 European flounder by local fishermen and by personnel from Sweco, an environmental consulting company. The samples were collected in August and September 2014.	Brown trout, European flounder, salmon	**Brown trout (stationary)** **Lake Langvatnet: n = 6, DF[a] 100%, mean[a] (range) = 1.05 (0.11–2.39) ng/g ww** **Lake Lavangsvatnet: n = 5, DF[a] 100%, mean[a] (range) = 0.7 (0.10–1.23) ng/g ww** **Lake Strandvatnet (reference): n = 1, point = 0.10 ng/g ww** **Brown trout (anadromous)** **Lake Lavangsvatnet: n = 3, DF[a] 100%, mean (range) = 0.06 (0.02–0.15) ng/g ww** **River Tårstadelva: n = 5, DF[a] 80%, range = <LOD–0.02 ng/g ww** **Lake Strandvatnet (reference): n = 2, DF[a] 100%, mean (range) = 0.007 (0.004–0.01) ng/g ww** **European flounder (catadromous)**

Study	Location and Source	Seafood Type	Results
			Lake Lavangsvatnet: n = 3, DF[a] 100%, mean[a] (range) = 0.153 (0.10–0.19) ng/g ww **Salmon (anadromous)** **Lake Lavangsvatnet: n = 5, DF[a] 60%, range = <LOD–0.07 ng/g ww** **River Tårstadelva: n = 5, DF[a] 60%, range = <LOD–0.02 ng/g ww** (LOD = 0.014–0.224 ng/g for all PFAS)
Hölzer et al. (2011)	Germany Germany Fish from Lake Möhne and River Möhne were caught by electric fishing or net fishing between June 2006 and October 2008. The River Möhne was contaminated with PFCs mainly by the application of polluted soil conditioner on agricultural lands between 2004 and 2006, which then drained into tributaries of the river. Fish samples for food monitoring were collected from retail trade, wholesale trade, supermarkets, and producers. Sampling year not provided.	Perch, pike, eel, cisco, and roach from Lake Möhne /River Möhne Eel, trout, pike/perch, and other from trade/markets	Lake Möhne: Perch: n = 15, DF 0% Pike: n = 6, DF 0% Eel: n = 5, DF 0% Cisco: n = 8, DF 0% Roach: n = 10, DF 0% Food monitoring: Eel: n = 2, DF 0% Trout: n = 73, DF 0% Pike/perch: n = 8, DF 0% Other: n = 34, DF 0% (LOD = 2.5 ng/g, LOQ = 5.1 ng/g)
Koponen et al. (2015)	Finland (Baltic Sea, Vanhankaupunginlahti bay, Lake Päijänne) A total of 296 individual fish samples were collected in 2009–2010 from five commercially and recreationally important fishing area across the Finnish coast of the Baltic Sea (Oulu, Pori, Turku, Hanko, and Kotka), Helsinki Vanhankaupunginlahti bay, a large freshwater Lake Päijänne, and four fish farming facilities. Most of the individual samples were pooled, each pool consisting of 2–10 individuals. Baltic herring, pike-perch, perch, burbot, whitefish, salmon, and vendace were collected from the Baltic sea; perch and pike-perch were collected from Helsinki Vanhankaupunginlahti bay; perch was collected from Lake Päijänne. Whitefish and rainbow trout were farmed fish. The selection of fish species was mainly based on the significance of fish in the Finnish diet.	Baltic herring, pike-perch, perch, burbot, whitefish, salmon, vendace, whitefish, rainbow trout	Baltic Sea: **Baltic herring: n = 58, DF NR, range = <0.21–2.7 ng/g** **Pike-perch: n = 30, DF NR, range = <0.28–0.35 ng/g** **Perch: n = 25, DF NR, range = <0.21–0.83 ng/g** **Burbot: n = 49, DF NR, range = <0.20–1.5 ng/g** **Whitefish: n = 27, DF NR, range = <0.26–0.63 ng/g** Salmon: n = 44, DF 0% (<0.39 ng/g) **Vendace: n = 20, DF[a] 100%, range = 0.35–0.36 ng/g** Vanhankaupunginlahti bay: **Pike-perch: n = 6, DF NR, range = <0.21–0.33 ng/g** **Perch: n = 7, DF NR, range = <0.23–0.24 ng/g** Lake Päijänne: Perch: n = 10, DF 0% (<0.18 ng/g) Farmed fish: Whitefish: n = 10, DF 0% (<0.37 ng/g)

Study	Location and Source	Seafood Type	Results
			Rainbow trout: n = 10, DF 0% (<0.35 ng/g) (LOQ = 0.18–0.39 ng/g)
Jörundsdóttir et al. (2014)	Iceland Samples were collected by the Icelandic Marine Research Institute in March 2011 during their biannual scientific survey. Cod and anglerfish were caught south-west of Iceland, blue whiting was caught south-east of Iceland, and lumpfish and pollock were caught north-west of Iceland, while ling, plaice, and lemon sole were caught west of Iceland. Each fish sample consisted of a pooled sample from the entire edible part from ten individuals of the same species.	Anglerfish, Atlantic cod, blue whiting, lemon sole, ling, lumpfish, plaice, pollock	Anglerfish (n = 1), Atlantic cod (n = 2), blue whiting (n = 2), lemon sole (n = 1), ling (n = 1), lumpfish (n = 4), plaice (n = 1), pollock (n = 1): DF 0% (LOD = 0.10 ng/g) *n represents number of composite samples
Yamada et al. (2014)	France Marine fish sampled were selected based on the fish consumption habits of the population of four areas – La Rochelle in Gironde-Charente Maritime Sud, Le Havre in Normandy-Baie de Seine, Lorient in South Brittany, and Toulon in Mediterranean-Var. Five primary samples of fish were bought from the fish market and/or supermarket in each region for each species in January–April 2005. Freshwater fish sampled were selected based on the individual dietary consumption analysis of anglers or their family members of the ICAR-PCB study. Freshwater fish were collected in six major French rivers with each river divided into three or four section in 2008–2009. Half of the samples were composite samples.	Freshwater fish, fresh or frozen marine fish	Results presented for lower bound and upper bound if LB value different from UB value **Fresh and frozen marine fish:** **Total LB: n = 95, DF NR, mean (range) = 0.09 (0–0.27) ng/g ww** **Total UB: n = 95, DF NR, mean (range) = 0.11 (0.04–0.27) ng/g ww** Anchovy: n = 1, LB–UB = 0–0.06 ng/g ww Monkfish: n = 4, LB–UB = 0.09–0.11 ng/g ww Catshark: n = 4, LB–UB = 0.06–0.08 ng/g ww Cod: n = 4, LB–UB = 0.15 ng/g ww Common dab: n = 4, LB–UB = 0.09–0.1 ng/g ww Orange roughy: n = 3, LB–UB = 0.19 ng/g ww Plaice/witch: n = 2, LB–UB = 0.16 ng/g ww Goatfish: n = 3, LB–UB = 0.15–0.19 ng/g ww Grenadier: n = 4, LB–UB = 0.05–0.07 ng/g ww Gurnard: n = 1, LB–UB =0–0.04 ng/g ww Haddock: n = 2, LB–UB = 0.17 ng/g ww Hake: n = 4, LB–UB = 0.05–0.06 ng/g ww

Study	Location and Source	Seafood Type	Results
			Halibut: n = 4, LB–UB = 0.03–0.06 ng/g ww John Dory: n = 2, LB–UB = 0.08–0.1 ng/g ww Ling: n = 4, LB–UB = 0.03–0.07 ng/g ww Mackerel: n = 4, LB–UB = 0.03–0.06 ng/g ww Pollack: n = 3, LB–UB = 0.19 ng/g ww Pout: n = 1, LB–UB = 0.12 ng/g ww Ray: n = 4, LB–UB = 0.13 ng/g ww Saithe: n = 4, LB–UB =0.04–0.06 ng/g ww Salmon: n = 4, LB–UB = 0.05–0.07 ng/g ww Sardine: n = 4, LB–UB = 0–0.07 ng/g ww Scorpionfish: n = 1, LB–UB = 0.12 ng/g ww Seabass: n = 4, LB–UB = 0.21–0.24 ng/g ww Sea bream: n = 4, LB–UB = 0.07–0.09 ng/g ww Sole: n = 4, LB–UB = 0.27 ng/g ww Swordfish: n = 4, LB–UB = 0–0.06 ng/g ww Tuna: n = 4, LB–UB = 0–0.04 ng/g ww Whiting: n = 4, LB–UB = 0.07–0.1 ng/g ww **Freshwater fish:** Barbel: n = 5, LB–UB = 1.19–1.21 ng/g ww Bleak: n = 9, LB–UB = 0.01–0.03 ng/g ww Brown trout: n = 31, LB–UB = 0.08–0.13 ng/g ww Chub: n = 9, LB–UB = 0.68–0.7 ng/g ww Common carp: n = 7, LB–UB = 0.03–0.1 ng/g ww Common roach: n = 67, LB–UB = 0.34–0.38 ng/g ww Minnow: n = 1, LB–UB = 0.19 ng/g ww European eel: n = 137, LB–UB =0.12–0.26 ng/g ww European perch: n = 9, LB–UB = 0.12–0.13 ng/g ww Freshwater bream: n = 34, LB–UB =0.11–0.13 ng/g ww Gudgeon: n = 5, LB–UB = 0.21–0.24 ng/g ww Northern pike: n = 8, LB–UB = 0.04–0.07 ng/g ww White bream: n = 22, LB–UB = 0.06–0.11 ng/g ww Thicklip grey mullet: n = 6, LB–UB = 0.15 ng/g ww Wels catfish: n = 14, LB–UB = 0.08–0.1 ng/g ww Western vairone: n = 1, LB–UB = 0–0.08 ng/g ww Pike-perch: n = 22, LB–UB = 0.03–0.08 ng/g ww

Study	Location and Source	Seafood Type	Results
			(LOD = 0.007–0.95 ng/g for PFAAs other than PFOA and PFOS) *Lower bound (LB) scenario defined as values <LOD replaced with 0 *Upper bound (UB) scenario defined as values <LOD replaced with LOD
Eriksson et al. (2013)	Denmark (Faroe Islands) Wild fish (cod and saithe) were sampled from the Faroe Shelf area; cod were sampled in October and August 2011, while saithe were sampled in April 2012. Three farmed salmon samples were collected from different fjords in Faroe Islands, sampling year not reported.	Farmed salmon, wild-caught cod, wild-caught saithe	Cod 1, n = 1, point = <0.035 ng/g Cod 2, n = 1, point = <0.035 ng/g Saithe 1, n = 1, point = <0.035 ng/g Saithe 2, n = 1, point = <0.035 ng/g Salmon 1, n = 1, point = <0.035 ng/g Salmon 2, n = 1, point = <0.035 ng/g Salmon 3, n = 1, point = <0.035 ng/g (LOD = 0.035 ng/g) *n represents number of pooled samples, each combining muscle tissue from five fish
Falandysz et al. (2006)	Poland Cod samples were collected from the Gulf of Gdañsk in the Baltic Sea (south coast of Poland) in February 2003.	Cod *(Gadus morhua)*	**n = 18, DF NR, mean, median (range) = 1.2, 1.1 (0.1–2.1) ng/mL** (LOD not reported) *Values reported for animal whole blood samples
Rivière et al. (2019)	France Samples collected between July 2011 and July 2012 in the center region of France. Food items were selected based on the results of a national consumption survey to obtain a representative and general view of children's (0–3 years old) food consumption. All analyzed samples were formed of 12 subsamples of the same food and of equal weight. The fish were cooked according to the practices reported in the survey of practices.	Fish (unspecified)	n = 1, DF 0% (LOD = 0.0002–3.7 ng/g fw for all PFAS) *n represents number of composite samples
Barbosa et al. (2018)	Belgium, France, The Netherlands, Portugal Fish were collected from different markets based on the assumption that the fish species were frequently consumed in European Union countries and the fish species contained high levels of contaminants of emerging concern. Sampling year not reported. The following fish species (origin, market country) were included: *P. platessa:* Channel, Belgium *M. australis:* South America, Portugal *M. capenis:* South Africa, Portugal *K. pelamis*: Azores, Portugal	Raw and steamed fish (*P. platessa, M. australis, M. capenis, K. pelamis*, and *M. edulis*)	*P. platessa*: n = 25, DF 0% *M. australis*: n = 25, DF 0% *M. capenis*: n = 25, DF 0% *K. pelamis*: n = 25, DF 0% *M. edulis*: The Netherlands: n = 50, DF 0% France: n = 50, DF 0% (LOD = <0.01 ng/g ww for all PFCs)

Study	Location and Source	Seafood Type	Results
	M. edulis: North Sea, The Netherlands; France, France		
Gebbink et al. (2015)	Sweden Food items were purchased at two major grocery store chains in four major Swedish cities in 1999 and 2005. In 2010, sampling was limited to Uppsala city since no systematic geographical differences in food contamination was observed in the two earlier market basket studies. The food items were selected based on Swedish food and production statistics and were not cooked before analysis. Homogenates of fish products (fresh and frozen fillets of fish, canned fish products, shellfish) were prepared for each collection year by mixing food items proportionally according to food consumption statistics. Results were not reported for the 2005 and 2010 fish product composite samples (only reported pooled with other food types).	Fish	**1999: n = 1, point = 0.07 ng/g** (MLOQ = 0.0003 ng/g) *n represents number of composite samples
Vassiliadou et al. (2015)	Greece Samples were obtained during the winter and early spring of 2011. Finfish, squids, and shrimps were purchased from the local fish market in Kallithea, Athens, while mussels were obtained from a mariculture farm within the Saronikos Gulf, Attika. Samples were analyzed raw as well as cooked in the ways favored in Greek cuisine (pan-fried in olive oil and/or grilled). Quadruplicate composite samples were created for each food type, each consisting of four to six items of raw or cooked fish or shellfish.	Anchovy, bogue, hake, picarel, sand smelt, sardine, striped mullet, mussel, shrimp, squid	**Striped mullet:** **Raw: n = 4, DF NR, mean = 0.60 ng/g ww** **Fried: n = 4, DF NR, mean = 0.57 ng/g ww** **Grilled: n = 4, DF NR, mean = 0.50 ng/g ww** **Shrimp:** **Raw: n = 4, DF NR, mean = 1.27 ng/g ww** **Fried: n = 4, DF NR, mean = 1.52 ng/g ww** Anchovy (raw, fried, grilled), bogue (raw, fried, grilled), hake (raw, fried, grilled), picarel (raw, fried), sand smelt (raw, fried), sardine (raw, fried, grilled), mussel (raw, fried), squid (raw, fried, grilled): n = 4, DF 0% (LOD = 0.42 ng/g; LOQ = 1.25 ng/g) *n represents number of composite samples
Carlsson et al. (2014)	Greenland (Nuuk) Seafood was purchased at the local fish market and grocery shops in June 2010. All items were originally caught in the vicinity of the Nuuk area and/or along the West coast of Greenland and represented the common food items consumed by the local Inuit population.	Salmon, halibut	Raw salmon fillet: n = 6, DF 0% Smoked salmon fillet: n = 6, DF 0% Smoked halibut fillet: n = 6, DF 0% (LOD = 0.014–0.224 ng/g for all PFAS)
Domingo et al. (2012)	Spain (Catalonia) Foods purchased from 4 shops/stores of each of the 12 representative cities of Catalonia (Barcelona,	Fish and seafood (sardine, tuna, anchovy, swordfish, salmon, hake, red mullet, sole,	**n = 2, DF NR, mean = 0.5 ng/g fw** (LOD not reported) *n represents number of composite samples

Study	Location and Source	Seafood Type	Results
	l'Hospitalet de Llobregat, Vilanova I la Geltrú, Matafr, Sabadell, Terrassa, Girona, Tarragona, Reus, Tortosa, Lleida and Manresa) in September 2011. Shops/stores included local markets, small stores, supermarkets, and big grocery stores. For each food item, two composite samples were prepared for analysis, where each composite sample consisted of 24 individual units. Only edible parts of each food item were included in the composites.	cuttlefish, clam, mussel, and shrimp)	
Vestergren et al. (2012)	Sweden (Malmoe, Gothenburg, Uppsala, Sundsvall) Purchasing locations of the two largest retail chains in Sweden were selected in each of four major Swedish cities. All purchases were made in spring/summer of 1999, 2005, and 2010. In 2010, the study was limited to the largest retail chains in Uppsala located in close vicinity to Stockholm. An equal amount of each food group from each of the four cities was combined into one sample pool to provide a representative sample for the Swedish urban population.	Fish products (fresh and frozen fillets of fish, canned fish products, shellfish)	**1999: n = 1, point = 0.090 ng/g** **2005: n = 1, point = 0.090 ng/g** **2010: n = 1, point = 0.072 ng/g** (MDL = 0.0025 ng/g; MQL = 0.0063 ng/g) *n represents number of composite samples
Noorlander et al. (2011)	The Netherlands Fish randomly purchased from several Dutch retail stores with nationwide coverage in November 2009. Fish samples were ground, homogenized, and pooled for analysis.	Fatty fish (herring, eel, mackerel, salmon), lean fish (cod, plaice, pollack, tuna), crustaceans (mussels, shrimp, crab)	**Fatty fish: n = 1, point = 0.005 ng/g** **Lean fish: n = 1, point = 0.077 ng/g** **Crustaceans: n = 1, point = 0.058 ng/g** (LOD not reported) *n represents number of composite samples
Jogsten et al. (2009)	Spain (Catalonia) Fish samples purchased from local markets, large supermarkets, and grocery stores from two different areas of Tarragona Province, Catalonia in January and February 2008. The cities of Tarragona and Reus were sampled in the northern area and L'Ametlla de Mar and Tortosa in the southern area. For each food item, two composite samples were analyzed (one composite for the northern area and one for the southern area). Each composite was formed of a minimum of six individual sub-samples of the same product.	Marinated salmon (homemade and packaged)	Homemade: n = 2, DF 0% Packaged: n = 2, DF 0% (LOD not reported) *n represents number of composite samples *Values of ND were replaced with 1/2×LOD
Ericson et al. (2008a)	Spain Food samples purchased from local markets, large supermarkets, and grocery stores within Tarragona County in July 2006. Food samples were randomly	White fish, seafood, tinned fish, blue fish	White fish: n = 2, DF 0% Seafood: n = 2, DF 0% Tinned fish: n = 2, DF 0% Blue fish: n = 2, DF 0%

Study	Location and Source	Seafood Type	Results
	purchased with origin source not specified. Each of the food samples were duplicated and combined to analyze a composite sample. Only the edible part of each food was included in the composite samples. Composite samples included the following: White fish: hake, whiting blue, sea bass, monkfish Seafood: mussel, shrimp Tinned fish: tuna, sardine, mussel Blue fish: salmon, sardine, and tuna		(LOD = 0.001–0.65 ng/g.fw) *n represents number of composite samples
Johansson et al. (2014)	Sweden Farmed rainbow trout (whole fish) were collected from fish farms along the Swedish Baltic Sea coast (brackish water). Only fish older than 12 months were sampled. Samples were collected annually from 1999 to 2010 within the Swedish National Food Agency's official food control program.	Rainbow trout	n = 36, DF 0% (MDL = 0.050 ng/g fw; MQL = 0.140 ng/g fw)
Multiple Continents			
Chiesa et al. (2019)	United States (Pacific Ocean) Wild-caught fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from USA-Pacific Ocean (Food and Agriculture Organization Area 67 and 77).	Wild-caught salmon (*Oncorhynchus kisutch* and *Oncorhynchus keta*)	*Oncorhynchus kisutch:* n = 5, DF 0% *Oncorhynchus keta*: n = 2, DF 0% (LOQ = 0.005 ng/g)
	Canada Wild-caught fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Canada (Food and Agriculture Organization Area 67).	Wild-caught salmon (*Oncorhynchus nerka)*	n = 15, DF 0% (LOQ = 0.005 ng/g)
	Norway Farmed fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Norway (Food and Agriculture Organization Area 27).	Farmed salmon (*Salmo salar)*	n = 25, DF 0% (LOQ = 0.005 ng/g)
	Scotland Wild-caught and farmed fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Scotland (Food and Agriculture Organization Area 27).	Wild-caught and farmed salmon (*Salmo salar)*	Wild-caught: n = 2, DF 0% Farmed: n = 17, DF 0% (LOQ = 0.005 ng/g)

Notes: DF = detection frequency; ww = wet weight, LOD = limit of detection; LOQ = limit of quantitation; MDL = method detection limit; ND = not detected.
[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

C.3.1.2. Other Food Types

PFNA was included in a suite of PFAS evaluated in FDA's 2019, 2021, and 2022 Total Diet Study Sampling (US FDA, 2022c, 2022b, 2022a, 2021b, 2021a, 2020b, 2020a); it was detected at concentrations of 233 ng/kg (0.233 ng/g) in baked cod and 50 ng/kg (0.050 ng/g) in frozen (oven-cooked) fish sticks or patties in 2021, but it was not detected in any of the other food samples tested. It should be noted that FDA indicated that the sample sizes used in the PFAS 2019, 2021, and 2022 Total Diet Study Sampling were limited and that the results should not be used to draw definitive conclusions about PFAS levels in the general food supply (US FDA, 2022c). PFNA was not detected in milk samples collected from a farm with groundwater known to be contaminated with PFAS; however, it was detected in produce (corn) collected from an area near a PFAS production plant in FDA studies of the potential PFAS exposure to the U.S. population (US FDA, 2021c, 2018). PFNA was detected in beef steak in the Canadian Total Diet studies from 1992 to 2004, but it was not detected in any of the other food samples tested (ATSDR, 2021; Tittlemier et al., 2007).

Several U.S. studies were identified that examined PFNA in breastmilk or food types other than breastmilk (Tipton et al., 2017; Blaine et al., 2014; Blaine et al., 2013; Young et al., 2012; Schecter et al., 2010; von Ehrenstein et al., 2009; Kuklenyik et al., 2004) (Table C-3). Few U.S. studies analyzed foods from any one origin – only two studies sampled from store- or market-bought meats, eggs, produce, and dairy, one studied wild alligator meat, two sampled from crops (produce and corn grain and stover) grown in biosolids-amended soils (and also control and municipal soils) as part of greenhouse and field studies, and two studied breastmilk.

Two studies purchased food items from stores and markets for evaluation (Young et al., 2012; Schecter et al., 2010). Schecter et al. (2010) assessed PFNA and other PFAS in food samples collected from five Dallas, Texas grocery stores in 2009. The origin or source of each food was not described. Food items included meat products (bacon, canned chili, chicken breast, ground beef, roast beef, ham, sausage, and turkey), dairy (butter, cheeses, frozen yogurt, ice cream, milk, and yogurt), eggs, and grains (cereal), fruits and vegetables (apples, potatoes), and fats/other (canola oil, margarine, olive oil, peanut butter). PFNA was not detected in any of the food samples. In Young et al. (2012), cow milk was purchased from retail markets across the continental United States representing 17 states; the sampling year was not reported. Cow milk samples included organic milk, vitamin D added milk, and ultra-pasteurized milk. PFNA was not detected in any of the 49 retail milk samples (method detection limit (MDL) = 0.28 ng/g).

One study investigated PFAS levels in wild meat (Tipton et al., 2017). Tipton et al. (2017) assessed alligator tail meat that was collected during the South Carolina recreational hunting season between September to October 2015. Tail meat samples were collected from four different public hunt units – Southern Coastal, Middle Coastal, Midlands, and Pee Dee. PFNA was detected in samples from all hunt units with the exception of the Midlands (n = 2), where PFNA was not detected. Median concentrations from Southern Coastal (n = 19), Middle Coastal (n = 17), and Pee Dee (n = 2) were 0.107 ng/g, 0.102 ng/g, and 0.117 ng/g wet mass, respectively.

Two studies by Blaine et al. (2014; 2013) evaluated PFNA in crops grown in greenhouse and field studies. In Blaine et al. (2014), PFAS levels were measured in celery root, pea fruit, and radish root grown in a greenhouse study with control (unamended) soil, industrially impacted soil, and municipal soil (n = 3–5). PFNA was detected in radish root and celery shoot from all three soils and pea fruit from only industrially impacted soil. Mean concentrations of PFNA in radish root for the control, industrially impacted, and municipal soil were 4.79 ng/g, 26.88 ng/g, and 5.99 ng/g, respectively. Mean concentrations of PFNA in celery shoot for the control, industrially impacted, and municipal soil were 1.89 ng/g, 13.81 ng/g, and 1.62 ng/g, respectively. The mean concentration of PFNA in pea fruit in the industrially impacted soil was 1.45 ng/g. Authors noted minor cross-contamination of the control soil due to the proximity of the unamended soil to biosolids-amended soil. In Blaine et al. (2013), authors studied the uptake of PFAS into edible crops in both field and greenhouse studies. In the field study, PFAS levels were measured in corn grain and corn stover grown with control (unamended), urban biosolids-amended, and rural biosolids-amended soil (n = 3–7). Mean PFNA concentrations were below the LOQ in both corn grain and corn stover grown in any field study plots (<0.10 ng/g for corn grain; <0.29 ng/g for corn stover). In the greenhouse study, lettuce and tomato plants were grown in control soil, industrially impacted soil, or municipal soil (n = 3–5). Mean PFNA concentrations were below the LOQ (2.96 ng/g) in any tomato plants but was detected in lettuce grown in industrially impacted soil and municipal soil at mean concentrations of 57.39 ng/g and 4.73 ng/g, respectively. PFNA was not detected above the LOQ (0.04 ng/g) in lettuce grown in control soil. Sampling year was not reported.

The remaining two studies evaluated the occurrence of PFNA in breastmilk (von Ehrenstein et al., 2009; Kuklenyik et al., 2004). von Ehrenstein et al. (2009) collected breastmilk samples between December 2004 and July 2005 from women between the ages of 18 and 38 at the time of recruitment as part of the pilot study Methods Advancement for Milk Analysis (MAMA). Women provided milk samples at two visits – the first visit was 2–7 weeks postpartum, and the second visit was 3–4 months postpartum. PFNA was not detected in any of the samples from the first visit (n = 18) or second visit (n = 20). Similarly, PFNA was below the LOD (1.0 ng/mL) in the samples reported by Kuklenyik et al. (2004). Kuklenyik et al. (2004) did not report information on the breastmilk donors or the sampling procedure as it was unavailable; PFNA was not detected in either of the two samples.

Results for all of the identified non-U.S. studies are presented in detail in Table C-5 and are summarized here. Among the European studies, many collected food samples of unknown origin from grocery stores (Scordo et al., 2020; Rivière et al., 2019; Sznajder-Katarzyńska et al., 2019; Sznajder-Katarzyńska et al., 2018; Papadopoulou et al., 2017; Surma et al., 2017; D'Hollander et al., 2015; Gebbink et al., 2015; Pérez et al., 2014; Herzke et al., 2013; Hlouskova et al., 2013; Domingo et al., 2012; Vestergren et al., 2012; Noorlander et al., 2011; Jogsten et al., 2009; Ericson et al., 2008a). A wide variety of items were analyzed including meats and seafood, dairy, fruits and vegetables, grains, pastries and other sweets, spices, sweeteners, and other beverages. Of these studies, five reported no detectable PFNA in any of the items sampled (Rivière et al., 2019; Sznajder-Katarzyńska et al., 2018; Surma et al., 2017; Jogsten et al., 2009; Ericson et al., 2008a). Among the studies that did report detectable PFNA in some food items, PFNA was found in all major food categories examined except spices, salts, and sweeteners (sugar and honey) though there was no apparent trend in specific food types with PFNA present in measurable quantities.

Three studies focused on PFNA in eggs. Ghelli et al. (2019) collected eggs from commercial laying hen farms in Italy. PFNA was detected at trace levels only in 5% of the 44 eggs sampled. Johansson et al. (2014) collected eggs from 20 to 25 commercial egg producers in Sweden every year between 1999-2010. Ten to 12 eggs were pooled for analysis and they reported detectable quantities of PFNA in 28% of 72 pooled samples, with a maximum observed value of 0.143 ng/g fw. Zafeiraki et al. (2016a) collected home and commercially produced eggs in Greece and the Netherlands. They reported detectable levels of PFNA in home produced eggs from both regions, with a maximum observed value of 2.0 ng/g. PFNA was not detected in any of the commercially produced eggs sampled.

Two studies focused on PFNA in milk and other dairy products. Johansson et al. (2014) collected milk from about 10 dairy farms in Sweden every year between 1999-2010. PFNA was not present at detectable levels in any of the samples. Still et al. (2013) collected commercially sold dairy product samples in Germany. PFNA was present at detectable levels in one of four milk samples, all three cheese samples and one butter sample, with a maximum observed value of 0.0094 ng/g. They reported non-detect results in all other products examined. Eriksson et al. (2013) identified PFNA concentrations in milk from two major dairy farms, with a maximum of 0.073 ng/g ww. They identified PFNA in 75% of four additional dairy samples from Faroe Island, but it was not detected in other samples of yogurt, créme fraiche, and potatoes. Vestergren et al. (2013) evaluated concentrations in milk and reported a mean of 0.0023 ng/mL. This study also evaluated concentrations in cow liver, blood, and muscle, finding the highest concentration in liver at a mean of 0,016 ng/g.

In addition, two non-U.S. studies focused on locally caught food items of importance to indigenous populations. Binnington et al. (2017) sampled whale blubber collected from two beluga whales captured off the West coast of Canada. PFNA was present at measurable concentrations in all solid samples (max = 0.1718 ng/g lipids) but was not detectable in rendered oil. Carlsson et al. (2014) assessed wild caught seal beef, narwhal, and whale beef in Greenland and found detectable concentrations of PFNA only in seal beef.

PFNA was also detected in some beverages. Stahl et al. (2014) measured PFNA in a selection of Hessian, Belgian, and Bavarian beers and found measurable concentrations in 14% of 93 samples. Eschauzier et al. (2013) measured PFNA in cola and brewed coffee samples collected from various locations in Amsterdam and found that PFNA was not present at measurable concentrations in any sample.

Of the thirteen non-U.S. studies that evaluated the occurrence of PFNA in breastmilk, five did not report detectable concentrations. These studies evaluated 13 women in Ontaria, Canada (Kubwabo et al., 2013); 10 women in Spain (Kärrman et al., 2010), 11 pooled samples obtained from 109 women in Ireland (Pratt et al., 2013), 61 women in France (Cariou et al., 2015), and 20 women in Spain (Llorca et al., 2010). However, the remaining European studies did report quantifiable concentrations of PFNA in breastmilk. Among these studies, detectable concentrations of PFNA were reported in 17% of samples measured from 12 individual Swedish women and 33% of nine composite samples composed of breastmilk from 25-90 Swedish women (Kärrman et al., 2007); 42.5% of breastmilk samples from 40 Belgian women (Croes et al., 2012); 6% of breastmilk samples from 67 French women (Motas Guzmàn et al., 2016); 100% of 31 composite samples composed of breastmilk from 5–116 Swedish women and 94% of samples measured from 46 individual Swedish women (Nyberg et al., 2018); 5% of samples

from 14 Spanish women (Beser et al., 2019); 69% of 16 samples from Irish women (Abdallah et al., 2020); 48% of samples from 50 women in the Czech Republic (Lankova et al., 2013); and 2% of samples from 48 French women (Antignac et al., 2013). Abdallah et al. (2020) reported the highest value for PFNA in breastmilk, at 0.1 ng/mL.

Several European studies focused on other sources of dietary exposures in children. Lankova et al. (2013) measured PFNA in infant formula samples purchased at Czech retail markets and reported a maximum observed value of 0.011 ng/g. Llorca et al. (2010) measured PFNA in baby cereals and infant formulas purchased from reatailers in Spain. They reported PFNA present in measureable quantities in all of the samples measured, with a maximum reported value of 0.1138 ng/g in baby cereal and 0.219 ng/g in infant formulas. Dellatte et al. (2013) measured PFNA in ready-to-eat meals collected from nursery and primary school canteens. PFNA was present at detectable levels in one of six samples, with a reported concentration of 0.0063 ng/g.

Table C-5. Summary of PFNA Occurrence in Other Food

Study	Location and Source	Food Types	PFNA Results
United States			
Schecter et al. (2010)	United States (Texas) Food samples from five different grocery stores in Dallas, Texas were collected in 2009. Ten individual samples were collected for each food type and combined to form composite samples. The origin/source of the food samples were not reported.	Dairy; fruits and vegetables; grains; meat; fats/other	Meat: Hamburger: n = 1, DF 0% Bacon: n = 1, DF 0% Sliced turkey: n = 1, DF 0% Sausages: n = 1, DF 0% Ham: n = 1, DF 0% Sliced chicken breast: n = 1, DF 0% Roast beef: n = 1, DF 0% Canned chili: n = 1, DF 0% Dairy and Eggs: Butter: n = 1, DF 0% American cheese: n = 1, DF 0% Other cheese: n = 1, DF 0% Whole milk: n = 1, DF 0% Ice cream: n = 1, DF 0% Frozen yogurt: n = 1, DF 0% Whole milk yogurt: n = 1, DF 0% Cream cheese: n = 1, DF 0% Eggs: n = 1, DF 0% Grains: Cereals: n = 1, DF 0% Fruits and Vegetables: Apples: n = 1, DF 0% Potatoes: n = 1, DF 0% Fats and Other: Olive oil: n = 1, DF 0% Canola oil: n = 1, DF 0% Margarine: n = 1, DF 0% Peanut butter: n = 1, DF 0% (LOD not reported for any food item) *n reflects number of composite samples, each composed of ~10 individual samples
Young et al. (2012)	United States (17 states) Retail cow's milk samples were all pasteurized whole milk, commercially available, and purchased at retail markets across the continental United States representing 17 states. Samples were organic milk,	Dairy	n = 49, DF 0% (MDL = 0.28 ng/g)

Study	Location and Source	Food Types	PFNA Results
	vitamin D added milk, or ultra-pasteurized milk. Sampling year not reported.		
Tipton et al. (2017)	United States (South Carolina) Alligator tail meat samples were collected from a local wild game meat processer during the South Carolina recreational hunt season between September to October 2015. Samples were from four different public hunt units—Southern Coastal, Middle Coast, Midlands, and Pee Dee.	Meat	**Alligator tail:** **Southern coastal: n = 19, DF[a] 74%, median (range) = 0.107 (<0.088–0.551) ng/g wet mass** **Middle coastal: n = 17, DF[a] 65%, median (range) = 0.102 (<0.073–0.553) ng/g wet mass** **Pee Dee: n = 2, DF[a] 100%, median (range) = 0.117 (0.100–0.135) ng/g wet mass** Midlands: n = 5, DF 0% (RL not reported)
Blaine et al. (2014)	United States (Midwest) Crops grown in in greenhouse study with control (unamended), industrially impacted soil, or municipal soil. Control soil had minor cross-contamination due to proximity to biosolids-amended fields. Industrially impacted soil was amended with industrially impacted biosolids, and municipal soil was amended with municipal biosolids for over 20 years. Crops grown in the greenhouse study were grown from seed in pots, which were randomly arranged within the greenhouse. Sampling year not reported.	Fruits and vegetables	**Radish root:** **Control: n = 3–5, DF NR, mean = 4.79 ng/g** **Industrially impacted; n = 3–5, DF NR, mean = 26.68 ng/g** **Municipal: n = 3–5, DF NR, mean = 5.99 ng/g** **Celery shoot:** **Control: n = 3–5, DF NR, mean = 1.89 ng/g** **Industrially impacted: n = 3–5, DF NR, mean = 13.81 ng/g** **Municipal: n = 3–5, DF NR, mean = 1.62 ng/g** Pea fruit: Control: n = 3–5, DF 0% **Industrially impacted: n = 3–5, DF NR, mean = 1.45 ng/g** Municipal: n = 3–5, DF 0% (LOQ = 0.07ng/g)
Blaine et al. (2013)	United States (Midwest) Crops grown in urban and rural full-scale field study with control (unamended) and biosolids-amended soil. Three agricultural fields were amended (0.5×, 1×, or 2×) with municipal biosolids. Urban biosolids (1× and 2×) were from a WWTP receiving both domestic and industrial waste. Rural biosolids (0.5×) were from a WWTP receiving domestic waste only. Control plots were proximal to the rural and urban amended corn grain and corn stover field sites; sampling year not provided.	Fruits and vegetables; grains	Field study: Corn grain: Urban nonamended: n = 3–7, DF NR, mean = <0.10 ng/g Urban 1×: n = 3–7, DF NR, mean = <0.10 ng/g Urban 2×: n = 3–7, DF NR, mean = <0.10 ng/g Rural nonamended: n = 3–7, DF NR, mean = <0.10 ng/g Rural 0.5×: n = 3–7, DF NR, mean = <0.10 ng/g Corn stover: Urban nonamended: n = 3–7, DF NR, mean = <0.29 ng/g

Study	Location and Source	Food Types	PFNA Results
	Crops grown in greenhouse study with control (nonamended) and biosolids-amended soil. Nonamended soil obtained from a field that received commercial fertilizers and had a similar cropping system as the nearby municipal soil site. Municipal soil was obtained from a reclamation site in Illinois where municipal biosolids were applied at reclamation rates for 20 years, reaching the cumulative biosolids application rate of 1,654 Mg/ha. Industrially impacted soil was created by mixing composted biosolids from a small municipal (but impacted by PFAA manufacturing) WWTP with control soil on a 10% mass basis. Sampling year not provided.		Urban 1×: n = 3–7, DF NR, mean = <0.29 ng/g Urban 2×: n = 3–7, DF NR, mean = <0.29 ng/g Rural nonamended: n = 3–7, DF NR, mean = <0.29 ng/g Rural 0.5×: n = 3–7, DF NR, mean = <0.29 ng/g (LOQ = 0.10 ng/g for corn grain; LOQ = 0.29 ng/g for corn stover) Greenhouse study: Lettuce: Nonamended: n = 3–5, DF NR, mean = <0.04 ng/g **Industrially impacted: n = 3–5, DF NR, mean = 57.39 ng/g** **Municipal: n = 3–5, DF NR, mean = 4.73 ng/g** Tomato: Nonamended: n = 3–5, DF NR, mean = <2.86 ng/g Industrially impacted: n = 3–5, DF NR, mean = <2.86 ng/g Municipal: n = 3–5, DF NR, mean = <2.86 ng/g (LOQ = 0.04 ng/g for lettuce; LOQ = 2.86 ng/g for tomato)
von Ehrenstein et al. (2009)	United States (North Carolina) As part of the Methods Advancement for Milk Analysis (MAMA) pilot study, 34 breastfeeding women aged 18 to 38 years at recruitment provided breastmilk samples at two visits. The first visit occurred 2–7 weeks postpartum, and the second visit occurred 3–4 months postpartum. Both visits were between December 2004 and July 2005.	Breastmilk	Visit #1: n = 18, DF 0% Visit #2: n = 20, DF 0% (LOQ = 0.30 ng/mL)
Kuklenyik et al. (2004)	United States (Georgia) Authors reported that no information was provided on the human milk donors or the sampling procedure.	Breastmilk	n = 2, DF 0% (LOD = 1.0 ng/mL)
Canada			
Binnington et al. (2017)	Canada (Tuktoyaktuk, Northwest Territory) Samples were collected from two beluga whales (HI-14-06 and HI-14-11) caught offshore of Tuktoyaktuk during summer 2014. The belugas	Meat	**Beluga whale blubber – muktuk** **Air dry: n = 4, DF NR, range of means = 0.0713–0.1288 ng/g lipids** **Hang dry: n = 4, DF NR, range of means = 0.1013–0.150 ng/g lipids**

Study	Location and Source	Food Types	PFNA Results
	were members of the Eastern Beaufort Sea population, with a home range including respective summering and wintering grounds in the Beaufort and Bering Seas. The selected individuals were harvested on the shore of nearby Hendrickson Island. Beluga whale blubber forms the basis for two distinct traditional food types: (i) muktuk designates food items composed of the outer layer of blubber and its attached skin and connective tissue, while (ii) uqsuq designates food items derived from the inner layers of blubber. Samples were analyzed in duplicate at each step in the preparation process.		**Boil drum: n = 4, DF NR, range of means = 0.0831–0.1268 ng/g lipids** **Boil pot: n = 4, DF NR, range of means = 0.0617–0.079 ng/g lipids** **Roast: n = 4, DF NR, range of means = 0.0813–0.1205 ng/g lipids** **Aged 2 days: n = 4, DF NR, range of means = <MDL–0.0914 ng/g lipids** **Aged 5 days: n = 4, DF NR, range of means = 0.0773–0.1004 ng/g lipids** **Beluga whale blubber – uqsuq** **Baseline: n = 4, DF NR, range of means = 0.0385–0.0769 ng/g lipids** **Aged 2 days: n = 4, DF NR, range of means = <MDL–0.1718 ng/g lipids** **Aged 5 days: n = 4, DF NR, range of means = 0.0582–0.1391 ng/g lipids** Oil: n = 4, DF 0% (MDL = 0.01 ng/mL)
Kubwabo et al. (2013)	Canada (Ontario) Breastmilk samples were collected in the Kingston region of Ontario in 2003–2004.	Breastmilk	n = 13, DF 0% (MDL = 0.062 ng/mL; LOQ = 0.208 ng/mL)
Europe			
Scordo et al. (2020)	Italy Commercially available strawberry and olive fruits were purchased in two Italian supermarkets in 2018.	Fruits	**Strawberries: n = 2, DF[a] 50%, range = <0.0013–0.047 ng/g dw** (MDL = 0.0013 ng/g; MQL = 0.0049 ng/g) **Olives: n = 2, DF[a] 100%, mean[a] (range) = 0.0385 (0.035–0.042) ng/g dw** (MDL = 0.0007 ng/g; MQL = 0.0026 ng/g)
Sznajder-Katarzyńska et al. (2019)	Poland Milk and milk products were purchased in Polish markets in 2017. Commercially available samples of each product were obtained from five different suppliers.	Dairy	**All dairy: n = 35, DF 45.7%, sum PFNA = 0.83 ng/g** **Milk: n = 5, DF[a] 80%, range = 0.04–0.09 ng/g** **Cottage cheese: n = 5, DF[a] 100%, range = 0.03–0.06 ng/g** **Natural yogurt: n = 5, DF[a] 60%, range = 0.07–0.10 ng/g** **Kefir/Bonny clabber: n = 5, DF[a] 20%, range = 0.07–0.07 ng/g**

Study	Location and Source	Food Types	PFNA Results
			Sour cream: n = 5, DF[a] 60%, range = 0.03–0.05 ng/g Camembert-type cheese (n = 5), butter (n = 5): DF 0% (LOD = 0.006 ng/g; LOQ = 0.019 ng/g) *Range reported for detected values
Rivière et al. (2019)	France Samples collected between July 2011 and July 2012 in the center region of France. Food items were selected based on the results of a national consumption survey to obtain a representative and general view of children's (0–3 years old) food consumption. All analyzed samples were formed of 12 subsamples of the same food and of equal weight. The products purchased were prepared in a way that reflected as closely as possible what is done in the home (preparation and cooking).	Meat; dairy; fruits and vegetables; fats/other	Infant-specific foods: Milk-based beverage (n = 8), cereals (n = 5), milk-based desserts (n = 6), growing-up milk (n = 9), soups and puree (n = 11), fruit puree (n = 4), vegetable-based ready-to-eat meal (n = 20), meat/fish-based ready-to-eat meal (n = 45), infant formula (n = 28), follow-on formula (n = 33): DF 0% Common foods: Non-alcoholic beverages (n = 1), dairy-based desserts (n = 1), milk (n = 1), mixed dishes (n = 1), ultra-fresh dairy products (n = 1), meat (n = 1), poultry and game (n = 1): DF 0% (LOD = 0.0002–3.7 ng/g for all PFAS) *n represents number of composite samples
Sznajder-Katarzyńska et al. (2018)	Poland Samples were purchased in Polish markets in 2017. Individual food items were selected among the most frequently consumed in Poland. Vegetables (potatoes, beetroots, carrots, white cabbage, tomatoes) and fruits (apples, cherries, strawberries) of Polish origin were bought in season when naturally ripe. Bananas, lemons, and oranges were bought after being imported to Poland. Five different samples of each fruit or vegetable were collected.	Fruits and vegetables	Apples, bananas, cherries, lemons, oranges, strawberries, beetroots, carrots, tomatoes, potatoes, and white cabbage: n = 5 for each, DF 0% (LOD = 0.007 ng/g; LOQ = 0.014 ng/g)
Surma et al. (2017)	Spain, Slovakia Spice samples were collected in powder form from Spain and Slovakia. Sampling year not reported.	Fats/other	Spain: Anise (n = 1), star anise (n = 1), white pepper (n = 1), fennel (n = 1), cardamom (n = 1), clove (n = 1), coriander (n = 1), nutmeg (n = 1), allspice (n = 1), cinnamon (n = 2), vanilla (n = 1), ginger (n = 1), peppermint (n = 1), parsley (n = 1), thyme (n = 1), laurel (n = 1), garlic (n = 1), cumin (n = 1), black

Study	Location and Source	Food Types	PFNA Results
			pepper (n = 1), mild hot pepper (n = 1), hot hot pepper (n = 1), oregano (n = 2): DF 0% Slovakia: Anise (n = 1), star anise (n = 1), white pepper (n = 1), fennel (n = 1), cardamom (n = 1), clove (n = 1), coriander (n = 1), nutmeg (n = 1), allspice (n = 1), cinnamon (n = 1), vanilla (n = 1), ginger (n = 1): DF 0% (LOD = 0.031g/g; LOQ = 0.093 ng/g)
Zafeiraki et al. (2016a)	Greece, The Netherlands Home and commercially-produced eggs were collected from different regions in the Netherlands and Greece in August 2013–August 2014. Home-produced eggs were voluntarily provided, and commercial eggs were purchased from supermarkets. The yolks of the same sample of eggs were pooled, homogenized, and then analyzed.	Fats/other	**Domestic eggs:** **The Netherlands: n = 73, DF 18%, median (range) = 0.9 (<0.5–2.0) ng/g** **Greece: n = 45, DF 20%, median (range) = 0.8 (<0.5–1) ng/g** Commercial eggs: The Netherlands: n = 22, DF 0% Greece: n = 31, DF 0% (LOD = 0.15 ng/g; LOQ = 0.5 ng/g) *Median calculated only on the concentrations above LOQ
Zafeiraki et al. (2016b)	The Netherlands Samples purchased from local markets and slaughterhouses in the Netherlands in 2014. Samples included liver samples of horse, sheep, bovine, pig, and chicken.	Meat	Horse: n = 19, DF 0% Sheep: n = 18, DF 0% Bovine: n = 22, DF 0% Pig: n = 20, DF 0% Chicken: n = 20, DF 0% (LOQ = 0.5 ng/g ww)
D'Hollander et al. (2015)	Czech Republic, Belgium, Norway, Italy The Czech Republic, Belgium, Norway, and Italy were selected to represent eastern, western, northern, and southern Europe. Sampling took place between spring and summer 2010 as part of the PERFOOD study. Individual items were randomly selected in three national retail stores covering different brands or countries of origin. Of each item, three to ten single samples were combined to create a pooled sample. Individual food items that were collected to create pooled samples were: Cereals: rice, wheat (white), oats, rye	Grains; fruits; fats/other	**Czech Republic:** Cereals: wheat (white), oats, rye: n = 1 each, point = <0.020 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.001 ng/g Fruits – berries: strawberries: n = 1, point = <0.001 ng/g Fruits – citrus fruit: oranges, tangerines: n = 1, point = <0.001 ng/g Fruits – pipe and stone fruit: **Apples: n = 1, point = 0.002 ng/g** **Pears: n = 1, points = 0.001 ng/g** Peaches: n = 1, point = <0.001 ng/g

Study	Location and Source	Food Types	PFNA Results
	Sweets: sugar (beet), sugar (cane), honey Fruits: berries – strawberries, citrus fruit – oranges, tangerines, lemons, grapefruits, pipe and stone fruit – apples, pears, peaches, plums, others and exotic fruit – melons, grape, bananas Miscellaneous: "rock" salt, "marine" salt		Fruits – others and exotic fruit: melons: n = 1, point = <0.004 ng/g Miscellaneous: rock salt: n = 1, point = <0.001 ng/g **Italy:** Cereals: Rice, maize: n = 1 each, point = <0.001 ng/g Wheat (white): n = 1, point = <0.020 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.001 ng/g Fruits – citrus fruit: lemons: n = 1, point = <0.001 ng/g Fruits – pipe and stone fruit: **Apples: n = 1, point = 0.016 ng/g** **Pears: n = 1, point = 0.002 ng/g** **Peaches: n = 1, point = 0.012 ng/g** Plums: n = 1, point = <0.001 ng/g Fruits – others and exotic fruit: Grapes: n = 1, point = <0.001 ng/g Bananas: n = 1, point = 0.003 ng/g Miscellaneous: marine salt: n = 1, point = <0.001 ng/g **Norway:** Cereals: wheat (white): n = 1, point = <0.020 ng/g Sweets: sugar (cane), honey: n = 1 each, point = <0.001 ng/g Fruits - berries: strawberries: n = 1, point = <0.001 ng/g Fruits – citrus fruit: Oranges: n = 1, point = <0.001 ng/g **Grapefruits: n = 1, point = 0.0248 ng/g** Fruits – pipe and stone fruit: apples, pears: n = 1 each, point = <0.001 ng/g **Fruits – others and exotic fruit: melons: n = 1, point = 0.0099 ng/g** Miscellaneous: rock salt: n = 1, point = <0.001 ng/g **Belgium:** Cereals: rice, wheat (white), wheat (dark), oats: n = 1 each, point = <0.001 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.001 ng/g

Study	Location and Source	Food Types	PFNA Results
			Fruits - berries: strawberries: n = 1, point = <0.001 ng/g Fruits – citrus fruit: oranges, lemons: n = 1 each, point = <0.001 ng/g Fruits – pipe and stone fruit: **Apples: n = 1, point = 0.205 ng/g** Pears: n = 1, point = <0.001 ng/g Plums: n = 1, point = <0.001 ng/g **Fruits – others and exotic fruit: grapes: n = 1, point = <0.007 ng/g** (LOD = 0.001 or 0.020 ng/g) *n represents number of composite samples
Gebbink et al. (2015)	Sweden Food items were purchased at two major grocery store chains in four major Swedish cities in 1999 and 2005. In 2010, sampling was limited to Uppsala city since no systematic geographical differences in food contamination was observed in the two earlier market basket studies. The food items were selected based on Swedish food and production statistics and were not cooked before analysis. The food items were divided into 12 groups and homogenates for each food group were prepared by mixing food items proportionally according to food consumption statistics. Results by food group were not reported for the 2005 and 2010 years. For all three sampling years, a homogenate was prepared by mixing proportional amounts of each food group according to consumption data for the respective year (includes fish samples).	Fruits and vegetables; meat; grains; fats/other	**1999:** **Dairy products: n = 1, point = 0.0005 ng/g** **Meat products: n = 1, point = 0.0067 ng/g** **Fats: n = 1, point = 0.0037 ng/g** **Pastries: n = 1, point = 0.0012 ng/g** **Egg: n = 1, point = 0.024 ng/g** **Fruit: n = 1, point = 0.0006 ng/g** **Soft drinks: n = 1, point = 0.0005 ng/g** Cereal products: n = 1, point = <0.0003 ng/g Vegetables: n = 1, point = <0.0003 ng/g Potatoes: n = 1, point = <0.0003 ng/g Sugar and sweets: n = 1, point = <0.0003 ng/g **Year pool: n = 12, point = 0.0077 ng/g** **2005:** **Year pool: n = 12, point = 0.016 ng/g** **2010:** **Year pool: n = 12, point = 0.015 ng/g** (MLOQ = 0.0003 ng/g) *n represents number of composite samples
Carlsson et al. (2014)	Greenland (Nuuk) Meat was purchased at the local fish market and grocery shops in June 2010. All items were originally caught in the vicinity of the Nuuk area and/or along the West coast of Greenland and represented the common food items consumed by the local Inuit population.	Meat	**Seal beef: n = 2, DF[a] 50%, range = <LOD–0.3 ng/g ww** Narwhal: n = 6, DF NR, median = <LOD Whale beef: n = 8, DF NR, median = <LOD (LOD = 0.014–0.224 ng/g for PFAS)

Study	Location and Source	Food Types	PFNA Results
Pérez et al. (2014)	Serbia (Belgrade and Novi Sad), Spain (Barcelona, Girona, and Madrid) Between September 2011 and February 2013, samples were purchased from different supermarkets and retail stores in representative cites around the world, including cities in Serbia and Spain in Europe. Foodstuffs were grouped into the following categories: cereals; pulses and starchy roots; tree-nuts, oil crops, and vegetable oils; vegetables and fruits; meat and meat products; milk, animal fats, dairy products, and eggs; fish and seafood; and other such as candies and coffee.	Grains; fruits and vegetables; fats/other; meat; dairy; seafood	**Spain: n = 174, DF 13.3%, mean, median (range) = 1.175, 0.43 (ND–13) ng/g** **Serbia: n = 36, DF 10.3%, mean, median (range) = 0.208, 0.185 (ND–0.43) ng/g** (MLOD = 0.039–0.412 ng/g, depending on food item) *Results not reported for individual food categories
Stahl et al. (2014)	Belgium Samples of 83 Hessian beers were obtained from the Hessian control authority, 4 Bavarian beers were provided by the Bavarian Health and Food Safety Authority, and 6 Belgian beers were obtained from Belgian retail stores. Sampling year not provided.	Fats/other	**All beer: n = 93, DF (frequency of quantification) 14%, mean, median (maximum) = 0.00377, <LOQ (0.101) ng/mL** **German (Hesse): n = 83, DF (frequency of quantification) 12%, mean, median (maximum) = 0.00343, <LOQ (0.101) ng/mL** **German (Bavaria): n = 4, DF (frequency of quantification) 50%, mean, median (maximum) = 0.0073, 0.00628 (0.0167) ng/mL** **Belgian: n = 6, DF (frequency of quantification) 17%, mean, median (maximum) = 0.00611, <LOQ (0.0367) ng/mL** (LOQ = 0.005 ng/mL)
Eschauzier et al. (2013)	The Netherlands (Amsterdam) Brewed coffee samples (n = 12) from different coffee machines were collected from all over the city. Coffee beans from four of these locations were collected to manually brew coffee. Post-mixed cola was collected (n = 4) together with corresponding tap water and an additional three cola samples from different parts of town. Sampling was conducted between February and April 2011 at various locations (cafés, universities, and supermarkets).	Fats/other	Post-mixed cola: n = 6, DF NR, mean = <0.07 ng/L Brewed coffee from coffee machines: n = 12, DF NR, mean = <0.11 ng/L Manually brewed coffee: n = 4, DF NR, mean = <0.11 ng/L (LOQ = 0.07 ng/L for cola; 0.11 ng/L for coffee)
Herzke et al. (2013)	Belgium, Czech Republic, Italy, Norway The Czech Republic, Belgium, Norway, and Italy were selected to represent eastern, western, northern, and southern Europe. Sampling took place between spring 2010 and 2011 as part of the PERFOOD	Vegetables	Belgium: n = 21, DF 0% Czech Republic: n = 16, **DF NR, mean = 0.0028 ng/g** Italy: n = 15, DF 0%

Study	Location and Source	Food Types	PFNA Results
	study. Individual items were randomly selected in three national retail stores covering different brands or countries of origin. Of each item, three to ten single samples were combined to create one pooled sample per country. The following items were sampled: Root vegetables: carrots Bulb vegetables: onions Fruiting vegetables: tomatoes, courgettes, cucumbers, aubergine, peppers Brassica vegetables: cauliflower, cabbage, broccoli Leaf vegetables: lettuce and other salads, spinaches, chicory, pre-packed lettuce mix, pre-packed and minced frozen spinach Stem vegetables: asparagus, celery, fennel, cultivated mushrooms Starchy root tubers: potatoes, prepacked ready-to-cook pommes frites Legumes, beans, dried: peas, beans		Norway: n = 17, **DF NR, mean = 0.00226 ng/g** (MQL = 0.002–0.050 ng/g) *n represents number of composite samples *Values below the MQL were substituted with the MQL value
Hlouskova et al. (2013)	Belgium, Czech Republic, Italy, Norway Food products were randomly purchased in several nationwide supermarkets in four European regions during summer 2010. Within the sampling campaign, the collection of at least one food item per subcategory (meat, fish, hen eggs, milk and dairy products, and butter) in all four countries was acquired. Food items within each subcategory included the following: Meat: beef, canned pork meat, poultry, pork, pig/bovine liver, rabbit, and/or sheep/lamb Fish: farmed freshwater fish, farmed marine fish, and/or seafood) Hen eggs Milk and dairy products: ultra-high temperature whole cow milk, ultra-high temperature skimmed cow milk, cheese (yellow, Gouda/Edamer, etc.), and butter Samples were pooled within a respective food category but not across food groups.	Pooled milk/dairy products; meat; fish; hen eggs	**n = 50, DF 16%, mean, median (range) = 0.0295, 0.0253 (0.00503–0.0701) ng/g** (MQL = 0.005 ng/g for fish and seafood, meat, hen eggs, and cheese; 0.002 ng/mL for milk, and 0.025 ng/g for butter) *n represents number of pooled samples *Results not reported for individual food groups

Study	Location and Source	Food Types	PFNA Results
Domingo et al. (2012)	Spain (Catalonia) Foods purchased from 4 shops/stores of each of the 12 representative cities of Catalonia (Barcelona, l'Hospitalet de Llobregat, Vilanova I la Geltrú, Matarr, Sabadell, Terrassa, Girona, Tarragona, Reus, Tortosa, Lleida and Manresa) in September 2011. Shops/stores included local markets, small stores, supermarkets, and big grocery stores. Analyzed samples included 40 items: Meat and meat products: veal steak, loin of pork, chicken breast, steak of lamb, boiled ham, "Frankfurt"-type sausage, cured ham Vegetables and tubers: lettuce, tomato, potato, carrot Fresh fruits: apple, orange, banana Milk and dairy products: whole and semi-skimmed milk, yogurt, cheese I – low fat, cheese II – medium fat, cheese III – extra fat Cereals: French bread, pasta Pulses: lentils Industrial bakery: cookies Eggs: hen eggs Oils and fats: olive oil For each food item, two composite samples were prepared for analysis, where each composite sample consisted of 24 individual units. Only edible parts of each food item were included in the composites.	Meat; vegetables; grains; fruits; dairy; fats/other	**Dairy products: n = 2, DF NR, mean = 0.04 ng/g fw** Meat and meat products: n = 2, DF NR, mean = <0.079 ng/g fw Vegetables: n = 2, DF NR, mean = <0.37 ng/g fw Tubers: n = 2, DF NR, mean = <0.095 ng/g fw Fruits: n = 2, DF NR, mean = <0.096 ng/g fw Eggs: n = 2, DF NR, mean = <0.1 ng/g fw Milk: n = 2, DF NR, mean = <0.13 ng/g fw Cereals: n = 2, DF NR, mean = <0.033 ng/g fw Pulses: n = 2, DF NR, mean = <0.068 ng/g fw Oils: n = 2, DF NR, mean = <0.038 ng/g fw Industrial bakery: n = 2, DF NR, mean = <0.029 ng/g fw (LOD not reported) *n represents number of composite samples
Vestergren et al. (2012)	Sweden (Malmoe, Gothenburg, Uppsala, Sundsvall) Purchasing locations of the two largest retail chains in Sweden were selected in each of four major Swedish cities. All purchases were made in spring/summer of 1999, 2005, and 2010. In 2010, the study was limited to the largest retail chains in Uppsala located in close vicinity to Stockholm. An equal amount of each food group from each of the four cities was combined into one sample pool to provide a representative sample for the Swedish urban population.	Dairy; meat; grains; fruits; vegetables; fats/other	**Meat products:** **1999: n = 1, point = 0.0071 ng/g** **2005: n = 1, point = 0.0092 ng/g** 2010: n = 1, point = 0.0058 ng/g (estimated) (MDL = 0.0025 ng/g; MQL = 0.0062 ng/g) **Egg:** **1999: n = 1, point = 0.0022 ng/g** 2005: n = 1, point = 0.0056 ng/g (estimated) 2010: n = 1, point = <MDL (MDL = 0.0025 ng/g; MQL = 0.0062 ng/g) Fruits: 1999: n = 1, point = 0.0019 ng/g (estimated) 2005: n = 1, point = <MDL 2010: n = 1, point = <MDL

Study	Location and Source	Food Types	PFNA Results
			(MDL = 0.0013 ng/g; MQL = 0.0032 ng/g) Dairy products: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0034 ng/g; MQL = 0.0085 ng/g) Fats: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0030 ng/g; MQL = 0.0074 ng/g) Pastries: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0013 ng/g; MQL = 0.0031 ng/g) Cereal products: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0026 ng/g; MQL = 0.0063 ng/g) Vegetables: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0013 ng/g; MQL = 0.0031 ng/g) Potatoes: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0013 ng/g; MQL = 0.0032 ng/g) Sugar and sweets: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0013 ng/g; MQL = 0.0031 ng/g) Soft drinks, lemonade: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0006 ng/g; MQL = 0.0016 ng/g) *n represents number of composite samples *Paper reported that estimated concentrations are between MDL and MQL; however, there are some instances when the estimated concentrations are <MDL

Study	Location and Source	Food Types	PFNA Results
Noorlander et al. (2011)	The Netherlands Food products randomly purchased from several Dutch retail stores with nationwide coverage in November 2009. Food samples were ground, homogenized, and pooled for analysis. Food items within each subcategory included the following: Flour: whole wheat flour, flour Pork: sausage, slice of bacon, pork chop, bacon, minced meat rolled in bacon Eggs: chicken eggs Bakery products: cake, almond paste cake, biscuits, brown spiced biscuit, pie Vegetables/fruit: apple, orange, grape, banana, onion, carrot, beet, chicory or leek, tomato, cucumber, paprika, mushroom, cauliflower, broccoli, white cabbage, red cabbage, brussel sprout, spinach, endive, lettuce, French beans Cheese: gouda cheese, edammer cheese, cheese (>48% fat, less salt), cheese (>30% fat), brie cheese Beef: ground beef, beefburger, stewing steak, braising steak, minced steak Chicken/poultry: chicken leg, quarter chicken, chicken filet, chicken burger, collared chicken Butter: butter salt-free, salted, low-fat Milk: half cream milk Vegetable oil: margarine (solid/fluid), low-fat margarine, frying fat (vegetable), frying oil (vegetable), sunflower oil Industrial oil: low-fat margarine, frying fat (industrial), frying oil (industrial)	Meat; dairy; fruits and vegetables; grains; fats/other	**Butter: n = 1, point = 0.002 ng/g** **Cheese: n = 1, point = 0.007 ng/g** **Eggs: n = 1, point = 0.006 ng/g** **Pork: n = 1, point = 0.002 ng/g** **Beef: n = 1, point = 0.004 ng/g** **Chicken/poultry: n = 1, point = 0.001 ng/g** **Bakery products: n = 1, point = 0.001 ng/g** **Vegetables/fruit: n = 1, point = 0.001 ng/g** **Flour: n = 1, point = 0.015 ng/g** Milk: n = 1, <0.001 ng/g Vegetable oil: n = 1, <0.0001 ng/g Industrial oil: n = 1, <0.0003 ng/g (LOD not reported) *n represents number of composite samples
Jogsten et al. (2009)	Spain (Catalonia) Food samples purchased from local markets, large supermarkets, and grocery stores from two different areas of Tarragona Province, Catalonia in January and February 2008. The cities of Tarragona and Reus were sampled in the northern area and L'Ametlla de Mar and Tortosa in the southern area. For each food item, two composite samples were analyzed (one composite for the northern area and one for the southern area). Each composite was	Fruits and vegetables; meat; fats/other	Lettuce; raw, cooked, and fried meat (veal, pork, and chicken); fried chicken nuggets; black pudding; lamb liver; pate of pork liver; foie gras of duck; "Frankfurt" sausages; common salt: n = 2 for each food item, DF 0% (LOD not reported) *n represents number of composite samples *Values of ND were replaced with 1/2×LOD

Study	Location and Source	Food Types	PFNA Results
	formed of a minimum of six individual sub-samples of the same product.		
Ericson et al. (2008a)	Spain Food samples purchased from local markets, large supermarkets, and grocery stores within Tarragona County in July 2006. Food samples were randomly purchased with origin source not specified. Each of the food samples were duplicated and combined to analyze a composite sample. Composite samples included the following: Vegetables: lettuce, tomato, green bean, spinach Pulses: lentils, beans, chickpeas Cereals: rice, spaghetti, bread Pork: sausage, hot dogs, steak, hamburger, ham Chicken: breast, thighs, sausage Veal: steak, hamburger Lamb: steak Dairy products: three different kinds of cheese, yogurt, "petit-Swiss" creamy yogurt, cream caramel, custard Fruits: apple, orange, pear, banana Oil: olive oil, sunflower oil, corn oil Fats: margarine Eggs	Meat; dairy; fruits; vegetables; grains; fats/other	Vegetables (n = 2), pulses (n = 2), cereals (n = 2), pork (n = 2), chicken (n = 2), veal (n = 2), lamb (n = 2), eggs (n = 2), dairy products (n = 2), whole milk (n = 2), semi-skimmed milk (n = 2), fruits (n = 2), margarine (n = 2), oil (n = 2): DF 0% (LOD = 0.001–0.65 ng/g fw) *n represents number of composite samples
Papadopoulou et al. (2017)	Norway Participants of the A-TEAM project collected a duplicate portion of all consumed foods and drinks, prepared as for consumption, over two consecutive weekdays. Only the samples collected in the first day were analyzed. Sampling year not reported.	Solid foods: cereals and cereal products, dairy products (not milk), fish and seafood, meat and meat products, sugar and sugar products, fats and oils, vegetables and nuts, fruits, salty snacks, eggs, potatoes; liquid foods: coffee, tea and cocoa, milk, water, alcoholic beverages, soft drinks	**Solid foods:** **n = 61, DF 2%, median (range) = 0 (0–0.001) ng/g** (LOQ = not available) **Liquid foods:** **n = 61, DF 16%, median (range) = 0 (0–0.00057) ng/g** (LOQ = 0.00001 ng/g) *Concentrations <LOQ were replaced by 0
Dellatte et al. (2013)	Italy (Genoa, Brescia, Ferrara, Perugia, Portici) Ready-to-eat meals were collected at nursery and primary school canteens as they were delivered to children aged 3–10 years during the spring of 2011. One canteen was selected from each city, except for Genoa which was represented by two canteens because in one, the internal school regulation	Food composite meals	**Portici: n = 1, point = 0.0063 ng/g** Genoa (n = 2), Brescia (n = 1), Ferrara (n = 1), Perugia (n = 1): DF 0% (LOQ = 0.0060 ng/g) *n represents number of composite samples

Study	Location and Source	Food Types	PFNA Results
	forbade the use of anti-stick cookware. For each canteen, lunch meals related to five school days (from Monday to Friday) were weighed, pooled, and homogenized. Beverages were not included in the composites.		
Ghelli et al. (2019)	Italy Egg samples were collected from commercial laying hen farms in 2017. Sampling was based on geographical origin of the eggs and rearing system (e.g., organic, aviary system, battery cage and barn). A total of 132 eggs were collected and eggs were boiled. Four pools (containing three homogenized yolks) were created for each of the following groups (geographical origin, rearing system), for a total of 44 samples analyzed: Group A: Pavia, barn Group B: Verona, organic Group C: Forlì-Cesena, battery cage Group D: Bologna, barn Group E: Forlì-Cesena, battery cage Group F: Ravenna, aviary system Group G: Ravenna, aviary system Group H: Bologna, organic Group I: Romagna, battery cage Group L: Romagna, organic Group M: Romagna, barn	Fats/other	Group A: n = 4, DF 0% Group B: n = 4, DF 0% Group C: n = 4, DF[a] 25%, range = ND–traces Group D: n = 4, DF[a] 25%, range = ND–traces Group E: n = 4, DF 0% Group F: n = 4, DF 0% Group G: n = 4, DF 0% Group H: n = 4, DF 0% Group I: n = 4, DF 0% Group L: n = 4, DF 0% Group M: n = 4, DF 0% (LOD = 0.1 ng/g for all PFAS; LOQ = 0.25 ng/g for all PFAS) *Traces defined as value between LOD and LOQ
Johansson et al. (2014)	Sweden Eggs from 20 to 25 producers were collected each year from 1999 to 2010 within the Swedish National Food Agency's official food control program. Each sample consisted of a pool of 10–12 eggs from one producer. The pooled samples comprised eggs from both conventional and organic production. Information on the number of organic eggs sampled was not available. Fresh milk was sampled from the tanks of milk transport vehicles between 1999 and 2009 as part of the food control program. The tanks generally contained milk from ten dairy farms. In 2010, milk samples were taken from the milk storage tanks on individual dairy farms. Between 10–25 milk samples	Dairy; fats/other	**Hen's eggs:** **Total: n = 36, DF[a] 28%, range = ND–0.143 ng/g fw** **1999: n = 3, DF[a] 33%, range = <0.020–0.062 ng/g fw** **2000: n = 3, DF[a] 33%, range = <0.020–0.025 ng/g fw** **2001: n = 3, DF[a] 100%, mean[a] (range) = 0.052 (0.020–0.075) ng/g fw** **2002: n = 3, DF[a] 33%, range = <0.020–0.143 ng/g fw** **2003: n = 3, DF[a] 33%, range = <0.020–0.087 ng/g fw** **2004: n = 3, DF[a] 33%, range = <0.020–0.020 ng/g fw**

Study	Location and Source	Food Types	PFNA Results
	were collected each year. The milk samples were extracted in two different batches.		2005: n = 3, DF 0% 2006: n = 3, DF 0% **2007: n = 3, DF[a] 33%, range = <0.020–0.020 ng/g fw** 2008: n = 3, DF 0% 2009: n = 3, DF 0% **2010: n = 3, DF[a] 33%, range = <0.020–0.026 ng/g fw** (MDL = 0.020 ng/g fw; MQL = 0.097 ng/g fw) Cow's milk (1st batch): Total: n = 18, DF 0% (MDL = 0.0073 ng/g fw; MQL = 0.0156 ng/g fw) Cow's milk (2nd batch): Total: n = 18, DF 0% (MDL = 0.0073 ng/g fw; MQL = 0.0156 ng/g fw)
Eriksson et al. (2013)	Denmark (Faroe Islands) Locally produced food items sampled in 2011–2012. Packaged dairy products were supplied by Faroe Islands, Meginfelag Búnaðarmanna – dairy products included samples of milk, low fat (0.5%), semi-skimmed (1.5%), yoghurt with banana and pear (3.4% fat), low fat (0.9%) plain yoghurt, and créme fraiche (18% fat). Yoghurt with banana and pear was sampled from two production batches, and the low fat plain yoghurt and créme fraiche was sampled from one production batch. Potatoes were sampled from two different farms.	Dairy; fruits and vegetables	**Milk:** **Farmer (Innan Glyvur): n = 1, point = 0.061 ng/g ww** **Farmer (Havnardal): n = 1, point = 0.073 ng/g ww** **Diary (Faroe Island): n = 4, DF[a] 75%, range = <0.048–0.058 ng/g ww** Yogurt (n = 3), creme fraiche (n = 1), potato (n = 2): DF 0% (LOD = 0.0048 ng/L for milk; LOD = 0.0014 ng/g for dairy; LOD = 0.0017 ng/g for potato)
Still et al. (2013)	Germany Fourteen commercially available samples of various dairy products and raw milk were provided from a cooperating dairy.	Dairy	**Milk products:** Raw milk: n = 1, point = <0.0016 ng/g Fresh milk: n = 1, point = <0.0016 ng/g Fresh whole milk: n = 1, point = <0.0016 ng/g **UHT milk: n = 1, point = 0.0034 ng/g** Yogurt: Yogurt (0.1% fat): n = 1, point = <0.0016 ng/g Yogurt (3.8% fat): n = 1, point = <0.0016 ng/g **Cheese:** **Semihard cheese: n = 1, point = 0.0094 ng/g** **Semisoft cheese: n = 1, point = 0.0062 ng/g** **Soft cheese: n = 1, point = 0.0081 ng/g**

Study	Location and Source	Food Types	PFNA Results
			Other dairy products: Whey drink: n = 1, point = <0.0016 ng/g Butter milk: n = 1, point = <0.0016 ng/g Cream yogurt: n = 1, point = <0.0016 ng/g Cream: n = 1, point = <0.0016 ng/g **Butter: n = 1, point = 0.0047 ng/g (between LOQ and LOD)** (LOD = 0.0016 ng/g)
Vestergren et al. (2013)	Sweden (Kårsta) Study was conducted at a dairy cattle farm that was selected to represent a background contaminated agricultural area with no known point sources of PFAS in the proximity. The farm had no history of sewage sludge application to the pasture land. Milk samples were collected between November 2010 and April 2011 from a milk tank, where milk from the entire farm is stored after milking. Muscle, liver, and whole blood samples were obtained from five individual cows from the slaughterhouse on two different occasions (April and June 2011).	Meat; dairy	**Milk: n = 6, DF NR, mean = 0.0023 ng/mL** **Liver: n = 5, DF NR, mean = 0.016 ng/g** **Blood: n = 5, DF NR, mean = 0.014 ng/g** **Muscle: n = 5, DF NR, mean = 0.0045 ng/g** (MDL = 0.0012 ng/mL in milk; 0.0023 ng/g in liver; 0.0020 ng/g in blood; 0.0013 ng/g in muscle)
Falandysz et al. (2006)	Poland Eider duck samples were collected from the Gulf of Gdañsk in the Baltic Sea (south coast of Poland) in February 2003.	Meat	**n = 16, DF NR, mean, median (range) = 0.4, 0.32 (0.3–0.9) ng/mL** (LOD not reported) *Values reported for animal whole blood samples
Lankova et al. (2013)	Czech Republic Breastmilk samples were obtained from 50 women living in the Olomouc region from April to August 2010. The age of participating mothers ranged from 20 to 43 years. Six types of infant formula from the Czech retail market were also examined: one powdered formula for infants, two formulas for toddlers, one formula for babies with lactose intolerance, one formula for premature babies, and one soya-based formula for babies with non-milk diets. Sampling year not provided.	Fats/other; breastmilk	**Breastmilk:** **n = 50, DF (frequency of quantification) 48%, range = <0.006–0.015 ng/mL** (LOQ = 0.006 ng/mL) **Infant formula:** **n = 6, DF NR, maximum = 0.011 ng/g** (LOQ = 0.009 ng/g)
Llorca et al. (2010)	Spain (Barcelona)	Breastmilk; grains; fats/other	Breastmilk: n = 20, DF 0% (frequency of quantification) (MLOD = 0.0035 ng/mL; MLOQ = 0.0115 ng/mL)

Study	Location and Source	Food Types	PFNA Results
	Individual breastmilk samples were collected from 20 women in Barcelona city in 2008. All samples were collected within 40 days postpartum. Samples of powdered infant formula (three brands) and dry cereal baby food (two brands) were collected from retail stores (sample year not reported).		**Baby cereals:** **n = 2, DF[a] 100%, mean[a] (range) = 0.091 (0.044–0.138) ng/g** (MLOD = 0.017 ng/g; MLOQ = 0.0575 ng/g) **Milk infant formulas:** **n = 3, DF[a] 100%, mean[a] (range) = 0.166 (0.118–0.219) ng/g** (MLOD = 0.012 ng/g; MLOQ = 0.039 ng/g)
Abdallah et al. (2020)	Ireland (Dublin) Breastmilk samples obtained from mothers recruited from breastfeeding clinics at two Irish maternity hospitals. Mothers provided samples between 3 and 8 weeks postpartum. Mothers were up to 41 years of age, primiparas, in good health, and exclusively feeding one infant. Sampling year not reported.	Breastmilk	**n = 16, DF 69%, mean, median (range) = 0.026, 0.014 (<0.01–0.1) ng/mL** (LOQ = 0.01 ng/mL) *Values <LOQ assumed to equal DF × LOQ
Beser et al. (2019)	Spain (Valencian region) Breastmilk samples were collected from 14 Spanish women (aged 30–39 years) living in the Valencian region and recruited by the perinatology group of the Health Research Institute La Fe in Valencia. Milk samples were collected at different stages after birth during 2015.	Breastmilk	**n = 20, DF[a] 5%, mean, median (range) = 0.070, 0.070 (ND–0.070) ng/mL** (MDL = 0.008 ng/mL; LOQ = 0.066 ng/mL) *Median and mean values calculated from values above LOQ
Nyberg et al. (2018)	Sweden (Gothenburg, Stockholm) Breastmilk samples were collected between two weeks and three months after delivery from healthy native Swedish mothers, who were predominately non-smokers and primiparous. There were a total of 20 pooled samples analyzed from Stockholm (1972–2016), containing 9–116 individual samples per pool, and 11 pooled samples from Gothenburg (2007–2015), containing 5–11 individuals per pool. In addition, samples collected in 2012 (16 from Gothenburg and 20 from Stockholm) and in 2016 (10 from Stockholm) were analyzed individually.	Breastmilk	**Stockholm (pooled): n = 20, DF[a] 100%, mean[a] (range) = 0.018 (0.003–0.051) ng/mL** **Gothenburg (pooled): n = 11, DF[a] 100%, mean[a] (range) = 0.017 (0.011–0.024) ng/mL** **Stockholm (2012, individual): n = 20, DF[a] 100%, mean[a] (range) = 0.015 (0.003–0.030) ng/mL** **Gothenburg (2012, individual): n = 16, DF[a] 100%, mean[a] (range) = 0.017 (0.005–0.038) ng/mL** **Stockholm (2016, individual): n = 10, DF[a] 70%, range = <0.002–0.018 ng/mL** (MDL = 0.002 ng/mL)
Motas Guzmàn et al. (2016)	Spain (Murcia) Individual breastmilk samples were collected from 67 women in Portman Bay, Murcia, Spain in May 2014. The area was one of the most degraded zones	Breastmilk	**n = 67, DF 6%, mean, median (range) = 0.041, 0.040 (0.015–0.070) ng/mL** (LOD = 0.0002 ng/mL) *For calculations, values <LOD were considered equal to zero

Study	Location and Source	Food Types	PFNA Results
	of the Mediterranean Sea due to mining activities. Samples were collected a few weeks postpartum.		*Range reported only for detected values
Antignac et al. (2013)	France (Seine-Saint Denis, Ardèche, Isère, Loire, Savoie counties) Breastmilk samples collected from mothers participating in the ELFE pilot study. Sampling year not reported, though all mothers gave birth in October 2007. Mothers were contacted by phone one month after leaving the maternity and provided with instructions on breastmilk collection. Milk samples could be collected during several lactation sessions. On average, 15 aliquot samples of 10 mL were collected for each participant and pooled into one sample for analysis.	Breastmilk	**n = 48, DF 2%, range = <0.05–0.064 ng/mL** (LOD = 0.05 ng/mL)
Croes et al. (2012)	Belgium (Flanders) Breastfeeding mothers were recruited from 9 maternities in 24 rural communities in East and West Flanders and Flemish Brabant in May 2009 – June 2010. Breastmilk samples were collected between two and eight weeks after delivery and a subset was analyzed for perfluorinated compounds.	Breastmilk	**n = 40, DF 42.5%, median (10^{th}–90^{th} percentile) = <LOQ (<LOQ–0.02) ng/mL** (LOQ = 0.01 ng/mL) *For all calculations, values <LOQ were treated as ½ LOQ
Kärrman et al. (2007)	Sweden (Uppsala, Göteborg, Lund, Lycksele) Individual breastmilk samples from 12 women in Uppsala, Sweden were collected in 2004. Composite samples were created from breastmilk samples collected from 25–90 women each year between 1996 and 2004 and pooled into an annual composite sample. Donors originated from four regions in Sweden (Uppsala 1996 -2000, 2002; Göteborg 2001; Lund 2003; Lycksele 2003-2004). All samples were collected from primiparous women (19–41 years old) during the third week after delivery.	Breastmilk	**Individual samples:** **2004: n = 12, DF[a] 17%, mean, median (range) = 0.017, NA (<0.005–0.020) ng/mL** **Composite samples:** **1996: n = 1, point = 0.028 ng/mL** 1997: n = 1, point = <0.005 ng/mL 1998: n = 1, point = <0.005 ng/mL 1999: n = 1, point = <0.005 ng/mL **2000: n = 1, point = 0.019 ng/mL** 2001: n = 1, point = <0.005 ng/mL 2002: n = 1, point = <0.005 ng/mL 2003: n = 1, point = <0.005 ng/mL **2003–2004: n = 1, point = 0.020 ng/mL** *n represents number of composite samples (DL = 0.005 ng/mL)
Cariou et al. (2015)	France (Toulouse) Breastmilk samples obtained from female volunteers hospitalized between June 2010 and January 2013 for planned caesarean delivery. Samples were	Breastmilk	n = 61, DF 0% (LOD = 0.01–0.04 ng/mL; LOQ = 0.05 ng/mL)

Study	Location and Source	Food Types	PFNA Results
	collected between the fourth and fifth day after delivery.		
Pratt et al. (2013)	Ireland Pooled breastmilk samples were collected from 109 first-time mothers at four centers across Ireland. Sampling year not reported.	Breastmilk	n = 11, DF 0% (LOD = 0.5–5 ng/mL for all PFAS) *n represents number of pooled samples
Kärrman et al. (2010)	Spain (Catalonia) Breastmilk samples were collected from healthy primiparae mothers aged 30–39 years who lived in Tarragona County for at least the last five years. Babies were aged 41–60 days when milk samples were collected in 2007.	Breastmilk	n = 10, DF 0% (LOQ = 0.03 ng/mL)

Notes: DF = detection frequency; LOD = limit of detection; LOQ = limit of quantitation; 0.5×, 1×, or 2× = ½, 1, or 2 times the agronomic rate of biosolids application to meet nitrogen requirements of the crop; MDL = method detection limit; NR = not reported; PFAA = perfluoroalkyl acids; RL = reporting limit; WWTP = wastewater treatment plant.
Bold indicates detected levels of PFNA in food.
[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

C.3.2. Food Contact Materials

In 2011, FDA reached a voluntary agreement with industry to remove from the market certain PFAS grease-proofing agents used in fast food packaging. As such, the occurrence of PFNA in fast food packaging in the U.S. may be declining over time. The EPA identified two studies reporting PFNA in food contact materials (FCM) conducted in the U.S. Liu et al. (2014) analyzed the occurrence of PFAS in treated food contact paper and other consumer products purchased from local retailers and online stores in the United States between March 2007 and September 2011. All treated food contact paper was manufactured in the United States. PFNA was detected in 33% of samples (n = 9), with two of the detects below 10 ng/g and the third detect at 212 ng/g. Sinclair et al. (2007) sampled microwave popcorn bags purchased in 2005 in New York City that may have originated from international retailers. PFNA was detected in one of three brands of microwave popcorn bags, both before and after cooking.

Several peer-reviewed studies conducted in Europe also monitored for PFNA in food contact materials. PFNA was not detected in any FCMs including paper packaging, cardboard, coated bakery release papers for oven baking, paper filters for coffee, microwave popcorn bags, and an ice cream tub in two studies (Vavrouš et al., 2016; Moreta and Tena, 2013). Two additional studies had the majority of samples reported as <MDL and <LOD (Vestergren et al., 2015; Zafeiraki et al., 2014). Vestergren et al. (2015) measured levels of PFNA in a baking mold, paper plates, a baking cover, and a paper cup, with PFNA only being detected in one paper plate (0.022 $\mu g/m^2$) and one paper cup (0.030 $\mu g/m^2$). Zafeiraki et al. (2014) reported <LOD for beverage cups, an ice cream cup, fast food paper boxes, paper materials for baking, microwave bags (before and after cooking), and aluminum foil bags/wrappers, but reported a range of <LOD–4.97 ng/g for fast food wrappers. Kotthoff et al. (2015) analyzed both recent FCM samples and archived samples purchased before 2010. Recent samples ranged in concentrations from <LOQ to 1.0 ng/g and archived samples were reported in higher concentrations ranging from 68.5 ng/g to 478.2 ng/g. Schlummer et al. (2015) evaluated PTFE-coated pans and a ceramic surface pan under "overheating scenarios" (250–370°C) and found PFNA detected in all samples at a range of 1.60–869 ng/h for the PTFE-coated pans and 0.09–0.26 ng/h for the ceramic surface pan. This study also evaluated a sandwich maker and waffle iron under normal applications (185–221°C) with the sandwich maker having no PFNA detected, and the waffle maker being detected at a range of <LOD–0.40 ng/h. A study from Poland analyzed wrapping papers, breakfast bags, baking papers, and roasting bags from three different brands. They found PFNA in all three wrapping papers at a range of 0.04–0.11 pg/cm^2, all three breakfast bags at a range of 0.02–0.07 pg/cm^2, one baking paper sample at 0.02 pg/cm^2, and no roasting bag samples.

In an analysis performed at the Department of Food Analysis and Nutrition of the University of Chemistry and Technology in Prague, Czech Republic, PFNA was not detected in 42 samples of disposable food packaging and tableware purchased from six different European countries between May and December 2020 (LOQ = 1.7 mg/kg) (Straková et al., 2021).

Table C-6. Summary of Studies Reporting the Occurrence of PFNA in Food Contact Materials

Study	Location	Site Details	Results
United States			
Liu et al. (2014)	United States (unspecified)	Treated food contact papers, including microwave cooking bags, were purchased between October 2007 and September 2011 from local retailers and online stores in the United States. All products originated from the United States. A total of nine samples were purchased.	N = 9, DF[a] 33%, range = BDL–212 ng/g (DL not reported)
Europe			
Vavrouš et al. (2016)	Czech Republic	Real samples of paper FCM (11 with direct food contact and 4 with indirect food contact) were acquired from a market. Samples included paper packages of wheat flour (n = 2), paper bags for bakery products (n = 2), sheets of paper for food packaging in food stores (n = 2), cardboard boxes for packaging of various foodstuffs (n = 3), coated bakery release papers for oven baking at temperatures up to 220°C (n = 3), and paper filters for coffee preparation (n = 3). Sampling year and country of origin for products not reported.	N = 15, DF 0% (LOQ = 0.0047 mg/kg)
Kotthoff et al. (2015)	Germany (Schmallenberg)	Thirty-three random samples of recent individual paper-based FCMs collected in the first until the third quarter of 2010 in Germany. Individual samples were bought from local retailers or collected by coworkers of the involved institutes. Sampled products spanned all quality levels from entry level to cutting edge products. The age of the samples ranged from a few years to decades. Country of origin not reported. "Archived" older samples of FCMs (baking paper purchased before 2010) were collected from the staff of the institutes. The age of these samples ranged from a few years to decade. Country of origin not reported.	Recent samples: n = 33, DF 24%, median (range) = <LOQ (<LOQ–1.0) ng/g Archived samples: n = 3, DF NR, median (range) = 284.9 (68.5–478.2) ng/g (LOQ = 0.5 ng/g) *Concentrations <LOQ were considered as zero *For recent samples, Table 1 reports n = 33, the Results text reports n = 36 according to the actual sampling plan, and Table 13 in Supplemental Appendix 3 reports n = 12
Schlummer et al. (2015)	Germany	Three PTFE coated pans (one low price product and two pans from the medium and upper quality range) and one pan with a ceramic surface were purchased from German stores in 2012. Producers of all four pans claimed their products were free of PFOA. Additionally, four electrically heated non-stick FCMs (three waffle irons and one sandwich maker) were acquired from households of members of the	Overheating scenario: PTFE-coated pans: n = 8, DF[a] 100%, mean[a] (range) = 221.7 (1.60–869) ng/h Ceramic surface pan: n = 2, DF[a] 100%, mean[a] (range) = 0.175 (0.09–0.26) ng/h (SDL = <0.8 ng/h) Application at normal temperatures: Sandwich maker: n = 3, DF[a] 0%

Study	Location	Site Details	Results
		Fraunhofer-Institut IVV. The authors estimated the institute-acquired items were purchased between pre-2000 to 2006. Two scenarios were tested: an "overheating scenario" for the pans (250–370°C) and a "normal application" scenario for 1 hour for all other food contact materials (185–221°C). Each product was tested 2–3 times.	Waffle irons: n = 9, DF[a] 22%, range = <LOD–0.40 ng/h (SDL = 0.1–0.5 ng/h)
Surma et al. (2015)	Poland	Three different brands of FCMs (A, B, C), including wrapping papers (n = 3), breakfast bags (n = 3), baking papers (n = 3), and roasting bags (n = 3), were obtained from typical, commercially available food contact products. Sampling year and country of origin for products not reported.	Wrapping paper: Brand A: n = 1, point = 0.11 pg/cm^2 Brand B: n = 1, point = 0.04 pg/cm^2 Brand C: n = 1, point = 0.06 pg/cm^2 Breakfast bag: Brand A: n = 1, point = 0.07 pg/cm^2 Brand B: n = 1, point = 0.03 pg/cm^2 Brand C: n = 1, point = 0.02 pg/cm^2 Baking paper: Brand A: n = 1, point = <LOQ Brand B: n = 1, point = <LOQ Brand C: n = 1, point = 0.02 pg/cm^2 Roasting bag: Brand A: n = 1, point = ND Brand B: n = 1, point = ND Brand C: n = 1, point = ND (LOD = 0.01 pg/cm^2; LOQ = 0.02 pg/cm^2)
Vestergren et al. (2015)	Norway (Tromsø, Trondheim)	Five samples of FCMs (one baking mold, two paper plates, one baking cover, and one paper cup) were purchased from major retail stores in November 2012. Sampling campaign designed to evaluate consumer products in product categories that were previously found to contain PFAS residuals and that were representative of products imported from China in large quantities. Individual products randomly selected without prior knowledge of surface treatment with PFAS. Year of manufacture not reported.	Baking mold: n = 1, point = <0.005 µg/m^2 Paper plates: n = 1, point = 0.022 µg/m^2 Baking cover: n = 1, point = <0.005 µg/m^2 Paper plates: n = 1, point = <0.005 µg/m^2 Paper cup: n = 1, point = 0.030 µg/m^2 (MDL = 0.005 µg/m^2)
Zafeiraki et al. (2014)	Greece	Forty-two samples of FCMs made of paper, paperboard, or aluminum foil were obtained randomly from retailers. Their exact composition was not stated and there was no information about perfluorochemicals used in their manufacturing process or not. Beverage and ice cream cups,	Beverage cups: n = 8, DF 0% Ice cream cup: n = 1, DF 0% Fast food paper boxes: n = 8, DF 0% Fast food wrappers: n = 6, DF NR, range = <LOD–4.97 ng/g

Study	Location	Site Details	Results
		wrappers, and paper boxes were collected in Athens from October to December 2012 from popular Greek fast food chain restaurants, coffee shops, and multiplex cinemas. Other FCMs (muffin cups, baking papers, and microwave popcorn and rice bags) were purchased from large supermarkets. All products except for microwave popcorn and rice bags were manufactured in Greece. Sampled packaging materials included unused items and used items (i.e., contained food products). A microwave popcorn bag was also analyzed before and after cooking.	Paper materials for baking: n = 2, DF 0% Microwave bags: n = 3, DF 0% Before cooking: n = 1, point = <LOD After cooking: n = 1, point = <LOD Aluminum foil bags/wrappers: n = 14, DF 0% (LOD = 0.42 ng/g; LOQ = 1.25 ng/g)
Moreta and Tena (2013)	Spain	Food-contact packings (microwave popcorn bag, ice cream tub, cardboard cup) were purchased from different local supermarkets between late 2011 and early 2012. Six microwave bags were purchased representing five different generic brands and one name brand; these corresponded to salty (n = 3), salty and buttered (n = 2), and sweet (n = 1) popcorn. The cardboard cup was made of printed cardboard and lined with a polymer layer. The ice cream tub was also made of printed cardboard and lined inside and outside with a polymer layer.	Popcorn bags: n = 6, DF 0% Cardboard cup: n = 1, DF 0% Ice cream tub: n = 1, DF 0% (LOD = 0.9 ng/g)

C.3.3. Consumer Products

Since the 1950's, PFNA has been used in industrial and consumer products, including fabric and carpet protective coatings, paper coatings, insecticide formulations, and surfactants (NCBI, 2022b). PFNA and other long-chain PFAS are found in aqueous film-forming foams, cosmetics, dental floss, floor polish, leather, food packaging materials, lithium batteries, ski wax, treated apparel, work apparel for medical staff, pilots, and firefighters, and in hair treatment products (NCBI, 2022b). Based on limited testing, PFNA has been detected in rinsates from fluorinated high-density polyethylene (HDPE) containers used by one pesticide product supplier (USEPA, 2022a). PFNA is not a registered pesticide under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), and the EPA does not set a 40 CFR Part 180 pesticide tolerance in food and feed commodities for PFNA (US GPO, 2022). Maximum residue levels for PFNA were not found in the Global Maximum Residue Level Database (Bryant Christie Inc, 2022).

Two studies based in the U.S. were identified that analyzed PFNA concentrations in a range of consumer products, including children's nap mats, household carpet/fabric-care liquids, and textiles (Zheng et al., 2020; Liu et al., 2014) (Table C-7). Of these two studies, the consumer products evaluated are likely used by adults (e.g., floor waxes), can come into contact with both adults and children (e.g., treated upholstery), or the user was not specified (e.g., clothing). Zheng et al. (2020) determined the occurrence of ionic and neutral PFAS in the childcare environment (dust and nap mats). Samples of children's nap mats were collected from seven Seattle childcare centers (n = 26; 20 polyurethane foam, 6 vinyl cover samples). PFNA was detected in 36% of nap mat samples with a mean concentration of 0.19 ng/g. Half of the analyzed mats were purchased as new products and the other half were used. The authors reported that total PFAS levels in the new versus Used mats were not significantly different. Total PFAS levels in mat foam versus Mat covers were also similar. Based on these results, the authors suggested that indoor air was not the major source of PFAS in mats and that PFAS in mats could be associated with the manufacturing process. Liu et al. (2014) analyzed the occurrence of PFAS in consumer products (including pretreated carpeting, commercial carpet-care liquids, household carpet/fabric-care liquids, treated apparel, treated home textiles and upholstery, treated non-woven medical garments, treated floor waxes and stone-wood sealants, membranes for apparel, and thread-sealant tapes and pastes) purchased between March 2007 and September 2001 from local retailers and online stores in the United States. The consumer products originated from the United States, England, Vietnam, China, Thailand, El Salvador, Bangladesh, Dominican Republic, Malaysia, and Indonesia. PFNA was detected in 44% of nine pretreated carpeting samples (ranging from below the detection limit (BDL) to 236 ng/g); in 58% of 12 commercial carpet/fabric-care liquid samples (BDL–8,860 ng/g); in 15% of 13 household carpet/fabric-care liquid and foam samples (BDL–37.3 ng/g); in 60% of 15 treated apparel samples (BDL–235 ng/g); in 100% of six treated home textile and upholstery samples with a mean of 42.6 ng/g; in 56% of nine treated non-woven medical garment samples (BDL–334 ng/g); in 88% of eight treated floor wax and stone/wood sealant samples (BDL–2,740 ng/g); and in 75% of eight membranes for apparel samples (BDL–12.8 ng/g). PFNA was not detected in thread-sealant tapes and pastes (n = 6). Detection limits were not reported in the study.

Results for all of the identified non-U.S. studies are presented in detail in Table C-7 and are summarized here. One study from Germany did not detect PFNA in cleaners and wood glue, but among other consumer products such as nanosprays/impregnation sprays, gloves, and ski wax

found medians ranging from 2.8–10.7 ng/g (Kotthoff et al., 2015). Additional consumer products from this study found outdoor textiles, carpets, leather, and awing cloth median concentrations ranging between <LOQ–3.7 μg/m^2. Notably, ski wax had a maximum concentration of 678.0 ng/g, much larger than all other samples which fell within the range. Favreau et al. (2016) also evaluated concentrations in various consumer products in Switzerland, including cleaners, polishes, and "other" which all had detection frequencies of 0%. Impregnation products (n = 60) were the only type with detectable levels of PFNA (range of 100–1,900 ng/g; DF = 5%). This study also measured PFNA in firefighting foams, the results for which are presented in Table C-7. Three other studies investigated PFNA concentrations in firefighting foam, two of which reported detection frequencies of 0% (Dauchy et al., 2017; Laitinen et al., 2014), while Høisæter et al. (2019) reported PFNA concentrations to be 3,100 ng/L in a 1:100 diluted foam sample.

van der Veen et al. (2020) identified PFNA in 13 samples of outdoor water repellant clothing where samples were cut into two pieces with one piece receiving elevated UV radiation, humidity, and temperature for 300 hours and the other remaining untreated. The untreated samples ranged in concentrations from ND–1.1 μg/m^2 and the treated samples ranged in concentrations from ND–100 μg/m^2. Another study analyzed clothing samples, as well as furniture textiles and carpets, and found cotton/leather clothes ranging in concentration from <0.005–0.008 μg/m^2 (n = 4), however outdoor clothing was excluded (Vestergren et al., 2015). Random samples of furniture textiles ranged in concentrations from <0.005–0.097 μg/m^2 (n = 27) and carpets/mats ranged in concentrations from <0.005–0.077 μg/m^2 (n = 9).

Cosmetic products were evaluated for PFNA concentrations in samples that reported PFAS as an ingredient and those that did not (Schultes et al., 2018). Among the samples that did not report PFAS as an ingredient, including moisturizing cream, foundation, powder and eye shadow, and shaving cream, the detection frequency of PFNA was 0%. Among PFAS-containing samples (the same type of cosmetics as previously noted with the addition of an eye pencil), moisturizing cream, eye pencil, and shaving cream reported 0% detection frequency. However, foundation samples (n = 6) ranged in concentrations from <3.45–651 ng/g and powder and eye shadow samples (n = 10) ranged in concentrations from <3.45–47.2 ng/g. Schlummer et al. (2015) evaluated the concentration of PFNA in three types of PTFE coated products: electric irons (n = 9), iron sole plates (n = 3), and hair straighteners (n = 3) under "normal applications" (180–230°C). Electric irons had a range of <LOD–0.10 ng/h, iron sole plates had a range of <LOD–0.03 ng/h, and hair straighteners had a detection frequency of 0%.

Table C-7. Summary of PFNA Consumer Product Data

Study	Location	Site Details	Results
United States			
Zheng et al. (2020)	United States (Seattle, Washington)	Children's nap mat samples (n = 26, finely cut) from seven Seattle childcare centers, including polyurethane foam (n = 20) and vinyl cover (n = 6) samples. Sampling year not reported.	N = 26, DF 36%, mean, median (range) = 0.19, 0.11 (ND–0.65) ng/g (MDL = 0.08 ng/g)
Liu et al. (2014)	United States (unspecified)	Consumer products commonly used indoors were purchased between March 2007 and September 2011 from local retailers and online stores in the United States. The samples analyzed for PFCAs included pre-treated carpeting, commercial carpet/fabric-care liquids, household carpet/fabric-care liquids and foams, treated apparel, treated home textile and upholstery (i.e., mattress pads), treated non-woven medical garments, treated floor waxes and stone-wood sealants, membranes for apparel, and thread-sealant tapes and pastes. The products originated from the United States, England, Vietnam, China, Thailand, El Salvador, Bangladesh, Dominican Republic, Malaysia, and Indonesia.	Pre-treated carpeting: n = 9, DF[a] 44%, range = BDL–236 ng/g Commercial carpet/fabric-care liquids: n = 12, DF[a] 58%, range = BDL–8,860 ng/g Household carpet/fabric-care liquids and foams: n = 13, DF[a] 15%, range = BDL–37.3 ng/g Treated apparel: n = 15, DF[a] 60%, range = BDL–235 ng/g Treated home textile and upholstery: n = 6, DF[a] 100%, mean[a] (range) = 42.6 (3.80–213) ng/g Treated non-woven medical garments: n = 9, DF[a] 56%, range = BDL–334 ng/g Treated floor waxes and stone-wood sealants: n = 8, DF 88%, range = BDL–2,740 ng/g Membranes for apparel: n = 8, DF[a] 75%, range = BDL–12.8 ng/g Thread-sealant tapes and pastes: n = 6, DF[a] 0% (DL not reported)
Europe			
van der Veen et al. (2020)	Sweden (unspecified)	Samples of durable water repellent outdoor clothing collected from six suppliers from the outdoor textile industry in Sweden (one pair of outdoor trousers, six jackets, and six fabrics for outdoor clothes*). Each sample was cut into two pieces – one exposed to elevated UV radiation, humidity, and temperature for 300 hours (assumed lifespan of outdoor clothing) and one untreated (not aged). Sampling year not reported. Year of manufacturing not reported for nine of the 13 samples; the remaining four samples (samples 4–7)	Point values presented as before aging (n =1), after aging (n = 1) Sample 1 (outdoor trousers): ND, 1.3 µg/m^2 Sample 2 (fabric for jacket): 0.05 µg/m^2, 0.14 µg/m^2 Sample 3 (fabric for jacket): ND, 0.13 µg/m^2 Sample 4 (men's jacket): ND, ND Sample 5 (men's jacket): 0.08 µg/m^2, 12 µg/m^2

Study	Location	Site Details	Results
		reported a manufacturing year of 2012/2013. Country of origin not reported. *The breakdown of the 13 items of outdoor clothing is reported differently in Section 2.2 and Table 1 of the article. Section 2.2 reports one pair of outdoor trousers, seven jackets, four fabrics for outdoor clothes, and one outdoor overall. Table 1 shows one pair of outdoor trousers, six jackets, and six fabrics for outdoor clothes	Sample 6 (fabric for outdoor clothes): 0.29 µg/m^2, 100 µg/m^2 Sample 7 (children's jacket): ND, ND Sample 8 (jacket): 1.1 µg/m^2, 3.5 µg/m^2 Sample 9 (fabric for outdoor clothes): ND, 0.63 µg/m^2 Sample 10 (fabric for outdoor clothes): ND, 0.15 µg/m^2 Sample 11 (fabric for outdoor clothes): ND, 0.49 µg/m^2 Sample 12 (fabric for outdoor clothes): ND, 0.11 µg/m^2 Sample 13 (fabric for outdoor clothes): ND, ND (LOD = 0.02–0.1 µg/m^2 for ionic PFAS)
Schultes et al. (2018)	Sweden (unspecified)	Thirty-one cosmetic products from five product categories (moisturizing cream, foundation, eye pencil, powder and eye shadow, shaving foam) purchased from the Swedish market in 2016–2017. Cosmetic products were selected based on (i) the 2015 KEMI survey which reported the most frequently reported PFAS in cosmetic products and (ii) a database of ingredient lists compiled by the Swedish Society for Nature Conservation. Twenty-four products listing nine different PFAS as active ingredients were purchased. In addition, seven products which did not list PFAS in their ingredients were also purchased from the same stores as control samples. Year of manufacture and country of origin not reported.	Control: Moisturizing cream: n = 1, DF 0% Foundation: n = 3, DF 0% Powder and eye shadow: n = 2, DF 0% Shaving cream: n = 1, DF 0% PFAS-containing: Moisturizing cream: n = 6, DF 0% Foundation: n = 6, DF[a] 16%, range = <3.45–651 ng/g Eye pencil: n = 1, DF 0% Powder and eye shadow: n = 10, DF[a] 30%, range = <3.45–47.2 ng/g Shaving cream: n = 1, DF 0% (LOD = 3.45 ng/g)
Favreau et al. (2016)	Switzerland (national)	Liquid consumer products, including impregnation agents, cleansers, polishes, lubricants, miscellaneous items, and commercial AFFFs purchased in 2012 and 2013 from stores and supermarkets throughout Switzerland. Products were purchased from 82 different producers and were selected based on their susceptibility to contain PFAS according to previous screenings. Miscellaneous "other" products included foam-suppressing agents for the chromium industry, paints, ski wax, inks, and tanning substances. AFFFs were divided into two sets based on the sampling source. AFFF set 1 was derived from stock solution in	Impregnation products: n = 60, DF 5%, mean, median (range) = 800, ND (100–1,900) ng/g Cleansers: n = 24, DF 0% Polishes: n = 18, DF 0% Others: n = 23, DF 0% AFFF set 1: n = 27, DF 70%, mean, median (range) = 3,400, 200 (100–37,900) ng/g AFFF set 2: n = 35, DF 3%, mean, median (range) = 200, ND (200–200) ng/g

Study	Location	Site Details	Results
		fire installation of industrial sites storing chemicals and petroleum products and samples may be the result of multiple AFFF fillings over the years (1990–2010 was the last documented filling date). AFFF set 2 came from commercially available AFFFs between 2012 and 2013 from six producers.	(LOQ = 0.5 ng/mL) *Mean and range values only include samples where PFNA was detected *ND treated as 0 for median calculations
Kotthoff et al. (2015)	Germany (Schmallenberg)	Forty-nine random samples of consumer products collected in the first until the third quarter of 2010 in Germany, including outdoor textiles, carpets, cleaning agents, impregnating agents, leather samples, and ski waxes. Individual samples were bought from local retailers or collected by coworkers of the involved institutes or local clubs (e.g., ski waxes from local skiing club). Sampled products spanned all quality levels from entry level to cutting edge products. The age of the samples ranged from a few years to decades. Country of origin not reported.	Cleaner: n = 6, DF = 0% Wood glue: n = 1, DF = 0% Nanosprays and impregnation sprays: n = 3, DF = 56%, median (maximum) = 2.8 (8.0) ng/g Outdoor textiles: n = 3, DF = 67%, median (maximum) = 1.0 (8.3) µg/m^2 Carpet: n = 6, DF = 20%, median (maximum) = <LOQ (1.2) µg/m^2 Gloves: n = 3, DF = 100%, median (maximum) = 2.9 (5.7) ng/g Ski wax: n = 13, DF = 73%, median (maximum) = 10.7 (678.0) ng/g Leather: n = 13, DF = 92%, median (maximum) = 1.9 (1.9) µg/m^2 Awing cloth: n = 1, DF = 100%, median (maximum) = 3.7 (3.9) µg/m^2 (LOQ = 0.5 ng/g or 0.5 µg/m^2) *Concentrations <LOQ were considered as zero
Schlummer et al. (2015)	Germany	Three types of PTFE coated products (three electric irons, one iron sole plate, and one electric hair straightener) were acquired from households of members of the Fraunhofer-Institut IVV. The sampling year was not provided but assumed to be 2012 based on the collection of other food contact materials. The authors estimated the institute-acquired items were purchased between pre-2008 to 2013. The products were tested during a “normal application” scenario for 1 hour at 180–230°C. Each product was tested three times.	Electric iron: n = 9, DF[a] 44%, range = <LOD–0.10 ng/h Iron sole plate: n = 3, DF[a] 66%, range = <LOD–0.03 ng/h Hair straightener: n = 3, DF[a] 0% (SDL = 0.1–0.5 ng/h)

Study	Location	Site Details	Results
Vestergren et al. (2015)	Norway (Tromsø, Trondheim)	Samples of furniture textile (samples included baby-related items such as baby mattress, baby blanket, and baby bed cover), carpet, and clothing samples were purchased from three major retail stores during November 2012–February 2013. Sampling campaign designed to evaluate consumer products in product categories that were previously found to contain PFAS residuals and that were representative of products imported from China in large quantities. Individual products randomly selected without prior knowledge of surface treatment with PFAS. Outdoor clothing was excluded. Year of manufacture not reported.	Furniture textiles: n = 27, DF[a] = 15%, range = <0.005–0.097 µg/m^2 Carpets/mats: n = 9, DF[a] = 33%, range = <0.005–0.077 µg/m^2 Cotton/leather clothes: n = 4, DF[a] = 50%, range = <0.005–0.008 µg/m^2 (MDL = 0.005 µg/m^2)
Høisæter et al. (2019)	Norway (unspecified)	AFFF concentrate, containing PFOS as the main PFAS, from the same supplier as assumed to have been used historically at a firefighting training facility where AFFF containing PFOS was used extensively from the early 1990s until 2001 when it was phased out and replaced by fluorotelomer containing AFFF until 2011. Concentrations reported in 1:100 diluted AFFF. Number of samples not reported but assumed to be 1. Sampling year assumed to be 2016 based on when samples were collected for groundwater and soil.	1:100 diluted foam: n = 1, point = 3,100 ng/L Relative contribution to total PFAS = <0.1% (LOQ = 0.3 ng/L)
Dauchy et al. (2017)	France (unspecified)	Nine firefighting foam concentrates were provided by a professional user and included alcohol-resistant film-forming fluoroprotein foams (n = 5), alcohol-resistant aqueous film-forming foams (n = 2), film-forming fluoroprotein foams (n = 1), and fluorine-free foams (n = 1). These concentrates were manufactured after 2002 by four different manufacturers. Concentrate sampling year was not reported, though water sampling in the same study was conducted in November 2014.	N = 9, DF (frequency of quantification) 0% (LOQ = 5,000 µg/L)
Laitinen et al. (2014)	Finland (Oulu)	Sthamex 3% AFFF liquid, manufactured in Germany and available commercially in Finland, used by firefighters during training in the simulation of aircraft accidents. Samples collected in 2010.	N = NR, DF (frequency of quantification) 0% *Low concentrations of PFNA were detected, but were below LOQ (LOQ = 20 µg/mL)
Origin Unspecified			
Bečanová et al. (2016)	Not specified	One hundred twenty-six samples of (1) household equipment (textiles, floor coverings, electrical and electronic equipment (EEE), and plastics; includes	Household equipment: n = 55, DF[a] 2%, range = <MQL–9.85 ng/g

Study	Location	Site Details	Results
		children-related items such as teddy bear filling, teddy bear cover, and plush); (2) building materials (oriented strand board, other composite wood and wood, insulation materials, mounting and sealant foam, I materials, polystyrene, air conditioner components); (3) car interior materials; and (4) wastes of electrical and electronic equipment (WEEE) purchased (for new materials) or collected from various sources (for older and used materials). Production year ranged from 1981 to 2010. Origin of production and location and year of purchase/collection not reported.	Building materials: n = 54, DF[a] 0% Car interior materials: n = 10, DF[a] 0% WEEE: n = 7, DF[a] 14%, range = <MQL–0.221 ng/g (IQL = 27 pg/mL; MQL = 0.16 ng/g)
Gremmel et al. (2016)	Not specified	Sixteen outdoor jackets (15 outdoor jackets and one working jacket) purchased during August 2011 to March 2012. Besides the working jacket and two other jackets (one arrived unpacked in shop and had been on sale for four weeks while the other had been on sale since February 2010), all other jackets were new and packed in a plastic shell. Jackets were selected considering factors such as origin of production (primarily Asia, with some origins not specified), price, market, and textile. Location of purchase and year of manufacture not reported.	J0: n = 2, DF NR, mean = <LOQ J1: n = 2, DF NR, mean = 0.14 ng/g J2: n = 2, DF NR, mean = 1.54 ng/g J3: n = 2, DF NR, mean = 0.33 ng/g J4: n = 2, DF NR, mean = 0.35 ng/g J5: n = 2, DF NR, mean = 0.22 ng/g J6: n = 2, DF NR, mean = 0.40 ng/g J7: n = 2, DF NR, mean = 0.33 ng/g J8: n = 2, DF NR, mean = 0.34 ng/g J9: n = 2, DF NR, mean = 1.17 ng/g J10: n = 2, DF NR, mean = 0.29 ng/g J11: n = 2, DF NR, mean = 6.87 ng/g J12: n = 2, DF NR, mean = 0.18 ng/g J13: n = 2, DF NR, mean = 0.47 ng/g J14: n = 2, DF NR, mean = 22.9 ng/g J15: n = 2, DF NR, mean = 62.9 ng/g (LOD = 0.05 ng/mL; LOQ = 0.01 µg/m^2)

Notes: BDL = below detection limit; DF = detection frequency; DL = detection limit; MDL = method detection limit; ND = not detected; PFCA = Perfluoroalkyl carboxylic acids.
[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

C.3.4. Indoor Dust

In a Wisconsin Department of Health Services study, Knobeloch et al. (2012) examined levels of 16 perfluoroalkyl chemicals in vacuum cleaner dust from 39 Wisconsin homes across 16 counties in March and April 2008 (Table C-8). Samples from these homes built between 1890 and 2005 were collected during a pilot study to assess residential exposure to persistent contaminants found in the Great Lakes Basin. PFNA was found in all samples at a median concentration of 12 ng/g. The number of rooms with synthetic, wall-to-wall carpeting and the square footage of the homes were both significantly positively correlated with dust concentrations of PFNA. Based on the results of this study, the authors suggested that perfluoroalkyl chemicals may be ubiquitous contaminants in U.S. homes. In an EPA study of 112 indoor dust samples collected from vacuum cleaner bags from homes and daycare centers in North Carolina and Ohio in 2000–2001 (EPA's Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) study), samples were collected from 102 homes and 10 daycare centers in North Carolina (49 homes, 5 daycare centers) and Ohio (53 homes, 5 daycare centers) (Strynar and Lindstrom, 2008). Results were not reported separately for homes and daycares. Overall, PFNA was detected in 42.9% of all samples (n = 112) with mean and median concentrations of 22.1 ng/g and 7.99 ng/g, respectively. The authors concluded that the study measured perfluorinated compounds in house dust at levels that may represent an important pathway for human exposure.

Additional peer-reviewed studies based in the U.S. were identified that evaluated the occurrence of PFNA and other PFAS in dust of indoor environments, primarily in homes, as well as in schools, childcare facilities, offices, and vehicles (Zheng et al., 2020; Scher et al., 2019; Byrne et al., 2017; Karásková et al., 2016; Wu et al., 2014; Fraser et al., 2013; Kato et al., 2009) (Table C-8). For those studies with results stratified for U.S. homes, PFNA levels and detection frequencies were lowest in a study of remote Alaska Native villages (35% detection, median below 0.2 ng/g), while in other U.S. locations, PFNA was detected in at least 65% of samples (some studies reporting 100% detection) at widely varying mean and median levels across the studies (from approximately 4 ng/g to 70 ng/g). Few studies sampled childcare centers, vehicles, and offices, and none of the reviewed studies reported measurements in other microenvironments (e.g., public libraries, universities).

Several studies reported results from dust samples collected only from homes (Scher et al., 2019; Byrne et al., 2017; Wu et al., 2014), with one study sampling from locations near a PFAS production facility. Scher et al. (2019) evaluated indoor dust in 19 homes in Minnesota within a groundwater contamination area (GCA) in the vicinity of a former 3M PFAS production facility. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota PWS or a private well previously tested and shown to have detectable levels of PFOA or PFOS. In the house dust samples, collected from July to September 2010, the detection frequencies for PFNA were 68% and 95% for entryways to the yard and interior living spaces such as the family or living rooms, respectively (n = 19 each), with median concentrations of 9.7 ng/g and 26 ng/g, respectively. PFAS concentrations in both sampling locations were higher than corresponding soil concentrations, suggesting that interior sources were the main contributors to PFAS in house dust.

Byrne et al. (2017) assessed exposure to PFNA and other PFAS among residents of two remote Alaska Native villages on St. Lawrence Island. PFAS concentrations were measured in dust collected from the surfaces of floors and furniture of 49 homes on St. Lawrence Island during February–April of 2013 and 2014. Residents were asked not to sweep or dust for one week prior to sampling. The authors described the overall PFAS levels in dust samples as "on the lower end of those reported worldwide in other studies." PFNA was detected in 35% of all samples (n = 49) with a median value below the LOD (0.1 ng/g–0.2 ng/g). Wu et al. (2014) measured concentrations of five PFCs in residential dust in California in 2008–2009. Dust samples were collected from the carpet or area rug in the main living area of the home. Homes of parents with young children and homes with older adults were differentiated to characterize the relationship between serum concentrations of PFCs and several other factors, including PFC concentrations in residential dust. PFNA was detected in 65% of samples from households with young children in Northern California (n = 82), with mean and median concentrations of 67.4 ng/g and 9.70 ng/g, respectively. PFNA was detected in 72% of samples from households of older adults in central California (n = 42), with mean and median concentrations of 58.5 ng/g and 11.85 ng/g, respectively.

Apart from the information reported by Strynar and Lindstrom (2008), one other study included childcare centers in the locations sampled (Zheng et al., 2020). Zheng et al. (2020) collected dust samples from seven childcare centers in Seattle, Washington (n = 14) and one childcare facility in West Lafayette, Indiana (n = 6 across six rooms); the sampling year was not reported. The included childcare facilities consisted of several building types, including multiple classrooms, a former church, and a former home. Because centers were vacuumed and mopped daily, dust samples were obtained from elevated surfaces (shelving, tops of bookcases/storage cubbies) along with floor dust. PFNA was detected in all samples at mean and median concentrations of 3.2 ng/g and 1.7 ng/g, respectively.

One study evaluated PFNA levels in vehicles and offices, in addition to homes. Fraser et al. (2013) collected dust samples between January and March 2009 from three microenvironments of 31 individuals in Boston, Massachusetts (offices (n = 31), homes (n = 30), and vehicles with sufficient dust for analysis (n = 13)). Study participants worked in separate offices located across seven buildings, which were categorized as Building A (n = 6), Building B (n = 17), or Other (n = 8). Building A was a newly constructed (approximately one year prior to study initiation) building with new carpeting and new upholstered furniture in each office; Building B was a partially renovated (approximately one year prior to study initiation) building with new carpeting throughout hallways and in about 10% of offices. The other buildings had no known recent renovation occurred. Study offices were not vacuumed during the sampling week and participants were asked not to dust or vacuum their homes and vehicles for at least one week prior to home sampling. PFNA was detected in 94%, 67%, and 85% of office, home, and vehicle dust samples, respectively, with geometric mean concentrations of 63.0 ng/g, 10.9 ng/g, and 14.7 ng/g, respectively. Geometric mean PFNA concentrations were statistically significantly higher in offices compared to homes and vehicles. The study also observed that PFNA concentration in house dust was significantly predictive of PFNA serum concentration.

Two studies evaluated dust samples collected across multiple continents (Karásková et al., 2016; Kato et al., 2009). Karásková et al. (2016) examined PFAS levels in house dust collected between April and August 2013 from the living rooms and bedrooms of 14 homes in the United

States, 15 homes in Canada, and 12 homes in the Czech Republic (locations unspecified). PFNA was detected in all U.S. samples (n = 20) at mean and median concentrations of 10.9 ng/g and 3.9 ng/g, respectively. The authors reported PFNA concentrations were significantly higher in North America compared to the Czech Republic, which they indicated may suggest a faster shift from long-chain PFAS to their shorter-chain homologues in Europe than in North America. Overall, no significant differences in total PFAS concentrations were found between the bedroom and living room in the same household although significant relationships were found based on type of floors, number of residents, and age of the house. A second multicontinental study (Kato et al., 2009) measured PFC concentrations in 39 household dust samples collected in 2004 from homes in the United States (Atlanta, GA) (n = 10), United Kingdom (n = 9), Germany (n = 10), and Australia (n = 10). Across all 39 homes, PFNA was detected in 25.6% of samples with a median concentration below the LOQ (2.6 ng/g). The authors did not report stratified PFNA data by country.

Results for all of the identified non-U.S. studies are presented in detail in Table C-8 and are summarized here. One study conducted in Canada evaluated dust samples in homes by using the vacuum cleaners of participants and found PFNA detected in 69% of samples (n = 48) with a mean concentration of 0.71 ng/g (Makey et al., 2017). Studies conducted in Europe evaluated indoor dust concentrations from homes, schools, workplaces, and/or cars. One of these studies, Huber et al. (2011) from Norway, also sampled a storage room which contained chemicals and highly contaminated samples and found an elevated concentration at 43.4 ng/g when compared to other locations within the study. Other samples within the study include living rooms, carpets, sleeping rooms, sofas, and offices and concentrations ranged between 0.2–26.7 ng/g. Four studies sampled floor dust exclusively in homes with results ranging from not detected to 37 ng/g (de la Torre et al., 2019; Winkens et al., 2018; Padilla-Sánchez and Haug, 2016; Jogsten et al., 2012). Another study in Norway evaluated dust from elevated surfaces such as bookshelves and windowsills (n = 41) and reported a range of 3.9–92 ng/g, notably higher than other studies (Haug et al., 2011). One study from Sweden assessed 20 dust samples from elevated surfaces in preschools and reported a median and 95th percentile of 1.09 ng/g and 56.0 ng/g, respectively (Giovanoulis et al., 2019). Harrad et al. (2019) evaluated dust concentrations from living rooms, cars, and classrooms in Ireland ranging from <0.05 ng/g to 14 ng/g and, notably, from offices which had concentrations ranging from <0.05 ng/g to 120 ng/g. A study in Belgium evaluated randomly selected homes (n = 43) and offices (n = 10) and found dust samples containing median (95th percentile) values of 0.1 (2.1) ng/g dw and 0.4 (62) ng/g dw, respectively (D'Hollander et al., 2010).

Table C-8. Summary of PFNA Indoor Dust Data

Study	Location	Site Details	Results
United States			
Scher et al. (2019)	United States (Twin Cities metropolitan region, Minnesota)	Nineteen homes in three cities within a GCA near former 3M PFAS production facility as well as from three homes in the Twin Cities Metro outside the GCA. Dust samples collected from an entryway to the yard and from an interior living space (e.g., family room, living room) in each home in July–September 2010. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. Results were not reported for homes outside the GCA.	Entryway: n = 19, DF 68%, median (range) = 9.7 (<RL–1,000) ng/g Living room: n = 19, DF 95%, median (range) = 26 (<RL–450) ng/g (RL = 5 ng/g)
Byrne et al. (2017)	United States (St. Lawrence Island, Alaska)	Dust samples collected from the surfaces of floors and furniture from 49 homes during February–April of 2013 and 2014. Participants were asked not to sweep or dust for one week prior to sampling.	n = 49, DF 35%, median (95th percentile) = <LOD (1.93) ng/g (MDL = 0.1–0.2 ng/g for all PFAS)
Wu et al. (2014)	United States (Central Valley area, California)	Distributions of PFC dust concentrations were determined for households with young children in Northern California (n = 82) and households of older adults in central California (n = 42). Dust samples were collected in 2008–2009 from the carpet or area rug in the main living area of the homes. Homes of parents with young children and homes with older adults were differentiated to characterize the relationship between serum concentrations of PFCs and PFC concentrations measured in residential dust.	Parents of young children: n = 82, DF 65%, mean, median (range) = 67.4, 9.70 (ND–1,910) ng/g Older adults: n = 42, DF 72%, mean, median (range) = 58.5, 11.85 (ND–883) ng/g (LOD = 0.10 ng/mL) *Data below LOQ replaced by LOD/√2
Knobeloch et al. (2012)	United States (Great Lakes Basin, Wisconsin)	Dust samples were collected by the Wisconsin Department of Health Services from 39 Wisconsin homes across 16 counties in March–April 2008. Vacuum bags were collected or bagless vacuums were emptied into sterilized glass jars. Homes were built between 1890 and 2005.	n = 39, DF 100%, median (range) = 12 (1.3–280) ng/g (RL = 1 ng/g)
Zheng et al. (2020)	United States (Seattle, Washington; West Lafayette, Indiana)	Seven childcare centers in Seattle (14 samples) and one center in Lafayette (6 samples); sampling year not reported. Since all centers were vacuumed and mopped daily, dust samples from elevated surfaces	n = 20; DF 100%, mean, median (range) = 3.2, 1.7 (0.11–13) ng/g (MDL = 0.08)

Study	Location	Site Details	Results
		(shelving, tops of bookcases/storage cubbies) were collected along with floor dust in the same sample.	
Strynar and Lindstrom (2008)	United States (North Carolina; Ohio)	Dust samples from vacuum cleaner bags were obtained in 2000–2001 during the EPA's Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) study from North Carolina (49 homes, 5 daycare centers) and Ohio (53 homes, 5 daycare centers). Vacuum cleaner bags were only collected if available at each site.	n = 112; DF 42.9%, mean, median (maximum) = 22.1, 7.99 (263) ng/g (LOQ = 11.3 ng/g) *Values below the LOQ assigned a value of LOQ/√2
Fraser et al. (2013)	United States (Boston, Massachusetts)	Dust samples were collected in January–March 2009 from offices (n = 31), homes (n = 30), and vehicles (n = 13) of 31 individuals. Study participants worked in separate offices located across seven buildings, which were categorized into Building A, Building B, and Other. Six samples were collected from Building A, a newly constructed (approximately one year prior to study initiation) building with new carpeting and new upholstered furniture in each office. Seventeen samples were collected from Building B, a partially renovated (approximately one year prior to study initiation) building with new carpeting throughout hallways and in about 10% of offices. Eight samples were collected from the other five remaining buildings where no known recent renovation occurred. Study offices were not vacuumed during the sampling week and homes and vehicles were not vacuumed for at least one week prior to sampling. Entire accessible floor surface areas and tops of immovable furniture were vacuumed in offices and the main living area of homes. Entire surface areas of the front and back seats of vehicles were vacuumed. Number of home dust samples was reduced to 30 because 1 participant lived in a boarding house with no main living area. Sufficient mass of dust for analysis was available from only 13 vehicles.	Homes: n = 30, DF 67%, GM (range) = 10.9 (6.21–1,420 ng/g) Offices: n = 31, DF 94%, GM (range) = 63.0 (10.9–639) ng/g Vehicles: n = 13, DF 85%, GM (range) = 14.7 (4.95–101 ng/g) (LOQ = 5 ng/g) *GM calculated by replacing values <LOQ with LOQ/√2 *Range of detected values reported
Canada			
Makey et al. (2017)	Canada (Vancouver)	Dust samples were collected from a subset of Chemicals, Health, and Pregnancy (CHirP) Study participants' vacuum cleaners in 2007–2008; indoor air and serum samples were also collected. Vacuum cleaner dust was sampled by collecting whole vacuum	n = 48, DF 69%, GM = 0.71 ng/g (DL = 0.06 ng/g)

Study	Location	Site Details	Results
		cleaner bags or subsampling bagless or central vacuums.	
Europe			
de la Torre et al. (2019)	Spain (unspecified), Belgium (unspecified), Italy (unspecified)	Sixty-five homes belonging to the partners of Test-Achats (Belgium), Altroconsumo (Italy), and OCU Ediciones SA (Spain). Home occupants vacuumed the entire floor of their home from September 2016 to January 2017 and vacuum bags were collected	Total: n = 65, DF 46%, median (range) = 0.04 (ND–9.04) ng/g Spain: n = 21, DF 48%, median (range) = 0.04 (ND–5.70) ng/g Belgium: n = 22, DF 36%, median (range) = 0.04 (ND–9.04) ng/g Italy: n = 22, DF 55%, median (range) = 0.10 (ND–6.54) ng/g (LOQ = 0.06 ng/g) *Values below LOQ replaced with LOQ/(square root of 2)
Winkens et al. (2018)	Finland (Kuopio)	Sixty-three private households from the birth cohort study, LUKAS2. Floor dust samples collected in 2014/2015 from the children's bedroom (entire floor). Participants were instructed not to vacuum clean the room at least a week before sampling. For 55 rooms, dust samples were collected at the end of a 3-week air sampling period (indoor air results reported in a different study).	n = 63, DF 52.4%, mean, median (range) = 1.76, 1.05 (BDL–14.8) ng/g (MDL = 0.73 ng/g) *Values <MDL were treated as MDL/(square root of two)
Padilla-Sánchez and Haug (2016)	Norway (Oslo)	Homes of staff from the Norwegian Institute of Public Health. Dust samples collected from vacuum cleaner bags provided by staff. Sampling year not provided.	n = 7, DF[a] 71%, range = ND–3 ng/g (MDL = 0.028 ng/g; MQL = 0.094 ng/g)
Jogsten et al. (2012)	Spain (Catalonia)	Dust sampling was performed in December 2009 from ten households using household vacuum cleaner dust bags. Samples were collected out of convenience and may not be representative of the entire Catalan population.	n = 10, DF[a] 100%, mean[a] (range) = 6.77 (0.4–37) ng/g (LOD = 0.038 ng/g)
Haug et al. (2011)	Norway (Oslo)	Forty-one homes of breastfeeding mothers recruited for a study on exposure pathways. House dust samples collected between February and May 2008 on two consecutive days while the residence was in regular use. Samples taken from elevated surfaces such as bookshelves and window sills (deposited dust) and not from the floor.	n = 41, DF (frequency of quantification)[a] 61%, mean, median (range) = 29, 23 (3.9–92) ng/g (LOQ = 4.9–31 ng/g) *Concentrations that were not detected or <LOQ were replaced by the LOQ divided by the square root of two

Study	Location	Site Details	Results
Giovanoulis et al. (2019)	Sweden (Stockholm)	Twenty preschools that had been previously sampled in 2015 and then participated in the "chemical smart preschool" initiative to reduce the presence of hazardous chemicals in the indoor environment; 2015 results are reported elsewhere. Samples for this study were collected during January to February 2018. One settled dust sample was collected from elevated surfaces (50–250 cm above the floor) from different areas of a play room at each preschool.	n = 20, DF 55%, medi[an] (95th percentile) = 1.09 (56.0) ng/g (LOD = 0.5 ng/g) *Values <LOD were replaced with ½×LOD
Harrad et al. (2019)	Ireland (Dublin, Galway, and Limerick)	Dust collected from homes (living rooms), offices, cars, and school classrooms; air samples also collected. Samples collected between August 2016 and January 2017. Sample numbers were split approximately equally from each of the three counties.	Homes: n = 32, DF 9.0%, mean, median (range) = <0.05, 0.52 (<0.05–14) ng/g Offices: n = 33, DF 34%, mean, median (range) = 8.6, <0.05 (<0.05–120) ng/g Cars: n = 31, DF 41%, mean, median (range) = 0.55, 0.05 (<0.05–3.1) ng/g Classrooms: n = 32, DF 6.0%, mean, median (range) = <0.05, <0.05 (<0.05–0.71) ng/g (LOD = 0.1 ng/g) *When analyte peaks are <LOD, concentrations were assumed to equal DF × LOD where DF is expressed as a fraction
Huber et al. (2011)	Norway (Tromsø)	Homes and workplaces sampled in winter 2007–2008. Home samples included seven different living rooms (L-1 to L-7), one sleeping room (S, related to L-3), one sofa (a stain repellant fabric, related to L-7), and one carpet (related to L-4). Workplace samples included an office and a storage room at Fram Center; old documents and chemicals and highly contaminated samples were stored in the storage room. Samples were taken from bookshelves, commodes, TVs, electrical heaters, picture frames, window sills and sun blinds. Dust from the floor was not sampled.	All homes: n = 7, DF NR, median = 7 ng/g Living room: n = 7, DF[a] 100%, mean, median (range) = 9.3, 7 (3.3–26.7) ng/g Carpet: n = 1, point = 0.2 ng/g Sleeping room: n = 1, point = 11.6 ng/g Sofa: n = 1, point = 2.4 ng/g Office: n = 1, point = 10.6 ng/g Storage room: n = 1, point = 43.4 ng/g (LOD on column = 0.001 ng; MDL = 0.25–4.92 ng'g)
D'Hollander et al. (2010)	Belgium (Flanders)	Forty-three randomly selected homes and ten randomly selected offices throughout Flanders. Samples collected using a vacuum from bare floor, possibly covered with carpet, in 2008. In homes, the living room, bedroom, kitchen, and working area were sampled.	Homes: n = 43, DF NR, median (95th percentile) = 0.1 (2.1) ng/g dw Offices: n = 10, DF NR, median (95th percentile) = 0.4 (62) ng/g dw (LOQ = 0.06 ng/g)

Study	Location	Site Details	Results
			*Concentrations <LOQ were replaced by DF × LOQ *For homes, both Table 3 and Section 2.1 reported n = 43; however, Table 2 reported n = 45
Multiple Continents			
Karásková et al. (2016)	United States (unspecified), Canada (unspecified), Czech Republic (unspecified)	Fifty-six dust samples from 14 homes in the United States, 15 homes in Canada, and 12 homes in the Czech Republic were collected between April and August 2013. Samples were collected in living rooms and bedrooms.	United States: n = 20, DF 100%, mean, median (range) = 10.9, 3.9 (1.1–62.9) ng/g Canada: n = 20, DF 95.0%, mean, median (range) = 19.4, 4.4 (<MQL–195) ng/g Czech Republic: n = 16, DF 50.0%, mean, median (range) = 3.0, <MQL (ND–11.0) ng/g (MDL = 0.27–1.33 ng/g; MQL = 0.72–3.48 ng/g; ranges represent lower bound and upper bound which were calculated by dividing the MDL/MQL by the biggest and smallest dust sample weight, respectively) *Mean calculated only from values >MQL *Median calculated by replacing values <MQL with √2/2*MQL
Kato et al. (2009)	United States (Atlanta, Georgia), Germany (unspecified), United Kingdom (unspecified), Australia (unspecified)	Thirty-nine household dust samples from the United States (n = 10), Germany (n = 10), United Kingdom (n = 9), and Australia (n = 10) collected in 2004 for method validation. Dust sampling procedures not described.	n = 39, DF 25.6%, median (range) = <LOQ (<LOQ–832) ng/g (LOQ = 2.6 ng/g)

Notes: CTEPP = Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants; GCA = groundwater contamination area; DF = detection frequency; RL = reporting limit; LOD = limit of detection; MDL = method detection limit; ND = not detected; LOQ = limit of quantitation; GM = geometric mean; MQL = method quantification limit; PFC = Perfluorochemicals

C.3.5. Air

Perfluoroalkyl chemicals have been released to air from wastewater treatment plants, waste incinerators, and landfills (Ahrens et al., 2011), though there is limited information on the detection levels or frequencies of PFNA in either indoor or ambient air. ATSDR (2021) notes perfluoroalkyl chemicals have been detected in air and they can be transported long distances via the atmosphere. For example, in a study performed from April 2007 to January 2009, PFNA was detected at an average concentration of 0.3 pg/m^3 in 8% of 141 atmospheric samples from Atlantic and Southern Oceans and coastal areas of the Baltic Sea (NCBI, 2022b; Dreyer et al., 2009). PFNA is not expected to be broken down directly by photolysis (NCBI, 2022b). PFNA can undergo hydroxylation in the atmosphere, with a (predicted average) atmospheric hydroxylation rate of 8.41×10^{-13} cm^3/molecule – second to a (derived) rate of 5.2×10^{-11} cm^3/molecule – second (with corresponding estimated half-life of 31 days for this reaction in air) (USEPA, 2022b; NCBI, 2022b). With a vapor pressure of 4.83×10^{-3} mm Hg at 20°C (extrapolated), 8.3×10^{-2} mm Hg at 25°C (estimated), 8.4 mm Hg at 99.63°C (measured), and a (measured) range of 4.80×10^{-3} mm Hg to 9.77×10^{-3} mm Hg, volatilization is not expected to be an important fate process for this chemical (USEPA, 2022b; NCBI, 2022b; ATSDR, 2021). The EPA's Toxics Release Inventory reported release data for PFNA in 2020, with a total onsite disposal, offsite disposal, and other releases concentration of 0 pounds from one facility (USEPA, 2022c). PFNA is not listed as a hazardous air pollutant (USEPA, 2022d).

C.3.5.1. Indoor Air

No studies from the U.S. reporting levels of PFNA in indoor air were identified from the primary or gray literature. However, the EPA identified studies from Canada and Europe that are summarized below and in Table C-9 (Harrad et al., 2019; Makey et al., 2017; Jogsten et al., 2012; Barber et al., 2007). All of these studies sampled from homes, while only one study also sampled from offices, cars, and classrooms and one study also sampled from a laboratory.

Two studies reported results from indoor air samples collected only from homes (Makey et al., 2017; Jogsten et al., 2012). In the Canadian study, Makey et al. (2017) collected indoor air, vacuum cleaner dust, and serum samples in 2007–2008 from the homes of women in the second trimester of pregnancy and analyzed the samples for levels of PFAAs. Participants were part of the Chemicals, Health, and Pregnancy (CHirP) Study. PFNA was detected in 42% of indoor air samples (n = 39), with a geometric mean of 1.5 pg/m^3. In Spain, Jogsten et al. (2012) sampled indoor air (n = 10) from selected homes in Catalonia in December 2009 and evaluated levels of 27 PFCs. PFNA was not detected (LOD = 3.1–280 pg/m^3 for all ionic PFAS).

The remaining two studies evaluated PFNA levels in offices, vehicles, and/or schools, in addition to homes. In Ireland, Harrad et al. (2019) collected air samples in homes (living rooms, n = 34), offices (n = 34), cars (n = 31), and school classrooms (n = 28) between August 2016 and January 2017. PFNA was detected in all four indoor microenvironments in 18%, 91%, 90%, and 93% of samples for homes, offices, cars, and classrooms, respectively. The mean (median) concentrations were 2.1 (1.7) pg/m^3 in homes, 3.7 (2.5) pg/m^3 in offices, 5.2 (2.1) pg/m^3 in cars, and 3.5 (2.5) pg/m^3 in classrooms. In Norway, neutral and ionic PFAS were analyzed in indoor air samples collected from three homes and one laboratory in Tromsø between April and June 2005 (Barber et al., 2007). The study detected PFNA in all four samples, with a mean concentration of 2.7 pg/m^3.

Table C-9. Summary of Studies Reporting the Occurrence of PFNA in Indoor Air

Study	Location	Site Details	Results
Canada			
Makey et al. (2017)	Canada (Vancouver)	Samples were collected from a subset of Chemicals, Health, and Pregnancy (CHirP) Study participants' homes in 2007–2008; dust and serum samples were also collected. Indoor air was sampled using passive samplers deployed in participants' bedrooms for four weeks.	n = 39, DF 42%, GM = 1.5 pg/m^3 (DL = 0.02 pg/m^3)
Europe			
Jogsten et al. (2012)	Spain (Catalonia)	Indoor air sampling was performed in December 2009 from ten households at approximately 1 m above the floor. Samples were collected out of convenience and may not be representative of the entire Catalan population. Both particulate and gas phases collected.	n = 10, DF 0% (LOD = 3.1–280 pg/m^3 for all ionic PFAS)
Harrad et al. (2019)	Ireland (Dublin, Galway, Limerick)	Air samples collected from homes (living rooms), offices, cars, and school classrooms; dust samples also collected. Samples collected between August 2016 and January 2017. Sample numbers were split approximately equally from each of the three counties. Gas or particulate phase not specified.	Homes: n = 34, DF 18%, mean, median (range) = 2.1, 1.7 (<0.3–13) pg/m^3 Offices: n = 34, DF 91%, mean, median (range) = 3.7, 2.5 (<0.3–18) pg/m^3 Cars: n = 31, DF 90%, mean, median (range) = 5.2, 2.1 (<0.3–24) pg/m^3 Classrooms: n = 28, DF 93%, mean, median (range) = 3.5, 2.5 (<0.3–15) pg/m^3 (LOD = 0.3 pg/m^3) *When analyte peaks are <LOD, concentrations were assumed to equal DF × LOD where DF is expressed as a fraction
Barber et al. (2007)	Norway (Tromsø)	Air samples taken from four indoor locations (three houses and one laboratory) in Tromsø in April–June 2005. PFNA was measured in the particulate phase.	n = 4, DF[a] 100%, mean (range) = 2.7 (0.9–4.7) pg/m^3 (MQL = 0.84 pg/m^3)

C.3.5.2. Ambient Air

A single U.S. study measured levels of PFNA in ambient air (Kim and Kannan, 2007). Kim and Kannan (2007) analyzed particle phase (n = 8) and gas phase (n = 8) concentrations of perfluorinated acids in ambient air samples collected in and around Albany, New York in May and July 2006 to examine the relative importance of certain media pathways to the contamination of urban lakes. PFNA was detected in all gas phase samples with mean and median concentrations of 0.21 pg/m^3 and 0.20 pg/m^3, respectively. PFNA was also detected in the particulate phase, but the detection frequency was not reported. Authors reported particulate phase mean and median concentrations of 0.13 pg/m^3 and below the LOQ (0.12 pg/m^3), respectively.

One Canadian PFNA study by Ahrens et al. (2011) analyzed the temporal ambient PFNA concentrations at two municipal solid waste (MSW) landfills and at one wastewater treatment plant. At the MSW landfills, they recorded a mean (range) concentration of 2.11 (0.97–3.24) pg/m^3 upwind of the site and 10.3 (4.82–15.8) pg/m^3 at the site. At the wastewater treatment plant, they measured the PFNA concentration within the facility in the primary clarifier, aeration tank, and secondary clarifier, finding mean concentrations between 2.97 and 3.62 pg/m^3. They also evaluated reference sites near to (<200m) and distant from (~600m) the facility, finding higher ambient PFNA concentrations at the near sites (mean of 1.64 pg/m^3) compared to the distant sites (single point estimate of 0.88 pg/m^3).

Among the European studies—conducted in Spain (Jogsten et al., 2012; Beser et al., 2011), Ireland (Harrad et al., 2019; Barber et al., 2007), Norway (Barber et al., 2007), and the United Kingdom (Barber et al., 2007)— each collected ambient (outdoor) air for PFNA measurements. Reported mean PFNA concentrations among these studies range from 0.08–3.8 pg/m^3. Beser et al. (2011) measured PFNA ($PM_{2.5}$-bound) in both residential and industrial areas (five sampling stations in total) of the Alicante Province in Spain in 2010, with mean concentrations ranging from 1.65–3.8 pg/m^3. Jogsten et al. (2012) did not detect PFNA in ten sites across Catalonia, Spain the year prior in 2009, however, they identified their limit of detection as 3.1 pg/m^3, which was higher than the mean PFNA concentration detected in four of the five sites in Spain evaluated by Beser et al. (2011). Harrad et al. (2020) compared PFNA concentrations both upwind and downwind of ten Irish MSW landfills in late 2018 and early 2019, finding comparable concentration ranges upwind (<0.08–0.31 pg/m^3) and downwind (0.08–0.52 pg/m^3). Barber et al. (2007) also detected ambient PFNA at detectable but not quantifiable levels (<3.3 pg/m^3 on average) in rural Mace Head, Ireland. United Kingdom samples (n = 15) from Hazelrigg and Manchester were reported in ranges of <0.06–0.9 pg/m^3 and two samples from Kjeller, Norway were reported at a range of 0.10–0.13 pg/m^3 (Barber et al., 2007).

Table C-10. Summary of Peer-Reviewed Studies Reporting the Occurrence of PFNA in Ambient Air

Study	Location	Site Details	Results
United States			
Kim and Kannan (2007)	United States (Albany, New York)	Roof of a lakehouse building located at Washington Park Lake in May and July 2006. Both particulate and gas phases collected.	Gas: n = 8, DF[a] 100%, mean, median (range) = 0.21, 0.20 (0.16–0.31) pg/m^3 Particle: n = 8, DF NR, mean, median (range) = 0.13, <LOQ (<LOQ–0.40) pg/m^3 (LOQ = 0.12 pg/m^3) *Non-detects were set to zero; values below the LOQ were set to ½ LOQ
Canada			
Ahrens et al. (2011)	Canada (Ontario)	Samples collected on and around one municipal WWTP for 63 days between July and September 2009. Samplers were placed at the primary clarifier, aeration tank, secondary clarifier, and at four reference sites (three near [within 200 m of the treatment tanks] and one distant [~600 m from the perimeter of the WWTP]). Samples also collected at two municipal solid waste landfills between June and August 2009 for 55 days. The two landfills were 60 km apart. Samplers were located upwind and onsite of the active zone of each landfill site and one field blank was collected at each site. Both sites collected landfill gas and the active area of the landfill was kept to a minimum by covering the waste with soil and a plastic film. The passive sampling configuration used resulted in the collection of mainly PFAS in the gas phase	WWTP: Reference sites (near): n = 3, DF[a] 100%, mean[a] (range) = 1.64 (1.10–2.11) pg/m^3 Primary clarifier: n = 2, DF[a] 100%, mean[a] (range) = 2.97 (1.10–4.84) pg/m^3 Aeration tank: n = 3, DF[a] 100%, mean[a] (range) = 3.63 (2.76–4.37) pg/m^3 Secondary clarifier: n = 2, DF[a] 100%, mean[a] (range) = 3.03 (2.24–3.81) pg/m^3 Reference site (distant): n = 1, point = 0.88 pg/m^3 Landfills: Upwind: n = 2, DF[a] 100%, mean[a] (range) = 2.11 (0.97–3.24) pg/m^3 On site: n = 2, DF[a] 100%, mean[a] (range) = 10.3 (4.82–15.8) pg/m^3 (MDL = 0.04–0.87 pg/m^3 for PFCAs, PFSAs, and PFOSA)
Europe			
Harrad et al. (2020)	Ireland (multiple cities)	Samples collected from ten municipal solid waste landfills upwind and downwind at each site between November 2018 and January 2019. Location of sampling sites based on wind direction data taken from the Irish Meteorological Service, with slight modification where necessary based on local information from site operators and ease of access. Sample sites were between 150 and 500 m of the center of the landfill. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction and demolition, and	Downwind: n = 10, DF[a] 100%, mean, median (range) = 0.23, 0.17 (0.08–0.52) pg/m^3 Upwind: n = 10, DF[a] 90%, mean, median (range) = 0.15, 0.13 (<0.08–0.31) pg/m^3 (LOD = 0.08 pg/m^3) *Non-detects replaced by ½ LOD

Study	Location	Site Details	Results
		biomedical waste. Gas or particulate phase not specified.	
Jogsten et al. (2012)	Spain (Catalonia)	Outdoor air sampling conducted in December 2009 for the purposes of comparison to indoor air and dust samples. Number of sites not specified but assumed to be ten because indoor air was sampled from ten homes. Samples were collected out of convenience and may not be representative of the entire Catalan population. Both particulate and gas phases collected.	n = 10; DF 0% (LOD = 3.1–280 pg/m^3 for all ionic PFAS)
Beser et al. (2011)	Spain (Alicante province)	Samples collected from April to July 2010 from five stations. Two stations were placed in Elche (one in a residential area and the other in an industrial area). The third station was placed in a residential area of Alicante City. The fourth station was in a rural area of Pinoso and the last station was in a residential area of Alcoy. Concentrations reported for PM2.5-bound PFNA.	Elche (residential): n = 11, DF[a] 55%, mean = 2.7 pg/m^3 Elche (industrial): n = 13, DF[a] 69%, mean = 3.8 pg/m^3 Alicante City: n = 11, DF[a] 36%, mean = 1.7 pg/m^3 Pinoso: n = 3, DF[a] 67%, mean = 1.65 pg/m^3 Alcoy: n = 3, DF[a] 100%, mean = 2.2 pg/m^3 (MQL = 1.4 pg/m^3) *Mean calculated from values >MQL
Barber et al. (2007)	United Kingdom (Hazelrigg, Manchester); Ireland (Mace Head); Norway (Kjeller)	Samples collected from four field sites in Europe: Hazelrigg (semirural) and Manchester (urban) were sampled in two sampling events in February–March 2005 and November 2005–January 2006; Mace Head (rural) was sampled in March 2006; and Kjeller (rural) was sampled in November–December 2005. PFNA was measured in the particulate phase.	Hazelrigg first sampling event: n = 2, DF NR, mean = <13.8 pg/m^3 (MQL = 13.9 pg/m^3) *The glass-fibre filters were analyzed in a batch of samples that showed contamination problems, so the high associated blank value used to calculate the MQL put most analytes <MQL. Hazelrigg second sampling event: n = 10, DF[a] = 90%, mean (range) = 0.9 (<0.06–1.7) pg/m^3 (MQL = 0.006 pg/m^3) Manchester first sampling event: n = 2, DF NR, mean = <26.6 pg/m^3 (MQL = 22.5 pg/m^3) Manchester second sampling event: n = 1, point = 0.8 pg/m^3 (MQL = 0.006 pg/m^3) Mace Head: n = 4, DF NR, mean = <3.3 pg/m^3

Study	Location	Site Details	Results
			(MQL = 3.32 pg/m^3) Kjeller: n = 2, DF[a] 100%, mean (range) = 0.12 (0.10–0.13) pg/m^3 (MQL = 0.10 pg/m^3) *Means calculated from values >MQL

C.3.6. Soil

The use and production of PFNA could result in its release to soils through various waste streams (NCBI, 2022b). When released to soil, based on its physico-chemical properties, PFNA is expected to have no mobility (NCBI, 2022b). PFNA has been measured in grass samples grown in soil containing PFNA and other PFAS near Decatur, Alabama (ATSDR, 2021; Yoo et al., 2011). In addition, PFNA has been found to accumulate in the roots of maize plants grown in soil containing PFNA and other PFAS (ATSDR, 2021; Krippner et al., 2014).

Seven U.S. studies were identified that evaluated the occurrence of PFNA and other PFAS in soil (Galloway et al., 2020; Nickerson et al., 2020; Zhu and Kannan, 2019; Eberle et al., 2017; Anderson et al., 2016; Venkatesan and Halden, 2014; Blaine et al., 2013) (Table C-11). Among these studies, three analyzed soils potentially impacted by past AFFF use. The PFNA detection frequencies varied widely (from less than 20% to over 90%) but mean concentrations tended to be below 5 ng/g. Few studies analyzed soils in the vicinity of fluoropolymer manufacturing facilities or by contaminated soil amendments. Other than control soils in two greenhouse and field studies and one reference site, the U.S. studies did not evaluate soils without amendments or without a nearby current or historical PFAS source.

Two studies analyzed soils in the vicinity of fluoropolymer manufacturing facilities (Galloway et al., 2020; Zhu and Kannan, 2019). Galloway et al. (2020) collected soil samples in December 2016 and March 2018 near a fluoropolymer production facility outside Parkersburg, West Virginia. The 2016 sampling included sites 4.0 km–48.1 km downwind to the north and northeast of the facility and the 2018 sampling included sites 1.3 km–45.4 km north of the facility. PFNA was detected in six of eight of the 2016 samples, however only one was above the LOQ with a concentration of 1.63 ng/g. PFNA was also detected in six of seven of the 2018 samples, however only one was above the LOQ with a concentration of 1.92 ng/g at a distance of 1.3 km. Both the 2016 and 2018 samples that were above the LOQ were reported at the site closest to the facility. In Zhu and Kannan (2019), authors studied PFAS concentrations in soil contaminated by a nearby fluoropolymer manufacturing facility in Little Hocking, Ohio, which had been manufacturing fluorochemicals for over five decades. The 45-acre well field located in a floodplain meadowland was divided into quadrants and surface soil samples were collected from multiple locations within each quadrant in October 2009. PFNA was detected in all 19 samples with mean and median concentrations of 2.7 ng/g and 2.5 ng/g, respectively.

Three studies analyzed soils potentially impacted by AFFF use (Nickerson et al., 2020; Eberle et al., 2017; Anderson et al., 2016). Anderson et al. (2016) assessed 40 sites across 10 active Air Force installations throughout the continental United States and Alaska between March and September 2014. Installations were included if there was known historic AFFF release in the period 1970–1990. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. The selected sites were not related to former fire training areas and were characterized according to volume of AFFF release – low, medium, and high. Across all sites, the PFNA detection frequency was 71.43% in 100 surface soil samples (median concentration of detects was 1.3 ng/g) and 14.42% in 112 subsurface soil samples (median concentration of detects was 1.5 ng/g). PFNA was detected more frequently at high-volume release sites (50.8% in 32 surface soil samples with mean concentration of 2.5 ng/g; 84.4% in 31 subsurface soil samples with mean concentration of

2.4 ng/g) than at low-volume sites (50.0% in 12 surface soil samples with mean concentration of 2.7 ng/g; 17.6% in 17 subsurface soil samples with mean concentration of 1.0 ng/g) and medium-volume sites (38.3% in 56 surface soil samples with mean concentration of 2.2 ng/g; 67.9% in 64 subsurface soil samples with mean concentration of 2.1 ng/g). Authors noted that given PFNA is not present in 3M AFFF formulations, there may be some degree of telomer-based AFFF contamination. Nickerson et al. (2020) developed a method to quantify anionic, cationic, and zwitterionic PFAS from AFFF-impacted soils. The method was applied to two soil cores collected from two different AFFF-impacted former fire training areas; the sampling year and geographic location were not provided. Eleven soil samples, corresponding to 11 depths ranging from 0.46 m to 15.1 m, were evaluated from Core E, and 12 soil samples, at depths ranging from 0.30 to 14.2 m, were evaluated from Core F. In Core E, PFNA was detected in 5 of 11 samples at depths both at the surface and further below ground with PFNA concentrations ranging from below the LOQ to 1.96 ng/g dw. In Core F, PFNA was detected in 5 of 12 samples at the five depths closest to the surface, with concentrations ranging from below the LOQ to 4.17 ng/g dw (LOQ not reported). Eberle et al. (2017) investigated the effects of an in situ chemical oxidation treatment for remediation of chlorinated volatile organic compounds and PFAAs co-contaminants. Soil samples were collected in 2012–2013 before and after a pilot scale field test at a former fire training site at Joint Base Langley-Eustis, Virginia. Monthly fire training activities were conducted at the site from 1968 to 1980 and irregular fire training activities continued until 1990. Impacted soil was excavated in 1982 but details were not provided. PFNA was detected in 1 of 5 pre-treatment samples and in 13 of 14 post-treatment samples. Of the available three paired pre- and post-treatment soil samples, PFNA was not detected pre-treatment in two pairings but detected post-treatment at 0.07 ng/g and 0.05 ng/g post-treatment. For the third pairing, PFNA was detected at 1.1 ng/g pre-treatment and below the LOQ (0.06 ng/g) post-treatment.

Of the remaining two studies conducted in the United States, Venkatesan and Halden (2014) conducted outdoor mesocosm studies to examine the fate of PFAS in biosolids-amended soil collected during 2005–2008. Biosolids were obtained from a wastewater treatment plant (WWTP) in Baltimore that primarily treated wastewater from domestic sources with only minor contribution (1.9%) from industry. The number of samples was not provided but PFNA was detected in the control (nonamended) soil at levels below 0.5 ng/g dw and in the biosolids-amended soil at a level not reported by the authors. In a field and greenhouse study, Blaine et al. (2013) studied the uptake of PFAS into edible crops grown in control and biosolids-amended soil. In the field study, urban biosolids were obtained from a WWTP receiving both domestic and industrial waste while rural solids were obtained from a WWTP receiving domestic waste only. PFNA was detected in soils from urban (mean = 0.20 ng/g, 0.28 n/g, and 0.40 ng/g in control, 1×[9] and 2× amended fields, respectively) and rural fields (mean = 0.06 ng/g and 0.75 ng/g in control and 0.5× amended fields, respectively). In the greenhouse study, three soils (nonamended control, industrially impacted, and municipal) were investigated. Industrially impacted soils contained composted biosolids from a small municipal WWTP that was impacted by PFAA manufacturing while municipal soils were obtained from a reclamation site in Illinois where municipal biosolids were applied for 20 years. PFNA was detected in all three soils at an average concentration of 0.30 ng/g, 20.15 ng/g, and 6.11 ng/g in control, industrially impacted,

[9] 0.5×, 1×, or 2× is defined as ½, 1, or 2 times the agronomic rate of biosolids application to meet nitrogen requirements of the crop.

and municipal soil, respectively. Authors noted that the trace levels of PFAS detected in the control soil may be due to minor cross-contamination from plowing, planting, or atmospheric deposition from the surrounding area where biosolids have been applied.

Results for all of the identified non-U.S. studies are presented in detail in Table C-11 and are summarized here. The EPA identified three Canadian studies, two were conducted at locations with prior AFFF use (Cabrerizo et al., 2018; Mejia-Avendaño et al., 2017). Cabrerizo et al. (2018) evaluated soil in two locations, one of which was relatively remote and largely did not have direct human contact and the other of which was previously used as a military training facility. The remote location (n = 19) had concentrations ranging from 0.0262 ng/g dw to 0.8749 ng/g dw and the historic military training location (n = 8) similarly had concentrations from 0.0836 ng/g dw to 0.7794 ng/g dw. Mejia-Avendaño et al. (2017) investigated soil samples at the site of the 2013 Lac-Mégantic train accident, where approximately 33,000 L of AFFF concentrates were used to put out fires. In 2013, 12 sample concentrations ranged from 0.138–19 ng/g dw and in 2015, two years after the incident, 11 sample concentrations ranged from 0.031–0.777 ng/g dw. In the third Canadian study, Dreyer et al. (2012) sampled bog peat cores in an undisturbed and well investigated location to determine historic atmospheric contamination. Estimated core segment dated back to 1912, with PFNA concentrations ranging between not detected and 0.412 ng/g.

Of the European studies, three were conducted at locations near firefighting facilities (Dauchy et al., 2019; Skaar et al., 2019; Hale et al., 2017). Dauchy et al. (2019) and Skaar et al. (2019) found varying results from not detected to 15 ng/g dw while Hale et al. (2017) reported a range of 2.8–41.3 ng/g for the 40% of samples that were detected. One study in Belgium (Groffen et al., 2019) evaluated soil samples at a perfluorochemical plant and at four sites increasing in distance from the plant. PFNA levels were detected at the plant ranging from not detected to 2.53 ng/g dw and ranging from not detected to 0.53 ng/g dw away from the plant in no discernable pattern. Harrad et al. (2020) investigated soil samples upwind and downwind from ten municipal solid waste landfills in Ireland. PFNA was found in all samples upwind of the landfills, ranging in concentrations from 0.0029–0.033 ng/g dw, and in 89% of samples downwind, ranging in concentrations from not detected to 0.0077 ng/g dw. A study in Norway (Grønnestad et al., 2019) found mean PFNA concentrations from a popular skiing location in Granåsen to be lower than that of the study's selected reference site in Jonsvatnet. Sammut et al. (2019) sampled soil from six random small urban fields in Malta and found concentrations ranging from 0.66–0.87 ng/g.

Table C-11. Summary of PFNA Data in Soil

Study	Location	Site Details	Results
United States			
Galloway et al. (2020)	United States (Parkersburg, West Virginia)	Soil samples collected near a fluoropolymer facility in two sampling trips in December 2016 and March 2018. The 2016 sampling trip included a collection radius 4.0–48.1 km downwind to the north and northeast of the facility. The 2018 sampling trip focused on samples collected to the north of the facility with a radius of 1.3–45.4 km.	2016 sampling: Drag Strip Road (4.0 km) = 1.63 ng/g Veto Lake (8.0 km) = <LOQ Veto Road (13.0 km) = ND Veto Road, dup. (13.0) = ND Strouds Run (15.3 km) = <LOQ Lookout Park (24.0 km) = <LOQ Archers Fork #1 (35.3 km) = <LOQ Archers Fork #2 (48.1 km) = <LOQ 2018 sampling: LHWA (1.3 km) = 1.92 ng/g Veto Lake (8.0 km) = <LOQ Veto Lake, dup. (8.0 km) = <LOQ Watertown (24.3 km) = ND Beverly (32.1 km) = <LOQ L. Olive Green Creek (39.9 km) = <LOQ Reinersville (45.4 km) = <LOQ (LOQ = 1 ng/g)
Zhu and Kannan (2019)	United States (Washington County, Ohio)	Surface soil (0–6 cm) samples collected in October 2009 from a 45-acre field located within a 1-mile radius of a fluoropolymer manufacturing facility in Little Hocking that had been manufacturing fluorochemicals for over five decades. The site was divided into quadrants and soil samples were collected from multiple locations within each quadrant.	n = 19, DF 100%, mean, median (range) = 2.7, 2.5 (1.6–6.3) ng/g dw (LOD = 0.1356 ng/g dw; LOQ = 0.452 ng/g dw)
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between 1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were not specifically controlled for in the site selection	Surface soil: Overall: n = 100, DF 71.43%, median (maximum) = 1.3 (23.0) ng/g *Breakdown by site:* Emergency Response (low-volume release): n = 12, DF 50.0%, mean (range) = 2.7 (1.5–4.1) ng/g Hangars and Buildings (medium-volume release): n = 56, DF 38.3%, mean (range) = 2.2 (0.21–12) ng/g

Study	Location	Site Details	Results
		process; active remedies had not been applied at any of the sites selected. Approximately ten samples were collected between March and September 2014 at each site for surface and subsurface soil; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases.	Testing and Maintenance (high-volume release): n = 32, DF 50.8%, mean (range) = 2.5 (0.24–23) ng/g (RL = 0.23 ng/g) Subsurface soil: Overall: n = 112, DF 14.42%, median (maximum) = 1.5 (6.49) ng/g *Breakdown by site:* Emergency Response (low-volume release): n = 17, DF 17.6%, mean (range) = 1.0 (0.5–1.5) ng/g Hangars and Buildings (medium-volume release): n = 64, DF 67.9%, mean (range) = 2.1 (0.21–12) ng/g Testing and Maintenance (high-volume release): n = 31, DF 84.4%, mean (range) = 2.4 (0.24–23) ng/g (RL = 0.24 ng/g) *Median calculated using quantified detections *Non-detects were substituted with ½ the reporting limit
Nickerson et al. (2020)	United States (unspecified)	Soil cores E and F from two different AFFF-impacted fire training areas; sampling year and geographic location not provided. Soil core E contained 11- 0.3 m increment samples from 0.3–15.2 m below ground surface and was collected in an area where the surficial soils were likely disturbed due to regrading and other soil redistribution activities. Soil core F contained 12- 0.61 m increment samples from 0–14.2 m below ground surface and was collected in an area where the surficial soils were highly permeable only within the upper 0.5 to 1 m, and the underlying impermeable clay layer exhibited a relatively high cation exchange capacity and organic carbon content.	Core E: 0.46 m = 1.96 ng/g dw 2.9 m = <LOQ 3.66 m = <LOQ 3.96 m = <LOQ 4.27 m = <LOQ 4.57 m = <LOQ 4.88 m = 0.22 ng/g dw 7.01 m = 0.26 ng/g dw 8.38 m = 0.73 ng/g dw 10.5 m = 1.09 ng/g dw 15.1 m = <LOQ Core F: 0.30 m = 0.70 ng/g dw

Study	Location	Site Details	Results
		The water table was relatively shallow (depth <3 m) at both sites.	1.22 m = 4.17 ng/g dw 1.83 m = 3.23 ng/g dw 2.44 m = 1.04 ng/g dw 3.05 m = 0.64 ng/g dw 4.11 m = <LOQ 7.62 m = <LOQ 8.84 m = <LOQ 9.45 m = <LOQ 10.5 m = <LOQ 11.9 m = <LOQ 14.2 m = <LOQ (LOQ not reported)
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, Virginia)	Pilot testing area in former fire training area (Training Site 15) at Joint Base Langley-Eustis where monthly fire training activities were conducted from 1968 to 1980 in a zigzag pattern burn pit. Facility was abandoned in 1980 but irregular fire training activities using an above-ground germed burn pit continued until 1990. Impacted soil was removed in 1982 but additional details of the excavation are not well known. Soil samples collected for pre- (April and September 2012) and post- (December 2013) in situ chemical oxidation treatment using a peroxone activated persulfate (OxyZone) technology. Treatment was conducted in Test Cell 1 over 113 days (April–August 2013). Soil samples were collected adjacent to wells; wells outside Test Cell 1 were used as sentry wells. Well IDs for pre- and post-sampling were not provided but the following three pairings were assumed based on Table 2 in the paper: U-20 with SB-106; U-16 with SB-112; and I-1 with SB-109.	Pre-treatment: I-1 (1.2–4.3 m) = 1.1 ng/g I-2 (1.2–4.3 m) = ND U-12 (2.1 m) = ND U-16 (3.0 m) = ND U-20 (1.8 m) = ND (LOQ = 0.68–0.72 ng/g) Post-treatment: SB-101 (4.3 m) = 0.07 ng/g SB-105 (1.8 m) = 0.02 ng/g SB-106/U-20 (1.8 m) = 0.07 ng/g SB-106 (4.3 m) = 0.14 ng/g SB-107 (1.8 m) = 0.03 ng/g SB-107 (4.3 m) = 0.2 ng/g SB-108 (1.8 m) = 0.03 ng/g SB-108 (4.3 m) = 0.15 ng/g SB-109/I-1 (3 m) = <LOQ SB-111 (4.3 m) = 0.29 ng/g SB-112 (1.8 m) = 0.06 ng/g SB-112/U-16 (3 m) = 0.05 ng/g SB-114 (1.8 m) = 0.3 ng/g SB-114 (4.3 m) = 0.33 ng/g (LOQ = 0.06 ng/g)
Venkatesan and Halden (2014)	United States (Baltimore, Maryland)	Archived agricultural soil (nonamended) collected during 2005–2008 at a depth of 0–20 cm from the United States Department of Agriculture-Agricultural Research Service Beltsville Agricultural Research	Nonamended: n = NR, DF NR, authors noted PFNA concentration was between 0.1–0.5 ng/g dw Amended: n = NR, DF NR, authors noted the detected levels of PFNA, along with PFOA,

Study	Location	Site Details	Results
		Center; number of sampling sites and number of samples not provided. Biosolids-amended soil obtained by mixing biosolids and soil at a volumetric ratio of 1:2. Biosolids were from Back River WWTP in Baltimore, a full-scale activated sludge treatment plant. Raw wastewater was primarily from domestic sources with only minor contribution (1.9%) from industry.	PFNA, PFDA, and PFUnA in the control soil accounted for 0.3–3% of their initial levels in the amended soil mix (MDL = 0.08 ng/g)
Blaine et al. (2013)	United States (Midwest)	Urban and rural full-scale field study with control (nonamended) and biosolids-amended plots. Three agricultural fields were amended (0.5×, 1×, or 2×) with municipal biosolids. Urban biosolids (1× and 2×) were from a WWTP receiving both domestic and industrial waste. Rural biosolids (0.5×) were from a WWTP receiving domestic waste only. Control plots were proximal to the rural and urban amended corn grain and corn stover field sites; sampling year not provided. Greenhouse study with control (nonamended) and biosolids-amended soil. Nonamended soil obtained from a field that received commercial fertilizers and had a similar cropping system as the nearby municipal soil site. Municipal soil was obtained from a reclamation site in Illinois where municipal biosolids were applied at reclamation rates for 20 years, reaching the cumulative biosolids application rate of 1,654 Mg/ha. Industrially impacted soil was created by mixing composted biosolids from a small municipal (but impacted by PFAA manufacturing) WWTP with control soil on a 10% mass basis. Sampling year not provided.	Field study: Urban non-amended: n = 3–7, DF NR, mean = 0.20 ng/g Urban 1×: n = 3–7, DF NR, mean = 0.28 ng/g Urban 2×: n = 3–7, DF NR, mean = 0.40 ng/g Rural non-amended: n = 3–7, DF NR, mean = 0.06 ng/g Rural 0.5×: n = 3–7, DF NR, mean = 0.75 ng/g (LOQ not reported) Greenhouse study: Nonamended: n = 3–5, DF NR, mean = 0.30 ng/g Industrially impacted: n = 3–5, DF NR, mean = 20.15 ng/g Municipal: n = 3–5, DF NR, mean = 6.11 ng/g (LOQ not reported)
Canada			
Cabrerizo et al. (2018)	Canada (Melville and Cornwallis Islands)	Catchment areas of lakes in the Cape Bounty Arctic Watershed Observatory on southern Melville Island (West, East, and Headwater lakes) during summer (late July-early August) 2015 and 2016, representing an environment largely unimpacted by direct human activity; data for 19 sampling sites available (S6, S11–S28).	Melville Island lakes: n = 19, DF[a] 100%, mean[a] (range) = 0.3248 (0.0262– 0.8749) ng/g dw Cornwallis Island lakes: n = 8, DF[a] 100%, mean[a] (range) = 0.3333 (0.0836–0.7794) ng/g dw (LOD = 0.0001–0.018 ng/g for all PFAS)

Study	Location	Site Details	Results
		Catchment areas of lakes on Cornwallis Island (Resolute, North, Small, Meretta, 9 Mile, and Amituk lakes) near the community of Resolute Bay during summer (late July-early August) 2015 and 2016. Resolute Bay has a military and civilian airport which discharged its wastewaters into the upper area of the catchment until 1997, three old solid waste landfills 1.5–2 km west of the airport used until the mid-1990s, and Arctic research and military training facilities close to the airport that support activities such as vehicle use, firefighting, and construction/demolition; eight sampling sites (S29–S36).	
Mejia-Avendaño et al. (2017)	Canada (Lac-Mégantic, Quebec)	Site of July 2013 Lac-Mégantic train accident where 63 out of 72 train cars carrying 8 million liters of crude oil derailed and a major oil fire ignited. Seven types of AFFFs and approximately 33,000 L of AFFF concentrates were used. Samples were collected in July 2013 weeks after the accident from the western shores of Chaudière River, at the point where the oil and AFFF runoff reached the river, approximately 500 m from the edge of the derailment site; in July 2015 from the fire burn site and adjacent area in downtown Lac-Mégantic where the soil was continuously excavated for remediation (the site was the closest to the accident site among the areas open to sampling); and from a background, nonimpacted area next to Chaudière River, about 5 km from the accident site, on the east shore of the river and on the opposite side of the accident.	Background: n = 3, DF NR, mean = 0.212 ng/g dw 2013: n = 12 (from 12 sites), DF[a] 100%, mean[a] (range) = 4.41 (0.138–19) ng/g dw 2015: n = 11 (from 9 sites), DF[a] 100%, mean[a] (range) = 0.274 (0.031–0.777) ng/g dw (LOD = 0.02 ng/mL; LOQ = 0.05 ng/mL)
Dreyer et al. (2012)	Canada (Ottawa, Ontario)	Two ombrotrophic Mer Bleue bog peat core samples collected in October 2009 and cut into 5-cm segments (nine segments for the first core, eight segments for the second core); Mer Bleue selected because it is undisturbed and well investigated and is located in a meltwater channel of the postglacial Ottawa River. Peat cores sampled to determine their suitability for determining historic atmospheric contamination; contaminants present due to atmospheric deposition only. The year for each segment was estimated through dating of Mer Bleue peat cores collected in the same year for a different study.	First core (first parallel; second parallel): 2009: 0.041; 0.033 ng/g 2006: 0.143; 0.192 ng/g 2001: 0.229; 0.223 ng/g 1992: 0.259; 0.271 ng/g 1983: 0.241; 0.234 ng/g 1973: 0.203; 0.322 ng/g 1962: 0.263; 0.297 ng/g 1945: 0.148; 0.156 ng/g 1927: 0.069; 0.089 ng/g 1912: 0.044; 0.044 ng/g

Study	Location	Site Details	Results
			Second core (first parallel; second parallel): 2009: 0.052; 0.062 ng/g 2006: 0.193; 0.166 ng/g 2001: 0.162; 0.234 ng/g 1992: 0.320; 0.319 ng/g 1983: 0.396; 0.412 ng/g 1973: 0.206; 0.206 ng/g 1962: 0.082; 0.081 ng/g 1945: 0.160; 0.149 ng/g 1927: <MQL; <MDL (IDL = 0.003 ng/g; MQL = 0.019 ng/g; MDL = 0.048 ng/g) *Authors estimated the year for each core segment using cores for a different study that underwent dating
Europe			
Groffen et al. (2019)	Belgium (Antwerp)	3M perfluorochemical plant and four sites with increasing distance from plant were selected based on prior biomonitoring studies in the vicinity of the plant. The four sites are: Vlietbos (1 km SE from 3M), Rot-Middenvijver (2.3 km ESE from 3M), Burchtse Weel (3 km SE from 3M), and Fort 4 (11 km SE from 3M). Samples collected in June 2016.	Plant: n = 13, DF 69%, mean, median (range) = 0.83, 0.34 (<LOQ–2.53) ng/g dw 1 km from plant: n = 10, DF 30%, mean, median (range) = <LOQ, <LOQ (<LOQ–0.44) ng/g dw 2.3 km from plant: n = 10, DF 0% 3 km from plant: n = 10, DF 10%, mean, median (range) = <LOQ, <LOQ (<LOQ–0.38) ng/g dw 11 km from plant: n = 14, DF 29%, mean, median (range) = <LOQ, <LOQ (<LOQ–0.53) ng/g dw (LOQ = 0.26 ng/g dw)
Dauchy et al. (2019)	France (unspecified)	Site where fluorosurfactant-based foams have been used extensively. From 1969 to 1984, the site was an oil refinery, with the exact location of the firefighting training area, frequency of training sessions, and history of firefighting training activities unknown. From 1987 to date, it has been a large training area for firefighters. Samples collected in six areas from two sampling campaigns. First sampling campaign collected 30 soil cores between 2 m and 4 m in June 2015 from areas 1–5 (composite soil samples collected	Area 1: SC-71, SC-72, SC-73, SC-74, SC-75 = <2 ng/g dw (1–2 m) SC-76, SC-77, SC-78 = <2 ng/g dw (0–1 m) Area 2: SC-58 = <20, <20, <20, <20, <2, <20 ng/g dw (0–0.25, 0.5–0.75, 1–1.5, 2–2.5, 3–3.5, 3.5–4 m) SC-59 = <20, <4, <4, <2, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 3–3.5 m)

Study	Location	Site Details	Results
		every 25 cm in the topmost meter and then every 50 cm). Second sampling campaign collected 14 soil cores between 4 m and 15 m from areas 1–6 (thickness of composite soil samples ranged from 25 cm to 100 cm) in October 2016. Area 1 stored raw oil products when the oil refinery was operating; a preliminary survey showed hydrocarbon traces in the area, suggesting that an incident had occurred and that fluorinated surfactants could have been used. Area 2 is one of the main areas used for firefighting activities since 1987; training sessions held directly on the ground before 10-cm thick concrete slab was built in the 1990s. Area 3 was used for firefighting activities since 1987 and is situated on a 1-meter thick concrete slab on the foundations of the former oil refinery. Area 4 corresponds to the site's WWTP where sludge and sediment from a lagoon were stored directly on the ground; influents of the WWTP are highly contaminated by PFAS. Area 5 was used for firefighting training exercises by the former oil refinery. Area 6 is used for firefighting exercises with tank trucks.	SC-60 = <20, <20, <4, <4, <2, <2, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 2–2.5, 2.5–3 m) SC-61 = <20, <20, <4, <4, <20, <4, <4 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m) SC-62 = <20, <20, <4, <20, <4 ng/g dw (0–0.25, 0.5–0.75, 1–1.5, 2–2.5, 3.5–4 m) SC-63 = <20, <20, <4, <2, <2, <2, <2, <2, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5 m) SC-64 = <2, <2, <4, <4, <2, <4 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m) SC-65 = <20, <20, <20, <4, <4, <2, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 3–3.5, 3.5–4 m) SC-66 = <4, <4, <4, <2, <2, <2, <2, <2, <2, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5, 3.5–4 m) SC-58b = <10 ng/g dw (4–5, 5–6, 9–10, 14–15 m) SC-59b = <10, <10, 2, <10, <10 ng/g dw (3–4, 4–5, 6–7, 9–10, 14–15 m) SC-65b = <10 ng/g dw (4–5, 7–9, 9–11, 14–15 m) SC-67 = <20, <20, <10, <10 ng/g dw (0–1, 1.3–2, 2–3, 4–5 m) Area 3: SC-40 = <4 ng/g dw (0–1 m) SC-43 = <2 ng/g dw (1–2 m) SC-45 = <2 ng/g dw (0–1 m) SC-47 = <20 ng/g dw (0–1 m) SC-48 = <4 ng/g dw (0–1 m) SC-41 = <10 ng/g dw (0–0.25, 1–2 m) SC-42 = <10 ng/g dw (0–0.25, 1–2, 3–4 m) Area 4: SC-33 = <20 ng/g dw (0–1 m) SC-34 = <4 ng/g dw (1–2 m)

Study	Location	Site Details	Results
			SC-35 = <2 ng/g dw (0–1m) SC-36 = <4 ng/g dw (0.3–1 m) SC-37 = <20 ng/g dw (0.1–1.1 m) SC-37b = <10 ng/g dw (0–0.25, 1–1.5, 3–4 m) SC-38 = <10 ng/g dw (0.25–1, 2–3 m) Area 5: SC-10 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5 m) SC-11 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m) SC-12 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75 m) Area 6: SC-21 = 5, <10, <10, <10, <10 ng/g dw (0–0.25, 0.25–1, 2–3, 8–9, 13–15 m) SC-22 = 3, 2, <10, <20 ng/g dw (0–0.25, 0.25–1, 1–2, 3–4 m) SC-23 = 15, 3, <10, <10 ng/g dw (0–0.25, 0.25–1, 1–2, 3–4 m) SC-24 = 10, <10, 3 ng/g dw (0–0.25, 1–2, 3–4 m) SC-25 = 3, <10 ng/g dw (0–0.25, 1.5–2 m) SC-26 = <10 ng/g dw (2–3, 4–5 m) (LOQ = 2 ng/g dw)
Skaar et al. (2019)	Norway (Ny-Ålesund)	Samples collected in June 2016 in and around the international research facilities (Ny-Ålesund) near local firefighting training site. Background soil samples were collected at representative locations.	Background: n = 8, DF 0% Contaminated: n = 2, DF[a] 50%, range = <0.005–0.73 ng/g dw (IDL = 0.026 ng; LOD = 0.005 ng/g dw; LOQ = 0.01 ng/g dw) *Table 1 and Table S2 reported a total of nine samples across background and contaminated sites; however, Tables S11 and S13 report a total of ten samples (two contaminated sites from Table S11 and eight background sites from Table S13
Hale et al. (2017)	Norway (Gardermoen)	Samples collected in June 2015 from six locations around a firefighting training facility west of the Oslo airport site. Samples were taken at 0–1 m, 1–2 m, 2–3	n = 22, DF 40%, range = 2.8–41.3 ng/g (LOD = 1 ng/g)

Study	Location	Site Details	Results
		m, and 3 to groundwater table level (which was in all cases above 4 m). Facility was established in 1989 and AFFF was used extensively. AFFF containing PFOS was banned at the facility in 2007 and a complete ban on organofluorine AFFF was enforced in 2011. The soil is known to be contaminated with a range of perfluorinated compounds.	*Range reported for detects *The DF and range extracted are reported in the results (Section 3.1); however, Table S2 of the individual sample data show all concentrations ranging from <1.8 to <2.5 ng/g
Harrad et al. (2020)	Ireland (multiple cities)	Samples collected from ten municipal solid waste landfills upwind and downwind at each site between November 2018 and January 2019. At each upwind/downwind location, nine sub-samples of soil were taken in a "W" formation. Samples were collected from the same areas as air samples and were taken within the boundaries of the landfill operational facility. Soil used as capping on landfill cells was not sampled to ensure soil samples were not collected from soil placed after landfill operations ceased and that farming activities would not influence concentrations found. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction & demolition, and biomedical waste.	Downwind: n = 9, DF[a] 89%, mean, median (range) = 0.0045, 0.0043 (<0.001–0.0077) ng/g dw Upwind: n = 7, DF[a] 100%, mean, median (range) = 0.014, 0.006 (0.0029–0.033) ng/g dw (LOD = <0.001 ng/g dw) *Non-detects replaced by ½ LOD *Soil samples from three upwind locations and one downwind location destroyed in transit from field to laboratory
Grønnestad et al. (2019)	Norway (Granåsen, Jonsvatnet)	Upper layer soil samples (3–10 cm in depth) collected in June 2017 and 2018 from Granåsen (skiing area) and Jonsvatnet (reference site). Five samples per year were analyzed for each site. Located 10 km from Trondheim city center, Granåsen is the main arena for winter sports in Trondheim and hosts an annual ski jumping World Cup event and regional, national, and international competitions in cross-country skiing. Located 15 km away from Trondheim city center, Jonsvatnet is a natural forest area not used for ski-sports and is in the vicinity of an ecological farm next to Lake Jonsvatnet. The two study areas have similar vegetation.	Reference area: n = 10, DF 70%, mean (range) = 0.198 (<LOQ–0.928) ng/g dw Skiing area: n = 10, DF 90%, mean (range) = 0.179 (<LOQ–0.602) ng/g dw (LOQ = 0.056 ng/g dw)
Sammut et al. (2019)	Malta	Six surface soil samples (#10, 14, 15, 18, 20, and 22) collected between June and August 2015 from random small urban fields.	n = 6, DF[a] 100%, mean[a] (range) = 0.75 (0.66–0.87) ng/g (LOD = 0.50 ng/g; LOQ = 0.60 ng/g)

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; dw = dry weight; LOQ = limit of quantitation; LHWA = Little Hocking Water Association; LOD = limit of detection; MDL = method detection limit; 0.5×, 1×, or 2× = ½, 1, or 2 times the agronomic rate of biosolids application to meet nitrogen requirements of the crop; ND = not detected; NR = not reported; PFAA = perfluoroalkyl acids ; RL = reporting limit; WWTP = wastewater treatment plant.

C.3.7. Sediment

When released into water, based on its physico-chemical properties, PFNA is expected to adsorb to suspended solids and sediments (NCBI, 2022b). The EPA did not identify studies conducted in the U.S. that reported the occurrence of PFNA in sediment. Concentrations of PFNA in sediment samples collected from the Hudson Bay region of northeast Canada ranged from <0.06 ng/g to 0.14 ng/g (dry weight) (NCBI, 2022b; Kelly et al., 2009).

C.4. Recommended RSC

The EPA followed the Exposure Decision Tree approach to determine the RSC for PFNA (USEPA, 2000b). The EPA first identified three potential populations of concern (Box 1): pregnant women and their developing fetuses, lactating women, and women of childbearing age (see Section 2.3.2). However, limited information was available regarding specific exposure of these populations to PFNA in different environmental media. The EPA considered exposures in the general U.S. population as likely being applicable to these two populations. Second, the EPA identified several relevant PFNA exposures and pathways (Box 2), including dietary consumption, incidental oral consumption via exposure to dust, consumer products, and soil, dermal exposure via soil, consumer products, and dust, and respiration via ambient air. Several of these may be potentially significant exposure sources. Third, the EPA determined that there was not adequate quantitative data to describe the central tendencies and high-end estimates for all of the potentially significant sources (Box 3). For example, studies from Canada and Europe indicate that indoor air may be a significant source of exposure to PFNA. At the time of the literature search, the EPA was unable to identify studies assessing PFNA concentrations in indoor air samples from the U.S. and therefore, the agency does not have adequate quantitative data to describe the central tendency and high-end estimate of exposure for this potentially significant source in the U.S. population. However, the agency determined there were sufficient data, physical/chemical property information, fate and transport information, and/or generalized information available to characterize the likelihood of exposure to relevant sources (Box 4). Notably, based on the studies summarized in the sections above, there are significant known or potential uses/sources of PFNA other than drinking water (Box 6), though there is not information available on each source to make a characterization of exposure (Box 8A). For example, there are several studies from the U.S. indicating that PFNA may occur in dust sampled from various microenvironments (e.g., homes, offices, daycare centers, vehicles). However, the majority of studies sampled in only one location and few studies examined dust samples outside of the home (e.g., one study assessed PFNA occurrence in dust sampled from vehicles). Additionally, though several studies from around the U.S. measured PFNA concentrations in dust from houses, the detection frequencies in these studies varied widely (from 35% to 100%) and may be a result of uncertainties including home characteristics, behaviors of the residents, and the presence or absence of PFNA-containing materials or products (Haug et al., 2011). Therefore, it is not possible to determine whether dust can be considered a major or minor contributor to total PFNA exposure. Similarly, it is not possible to determine whether the other potentially significant exposure sources such as seafood and consumer products should be considered major or minor contributors to total PFNA exposure. Given these considerations, following recommendations of the Exposure Decision Tree (USEPA, 2000b), the EPA recommends an RSC of 20% (0.20) for PFNA.

Appendix D. PFHxS: Summary of Occurrence in Water and Detailed Relative Source Contribution

D.1. Occurrence in Water

The production of PFHxS and its use as a raw material or precursor for manufacturing PFAS-based products, as well as its previous use in firefighting foam and carpet treatment solutions, could result in its release to the aquatic environment through various waste streams (NCBI, 2022a). PFHxS has an estimated water solubility of 6,200 µg/L (6.2 mg/L) at 25°C and when released to surface water, it is not expected to adsorb to suspended solids and sediment (NCBI, 2022a). Volatilization from water surfaces is not expected to be an important fate process for PFHxS (NCBI, 2022a).

D.1.1. Groundwater

Several studies have evaluated the occurrence of PFHxS in groundwater in both the United States and Europe. PFHxS was detected in at least one groundwater sample site in each study in the U.S (Table D-1). Lindstrom et al. (2011) analyzed well water samples in Decatur, Alabama. The samples were collected in February 2009 from farms that had applied PFC-contaminated biosolids to local agricultural fields as a soil amendment for at least 12 years. PFHxS was detected in two wells at concentrations of 56.5 and 87.5 ng/L. In another study, median and maximum groundwater of 870 ng/L and 290,000 ng/L (0.870 µg/L and 290 µg/L), respectively, were detected at 10 U.S. military installations (Anderson et al., 2016). Three other studies of groundwater known to be impacted by nearby AFFF use similarly had PFHxS concentrations ranging from 36-120,000 ng/L and detection frequencies of 100% (Steele et al., 2018; Eberle et al., 2017; Moody et al., 2003).

Post et al. (2013) assessed raw water from public drinking water system intakes that were chosen to represent New Jersey geographically but were not necessarily associated with any known PFAS release. PFHxS was found in 2 of 18 systems at levels ≤10 ng/L. Appleman et al. (2014) evaluated groundwater contaminated by wastewater effluent discharge. At this site, detection frequency was 100%, but PFHxS levels did not exceed 11 ng/L. Procopio et al. (2017) collected groundwater from areas downstream of a manufacturer of PFAS-containing products but found minimal PFHxS in only 5% of samples, all of which ranged from non-detects to 5.5 ng/L. Boone et al. (2019) evaluated 17 PFAS in source and treated waters collected in 2010–2012. Of the three groundwater sources evaluated, PFHxS was detected in two out of three samples at levels of 1.88 and 44.8 ng/L. In the final U.S.-based study, Quiñones and Snyder (2009) examined levels of eight PFAS at two sites in Las Vegas Wash, Nevada that were highly impacted from treated wastewater. Samples were collected in 2008 as part of a study to assess both raw and treated water from utilities producing at least 75 megaliters of finished water per day. Mean PFHxS levels at the two sites were 6.8 and 5.6 ng/L at sites 1 (n = 7) and 2 (n = 8), respectively.

Of the studies conducted in Europe, 4 studies (Barreca et al., 2020; Sammut et al., 2019) were conducted in areas not associated with any known PFAS release. At these sites, the detection frequency of PFHxS was 0–40% with a maximum level of 32 ng/L. The remaining nine

European studies evaluated groundwater samples from sites with known or suspected PFAS releases associated with fluorochemical manufacturing (Boiteux et al., 2012; Loos et al., 2010) or AFFF use (Boiteux et al., 2017; Dauchy et al., 2012). Source categories for Gobelius et al. (2018) included fire training sites, but also included landfill/waste disposal sites, skiing areas, urban areas, and areas of unspecific industries. Of the sites with known sources of contamination, higher detection frequencies (up to 100%) and greater PFHxS levels (up to 3,470 ng/L) were reported at sites with AFFF use. Two related studies (Boiteux et al., 2017; Dauchy et al., 2012) sampled alluvial wells downstream of a fluorochemical manufacturing facility in France. Preliminary results showed low levels of PFHxS (up to 11 ng/L; (Dauchy et al., 2012)), but PFHxS was not detected in samples from the latter study (Boiteux et al., 2017).

Table D-1. Studies Reporting Occurrence of PFHxS in Groundwater

Study	Location	Site Details	Results
United States			
Procopio et al. (2017)	United States (New Jersey)	Groundwater from an industrial/business park located within the South Branch Metedeconk River watershed, where there was suspected illicit discharge to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs. Samples were collected following the installation of 16 temporary monitoring wells by the NJ Geological and Water Survey or a contract driller during August 2013 (sampling event #7) and June–July 2014 (sampling event #8). Samples were taken from the upper 1.5 m (5 ft) of the water table from each well, except for one "profile well" in which samples were collected at three different depths (3.7–4.6, 6.7–7.6, and 10.7–11.6 m below grade; 12–15, 22–25, and 35–38 ft below grade, respectively).	n = 19, DF[a] 5%, range = <5–5.5 ng/L (Minimum RL = 5 ng/L)
Post et al. (2013)	United States (New Jersey)	Raw water collected from public drinking water system intakes in two sampling campaigns. In the first sampling campaign, samples from 18 drinking water systems were collected between August 2009 and February 2010 from 1 confined well (sunk into an aquifer located between two impermeable strata) and 17 unconfined wells in the upper unconfined aquifer; sites were chosen to represent NJ geographically and included 1 site with a nearby industrial facility that previously used large quantities of PFNA (site 5). In the second sampling campaign, samples from two drinking water systems (PWS-A and PWS-B) were collected in 2010–2013 from five unconfined wells. Groundwater at these two systems were known to be contaminated by PFOA.	1st sampling campaign: n = 18, DF 11%, range = ND–10 ng/L 2nd sampling campaign: PWS-A, WF1A: n = 5, DF 0% PWS-A, WF1B: n = 4, DF 0% PWS-A, WF2A: n = 9, DF[a]14%, range = ND–6 ng/L PWS-A, WF2B: n = 9, DF 0% PWS-B: n = 8, DF 0% (Minimum RL = 5 ng/L)
Lindstrom et al. (2011)	United States (Decatur, Alabama)	Thirteen samples collected in February 2009 from 13 wells located on farms with historical land application of PFC-contaminated biosolids to local agricultural fields between 1995 and 2008. Biosolids obtained from local municipal WWTP where sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers,	n = 13, DF (frequency of quantification)[a] 15%, range = <LOQ–87.5 ng/L (LOQ = 10 ng/L)

Study	Location	Site Details	Results
		fluorocarbon fibers, polymers, polymer films, and resins.	
Boone et al. (2019)	United States (unspecified)	Three groundwater sites used as source waters for three DWTPs, collected in 2010−2012; some locations with known or suspected sources of wastewater in the source water, but study did not differentiate which locations had known or suspected sources.	n = 3, DF[a] 67%, range = ND–44.8 ng/L (LCMRL = 0.034 ng/L)
Appleman et al. (2014)	United States (New Jersey)	Groundwater source water for five DWTPs, sampled November 2011 to September 2012. Majority of the utilities were selected because they were either known from previous monitoring or expected based on their source waters to contain detectable PFAS (i.e., impacted by upstream wastewater effluent discharge). Two sites were sampled twice and three sites were sampled only Once.	n = 7, DF[a] 100%, mean[a] (range) = 5.5 (0.48–11) ng/L (Method RL = 0.25 ng/L)
Quiñones and Snyder (2009)	United States (Nevada)	Samples collected in 2008 from two groundwater sites in Las Vegas Wash, Nevada that were highly impacted from treated wastewater.	Site 1: n = 7, DF NR, mean (maximum) = 6.8 (24) ng/L Site 2: n = 8, DF NR, mean (maximum) = 5.6 (13) ng/L (Method RL = 1.0 ng/L)
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between 1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were not specifically controlled for in the site selection process; active remedies had not been applied at any of the sites selected. Approximately ten samples were collected between March and September 2014 at each site; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases. Groundwater	Overall: n = 149, DF 94.93%, median (maximum) = 870 (290,000) ng/L *Breakdown by site group:* Emergency Response (low-volume release): n = 24, DF 87.5%, mean (range) = 20,100 (10–270,000) ng/L Hangars and Buildings (medium-volume release): n = 100, DF 90.6%, mean (range) = 71,400 (23–910,000) ng/L Testing and Maintenance (high-volume release): n = 25, DF 100.0%, mean (range) = 400 (390–420) ng/L (Median RL = 7 ng/L) *Minimum of detected values reported

Study	Location	Site Details	Results
		samples were collected from existing monitoring wells and temporary monitoring wells installed with direct push technology.	*Median calculated using quantified detections *Non-detects were substituted with ½ the reporting limit
Steele et al. (2018)	United States (Alaska)	Monthly samples collected from a military installation during July 2016–March 2017; six wells from around the installation were sampled each month, along with a seventh well that was only sampled in July 2016. PFAS contamination predominately from prior legacy AFFF use. Wells selected based on historical sample data indicating PFAS contamination.	Data for July, August, September, October, November, December, January, and February, respectively: Well A: 170, 140, 130, 180, 150, 96, 120, 100 ng/L Well B: 220, 360, 360, 370, 410, 300, 400, 370 ng/L Well D: 120 ng/L (for July only; no values provided for other months) Well E: 36, 39, 48, 240, 210, 94, 85, 77 ng/L Well F: 82, 110, 110, 240, 150, 110, 100, 110 ng/L DK: 460, 590, 590, 700, 700, 530, 690, 740 ng/L FG: 60, 69, 75, 61, 93, 64, 67, 59 ng/L (Minimum DL not reported)
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, Virginia)	Pilot testing area in former fire training area (Training Site 15) at Joint Base Langley-Eustis where monthly fire training activities were conducted from 1968 to 1980 in a zigzag pattern burn pit. Facility was abandoned in 1980 but irregular fire training activities using an above-ground germed burn pit continued until 1990. Groundwater samples collected for screening/site characterization (April and December 2012), and for pre- (April 2013) and post- (October 2013 and February 2014) in situ chemical oxidation treatment using a peroxone activated persulfate (OxyZone) technology. Treatment was conducted in Test Cell 1 over 113 days (April through August 2013). Pre-treatment samples were collected from 14 wells screened in the deep zone, and 3 wells screened in the shallow zone. Post-treatment samples were collected from the same wells as the pre-treatment samples with an additional three wells (two shallow, one deep) sampled. Wells EC-1, EC-2, EC-3, EC-4, I-	Screening/site characterization: EC-1 (deep, sentry): 19,000 ng/L EC-2 (deep, sentry): 39,000 ng/L I-1 (deep):32,000 ng/L I-2 (deep, sentry): 57,000 ng/L I-4 (deep): 80,000 ng/L I-5 (shallow): 13,000 ng/L I-6 (shallow): 13,000 ng/L MW-2904 (deep): 9,400 ng/L U-16D (deep): 44,000 ng/L U-16S (shallow): 19,000 ng/L

Study	Location	Site Details	Results
		2, and I-3 were sentry wells to monitor the possible migration of oxidants and contaminants outside Test Cell 1. Limited data reported for post-treatment samples.	Pre-treatment: (values reported for two different laboratories: TA; CSM) EC-2 (deep, sentry): 40,000; 48,000 ng/L EC-3 (deep, sentry): 46,000; 59,400 ng/L I-1 (deep): 25,000; 24,900 ng/L I-2 (deep, sentry): 23,000; 20,400 ng/L I-4 (deep): 64,000; 66,400 ng/L Post-treatment: I-4 (deep), U-16D (deep), U-17D (deep), and U-20D (deep) showed a 56% reduction in PFHxS compared to pre-treatment values (LOQ = 1 ng/L)
Moody et al. (2003)	United States (Oscoda, Michigan)	Groundwater samples collected from ten wells during November 1998 and June 1999 from plume at Fire Training Area Two (FTA-02) at the former Wurtsmith Air Force Base; FTA-02 was used from 1952 to 1993 to train U.S. military personnel in firefighting procedures and included flooding a concrete pad with flammable liquids, igniting the fluids, and extinguishing the fire with firefighting agents including AFFF. Minimum of five years since active firefighting activity.	Well ID (distance from fire training pad): FT2 (17 m): n = 1, point = 120,000 ng/L FT3 (18 m): n = 1, point = 104,000 ng/L ML3 (114 m): n = 1, point = 70,000 ng/L ML8 (121 m): n = 1, point = 39,000 ng/L FT8 (183 m): n = 1, point = 30,000 ng/L FT9 (183 m): n = 1, point = 46,000 ng/L FT12 (305 m): n = 1, point = 23,000 ng/L FT14 (305 m): n = 1, point = 27,000 ng/L FT18 (518 m): n = 1, point = 33,000 ng/L FT17 (540 m): n = 1, point = 9,000 ng/L (LOD = 3,000 ng/L; LOQ = 13,000 ng/L) *Point value reported is from five replicate analyses of one sample
Europe			
Bach et al. (2017)	France (southern)	Samples were collected from alluvial wells that provide source water for two DWTPs. The two DWTPs are located on both sides of a river, ~15 km downstream from an industrial site where two facilities produce fluoropolymers; the industrial site	Alluvial wells for DWTP A: April 2013: n = 7, DF[a] 29%, range = <4–7 ng/L

Study	Location	Site Details	Results
		discharges its effluents at three points along a river. The alluvial wells are located along the river, with wells for the first DWTP (DWTP A) located on the left shore and alluvial wells for the second DWTP (DWTP B) located on the right shore, on an island formed by a backwater. Sample collection occurred in April, July, October, and December 2013.	July 2013: n = 7, DF[a] 43%, range = <4–8 ng/L October 2013: n = 7, DF[a] 43%, range = <4–8 ng/L December 2013: n = 7, DF[a] 57%, range = <4–6 ng/L Alluvial wells for DWTP B: April 2013: n = 8, DF[a] 50%, range = <4–7 ng/L July 2013: n = 8, DF[a] 63%, range = <4–8 ng/L October 2013: n = 7, DF[a] 57%, range = <4–8 ng/L December 2013: n = 8, DF[a] 63%, range = <4–6 ng/L (LOQ = 4 ng/L) *DF represents frequency of quantification
Boiteux et al. (2017)	France (northern)	Samples were collected in four sampling campaigns (May, July, October, and December 2013) from alluvial wells that provide source water for two DWTPs. The two DWTPs (A and B) are located downstream of an industrial WWTP that processes raw sewage from a facility that manufactures fluorotelomer-based products and side-chain-fluorinated polymers used in firefighting foams and stain repellents. DWTP A is located 15 km downstream from the WWTP and is supplied by five alluvial wells. DWTP B is located 20 km downstream of the WWTP and is supplied by four alluvial wells.	DWTP A: May 2013: n = 5, DF 0% July 2013: n = 5, DF 0% October 2013: n = 5, DF 0% December 2013: n = 5, DF 0% DWTP B: May 2013: n = 4, DF 0% July 2013: n = 4, DF 0% October 2013: n = 4, DF 0% December 2013: n = 4, DF 0% (LOQ = 4 ng/L) *DF represents frequency of quantification

Study	Location	Site Details	Results
Dauchy et al. (2012)	France (unspecified)	Raw water sampled in June 2010 from four monitoring wells at a fluoropolymer manufacturing plant (P13, P14, P15, P01). Groundwater flowed from well P14 to P01 and well P15 is nearest to the polyvinylidene fluoride production area. Raw water resources also collected from two DWTPs (five sampling sites – DWA-1, DWA-2, DWA-3, DWA-4, DWB-1); the first DWTP (DWA) is supplied by four alluvial wells, and the second DWTP (DWB) is supplied by one alluvial well. The two DWTPs are located on both sides of a river, 15 km downstream of fluorochemical manufacturing facility. The river receives wastewater from many domestic and industrial activities.	Fluoropolymer manufacturing plant: P13: n = NR, DF NR, 68 ng/L P14: n = NR, DF NR, 8 ng/L P15 and P01 not quantifiable due to dilution or matrix effects DWA: DWA-1: n = NR, DF NR, 11 ng/L DWA-2: n = NR, DF NR, 8 ng/L DWA-3: n = NR, DF NR, 4 ng/L DWA-4: n = NR, DF NR, <4 ng/L DWB: DWB-1: n = NR, DF NR, 5 ng/L (LOQ = 4 ng/L) *Study did not indicate whether concentrations reported were point values or means
Harrad et al. (2020)	Ireland (multiple cities)	Groundwater samples collected between November 2018 and January 2019 from ten municipal solid waste landfills at two sampling points down-gradient from the main body of each landfill. Each sampling point consisted of a borehole leading down to water reservoirs at a minimum depth of 5 m below ground level. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction and demolition, and biomedical waste.	n = 10, DF 20%, mean, median (range) = <0.1, <0.1 (<0.1–0.28) ng/L (LOD = <0.1 ng/L) *Non-detects replaced by ½ LOD
Gobelius et al. (2018)	Sweden (national)	Sampling conducted between May and August 2015, with the majority in July and a few samples in September and November 2015. Samples were collected in 21 regional counties by the County Administration Boards. Sampling locations selected based on potential vicinity of PFAS hot spots (i.e., fire training sites, unspecific industry, sewage treatment plant effluent, landfill/waste disposal, skiing, and urban areas) and/or importance as a drinking water source. Sample numbers varied for each county and sampling sites were spread unevenly across Sweden.	n = 161, DF[a] 37%, range = <0.15–80 ng/L (MDL = 0.15 ng/L)

Study	Location	Site Details	Results
Boiteux et al. (2012)	France (national)	Raw water from DWTPs distributed across 100 French departments to represent ~20% of the national water supply flow; samples collected during two sampling campaigns in July–September 2009 (first campaign) and June 2010 (second cam–aign - focused on sites from first sampling campaign that had PFC levels >LOQ). Some sites possibly affected by commercial/industrial releases.	Overall: n = 196, DF (frequency of quantification) 18%, maximum = 32 ng/L *1st Sampling Campaign* n = 163, DF 31%, mean, median (maximum) = 1, <1 (32) ng/L 2nd Sampling Campaign n = 33, results not reported (LOD = 1.3 ng/L, LOQ = 4 ng/L)
Loos et al. (2010)	23 European countries	Groundwater collected from 164 groundwater monitoring stations of participating European Union Member State laboratories during an 8-week window in Fall 2008. There were no strict selection criteria for the sampling sites such as “representative” or “contaminated”. Most monitoring stations were “official” monitoring stations also used for drinking water abstraction.	n = 164, DF 34.8%, mean, median (maximum) = 1, 0 (19) ng/L (LOD = 0.4 ng/L)
Dauchy et al. (2019)	France (unspecified)	Samples collected in two sampling campaigns in and around site where fluorosurfactant-based foams have been used extensively. From 1969 to 1984, the site was an oil refinery, with the exact location of the firefighting training area, frequency of training sessions, and history of firefighting training activities unknown. From 1987 to date, it has been a large training area for firefighters. First sampling campaign collected 13 samples from 9 monitoring wells and 4 springs in June 2015. Second sampling campaign collected from four monitoring wells in October 2016. Monitoring wells MW-1 to MW-5 were located upgradient from the firefighter training site around a landfill site. Monitoring well MW-11 and springs SW-A, SW-B, and SW-D located downgradient from the landfill or firefighter training site but not in the direction of groundwater flow. Monitoring wells MW-6 to MW-13 and spring SW-C were located downgradient from the firefighter training site in the direction of groundwater flow.	Upgradient: Monitoring wells: n = 5, DF[a] 40%, range = <4–42 ng/L Downgradient but not in the direction of groundwater flow: Monitoring wells: n = 1, point = 42 ng/L Spring water: n = 3, DF[a] 33%, range = <4–7 ng/L Downgradient in the direction of groundwater flow: Monitoring wells: n = 7, DF[a] 100%, mean[a] (range) = 660 (26–2,860) ng/L Spring water: n = 1, point = 122 ng/L (LOQ = 4 ng/L)
Høisæter et al. (2019)	Norway (unspecified)	Firefighting training site with an airport that extensively used AFFF containing PFOS since the early 1990s until 2001 when it was replaced by fluorotelomer containing AFFF. All PFAS containing	n = 19, DF NR, mean* = 2,900 ng/L (LOD/LOQ not reported) *Mean estimated from Figure 4b in the paper

Study	Location	Site Details	Results
		firefighting foams was banned at the airport in 2011. Groundwater samples collected in 2016 at five pumping wells installed down gradient of the site to intercept and pump and treat the plume spreading fr10irefightrefighting training site. A total of 19 sampling campaigns were performed.	
Dauchy et al. (2017)	France (unspecified)	Samples collected in the vicinity of three sites (A, C, D) where fluorosurfactant-based foams are or were being heavily used. Site A is an oil storage depot located in a river port. In June 1987, a large explosion occurred in the depot and the fire was extinguished by applying a large amount of fluorosurfactant-based foams. Two groundwater samples were collected in October 2014 and March 2015 from a monitoring well located in the center of the depot. The water table lies 2.5 – 3.5 m below the ground. Site C is a military airport, with the exact location of the training area, frequency of the training sessions, and history of the firefighting training activities unknown. The well supplying the DWTP was sampled in March 2015. Site D is a training center for firefighters. From 1969 to 1984, the site was an oil refinery. Starting in 1987, the site became a training area for firefighters, with exercises carried out directly on the soil. From the 1990s, some exercise areas were covered with concrete. In November 2014, groundwater samples were collected from five springs.	Site A: October 2014: n = 1, point = 139 ng/L March 2015: n = 1, point = 136 ng/L Site C: n = 1, point = 25 ng/L Site D: n = 5, DF[a] 20%, range = <4–132 ng/L (LOQ = 4 ng/L)
Filipovic et al. (2015)	Sweden (Stockholm)	Groundwater samples collected at the airfield and in the vicinity of the airport of the closed air force base F18 in Tullinge Riksten, 19 km south of Stockholm city center, where AFFFs were used. Samples collected in two sampling campaigns in December 2011 and May 2012. The air force base was formally demobilized in 1986 but continued to be used as an air force school for combat command and air surveillance until 1994. Of note, the air force base encountered numerous accidents and incidents during the transfer from propeller era to the jet engine era, including planes crashing upon takeoff and landing, fire incidents, accidental dispersion of jet engine starting	n = 16, DF 69%, range = <0.5–3,470 ng/L *Highest concentrations of 2,960 and 3,470 ng/L detected at sites G5 and G6 (MDL = <0.5 ng/L)

Study	Location	Site Details	Results
		fuel. The area was sold to a land developer in 1996 and is in the process of being transformed into a municipal area. Groundwater flow is directed from the military airfield towards Lake Tullingesjön. Sampling sites included locations under the main firefighting training facility (sites G5 and G6). Other groundwater sampling sites were not mapped to specific locations (note that soil samples were collected at the main firefighting training facility, intermediate soil depot, J34 Hawker Hunter site, old fire station, and soil depot).	
Gyllenhammar et al. (2015)	Sweden (Uppsala)	Three observation well sites (Tuna backar: n = 3 wells; Svartbäcken: n = 1 well, Librobäck: n = 2 wells;) were sampled from September 2012 to January 2013. Four DWTP production well sites (Storvad: n = 9 wells; Galgbacken: n = 1 well; Stadsträdgården and Kronåsen: n = 6 wells; Sunnersta: n = 5 wells) were sampled from July 2012 to February 2014. One private well (Klastorp) was sampled in September 2012. All wells located downstream of a military airport with firefighting training activities up to the year 2003. It is not known when the usage of AFFF started.	Observation wells: Tuna backar: n = 3, DF 100%, median = 690 ng/L Librobäck: n = 4, DF 0% Svartbäcken: n = 3, DF 100%, median = 250 ng/L Production wells: Storvad: n = 12, DF 0% Galgbacken: n = 7, DF 0% Stadsträdgården and Kronåsen: n = 103, DF 100%, median = 83 ng/L Sunnersta: n = 50, DF[a] 52%, median = 8 ng/L Private well: Klastorp: n = 1, point = 16 ng/L (MDL = 10 ng/L)
Wagner et al. (2013)	Germany (unspecified)	Groundwater samples collected downstream from a site contaminated by PFC-based AFFFs from firefighting activities. Sampling year not provided. Samples used to test out a new analytical protocol to determine trace levels of adsorbable organic fluorine.	n = 3, DF[a] 100%, mean (range) = 368 (230–510) ng/L (LOD = 50 ng/L F^-; LOQ = 150 ng/L F^-) *PFHxS concentrations were calculated using the fluorine concentrations reported in Table 4

Study	Location	Site Details	Results
Barreca et al. (2020)	Italy (Lombardia region)	Fifty-seven groundwater sampling stations throughout the region. Samples collected in 2018.	n = 130, DF 0% (LOQ = 1 ng/L)
Sammut et al. (2019)	Malta	Groundwater collected from ten boreholes at different areas on the island during November and December 2015 and January 2016. Collection sites were the most commonly used extraction sites by the Malta Water Services Corporation for water extraction as well as for sampling for water quality analysis.	n = 10, DF 70%, range = ND–2.22 ng/L (LOD = 0.02 ng/L; LOQ = 0.04 ng/L) *Of the ten samples, three were ND and three were <LOQ
Ciofi et al. (2018)	Italy (Tuscany)	Groundwater samples were collected at 12 locations. Sampling year was not reported. Each sample was collected from the phreatic layer with a mean depth between 10–75 m: GW-1: Siena, 10 m GW-2, GW-3, GW-4: Florence, 15 m GW-5: Prato, 75 m GW-6: Prato, 71 m GW-7: Prato, 70 m GW-8: Prato, 61 m GW-9: Florence, 17 m GW-10, GW-11, GW-12: Florence, 10 m	GW-1: n = 1, point = <0.014 ng/L GW-2: n = 1, point = <0.015 ng/L GW-3: n = 1, point = <0.016 ng/L GW-4: n = 1, point = <0.018 ng/L GW-5: n = 1, point = <0.015 ng/L GW-6: n = 1, point = 1.8 ng/L GW-7: n = 1, point = 1.0 ng/L GW-8: n = 1, point = <0.016 ng/L GW-9: n = 1, point = 1.8 ng/L GW-10: n = 1, point = <0.014 ng/L GW-11: n = 1, point = <0.013 ng/L GW-12: n = 1, point = 0.8 ng/L (MDL = 0.013–0.018 ng/L) *MDL varied by sample. MDL provided for 8 of 12 samples
Gellrich et al. (2013)	Germany (Hesse); France; Italy	Untreated water samples for preparation of mineral water included seven from Hesse, three from France, and four from Italy. The supplying waterworks obtain their untreated water either from Rhine river filtrate, a mixture of ground water and percolation water from the Rhine riverbed, drawn from wells 30–50 m deep or from wells in their closer vicinity. Sampling year not reported.	n = 14, DF (frequency of quantification) 0% (LOQ = 1 ng/L)
Llorca et al. (2012)	Spain (Barcelona)	Well water samples from two different sites were collected from the North of Barcelona metropolitan area in 2011.	n = 2, DF[a] 50%, range = <MLOQ–0.50 ng/L (MLOD = 0.27; MLOQ = 0.90 ng/L)

D.1.2. Surface Water

Overall, almost all U.S. studies reported PFHxS detected in at least one surface water sample site in each study. Three studies investigated surface water upstream and downstream of fluoropolymer facilities, with some sites also downstream of other potential PFAS sources (e.g., landfills, WWTP) (Galloway et al., 2020; Newsted et al., 2017; Newton et al., 2017). Galloway et al. (2020) assessed several rivers and tributaries along the Ohio River in three sampling trips in 2016. The sampling sites ranged from upstream, downstream, and north/northeast of a fluoropolymer facility and known PFAS containing landfills. In June 2016, samples were collected on a 188 km stretch of the Ohio River, from 130 km downstream to 58 km upstream of the facility, and tributaries that pass near known PFAS-containing landfills. In July 2016, samples were collected from lakes, rivers, and creeks to the north and northeast of the facility as far as 16 km downwind. The December 2016 trip expanded the collection radius to more than 48 km downwind to the north and northeast of the facility. PFHxS was detected in 96% of samples (n = 26) in June 2016, however all detects were below the LOQ (10 ng/L). Of the second sampling trip, PFHxS was detected at levels above the LOQ in three samples in July 2016 (n = 25), ranging from 64.5–79.0 ng/L. Finally, in December 2016, PFHxS was detected at levels above the LOQ in three samples ranging from 10.1–14.4 ng/L, detected but below the LOQ in 16 samples, and not detected in 21 samples. In Newsted et al. (2017), surface water samples were collected in August 2011 from a 3-mile section of the Upper Mississippi River: ten sampling reaches (three samples each) in an area between Ford Dam (between Minneapolis and St. Paul) and Hastings Dam (near Hastings) and which had been subject to 10–15 years of actions to reduce PFAS contamination from 3M Cottage Grove plant and other commercial/industrial entities. PFHxS was detected in one sample from reach 10, immediately downstream the 3M Cottage Grove facility outfall, at a concentration of 60.5 ng/L. PFHxS in all other samples was below the LOQ of 2.0 ng/L. Authors were not able to observe a clear and consistent time trend in water concentrations. Newton et al. (2017) investigated surface water upstream and downstream of facilities that manufactured or used fluorinated materials along the Tennessee River near Decatur, Alabama. Six sampling sites were located upstream of the manufacturing facilities and three sites were downstream. Among the upstream sites, three were also upstream of a WWTP. All samples were collected in October 2015. PFHxS was detected at one downstream site at a concentration of 39 ng/L; authors suggested elevated PFAS concentrations at downstream sites resulted from infiltration from groundwater or runoff from soil.

In five studies, sampling locations included surface waters potentially impacted by current and/or historic use of AFFFs (Genualdi et al., 2017; Anderson et al., 2016; Post et al., 2013; Nakayama et al., 2010; Nakayama et al., 2007). Genualdi et al. (2017) investigated a cranberry bog in Massachusetts approximately 10 miles from a military base with a history of AFFF usage. Bog water samples were collected in November 2016 and PFHxS was detected in all four samples (six total samples collected, two samples were lost due to evaporation) with a mean concentration of 10.98 ng/L. Authors concluded that given the presence and ratio of PFHxS and PFOS in the bog water samples, it was likely the surface water contamination was related to the previous AFFF usage. Anderson et al. (2016) assessed 40 sites across 10 active Air Force installations throughout the continental United States and Alaska between March and September 2014. Installations were included if there was known historic AFFF release in the period 1970–1990. The selected sites were not related to former fire training areas and were characterized according to volume of AFFF release—low (n = 2), medium (n = 32), and high (n = 2). PFHxS

was detected at both sites characterized to have high-volume AFFF releases, with mean concentration of 5,600 ng/L. Detection frequencies for sites with low- and medium-volume AFFF releases were 50.0% and 74.3%, respectively, and mean concentrations were 7.1 and 196,800 ng/L, respectively. Across all sites, the median concentration of detects was 710 ng/L. Post et al. (2013) evaluated raw surface water samples from 12 public drinking water system intakes collected between August 2009 and February 2010. Six rivers and six reservoirs, including two reservoirs in Atlantic County near a civil-military airport with possible AFFF use, were selected to represent New Jersey geographically. PFHxS was below the minimum RL (5 ng/L) in all six river samples. In reservoir samples, PFHxS was detected at two of six sites (n = 16) at concentrations of 44 and 46 ng/L. These two sites corresponded to the sites near the civil-military airport; given the presence of PFHxS and other PFCs, authors reported the contamination to be indicative of AFFF usage. Two studies from Nakayama et al. (2010; 2007) assessed surface water samples from the Upper Mississippi River, Missouri River, and Cape Fear River Basins. In Nakayama et al. (2010), a large-scale evaluation of the Upper Mississippi River Basin and portion of the Missouri River Basin was conducted to provide preliminary PFC data given the importance of the two basins in supplying drinking water. Between the two basins, 173 samples were collected across 88 sampling sites in March–August 2008 by several different agencies—Minnesota Pollution Agency, Wisconsin Department of Natural Resources, Illinois Environmental Protection Agency, and U.S. EPA Region 7 Water Quality Monitoring Team. Overall, the detection frequency of PFHxS was 89% with a median concentration of 0.71 ng/L. Authors reported higher PFC concentrations adjacent to chemical manufacturers, downstream of WWTPs receiving waste from those types of manufacturers, and near an airport with historic use of firefighting foams. In Nakayama et al. (2007), authors evaluated the performance of a new method for the collection and analysis of PFCs using samples collected from the Cape Fear River Basin, North Carolina during spring 2006. Authors noted possible sources of PFCs to the basin included firefighting foam from nearby air force bases and commercial/industrial facilities. One hundred surface water samples were taken from 80 sites selected to reflect water quality throughout the basin. PFHxS was detected in 98.7% of samples with mean and median concentrations of 7.29 and 5.66 ng/L, respectively. The highest concentrations were found in the middle reaches of the Cape Fear River and its two major tributaries.

Two studies examined surface water near or downstream of land application sites where PFC-contaminated WWTP effluent or biosolids were applied (Lasier et al., 2011; Lindstrom et al., 2011). In the first study, Lasier et al. (2011) sampled the Coosa River, Georgia during summer 2008; samples included two sites upstream (sites 1 and 2) and six sites downstream (sites 3–8) of a land application site, where treated effluent from a WWTP was sprayed. The WWTP processed effluents from multiple carpet manufacturers who were reported to use significant quantities of PFCs. Additionally, site 2 was downstream of a local airport, and site 4 was downstream of a manufacturing facility of latex and polyurethane back material—inputs for the carpet manufacturers. PFHxS was below the MDL (0.014 ng/g) at both of the upstream sites and at the most downstream sites (sites 7 and 8). Mean concentrations for sites 3–6 were 30, 31, 17, and 13 ng/L, respectively. Authors reported highest concentrations downstream of the land application and backing-material sites and then decreased concentrations increasingly downstream as a result of dilution. In the second study, Lindstrom et al. (2011) analyzed surface water samples from ponds and streams in Decatur, Alabama. The samples were collected in February 2009 from farms that had applied PFC-contaminated biosolids to local agricultural fields as a soil amendment for at least 12 years. The biosolids were obtained from a local municipal WWTP

where authors noted that sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers, fluorocarbon fibers, polymers, polymer films, and resins, although specific sources could not be characterized. PFHxS was detected in 22% of samples (n = 32), with levels ranging from below the LOQ (10 ng/L) to 218 ng/L.

Three studies evaluated surface water potentially impacted by wastewater (Boone et al., 2019; Subedi et al., 2015; Appleman et al., 2014). Boone et al. (2019) evaluated 17 PFAS in source and treated waters collected in 2010–2012. Authors attempted to select locations with known or suspected sources of wastewater in the source water, but ultimately the site selection was dependent upon the willingness of DWTPs to participate. The study did not differentiate which locations had known or suspected sources. Of the 22 surface water sources evaluated (16 river and 6 lake/reservoir), PFHxS was detected in 95% of samples (n = 22), ranging from not detected (LCMRL = 0.034 ng/L) to 19.7 ng/L. Subedi et al. (2015) collected 28 lake water samples from 3 sampling events in August–September 2012 and four sampling events in May–September 2013 from Skaneateles Lake. Sites were selected to be along the shoreline of homes that use an enhanced treatment unit for onsite wastewater treatment. Wastewater effluents were identified as a source of contamination to the lake. PFHxS was detected in 79% of samples with mean and median concentrations of 0.56 and 0.28 ng/L, respectively. Appleman et al. (2014) assessed source water from 11 utilities in Alaska, Alabama, Colorado, Nevada, New Jersey, Ohio, Oklahoma, and Wisconsin from August 2011 to May 2012, the majority of which were selected because they were either known from previous monitoring or expected to contain detectable PFAS because they were impacted by upstream wastewater effluent discharge. Authors evaluated the utilities and their effectiveness for removing PFAS. The study did not report an average concentration for PFHxS, but PFHxS was detected in 18 of 25 samples (from 8 of 11 utilities) with a maximum concentration of 13 ng/L.

In three studies, surface water samples were collected from locations with potential sources of PFAS that were not related to AFFF use (Procopio et al., 2017; Zhang et al., 2016; Sinclair et al., 2006). Procopio et al. (2017) evaluated samples collected between September 2011 and July 2014 from the Metedeconk River. Eight sampling events were conducted as part of a source trackdown study to identify potential sources of PFAS contamination after elevated PFOA levels were discovered at a raw surface water intake of the Brick Township Municipal Utilities Authority. In all 56 samples, PFHxS was below the minimum laboratory RL of 5 ng/L. Zhang et al. (2016) conducted analyses to determine major sources of surface water PFAS contamination. Freshwater sample collection sites included 22 sites in the state of Rhode Island (sampled June 2014) and 6 sites in the New York Metropolitan Area (sampled October 2014). Surface water sites were creeks and rivers in urban and rural locations. PFHxS was detected in 89% of samples (n = 28) ranging from below the limit of detection (LOD) to 35.022 ng/L. Authors identified potential PFAS sources at these sites to be metal coating plating; paint, coating, adhesive manufacturing; paper manufacturing; petroleum coal products manufacturing; printing activity; printing ink manufacturing; semiconductor manufacturing; sewage treatment; textile mills; waste management including landfills, and airports. PFHxS levels were below the LOD (0.06 ng/L) at three rural sites corresponding to a background site with no recorded upstream industrial facilities; a Pawcatuck River site 1 km upstream of a military, tactical, and performance synthetic and synthetic blend textiles manufacturer; and a Secret Lake-Oak Hill Brook site 2 km east of a legacy landfill site. Authors reported significantly higher concentrations in urban regions, with the highest being possibly attributed mainly to T.F. Green Airport near Mill Cove, Rhode Island.

In July 2004, Sinclair et al. (2006) collected 51 samples from nine major water bodies of New York to assess the distribution of PFAS. Water bodies included Lake Ontario, Niagara River, Lake Erie, Finger Lakes, Lake Onondaga, Lake Oneida, Erie Canal, Hudson River, and Lake Champlain. PFHxS was detected in 50 samples with median concentrations across all lakes ranged from 0.9 to 7.4 ng/L. The highest concentrations were detected at Lake Onondaga and Erie Canal with median concentrations of 7.4 and 2.6 ng/L, respectively. Authors noted that Lake Onondaga is a Superfund site, is influenced by several industrial sources located along the lake, and also receives effluent from the Metropolitan Syracuse sewage treatment plant. Based on the results of other PFAS detected, the authors also suggested that there may be greater industrial use of fluoropolymers and telomer-alcohol in the region, including the Erie Canal.

Of the remaining studies (Bradley et al., 2020; Pan et al., 2018; Boone et al., 2014; Quiñones and Snyder, 2009; Kim and Kannan, 2007), Bradley et al. (2020) analyzed samples of Lake Michigan untreated intake water as part of a study that also analyzed home tap water samples. Samples were collected in 2017 at four intake sites. PFHxS was detected in all seven samples with a mean concentration of 1.0 ng/L. Boone et al. (2014) developed and tested the accuracy and precision of an analytical method to determine PFCs in environmental and drinking waters. The authors analyzed PFCs in water samples collected from the surface of the Mississippi River at a low flow level (2.95 ft) in September 2010 and a high flow level (8.32 ft) in June 2009. Results were presented as means based on an average of primary and duplicate samples or an average of four replicates. PFHxS levels were 1.315 and 1.07 ng/L at low and high flow levels, respectively. In Quiñones and Snyder (2009), surface water samples from the Boulder Basin, Hoover Dam, and the lower Colorado River were collected in 2008, as part of a study to assess both raw and treated water from utilities producing at least 75 megaliters of finished water per day. PFC occurrence had not been previously determined or reported for these sites. Mean PFHxS levels at all sites were below the method RL (1.0 ng/L). Kim and Kannan (2007) sampled two urban lakes in Albany, New York during five sampling trips from February–November 2006. The lakes, Washington Park Lake are Rensselaer Lake, are located in downtown Albany and receive surface runoff from nearby roadways and residential areas during stormwater runoff. PFHxS was detected in Washington Park Lake (n = 6) at a mean and median concentration of 0.33 ng/L. PFHxS was detected in Rensselaer Lake (n = 5) at mean and median concentrations of 3.09 and 3.25 ng/L, respectively. Overall, PFHxS was detected in 81.8% of the 11 total samples. Finally, in a multicontinental study, Pan et al. (2018) assessed surface water samples from several countries including the United States (Delaware River), United Kingdom (Thames River), Germany and the Netherlands (Rhine River), and Sweden (Mälaren Lake). Twelve samples were collected in September–December 2016 along the Delaware River that spanned seven cities—Trenton, Bristol, Philadelphia, Chester, Delaware, Smyrna, and Frederica. Authors noted that all sampling sites were along the main stream of the studied rivers and not proximate to known point sources of any fluorochemical facilities. PFHxS was detected in all samples from the Delaware River with a mean concentration of 1.68 ng/L and were similar to levels found in the Rhine River and Mälaren Lake.

Detailed results of the occurrence of PFHxS in European surface waters are presented in Table D-2. Nine studies conducted in Europe evaluated sites with no known point fluorochemical source (Barreca et al., 2020; Munoz et al., 2018; Pan et al., 2018; Shafique et al., 2017; Eriksson et al., 2013; Wagner et al., 2013; Ahrens et al., 2009b; Ahrens et al., 2009a; Ericson et al., 2008b). Pan et al. (2018) performed a study that included surface water sampling

sites in the United Kingdom (Thames River), Germany and the Netherlands (Rhine River), and Sweden (Mälaren Lake). None of the sites sampled were proximate to known sources of PFAS, but for all three water bodies, detection frequency for PFHxS was 100%. The highest PFHxS levels were detected in the Thames River (maximum = 11.3 ng/L), which was about 3 to 4 times greater than the maximum levels found in the other water bodies. For the remaining nine studies, most reported PFHxS levels were relatively low (≤7.8 ng/L) or were not detected at all. Eight studies in Europe evaluated urban areas possibly affected by industrial activities (Lorenzo et al., 2015; Zhao et al., 2015; Boiteux et al., 2012; Eschauzier et al., 2012; Kovarova et al., 2012; Rostkowski et al., 2009) or wastewater effluent discharges (Lorenzo et al., 2015; Labadie and Chevreuil, 2011; Möller et al., 2010). PFHxS occurrence in these studies varied, with some studies reported 0% detections and some reporting detectable levels in all samples. The remaining studies conducted in Europe evaluated surface water samples from sites with known or suspected PFAS releases associated with fluorochemical manufacturing (Bach et al., 2017; Boiteux et al., 2017; Gebbink et al., 2017; Valsecchi et al., 2015) or AFFF use (Mussabek et al., 2019; Gobelius et al., 2018; Dauchy et al., 2017; Filipovic et al., 2015). Of the four studies potentially impacted by nearby fluorochemical manufacturing sites, two conducted in France found no PFHxS (Bach et al., 2017; Boiteux et al., 2017). PFHxS levels were not detected or relatively low at most sampling locations of the remaining two studies (<3 ng/L) except for River Brenta in Italy with an approximately 10-fold higher maximum level (35.6 ng/L). This site is also impacted by nearby textile and tannery manufacturers (Valsecchi et al., 2015). Consistent with U.S.-based studies, the highest PFHxS levels were found at the AFFF-impacted sites (up to 7,550 ng/L).

Table D-2. Studies Reporting PFHxS Occurrence in Surface Water

Study	Location	Site Details	Results
United States			
Galloway et al. (2020)	United States (Ohio; West Virginia)	Rivers and tributaries near a fluoropolymer facility sampled throughout three trips on June, July, and December 2016. In June 2016, samples were collected on a 188 km stretch of the Ohio River, from 130 km downstream to 58 km upstream of the facility, and tributaries that pass near known PFAS-containing landfills. In July 2016, samples were collected from lakes, rivers, and creeks to the north and northeast of the facility as far as 16 km downwind. The December 2016 trip expanded the collection radius to more than 48 km downwind to the north and northeast of the facility.	June 2016: n = 26, DF[a] = 96%* *PFHxS was detected but below the LOQ in 25 samples, and ND in 1 sample July 2016: n = 25, DF[a] = 12%*, range = ND–79.0 ng/L December 2016: n = 40; DF[a] = 48%*, range[a] = <LOQ–14.4 ng/L *PFHxS was detected above the LOQ in 3 samples, detected but below the LOQ in 16 samples, and ND in 21 samples (LOQ = 10 ng/L)
Newsted et al. (2017)	United States (Minnesota)	Ten sampling reaches (three samples each) spanned 3 miles of the Mississippi River within Pool 2, an area between Ford Dam (between Minneapolis and St. Paul) and Hastings Dam (near Hastings) and subject to ongoing PFAS reduction efforts by the Minnesota Pollution Control Agency for 10–15 years. Surface water samples were collected in August 2011.	Upstream of 3M Cottage Grove facility: n = 27, DF 0% Downstream of 3M Cottage Grove facility: n = 3, DF[a] 33%, range= <LOQ–60.5 ng/L (LOQ = 2.0 ng/L)
Newton et al. (2017)	United States (Decatur, Alabama)	Samples collected in October 2015 from nine sites along the Tennessee River. Three sites were downstream of facilities that manufacture or use fluorinated materials and six sites were upstream. Among the upstream sites, three were also upstream of a WWTP.	Upstream: n = 6, DF 0% Downstream: n = 3, DF[a] 33%, range = <LOQ–39 ng/L (LOQ = 10 ng/L)
Genualdi et al. (2017)	United States (Massachusetts)	Cranberry bog with surface water contaminated with PFAS—likely due to proximity to a military base with a history of AFFF usage. The bog was located approximately 10 miles from the military base. Bog water samples were collected in November 2016.	n = 6, DF 100%, mean[a] (range) = 10.98 (8.1–14) ng/L (MDL= 15 ng/L) *2 sample extracts were not available for analysis due to sample loss from evaporation *Table 3 reports the MDL as 15 ng/uL but four of the six detected samples have values less than 15 ng/L
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between	Overall: n = 36, DF 88.00%, median (range) = 710 (ND–815,000) ng/L

Study	Location	Site Details	Results
		1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were not specifically controlled for in the site selection process; active remedies had not been applied at any of the sites selected. Approximately ten samples were collected between March and September 2014 at each site; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases. Surface water sample locations included engineered storm water channels, engineered AFFF ponds, and natural streams.	*Breakdown by site:* Emergency Response (low-volume release): n = 2, DF 50.0%, mean (range) = 7.1 (7.1–7.1) ng/L Hangars and Buildings (medium-volume release): n = 32, DF 74.3%, 196,800 (360–2,700,000) ng/L Testing and Maintenance (high-volume release): n = 2, DF 100.0%, mean (range) = 5,600 (4,400–6,700) ng/L (Median RL = 7 ng/L) *Median calculated using quantified detections *Non-detects were substituted with ½ the reporting limit
Post et al. (2013)	United States (New Jersey)	Raw water collected from 12 public drinking water system intakes between August 2009 and February 2010 from 6 rivers and 6 reservoirs. Sites were chosen to represent NJ geographically and included two reservoir sites near a civil-military airport with possible AFFF use.	Overall n = 12, DF 17%, range = ND–46 ng/L Rivers: n = 6, DF 0% Reservoirs: n = 6, DF 33%, range = <5–46 ng/L (RL = 5 ng/L)
Nakayama et al. (2010)	United States (Illinois; Iowa; Minnesota; Missouri; Wisconsin)	Eighty-eight sampling sites collected between March and August 2008 from tributaries and streams in the Upper Mississippi River Basin and a portion of the Missouri River Basin. Samples were collected by the Minnesota Pollution Agency, Wisconsin Department of Natural Resources, Illinois Environmental Protection Agency, and U.S. EPA Region 7 Water Quality Monitoring Team. Each agency selected sampling sites with the intention of providing preliminary PFC data to the individual regions. Sampling sites included locations adjacent to chemical manufacturers, downstream of WWTPs receiving waste from those types of manufacturers, and near an airport with historic use of firefighting foams.	n = 173, DF 89%, median (range) = 0.71 (ND–169) ng/L (LOD = 0.02 ng/L) *ND data points were substituted with LOD/sqrt(2) = 0.014 ng/L *The maximum concentration of 169 ng/L was collected from a waterway that potentially receives a run-off from a historical fire training site at the Duluth International Airport. Excluding this location, the maximum PFHxS level in surface water is 14.5 ng/L

Study	Location	Site Details	Results
Nakayama et al. (2007)	United States (North Carolina)	Eighty sampling sites in river basin during spring 2006. The sites were selected to reflect water quality throughout the basin. Possible sources of PFCs include use of firefighting foam from Fort Bragg and Pope Air Force, metal-plating facilities, textile, and paper production, and other industries.	n = 100, DF 98.7%, mean, GM, median (range) = 7.29, 5.73, 5.66 (<LOQ–35.1) ng/L (LOQ = 1 ng/L, LOD = 0.05 ng/L) *Values below the LOQ were excluded from the calculation of the mean and GM * Table 2 reports a maximum concentration of 35.1 ng/L but the text reports the highest PFHxS concentration was 26.4 ng/L at Little Rock
Lasier et al. (2011)	United States (Georgia)	Upstream (sites 1 and 2) and downstream (sites 3–8) of a land application site were sampled along the Conasauga, Oostanaula, and Coosa Rivers during summer 2008, where effluents from carpet manufacturers (suspected of producing wastewaters containing perfluorinated chemicals) are processed at a WWTP and the treated WWTP effluent is sprayed onto the site. Additionally, site 2 was downstream a local airport and site 4 was downstream of a manufacturing facility for latex and polyurethane backing material.	Upstream: Sites 1 and 2: DF 0% Downstream: Site 3: DF NR, mean = 30 ng/L Site 4: DF NR, mean = 31 ng/L Site 5: DF NR, mean = 17 ng/L Site 6: DF NR, mean = 13 ng/L Site 7: DF 0% Site 8: DF 0% (MDL = 0.014 ng/g; LOQ = 0.031 ng/g) *Half of the MDL was used when measured concentrations were below the MDL *Concentrations measured in triplicate samples
Lindstrom et al. (2011)	United States (Decatur, Alabama)	Samples collected in February 2009 from ponds and streams located on farms with historical land application of PFC-contaminated biosolids to local agricultural fields between 1995 and 2008. Biosolids obtained from local municipal WWTP where sources discharging to the WWTP included facilities involved in the production and use of fluoropolymers, fluorocarbon fibers, polymers, polymer films, and resins.	n = 32, DF (frequency of quantification)[a] 22%, range = <LOQ–218 ng/L (LOQ = 10 ng/L)
Boone et al. (2019)	United States (unspecified)	Twenty-two surface waters (16 rivers and 6 lakes/reservoirs) used as source waters for 22 DWTPs, collected in 2010−2012; some locations with known or suspected sources of wastewater in the source water, but study did not differentiate which locations had known or suspected sources.	n = 22, DF[a] 95%, range = ND–19.7 ng/L (LCMRL = 0.034 ng/L)

Study	Location	Site Details	Results
Subedi et al. (2015)	United States (New York)	Lake water along the shoreline of residences that use an enhanced treatment unit for onsite wastewater treatment; samples were collected ~40 ft from the lakeshore about 2 ft below surface. Sampling occurred August–September 2012 (three sampling events) and May–September 2013 (four sampling events). Wastewater effluents identified as source of contamination.	n = 28, DF[a] 79%, mean, median, (maximum) = 0.56, 0.28 (2.57) ng/L (LOQ = 0.2 ng/L) *Data points <LOQ were substituted with ½ LOQ and NDs were substituted with zero
Appleman et al. (2014)	United States (Wisconsin; Oklahoma; Alaska; Alabama; Colorado; Ohio; Nevada; New Jersey)	Surface water source water for 11 DWTPs, sampled from August 2011 to May 2012; majority of the utilities selected because they were either known from previous monitoring or expected based on their source waters to contain detectable PFAS (i.e., impacted by upstream wastewater effluent discharge). Each site was sampled between one and four times.	n = 25, DF[a] 72%, range = <0.25–13 ng/L (Method RL = 0.25 ng/L)
Procopio et al. (2017)	United States (New Jersey)	Surface water from the Metedeconk River, where there was suspected illicit discharge to soil and groundwater from a manufacturer of industrial fabrics, composites, and elastomers that use or produce products containing PFAAs. Majority of samples collected from a 4.0 km (2.5 mile) reach of the South Branch Metedeconk River, although samples from the North Branch were also collected in sampling event 1. Samples were collected in eight sampling events in September and December 2011; February, July, September, and December 2012; June and August 2013; and June and July 2014.	n = 56, DF 0% (Minimum RL = 5 ng/L)
Zhang et al. (2016)	United States (Rhode Island; New York)	River and creek samples from 22 sites in Rhode Island collected in June 2014 and from 6 sites in the NY Metropolitan Area collected in October 2014. Both urban and rural locations were sampled.	Overall n = 28, DF[a] 89%, range = <LOD–35.022 ng/L Rhode Island: n = 22, DF[a] 86%, range = ND–35.022 ng/L Urban: n = 10, DF[a] 100%, mean[a] (range) = 6.471 (0.864–35.022) ng/L Rural: n = 12, DF[a] 75%, range = <LOD–0.645 ng/L New York Metropolitan Area (all urban sites): n = 6, DF[a] 100%, mean[a] (range) = 3.019 (0.224–8.526) ng/L (LOD = 0.06 ng/L)

Study	Location	Site Details	Results
Sinclair et al. (2006)	United States (New York)	Nine major water bodies—Lake Ontario, Niagara River, Lake Erie, Finger Lakes, Lake Onondaga, Lake Oneida, Erie Canal, Hudson River, and Lake Champlain—were sampled in July 2004 to represent the major water bodies of New York State. Lake Onondaga is a Superfund site, is influenced by industrial sources along the lake, and receives WWTP effluent.	Lake Ontario: n = 13, DF 100%, median (range) = 1.4 (1.2–2.8) ng/L Niagara River: n = 3, DF 100%, median (range) = 1.2 (1.2–1.4) ng/L Lake Erie: n = 3, DF 100%, median (range) = 1.2 (1.2–1.6) ng/L Finger Lakes: n = 13, DF 100%, median (range) = 0.9 (0.7–1.3) ng/L Lake Onondaga: n = 3, DF 100%, median (range) = 7.4 (4.2–8.5) ng/L Lake Oneida: n = 1, point = 0.9 ng/L Erie Canal: n = 3, DF 100%, median (range) = 2.6 (2.5–5.6) ng/L Hudson River: n = 8, DF 100%, median (range) = 0.9 (0.7–1.6) ng/L Lake Champlain: n = 4, DF[a] 75%, median (range) = 1.3 (ND–1.6) ng/L (DL = 0.5 ng/L)
Bradley et al. (2020)	United States (Chicago, Illinois; East Chicago, Indiana)	Lake Michigan untreated intake water at four intake sites. Samples collected in July and November 2017 at the intakes of the Chicago North and Chicago South WTPs and in July and November 2017 at the intakes of the two East Chicago DwTPs.	n = 7, DF[a] 100%, mean[a] (range) = 1.0 (0.8–1.3) ng/L (LOQ = 0.120–0.580 ng/L) *Quantitative (≥LOQ) and semiquantitative (between LOQ and MDL) results treated as detections
Boone et al. (2014)	United States (New Orleans, Louisiana)	Surface samples from the Mississippi River collected in June 2009 when the river was at a high flow level (8.32 ft) and in September 2010 when the river was at a low flow level (2.95 ft).	Low flow (2.95 ft): mean based on a primary and duplicate sample = 1.315 ng/L High flow (8.32 ft): mean based on four replicates = 1.07 ng/L (DL = 0.016 ng/L; LCMRL = 0.034 ng/L)
Quiñones and Snyder (2009)	United States (Arizona; Nevada)	Samples collected in 2008 from three sites in Boulder Basin, one site in Hoover Dam, and two sites from the lower Colorado River. PFC occurrence had not been previously determined or reported for these sites.	n = 40, DF* 0% (Method RL = 1.0 ng/L) *Mean values at all sites were <Method RL; Figure 3 in the paper shows all quantifiable values were <Method RL
Kim and Kannan (2007)	United States (Albany, New York)	Samples collected from two urban lakes—Washington Park Lake and Rensselaer Lake—during five sampling	Total: n = 11, DF 81.8%, mean, median (range) = 1.58, 0.53 (<LOQ–4.05) ng/L

Study	Location	Site Details	Results
		trips from February–November 2006. Both lakes are located in downtown Albany and receive surface runoff from nearby roadways and residential areas during stormwater runoff.	Washington Park Lake: n = 6, DF NR, mean, median (range) = 0.33, 0.33 (<LOQ–0.53) ng/L Rensselaer Lake: n = 5, DF NR, mean, median (range) = 3.09, 3.25 (2.35–4.05) ng/L (LOQ = 0.25 ng/L) *Non-detects were set to zero; values below the LOQ were set to ½ LOQ
Canada			
Lescord et al. (2015)	Canada (Cornwallis Island, Nunavut)	Six lakes (Meretta, Resolute, Char, Small, North, and 9 Mile) located on Cornwallis Island and near the Inuit community of Resolute Bay were sampled weekly in July to August in 2010 and biweekly in July to August 2011. Two lakes (Meretta and Resolute) are approximately 0.5 km downstream from a local airport, where wastewater from the airport and military base was discharged with little treatment into the Meretta catchment from 1949 to 1998.	Meretta: n = 5, DF NR, mean = 30 ng/L Resolute: n = 5, DF NR, mean = 19.7 ng/L Char: n = 5, DF NR, mean = 0.12 ng/L Small: n = 5, DF NR, mean = 0.11 ng/L North: n = 5, DF NR, mean = 0.01 ng/L 9 Mile: n = 5, DF NR, mean = 0.02 ng/L (MDL = 0.0017 ng/L)
Yeung et al. (2017)	Canada (Ontario)	River water samples were collected from Mimico Creek and Rouge River in November 2014 and analyzed using two methods. Reference method used ultra performance liquid chromatograph and a newly developed method used ultra performance convergence chromatograph for separation.	Results presented as reference method, new method: Mimico: n = 1, point = 24, 20 ng/L Rouge: n = 1, point = <2, <5 ng/L (MLOQ for reference method not reported; MLOQ for new method reported in Table S3 as 200 ng/L but based on Table S7, this should likely be 2 ng/L) *Results in Table S7 are presented in ng/L but based on a comparison to Figure 4, Table S7 should be in ng/mL
Veillette et al. (2012)	Canada (Ellesmere Island, Nunavut)	Lake catchment area located on the northwest coast of the island. Surface water was collected from the center of the lake, the littoral zone (30 m from the delta), the delta, and lake inflow and outflow in July 2007, May 2008, and August 2008. Samples were collected at depths of 2 m (underneath the ice cover), 10 m (the bottom of the mixed layer), and 32 m (in the monimolimnion).	n = 11, DF[a] 100%, mean (range) = 0.009 (0.003–0.024) ng/L (MDL = 0.0007 ng/L)

Study	Location	Site Details	Results
Europe			
Bach et al. (2017)	France (southern)	Grab water samples were collected from six locations along the shore of a river in April, July, October, and December 2013. The river selected for the study receives effluent at three points along the river from an industrial site where two facilities produce fluoropolymers. The first facility has been active since the 1960s, with production including PTFE synthesis from the beginning of the 1960s to 1985 with PFOA as a processing aid; more recently, PVDF has been synthesized since the early 1970s with fluorotelomer sulfonic acid (6:2 FTSA) or PFNA as a processing aid. The second facility, established in 2002, produced fluoropolymers with PFOA as a processing aid until 2008 when it was replaced with PFHxA. Samples were collected starting ~1.3 km upstream from the industrial site and covered ~15 km of the river. Samples (point #1 to #6) were collected from upstream to downstream.	Upstream: Sampling point #1: n = 1, DF 0% for April, July, October, and December 2013 Downstream: Sampling points #2–6: n = 1, DF 0% for April, July, October, and December 2013 (LOQ = 4 ng/L) *DF represents frequency of quantification
Boiteux et al. (2017)	France (northern)	Grab water samples were collected from seven locations along a river in May, July, October, and December 2013. The river selected for the study receives wastewater from an industrial WWTP that treats raw sewage coming from a facility that manufactures fluorotelomer-based products and side-chain-fluorinated polymers used in firefighting foams and stain repellents. Samples were collected starting ~1.2 km upstream of the WWTP discharge and encompassed ~65 km of the river. Samples were collected from upstream to downstream.	Upstream: Sampling point #1: n = 1, DF 0% for May, July, October, and December 2013 Downstream: Sampling points #3, 4, 5, 7, 9, 11: n = 1, DF 0% for May, July, October, and December 2013 (LOQ = 4 ng/L) *DF represents frequency of quantification
Gebbink et al. (2017)	The Netherlands (Dordecht)	River water samples collected in October 2016 at sites downstream (R1−R13) and upstream (R14−R16) of the Dordrecht fluorochemical production plant. Samples (R17−R18) were also collected from different waterbodies at control sites.	Control sites: n = 2, DF[a] (frequency of quantification) 100%, mean[a] (range) = 1.85 (1.7–2.0) ng/L Upstream: n = 3, DF[a] (frequency of quantification) 100%, mean[a] (range) = 2.1 (2.0–2.2) ng/L Downstream: n = 13, DF[a] (frequency of quantification) 100%, mean[a] (range) = 2.0 (1.5–2.2) ng/L (MQL = 0.02 ng/L)

Study	Location	Site Details	Results
Valsecchi et al. (2015)	Italy (River Basins Po, Brenta, Adige, Tevere, and Arno)	Five river basins were sampled between 2008 and 2013. Two river basins (Po and Brenta) receive discharges from two chemical plants that produce fluorinated polymers and intermediates; two river basins (Tevere and Adige) are not impacted by relevant industrial activities; and one river basin (Arno) has textile and tannery districts located along parts of the river. In total, 20 rivers were sampled at their basin closure stations. Rivers Arno, Tevere, and Po were also sampled along the course of the river.	Po: n = 105, DF 0% Brenta: n = 5, DF[a] 20%, range = <LOD–35.6 ng/L Adige: n = 5, DF 0% Tevere: n = 7, DF 0% Arno: n = 19, DF 0% (LOD = 5 ng/L)
Skaar et al. (2019)	Norway (Ny-Ålesund; Lake Linnévatnet area)	Freshwater samples were collected from Ny-Ålesund (research facility) in June 2016 and from the Lake Linnévatnet area (background site) in March 2014 and from April to June 2015. Surface water in Ny-Ålesund was contaminated by a local firefighting training site. Lake Linnévatnet receives water from meltwater of the adjacent glaciers and has few potential pollution sources.	Ny-Ålesund: Background: n = 7, DF[a] 43%, range = <0.005–2.65 ng/L Contaminated: n = 3, DF[a] 100%, mean[a] (range) = 150.83 (30.36–307.51) ng/L Lake Linnévatnet: Background: n = 20, DF[a] 55%, range = <0.005–0.023 ng/L (LOD = 0.005 ng/L; LOQ = 0.006 ng/L) *Values reported for sum of branched and linear PFHxS isomers
Mussabek et al. (2019)	Sweden (Luleå)	Samples from a man-made lake and pond approximately 500 m southwest from a firefighting training facility at the Norrbotten Air Force Wing collected in October 2015. The training facility has been active since 1941 and has used PFAS-containing AFFFs in the last decades. The lake and pond lie above a groundwater reservoir with high permeable soil and were selected because they are isolated water bodies receiving PFAS contamination and can potentially impact groundwater.	Lake: n = 2, DF NR, mean = 570 ng/L Pond: n = 2, DF NR, mean = 500 ng/L (LOD = 0.5 ng/L)
Gobelius et al. (2018)	Sweden (national)	Sampling conducted between May and August 2015, with the majority in July and a few samples in September and November 2015. Samples were collected in 21 regional counties by the County Administration Boards. Sampling locations selected based on potential vicinity of PFAS hot spots (i.e., fire training sites, unspecific industry, sewage treatment plant effluent, landfill/waste disposal, skiing, and urban areas) and/or importance as a drinking water	n = 281, DF[a] 61%, range = <0.15–7,550 ng/L (MDL = 0.15 ng/L) *Two types of water (i.e., surface water and recipient water [surface water]) included

Study	Location	Site Details	Results
		source. Sample numbers varied for each county and sampling sites were spread unevenly across Sweden. Surface water samples collected approximately 10 cm below the water surface.	
Dauchy et al. (2017)	France (unspecified)	Samples collected in the vicinity of three sites (B, C, D) where fluorosurfactant-based foams are or were being heavily used. Site B is an international civilian airport built in 1974. The exact location of the training area, frequency of training sessions, and history of firefighting training activities are unknown. In November 2014, surface water samples were collected in the only river running alongside the airport. Downstream from the airport, this river joins two other rivers, which were also sampled. Site C is a military airport, with the exact location of the training area, frequency of the training sessions, and history of the firefighting training activities unknown. In April 2014, surface water samples were collected in several rivers surrounding the military base. Site D is a training center for firefighters. From 1969 to 1984, the site was an oil refinery. Starting in 1987, the site became a training area for firefighters, with exercises carried out directly on the soil. From the 1990s, some exercise areas were covered with concrete. In November 2014, two surface water samples were collected from the river receiving effluent from a WWTP at the site, one upstream and one downstream of the discharge pipe.	Site B: n = 5, DF[a] 100%, mean[a] (range) = 11.2 (5–18) ng/L Site C: n = 9, DF 0% Site D: n = 2, DF[a] 50%, range = <4–251 ng/L (LOQ = 4 ng/L)
Filipovic et al. (2015)	Sweden (Stockholm)	Fourteen lakes and ponds surrounding the closed air force base F18 in Tullinge Riksten, 19 km south of Stockholm city center, where AFFFs were used. Samples collected in two sampling campaigns in December 2011 and April 2012. The air force base was formally demobilized in 1986 but continued to be used as an air force school for combat command and air surveillance until 1994. Of note, the air force base encountered numerous accidents and incidents during the transfer from propeller era to the jet engine era, including planes crashing upon takeoff and landing, fire incidents, accidental dispersion of jet engine	n = 14, DF 64%, range = <0.5–25.1 ng/L (MDL = <0.5 ng/L)

Study	Location	Site Details	Results
		starting fuel. The area was sold to a land developer in 1996 and is in the process of being transformed into a municipal Area.	
Ciofi et al. (2018)	Italy (Tuscany)	Surface water samples were collected at 13 locations. Sampling year was not reported. SW-1: Arno river before the "Canale Maestro della Chiana" (Arezzo), receiving agricultural runoff and untreated urban wastewater SW-2: Arno river after the "Canale Maestro della Chiana" (Arezzo) SW-3: Arno river before entering in Florence SW-4: Arno river after the discharge of the Florence WWTP SW-5: Arno river after the confluence of the Bisenzio river SW-6: Arno river after the city of Empoli (Florence) SW-7: Arno river after receiving the WWTP effluent from the leather industrial district of Santa Croce (Pisa) SW-8: Arno river in the proximity of the mouth (Pisa) SW-9: Bisenzio river before the confluence with Arno river (Florence) SW-10: Serchio river in the proximity of the mouth (Lucca) SW-11: East area of the coastal lake "Massaciuccoli" (Lucca) SW-12: West area of the coastal lake "Massaciuccoli" (Lucca) SW-13: Central area of the artificial lake "Bilancino" (Florence)	SW-1: n = 1, point = <0.010 ng/L SW-2: n = 1, point = <0.010 ng/L SW-3: n = 1, point = <0.011 ng/L SW-4: n = 1, point = <0.010 ng/L SW-5: n = 1, point = <0.010 ng/L SW-6: n = 1, point = <0.026 ng/L SW-7: n = 1, point = <0.010 ng/L SW-8: n = 1, point = <0.010 ng/L SW-9: n = 1, point = <0.010 ng/L SW-10: n = 1, point = <0.010 ng/L SW-11: n = 1, point = <0.010 ng/L SW-12: n = 1, point = <0.026 ng/L SW-13: n = 1, point = <0.013 ng/L (MDL = 0.010–0.013 ng/L; MQL = 0.026 ng/L) *MDL/MQL varied by sample. MDL provided for 11 of 13 samples; MQL provided for 2 of 13 samples
Munoz et al. (2018)	France (Marnay-sur-Sein; Bougival; Triel-sur-Seine)	Surface water along the Seine River was collected during four sampling campaigns between September 2011 and December 2012, each conducted in a different season. For each campaign, two to four samples were collected over a one-month period. Three sampling sites were investigated: Marnay-sur-Sein, located 200 km upstream from Paris, was selected as a reference site, non-affected by the Greater Paris region; Bougival, situated 40 km downstream from Paris, was chosen to investigate the impact of Greater Paris on PFAS levels; Triel-sur-	n = 36, DF 100%, range = 0.28–7.8 ng/L (LOD = 0.02–0.3 ng/L for all PFAS) *PFHxS concentrations were not reported in text or table by site but relative abundance by site is available in Figure 1

Study	Location	Site Details	Results
		Seine, another 40 km further downstream, was selected to assess the global influence of the Paris urban area, including other inputs such as WWTps.	
Lorenzo et al. (2015)	Spain (Guadalquivir River Basin; Ebro River Basin)	Surface water was collected from the Guadalquivir River and its main tributaries and from the Ebro River and its main tributaries in October 2010. Guadalquivir sampling locations included downstream of WWTPs, near industrial areas, near a military camp, or through major cities; Ebro sampling locations included nearby ski resorts and downstream of WWTP and industrial areas.	Guadalquivir: n = 24, DF 13%, mean (range) = 4.1 (1.5–88.5) ng/L Ebro: n = 24, DF 13%, mean (range) = 0.5 (1.1–5.8) ng/L *Minimum reported is the lowest amount quantified *Mean was calculated with not detected concentrations as zeros (MQL = 0.004 ng/L)
Zhao et al. (2015)	Germany (Elbe River)	Four sampling campaigns conducted in February, April, August, and October 2011 to represent the four seasons. Freshwater samples (sites E619 to E689 with salinity <1 PSU) were collected at nine locations in the river Elbe. Some sampling sites were near Hamburg city and experienced occasional discharge of wastewater from industrial plants.	February: n = 8, DF[a] 87.5%, range = <0.08–0.96 ng/L April: n = 9, DF[a] 100%, mean[a] (range) = 0.46 (0.08–0.77) ng/L August: n = 9, DF[a] 100%, mean[a] (range) = 0.60 (0.42–1.0) ng/L October: n = 8, DF[a] 100%, mean[a] (range) = 0.38 (0.18–0.52) ng/L (MDL = 0.03 ng/L)
Boiteux et al. (2012)	France (national)	Raw water from rivers used as source water for DWTPs. Sites distributed across 100 French department to represent ~20% of the national water supply flow; samples collected during two sampling campaigns in July–September 2009 (first campaign) and June 2010 (second campaign – focused on sites from first sampling campaign that had PFC levels >LOQ). Some sites possibly affected by commercial/industrial releases.	Overall: n = 135, DF (frequency of quantification) 7%, maximum = 8 ng/L *1st Sampling Campaign* n = 99, DF 48%, mean, median (maximum) = 1, <1 (8) ng/L *2nd Sampling Campaign* n = 36, results not reported (LOD = 1.3 ng/L, LOQ = 4 ng/L)
Eschauzier et al. (2012)	The Netherlands (Amsterdam)	Intake water from the Lek canal (n = 2) was collected in January and September 2010 to determine the behavior of PFAAs during the drinking water treatment processes. The Lek canal, a tributary of the river Rhine, is the source of drinking water for the city of Amsterdam and is downstream of an industrial point source in the German part of the Lower Rhine.	n = 2, DF[a] 100%, mean (range) = 2.0 (1.9–2.2) ng/L (LOQ = 0.55 ng/L)

Study	Location	Site Details	Results
Kovarova et al. (2012)	Czech Republic (Brno)	Seven locations in the Svitava and Svratka Rivers upstream and downstream of Brno, a city with highly developed chemical, engineering, textile, and food-processing industries. A sampler was installed at each site for 30 days twice a year (May and September 2008). Due to technical problems, samples were produced from only four of seven sites in May and from five of seven sites in September.	n = 9, DF 0% (LOD not reported)
Llorca et al. (2012)	Germany (Hesse), Spain (national)	Forty-eight surface river waters were sampled in 2010–2011 (24 from Spain and 24 from Germany). Samples from Germany were collected from agriculturally or industrially influenced streams. Samples from Spain were collected from the Xúquer River Basin, Llobregat River Basin, and Ebro River Basin.	Germany: n = 24, DF 21%, mean, median (range) = 1.9, 0.5 (0.06–5.6) ng/L Spain: n = 24, DF 21%, mean, median (range) = 16, 5.8 (0.06–37) ng/L (MLOQ = 0.06 ng/L) *MLOQ reported above is from Table 3; Table 2 reports MLOD = 0.27 ng/L and MLOQ = 0.90 ng/L
Labadie and Chevreuil (2011)	France (Paris)	Samples collected weekly in January–May 2010 in an urban stretch of the River Seine at the Austerlitz Quay, downtown Paris during a flood cycle. The sampling station is under the influence of two major WWTPs and two major combined sewer overflow outfalls.	n = 16, DF 100%, mean, median (range) = 7.1, 6.8 (3.9–12.0) ng/L (LOQ = 0.15 ng/L)
Möller et al. (2010)	Germany (Rhine River watershed)	Raw freshwater samples collected in September–October 2008 along the River Rhine (stations 1–36) and major tributaries of the River Rhine (e.g., Rivers Neckar, Main, Rhur, stations 37–48). Along the River Rhine, samples were taken upstream and downstream of Leverkusen, where effluent of a WWTP treating industrial wastewater was discharged. All samples taken at a water depth ≤1 m.	Rhine upstream Leverkusen: n = 27, DF NR, mean (range) = 3.04 (<0.51–14.5) ng/L Rhine downstream Leverkusen: n = 9, DF 100%, mean (range) = 1.93 (1.66–2.44) ng/L River Ruhr: n = 3, DF NR, mean (range) = 0.18 (<0.51–0.53) River Moehne: n = 1, point = 1.03 ng/L Other tributaries: n = 8, DF NR, mean (range) = 1.41 (<0.51–2.93) ng/L (MDL = 0.51 ng/L)
Rostkowski et al. (2009)	Poland (national)	Inland surface water samples were collected at 12 locations in the southern part of Poland and 14 locations in the northern part of Poland in October and December 2004. Inland surface waters included rivers, lakes, and streams. The northern locations flowed through forested, agricultural, and rural areas; these	North: n = 14, DF[a] 79%, range = <0.02–2.8 ng/L South: n = 11, DF[a] 91%, range = <0.67–113 ng/L (LOQ = 0.1 ng/L)

Study	Location	Site Details	Results
		areas are considered unpolluted with industrial chemicals. Some southern locations were near chemical industrial activities.	
Barreca et al. (2020)	Italy (Lombardia Region)	Fifty-two surface water sampling stations (rivers and streams) throughout the region. Samples collected in 2018.	n = 286, DF[a] 4% (LOQ = 1 ng/L)
Shafique et al. (2017)	Germany (River Elster; River Pleiße; River Saale; and River Elbe)	Surface water samples were collected from the River Elster, River Pleiße, River Saale, and River Elbe at the start of 2015.	Elster: n = 4, DF NR, mean = 0.42 ng/L Pleiße: n = 2, DF NR, mean = 0.37 ng/L Saale (Site A): n = 10, DF NR, mean = 0.13 ng/L Saale (Site B): n = 10, DF NR, mean = 0.60 ng/L Saale (Site C): n = 10, DF NR, mean = 0.07 ng/L Elbe: n = 2, DF NR, mean = 0.61 ng/L (MDL = 0.11 ng/L) *Values extracted from SI, which provides a more detailed breakdown of sites compared to that reported in the main text (where Elster and Pleiße sites were combined and Saale sites were combined)
Eriksson et al. (2013)	Denmark (Faroe Islands)	Grab samples collected in April–May 2012 from Lakes Leitisvatn, Havnardal , Kornvatn, and Á Mýranar.	Leitisvatn: n = 1, point = <0.058 ng/L Havnardal Lake: n = 1, point = <0.024 ng/L Kornvatn Lake: n = 1, point = <0.027 ng/L Á Mýranar: n = 1, point = <0.024 ng/L (LOD = 0.035 ng/L)
Wagner et al. (2013)	Germany (Rhine River)	Surface water samples were collected from the Rhine River. Sampling year not provided. Samples used to test out a new analytical protocol to determine trace levels of adsorbable organic fluorine.	n = 2, DF[a] 100%, mean (range) = 2.4 (1.6–3.2) ng/L (LOD = 50 ng/L F^-; LOQ = 150 ng/L F^-) *PFHxS concentrations were calculated using the fluorine concentrations reported in Table 4
Ahrens et al. (2009a)	Germany (Hamburg; Laurenburg)	Nine samples collected from the river Elbe in Hamburg city (sites 16-18) and from Laurenburg to Hamburg (sites 19-24) in August 2006. Samples were collected at a water depth of 1 m. Dissolved and	Hamburg: Dissolved: n = 3, DF[a] 100%, mean (range) = 0.60 (0.56–0.67) ng/L Particulate: n = 3, DF 0%

Study	Location	Site Details	Results
		particulate phases were analyzed for each of the water samples.	Laurenburg to Hamburg: Dissolved: n = 6, DF[a] 100%, mean (range) = 0.36 (0.24–0.49) ng/L Particulate: n = 6, DF[a] 33%, mean (range) = 0.029 (ND–0.098) ng/L (MDL = 0.140 ng/L for dissolved phase; 0.045 ng/L for particulate phase)
Ahrens et al. (2009b)	Germany (Elbe River)	Samples collected at 53 to 122 km (sites 1 to 9) upstream of estuary mouth of Elbe River in June 2007. *Only locations with conductivity <1.5 mS/cm were assumed to be freshwater and extracted	Site 1 (122 km): n = NR, DF NR, mean = 0.85 ng/L Site 2 (118 km): n = NR, DF NR, mean = 0.9 ng/L Site 3 (115 km): n = NR, DF NR, mean = 1.0 ng/L Site 4 (103 km): n = NR, DF NR, mean = 1.2 ng/L Site 5 (90 km): n = NR, DF NR, mean = 0.8 ng/L Site 6 (80 km): n = NR, DF NR, mean = 1.3 ng/L Site 7 (74 km): n = NR, DF NR, mean = 0.9 ng/L Site 8 (64 km): n = NR, DF NR, mean = 1.1 ng/L Site 9 (53 km): n = NR, DF NR, mean = 0.9 ng/L (MDL = 0.17 ng/L; MQL = 0.57 ng/g)
Ericson et al. (2008b)	Spain (Tarragona Province)	River water samples collected from the Ebro (at two points, Garcia and Mora), Francolí, and Cortiella Rivers in February 2007.	Ebro site 1: n = 1, point = 0.40 ng/L Ebro site 2: n = 1, point = 0.43 ng/L Francolí: n = 1, point = 0.78 ng/L Cortiella: n = 1, point = <0.18 ng/L (LOD = 0.18 ng/L)
Multiple Continents			
Pan et al. (2018)	United States (Delaware River)	Samples were collected from the Delaware River between September and December 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Trenton,	n = 12, DF[a] 100%, mean, median (range) = 1.68, 1.72 (0.65–2.63) ng/L (MDL = 0.05 ng/L)

Study	Location	Site Details	Results
		Bristol, Philadelphia, Chester, Delaware, Smyrna, and Frederica.	
	United Kingdom (Thames River)	Samples were collected from the Thames River in October 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Oxford and London.	n = 6, DF[a] 100%, mean, median (range) = 7.14, 6.42 (4.96–11.3) ng/L (MDL = 0.05 ng/L)
	Germany and The Netherlands (Rhine River)	Samples were collected from the Rhine River in December 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities in Germany included Offenbach, Frankfurt, Goarshausen, Rheinbrohl, Bonn, Cologne, Leverkusen, Dormagen, Düsseldorf, Duisburg, Wesel, and Emmerich. Cities in The Netherlands included Arnhem, Lienden, Duurstede, Nijmegen, Wamel, and Zaltbommel.	n = 20, DF[a] 100%, mean, median (range) = 1.98, 2.03 (0.12–3.90) ng/L (MDL = 0.05 ng/L)
	Sweden (Mälaren Lake)	Samples were collected from Mälaren Lake in September 2016. Sampling sites were not proximate to known point sources of any fluorochemical facilities. Cities included Örebro and Stockholm.	n = 10, DF[a] 100%, mean, median (range) = 1.30, 0.97 (0.56–2.79) ng/L (MDL = 0.05 ng/L)

D.2. RSC for PFHxS, Literature Search and Screening Methodology

The EPA applies an RSC to the RfD when calculating an MCLG based on noncancer effects or for carcinogens that are known to act through a nonlinear mode of action to account for the fraction of an individual's total exposure allocated to drinking water (USEPA, 2000b). The EPA emphasizes that the purpose of the RSC is to ensure that the level of a chemical allowed by a criterion (e.g., the MCLG for drinking water) or multiple criteria, when combined with other identified sources of exposure (e.g., diet, ambient and indoor air) common to the population of concern, will not result in exposures that exceed the RfD. In other words, the RSC is the portion of total daily exposure equal to the RfD that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee tea or soup, as well as from transfer to dietary items prepared with drinking water) relative to other exposure sources; the remainder of the exposure equal to the RfD is allocated to other potential exposure sources. For example, if for a particular chemical, drinking water were to represent half of total exposure and diet were to represent the other half, then the drinking water contribution (or RSC) would be 50%. The EPA considers any potentially significant exposure source when deriving the RSC.

The RSC is derived by applying the Exposure Decision Tree approach published in the EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000b). The Exposure Decision Tree approach allows flexibility in the RfD apportionment among sources of exposure and considers several characteristics of the contaminant of interest, including the adequacy of available exposure data, levels of the contaminant in relevant sources or media of exposure, and regulatory agendas (i.e., whether there are multiple health-based criteria or regulatory standards for the contaminant). The RSC is developed to reflect the exposure to the U.S. general population or a sensitive population within the U.S. general population and may be derived qualitatively or quantitatively, depending on the available data.

A quantitative RSC determination first requires "data for the chemical in question… representative of each source/medium of exposure and… relevant to the identified population(s)" (USEPA, 2000b). The term "data" in this context is defined as ambient sampling measurements in the media of exposure, not internal human biomonitoring metrics. More specifically, the data must adequately characterize exposure distributions including the central tendency and high-end exposure levels for each source and 95% confidence intervals for these terms (USEPA, 2000b). Frequently, an adequate level of detail is not available to support a quantitative RSC derivation. When adequate quantitative data are not available, the agency relies on the qualitative alternatives of the Exposure Decision Tree approach. A qualitatively-derived RSC is an estimate that incorporates data and policy considerations and thus, is sometimes referred to as a "default" RSC (USEPA, 2000b). Both the quantitative and qualitative approaches recommend a "ceiling" RSC of 80% and a "floor" RSC of 20% to account for uncertainties including unknown sources of exposure, changes to exposure characteristics over time, and data inadequacies (USEPA, 2000b).

In cases in which there is a lack of sufficient data describing environmental monitoring results and/or exposure intake, the Exposure Decision Tree approach results in a recommended RSC of

20%. In the case of MCLG development, this means that 20% of the exposure equal to the RfD is allocated to drinking water and the remaining 80% is reserved for other potential sources, such as diet, air, consumer products, etc. This 20% RSC value can be replaced if sufficient data are available to develop a scientifically defensible alternative value. If scientific data demonstrating that sources and routes of exposure other than drinking water are not anticipated for a specific pollutant, the RSC can be raised as high as 80% based on the available data, allowing the remaining 20% for other potential sources (USEPA, 2000b). Applying a lower RSC (e.g., 20%) is a more conservative approach to public health and results in a lower MCLG.

D.2.1. Literature Search and Screening

In 2020, the EPA conducted a literature search to evaluate evidence for pathways of human exposure to eight PFAS chemicals (PFOA, PFOS, PFBA, PFBS, PFDA, PFHxA, PFHxS, and PFNA) (Holder et al., 2023). This search was not date limited and spanned the information collected across the WOS, PubMed, and ToxNet/ToxLine (now ProQuest) databases. The results of the PFHxS literature search of publicly available sources are available through the EPA's Health & Environmental Resource Online website at https://hero.epa.gov/hero/index.cfm/project/page/project_id/2630.

The 950 literature search results for PFHxS were imported into SWIFT-Review (Sciome, LLC, Research Triangle Park, NC) and filtered through the Evidence Stream tags to identify human studies and nonhuman (i.e., those not identified as human) studies (Holder et al., 2023). Studies identified as human studies were further categorized into seven major PFAS pathways (Cleaning Products, Clothing, Environmental Media, Food Packaging, Home Products/Articles/Materials, Personal Care Products, and Specialty Products) as well as an additional category for Human Exposure Measures. Nonhuman studies were grouped into the same seven major PFAS pathway categories, except that the Environmental Media category did not include soil, wastewater, or landfill. Only studies published between 2003 and 2020 were considered. Application of the SWIFT-Review tags identified 654 peer-reviewed papers matching these criteria for PFHxS.

Holder et al. (2023) screened the 654 papers to identify studies reporting measured occurrence of PFHxS in human matrices and media commonly related to human exposure (human blood/serum/urine, drinking water, food, food contact materials, consumer products, indoor dust, indoor and ambient air, and soil). For this synthesis, additional screening was conducted to identify studies relevant to surface water (freshwater only) and groundwater using a keyword[10] search for water terms.

Following the PECO criteria outlined in Table D-3, the title and abstract of each study were independently screened for relevance by two screeners using *litstream*^{TM}. A study was included as relevant if it was unclear from the title and abstract whether it met the inclusion criteria. When two screeners did not agree whether a study should be included or excluded, a third reviewer was consulted to make a final decision. The title and abstract screening of Holder et al. (2023) and of this synthesis resulted in 494 unique studies being tagged as relevant (i.e., having data on occurrence of PFHxS in exposure media of interest) that were further screened with full-text

[10] Keyword list: water, aquifer, direct water, freshwater, fresh water, groundwater, ground water, indirect water, lake, meltwater, melt water, natural water, overland flow, recreation water, recreational water, river, riverine water, riverwater, river water, springwater, spring water, stream, surface water, total water, water supply

review using the same inclusion criteria. After additional review of the evidence collected by Holder et al. (2023), 109 studies originally identified for other PFAS also contained information relevant to PFHxS. Based on full-text review, 172 studies were identified as having relevant, extractable data for PFHxS from the United States, Canada, or Europe for environmental media, not including studies with only human biomonitoring data. Of these 172 studies, 161 were identified from Holder et al. (2023), where primary data were extracted into a comprehensive evidence database. Parameters of interest included: sampling dates and locations, numbers of collection sites and participants, analytical methods, limits of detection and detection frequencies, and occurrence statistics. Eleven of the 172 studies were identified in this synthesis as containing primary data on only surface water and/or groundwater.

Table D-3. Populations, Exposures, Comparators, and Outcomes (PECO) Criteria

PECO Element	Inclusion Criteria
Population	Adults and/or children in the general population and populations in the vicinity of PFAS point sources from the United States, Canada, or Europe
Exposure	Primary data from peer-reviewed studies collected in any of the following media: ambient air, consumer products, drinking water, dust, food, food packaging, groundwater[a], human blood/serum/urine, indoor air, landfill, sediment, soil, surface water[a] (freshwater), wastewater/biosolids/sludge
Comparator	Not applicable
Outcome	Measured concentrations of PFHxS (or measured emissions from food packaging and consumer products only)

Notes: PFHxS = perfluorohexanesulfonic acid.
[a] Surface water and groundwater were not included as relevant media in Holder et al. (2023). Studies were re-screened for these two media in this synthesis.

The evidence database of Holder et al. (2023) additionally identified 18 studies for which the main article was not available for review. As part of this synthesis, 17 of the 18 studies could be retrieved. An additional three peer-reviewed references were identified through gray literature sources, described below, that were included to supplement the search results. The combined 20 studies underwent full-text screening using the inclusion criteria in Table D-3. Based on full-text review, five studies were identified as relevant.

Using the screening results from the evidence database and this synthesis, a total of 177 peer-reviewed studies were identified as relevant. Fifty of these contained information relevant to the United States.

D.2.2. Additional Screening

The EPA also searched the following publicly available gray literature sources for information related to relative exposure of PFHxS for all potentially relevant routes of exposure (oral, inhalation, dermal) and exposure pathways relevant to humans:

- ATSDR's *Toxicological Profiles*;
- CDC's national reports on human exposures to environmental chemicals;
- EPA's CompTox Chemicals Dashboard;

- EPA's fish tissue studies;
- EPA's Toxics Release Inventory;
- Relevant documents submitted under the Toxic Substances Control Act and relevant reports from EPA's Office of Chemical Safety and Pollution Prevention;
- U.S. Food and Drug Administration's (FDA's) *Total Diet Studies* and other similar publications from FDA, U.S. Department of Agriculture, and Health Canada;
- NOAA's National Centers for Coastal Ocean Science data collections;
- National Science Foundation direct and indirect food and/or certified drinking water additives;
- *Throwaway Packaging, Forever Chemicals: European wide survey of PFAS in disposable food packaging and tableware* (Straková et al., 2021);
- PubChem compound summaries;
- Relevant sources identified in the relative source contribution discussions (Section 5) of EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA)/Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water*; and
- Additional sources, as needed.

The EPA has included available information from these gray literature sources for PFHxS relevant to its uses, chemical and physical properties, and for occurrence in ambient or indoor air, foods (including fish and shellfish), soil, dust, and consumer products. The EPA has included available information specific to PFHxS below on any regulations that may restrict PFHxS levels in media (e.g., water quality standards, air quality standards, food tolerance levels).

D.3. Summary of Potential Exposure Sources of PFHxS Other than Water

D.3.1. Dietary Sources

D.3.1.1. Seafood

PFHxS was detected in 71 of 157 fish tissue composite samples collected during the EPA's National Lake Fish Tissue Study, with a maximum concentration of 3.50 ng/g an[d] a 50th percentile concentration of <0.12 ng/g (Stahl et al., 2014). It was not detected in the 162 fish tissue composite samples collected during the EPA's 2008–2009 National Rivers and Streams Assessment (NRSA) (Stahl et al., 2014). More recently, PFHxS was detected in 32 of 349 fish tissue composite samples at concentrations ranging from 0.121 ng/g to 0.980 ng/g in the EPA's 2013–2014 NRSA (USEPA, 2020a). PFHxS was also detected in 1 of 152 fish tissue composite samples at a concentration of 0.96 ng/g in the EPA's 2015 Great Lakes Human Health Fish Fillet Tissue Study (USEPA, 2021g). PFHxS has been detected in a mixture of fish fillet samples collected from Mississippi River sites in Minnesota at a concentration of 0.47 ng/g (ATSDR, 2021; Delinsky et al., 2010). PFHxS has been detected in Irish pompano (*Diapterus auratus*), silver porgy (*Diplodus argenteus*), and gray snapper (*Lutjanus griseus*) from the St. Lucie Estuary in in NOAA's National Centers for Coastal Ocean Science, National Status and Trends Data (NOAA, 2022). Burkhard (2021) identified 47 studies reporting BAFs for PFHxS and

calculated a median (standard deviation) bioaccumulation factor (BAF) in muscle tissue/fillet of 19.95 ± 7.94 L/kg wet weight (reported as a logBAF of 1.30 ± 0.90 L/kg).

Five additional U.S. studies were identified that evaluated PFHxS levels in seafood (Young et al., 2022; Chiesa et al., 2019; Byrne et al., 2017; Young et al., 2013; Schecter et al., 2010) (Table D-4). One study evaluated fish samples collected directly from rivers and lakes (Byrne et al., 2017). As part of a study to assess exposure to PFHxS and other PFAS among residents of two remote Alaska Native villages on St. Lawrence Island, Byrne et al. (2017) measured PFAS concentrations in stickleback and Alaska blackfish, resident fish used as sentinel species to detect accumulation of PFAS in the local environment. Stickleback were collected from three locations – Suqitughneq (Suqi) River watershed (n = 9 composite samples), Tapisaggak (Tapi) River (n = 2 composite samples), and Troutman Lake (n = 3 composite samples). Blackfish were collected from the Suqi River (n = 29) but were not found in the other water bodies. Authors reported that the Suqi River watershed was upstream and downstream of a formerly used defense site and Tapi River was approximately 5 km east of a military site, however at the start of the study none of the sites were known to be contaminated with PFAS. The sample dates were not reported. PFHxS was not detected in any of the stickleback and blackfish samples, despite the authors noting that stickleback from Troutman Lake had "exceptionally high" total PFAS concentrations.

The remaining four studies purchased seafood from stores and fish markets (Young et al., 2022; Chiesa et al., 2019; Young et al., 2013; Schecter et al., 2010). Young et al. (2013) assessed fish and shellfish collected in 2010–2012 from retail markets across the continental United States. Retail markets in California, Florida, Illinois, Mississippi, New Jersey, New York, Tennessee, Texas, and Washington, D.C., were represented. Authors selected the 10 most consumed fish and shellfish in the United States that were farm raised, wild caught, or had unknown origin. Among the crab meat, shrimp, striped bass, catfish, clams, cod, flounder, pangasius, pollock, tuna, salmon, scallops, and tilapia, PFHxS was only detected in 1 of 10 samples of striped bass at a concentration of 0.66 ng/g. Young et al. (2022) evaluated fish and shellfish purchased from retail markets in the Washington. D.C. metropolitan area and online markets (clams only) from March 2021 through May 2022. Seafood samples represented 8 of the top 10 consumed fish and shellfish in the United States. Seafood samples were farm raised, wild caught, or of unknown origin, and location of harvest was provided when known. PFHxS was only detected in two seafood types, crab and clam meat. All samples of clam meat (n = 10) had detectable concentrations of PFHxS, ranging from 51 to 605 ng/kg. Only two samples of crabs (n = 11) had detectable levels of 112 and 242 ng/kg. Authors also analyzed food packaging for PFAS analytes and did not identify any packaging samples with detectable levels of PFAS. Schecter et al. (2010) evaluated PFHxS and other PFAS in seafood collected from five Dallas, Texas grocery stores in 2009. The origin or source of seafood was not described. Seafood included canned sardines in water, canned tuna, fresh catfish fillet, cod, frozen fish sticks, salmon, and tilapia (n = 1 composite sample for each seafood type). PFHxS was only detected in cod at a concentration of 0.07 ng/g ww. Finally, in a multicontinental study, Chiesa et al. (2019) collected salmon from a wholesale fish market in Milan, Italy; the sampling year was not reported. Wild-caught salmon samples originated from the United States (n = 7), Canada (n = 15), and Scotland (n = 2), while farmed salmon samples originated from Norway (n = 25) and Scotland (n = 17). Among the salmon that originated from the United States Pacific Ocean (FAO 67 and 77), two species – *Oncorhynchus kisutch* and *Oncorhynchus keta* – were analyzed, with PFHxS not

detected in either species (LOQ = 0.015 ng/g). PFHxS was also not detected in wild-caught salmon from Canada and Scotland.

In studies outside of the U.S., PFHxS was detected in multiple fish and shellfish species. Results for all of the identified non-U.S. studies are presented in detail in Table D-4and are summarized here. Approximately half of the studies including samples from outside the U.S. reported at least one sample with detectable levels of PFHxS. Bhavsar et al. (2014) reported detections of PFHxS in both cooked and raw samples of carp, lake trout, and walleye, but did not find detectable levels in raw or cooked Chinook salmon, all of which were caught recreationally from rivers in Ontario, Canada. PFHxS was also detected in several European studies that examined fresh-caught or farmed seafood, including market-bought samples (Hansen et al., 2016; Carlsson et al., 2014; Johansson et al., 2014; Yamada et al., 2014; Falandysz et al., 2006). The maximum PFHxS level detected in seafood was 2.22 ng/g ww in brown trout collected from Norway (Hansen et al., 2016). Though several studies reported on a variety of different species, many presented results for relatively small sample sizes ($n \leq 5$).

Table D-4. Summary of PFHxS Data in Seafood

Study	Location and Source	Seafood Type	Results
United States			
Byrne et al. (2017)	United States (Alaska) Stickleback collected from three locations on St. Lawrence Island: Suqitughneq (Suqi) River watershed (upstream and downstream of a formerly used defense site), Tapisaggak (Tapi) River (located approximately 5 km east of military site), and Troutman Lake, a coastal lake situated adjacent to the village of Gambell. Alaska blackfish collected from the Suqi River but were absent from the other water bodies. Sampling year not reported. No sites were known to be contaminated with PFASs at the initiation of the study.	Stickleback and Alaska blackfish	Stickleback: Troutman Lake: n = 3*; DF 0% Suqi River: n = 9*; DF 0% Tapi River: n = 2*; DF 0% Blackfish: n = 29; DF 0% (LOQ = 0.5–1 ng/g ww for all PFAS) *Number of composite samples, each composed of ~10 stickleback fish
Young et al. (2013)	United States (California; Illinois; Mississippi; Tennessee; Florida; New Jersey; New York; Texas; Washington, D.C.) Fish and shellfish collected from retail markets in 11 areas across the continental United States from 2010–2012. The fish and shellfish included farm raised, wild caught, and unknown origin, as well as freshwater fish, saltwater fish, and euryhaline fish. Crab meat, clams, cod, flounder, pangasius, salmon, scallops, and tilapia purchased from Washington, D.C. Shrimp purchased from Orlando, Florida; Memphis, Tennessee; and Nashville, Tennessee. Striped bass purchased from New York, New York and Cherry Hill, New Jersey. Catfish purchased from Indianola, Mississippi; Dallas, Texas; Tampa, Florida; and Orlando, Florida. Pollock purchased from Huntington Beach, California. Tuna purchased from Chicago, Illinois.	Crab, shrimp, striped bass, catfish, clams, cod, flounder, pangasius, pollock, tuna (can and pouch), salmon, scallops (bay and sea), tilapia	**Striped bass: n = 10, DF[a] 10%, range = ND–0.66* ng/g** Crab meat: n = 1, DF 0% Shrimp: n = 9, DF 0% Catfish: n = 13, DF 0% Clams: n = 1, DF 0% Cod: n = 1, DF 0% Flounder: n = 1, DF 0% Pangasius: n = 1, DF 0% Pollock: n = 1, DF 0% Tuna: n = 3, DF 0% Salmon: n = 2, DF 0% Scallops: n = 2, DF 0% Tilapia: n = 1, DF 0% (MDL = 0.55 ng/g for all seafood) *This value was above the MDL but below the LOQ; LOQ is estimated as 3x the MDL
Schecter et al. (2010)	United States (Texas) Seafood samples from five different grocery stores in Dallas, Texas were collected in 2009. Ten individual samples were collected for each food type and combined to form composite samples. The origin/source of the food samples were not reported.	Salmon, canned tuna, fresh catfish fillet, tilapia, cod, canned sardines, frozen fish sticks	**Cod: n = 1, point = 0.07 ng/g ww, LOD = NR** Salmon: n = 1, DF 0%, LOD = 0.07 ng/g ww Canned tuna: n = 1, DF 0%, LOD = 0.05 ng/g ww Fresh catfish fillet: n = 1, DF 0%, LOD = 0.06 ng/g ww Tilapia: n = 1, DF 0%, LOD = 0.04 ng/g ww Canned sardines: n = 1, DF 0%, LOD = 0.06 ng/g ww Frozen fish sticks: n = 1, DF 0%, LOD = 0.09 ng/g ww *Number of composite samples, each composed of ~10 individual samples

Study	Location and Source	Seafood Type	Results
Canada			
Bhavsar et al. (2014)	Canada (Ontario) Recreationally caught fish from four rivers – Credit River, Thames River, Niagara River, Welland River – in summer and fall of 2010 and 2011. Chinook salmon were caught from Credit River, common carp from Thames River, lake trout from Niagara River, and walleye from Welland River. Elevated PFASs concentrations were expected in the fish based on nearby industrial activities or previous monitoring work conducted by the Ontario Ministry of Environment. Raw fish were analyzed, as well as cooked fish using three different cooking methods (baking, broiling, and frying).	Raw and cooked fish (chinook salmon, common carp, lake trout, walleye)	Chinook salmon: Raw: n = 5, DF NR, mean = <0.006 ng/g ww Baked: n = 5, DF NR, mean = <0.006 ng/g ww Broiled: n = 5, DF NR, mean = <0.006 ng/g ww Fried: n = 5, DF NR, mean = <0.006 ng/g ww **Common carp:** **Raw: n = 5, DF NR, mean = 0.292 ng/g ww** **Baked: n = 5, DF NR, mean = 0.341 ng/g ww** **Broiled: n = 5, DF NR, mean = 0.291 ng/g ww** **Fried: n = 5, DF NR, mean = 0.359 ng/g ww** **Lake trout:** **Raw: n = 4, DF NR, mean = 0.258 ng/g ww** **Baked: n = 4, DF NR, mean = 0.248 ng/g ww** **Broiled: n = 4, DF NR, mean = 0.263 ng/g ww** **Broiled; n = 4, DF NR, mean = 0.245 ng/g ww** **Walleye:** **Raw: n = 5, DF NR, mean = 0.080 ng/g ww** **Baked: n = 5, DF NR, mean = 0.098 ng/g ww** **Broiled; n = 5, DF NR, mean = 0.088 ng/g ww** **Fried; n = 5, DF NR, mean = 0.083 ng/g ww** (LOQ not reported)
Europe			
Hansen et al. (2016)	Norway (Evenes; Skånland) Fish were sampled from Lake Langvatnet, Lake Lavangsvatnet, River Tårstadelva, and the reference Lake Strandvatnet. A civilian airport (location also shared with the Air Station of the Royal Norwegian Air Force) is situated on a ridge between Lake Langvatnet and Lake Lavangsvatnet. These waters are affected by PFAS due to AFFF emissions from the airport. Lake Lavangsvatnet drains into the river Tårstadelva and Lake Strandvatnet is ~15 km away from the airport with no connection to the airport runoff. Samples of the dorsolateral muscle were taken from 10 salmon, 10 anadromous brown trout, 12 stationary brown trout, and 3 European flounder by local fishermen and by personnel from Sweco, an environmental consulting company. The samples were collected in August and September 2014.	Brown trout, European flounder, salmon	**Brown trout (stationary):** **Lake Langvatnet: n = 6, DF[a] 83%, range = <LOD–0.15 ng/g ww** **Lake Lavangsvatnet: n = 5, DF[a] 100%, mean[a] (range) = 0.952 (0.21–2.22) ng/g ww** Lake Strandvatnet (reference): n = 1, point = <LOD **Brown trout (anadromous):** **Lake Lavangsvatnet: n = 3, DF[a] 67%, range = <LOD–0.86 ng/g ww** **River Tårstadelva: n = 5, DF[a] 60%, range = <LOD–0.01 ng/g ww** Lake Strandvatnet (reference): n = 2, DF 0% **European flounder (catadromous):** **Lake Lavangsvatnet: n = 3, DF[a] 100%, mean[a] (range) = 0.27 (0.10–0.53) ng/g ww** **Salmon (anadromous):** **Lake Lavangsvatnet: n = 5, DF[a] 60%, range = <LOD–0.03 ng/g ww**

Study	Location and Source	Seafood Type	Results
			River Tårstadelva: n = 5, DF[a] 20%, range = <LOD–0.01 ng/g ww (LOD = 0.014–0.224 ng/g for all PFAS)
Hölzer et al. (2011)	Germany Fish from Lake Möhne and River Möhne were caught by electric fishing or net fishing between June 2006 and October 2008. The River Möhne was contaminated with PFCs mainly by the application of polluted soil conditioner on agricultural lands between 2004 and 2006, which then drained into tributaries of the river. Fish samples for food monitoring were collected from retail trade, wholesale trade, supermarkets, and producers. Sampling year not provided.	Perch, pike, eel, cisco, and roach from Lake Möhne /River Möhne Eel, trout, pike/perch, and other from trade/markets	Lake Möhne: Perch: n = 15, DF 0% Pike: n = 6, DF 0% Eel: n = 5, DF 0% Cisco: n = 8, DF 0% Roach: n = 10, DF 0% Food monitoring: Eel: n = 2, DF 0% Trout: n = 73, DF 0% Pike/perch: n = 8, DF 0% Other: n = 34, DF 0% (LOD = 0.1 ng/g, LOQ = 0.2 ng/g)
Koponen et al. (2015)	Finland (Baltic Sea, Vanhankaupunginlahti bay, Lake Päijänne) A total of 296 individual fish samples were collected in 2009–2010 from five commercially and recreationally important fishing area across the Finnish coast of the Baltic Sea (Oulu, Pori, Turku, Hanko, and Kotka), Helsinki Vanhankaupunginlahti bay, a large freshwater Lake Päijänne, and four fish farming facilities. Most of the individual samples were pooled, each pool consisting of 2–10 individuals. Baltic herring, pike-perch, perch, burbot, whitefish, salmon, and vendace were collected from the Baltic sea; perch and pike-perch were collected from Helsinki Vanhankaupunginlahti bay; perch was collected from Lake Päijänne. Whitefish and rainbow trout were farmed fish. The selection of fish species was mainly based on the significance of fish in the Finnish diet.	Baltic herring, pike-perch, perch, burbot, whitefish, salmon, vendace, whitefish, rainbow trout	Baltic Sea: Baltic herring (n = 58), pike-perch (n = 30), perch (n = 25), burbot (n = 49), whitefish (n = 27), salmon (n = 44), vendace (n = 20): DF 0% Vanhankaupunginlahti bay: Pike-perch (n = 6), perch (n = 7): DF 0% Lake Päijänne: Perch (n = 10): DF 0% Farmed fish: Whitefish (n = 10), rainbow trout (n = 10): DF 0% (LOQ = 0.18–0.39 ng/g)
Jörundsdóttir et al. (2014)	Iceland Samples were collected by the Icelandic Marine Research Institute in March 2011 during their biannual scientific survey. Cod and anglerfish were caught south-west of Iceland, blue whiting was caught south-east of Iceland, and lumpfish and pollock were caught north-west of Iceland, while ling, plaice, and lemon sole were caught west of Iceland. Each fish sample consisted of a pooled sample from the entire edible part from ten individuals of the same species.	Anglerfish, Atlantic cod, blue whiting, lemon sole, ling, lumpfish, plaice, and pollock	Anglerfish (n = 1), Atlantic cod (n = 2), blue whiting (n = 2), lemon sole (n = 1), ling (n = 1), lumpfish (n = 4), plaice (n = 1), pollock (n = 1): DF 0% (LOD = 0.15 ng/g) *n represents number of composite samples

Study	Location and Source	Seafood Type	Results
Yamada et al. (2014)	France Marine fish sampled were selected based on the fish consumption habits of the population of four areas – La Rochelle in Gironde-Charente Maritime Sud, Le Havre in Normandy-Baie de Seine, Lorient in South Brittany, and Toulon in Mediterranean-Var. Five primary samples of fish were bought from the fish market and/or supermarket in each region for each species in January–April 2005. Freshwater fish sampled were selected based on the individual dietary consumption analysis of anglers or their family members of the ICAR-PCB study. Freshwater fish were collected in six major French rivers with each river divided into three or four section in 2008–2009. Half of the samples were composite samples.	Freshwater fish, fresh or frozen marine fish	Results presented for lower bound and upper bound if LB value different from UB value **Fresh and frozen marine fish:** **Total LB: n = 95, DF NR, mean (range) = 0.00 (0–0.03) ng/g ww** **Total UB: n = 95, DF NR, mean (range) = 0.03 (0.02–0.04) ng/g ww** Anchovy: n = 1, LB–UB = 0–0.03 ng/g ww Monkfish: n = 4, LB–UB = 0–0.02 ng/g ww Catshark: n = 4, LB–UB = 0.02–0.04 ng/g ww Cod: n = 4, LB–UB = 0–0.02 ng/g ww Common dab: n = 4, LB–UB = 0–0.02 ng/g ww Orange roughy: n = 3, LB–UB = 0–0.03 ng/g ww Plaice/witch: n = 2, LB–UB = 0–0.02 ng/g ww Goatfish: n = 3, LB–UB = 0–0.03 ng/g ww Grenadier: n = 4, LB–UB = 0–0.02 ng/g ww Gurnard: n = 1, LB–UB = 0.03 Haddock: n = 2, LB–UB = 0–0.02 ng/g ww Hake: n = 4, LB–UB = 0–0.02 ng/g ww Halibut: n = 4, LB–UB = 0–0.03 ng/g ww John dory: n = 2, LB–UB = 0.02–0.04 ng/g ww Ling: n = 4, LB–UB = 0–0.02 ng/g ww Mackerel: n = 4, LB–UB = 0–0.03 ng/g ww Pollack: n = 3, LB–UB = 0–0.02 ng/g ww Pout: n = 1, LB–UB = 0–0.03 ng/g ww Ray: n = 4, LB–UB = 0–0.02 ng/g ww Saithe: n = 4, LB–UB = 0–0.02 ng/g ww Salmon: n = 4, LB–UB = 0–0.03 ng/g ww Sardine: n = 4, LB–UB = 0–0.03 ng/g ww Scorpionfish: n = 1, LB–UB = 0–0.03 ng/g ww Seabass: n = 4, LB–UB = 0–0.03 ng/g ww Sea bream: n = 4, LB–UB = 0–0.03 ng/g ww Sole: n = 4, LB–UB = 0–0.02 ng/g ww Swordfish: n = 4, LB–UB = 0–0.03 ng/g ww Tuna: n = 4, LB–UB = 0–0.03 ng/g ww Whiting: n = 4, LB–UB = 0–0.02 ng/g ww **Freshwater fish:** Barbel: n = 5, LB–UB = 0.19–0.22 ng/g ww Bleak: n = 9, LB–UB = 0.02–0.41 ng/g ww Brown trout: n = 31, LB–UB = 0.06–0.17 ng/g ww Chub: n = 9, LB–UB = 0.05–0.16 ng/g ww Common carp: n = 7, LB–UB = 0.67–0.76 ng/g ww

Study	Location and Source	Seafood Type	Results
			Common roach: n = 67, LB–UB = 0.06–0.18 ng/g ww Minnow: n = 1, LB–UB = 0.4 ng/g ww European eel: n = 137, LB–UB = 0.67–0.77 ng/g ww European perch: n = 9, LB–UB = 0.11–0.2 ng/g ww Freshwater bream: n = 34, LB–UB = 0.1–0.24 ng/g ww Gudgeon: n = 5, LB–UB = 0.36–0.39 ng/g ww Northern pike: n = 8, LB–UB = 0.02–0.18 ng/g ww White bream: n = 22, LB–UB = 0.21–0.3 ng/g ww Thicklip grey mullet: n = 6, LB–UB = 0.01–0.16 ng/g ww Wels catfish: n = 14, LB–UB = 0.01–0.12 ng/g ww Western vairone: n = 1, LB–UB = 0–0.17 ng/g ww Pike-perch: n = 22, LB–UB = 0.06–0.16 ng/g ww (LOD = 0.007–0.95 ng/g for PFAAs other than PFOA and PFOS) *Lower bound (LB) scenario defined as values <LOD replaced with 0 *Upper bound (UB) scenario defined as values <LOD replaced with LOD
Eriksson et al. (2013)	Denmark (Faroe Islands) Wild fish (cod and saithe) were sampled from the Faroe Shelf area; cod were sampled in October and August 2011, while saithe were sampled in April 2012. Three farmed salmon samples were collected from different fjords in Faroe Islands, sampling year not reported.	Farmed salmon, wild-caught cod, wild-caught saithe	Cod 1, n = 1, DF 0% Cod 2, n = 1, DF 0% Saithe 1, n = 1, DF 0% Saithe 2, n = 1, DF 0% Salmon 1, n = 1, DF 0% Salmon 2, n = 1, DF 0% Salmon 3, n = 1, DF 0% (LOD = 0.018 ng/g) *n represents number of pooled samples, each combining muscle tissue from five fish
Falandysz et al. (2006)	Poland Cod samples were collected from the Gulf of Gdañsk in the Baltic Sea (south coast of Poland) in February 2003.	Cod *(Gadus morhua)*	**n = 18, DF NR, mean, median (range) = 0.1, 0.1 (0.05–0.8) ng/mL** (LOD not reported) *Values reported for animal whole blood samples
Rivière et al. (2019)	France Samples collected between July 2011 and July 2012 in the center region of France. Food items were selected based on the results of a national consumption survey to obtain a representative and general view of children's (0–3 years old) food consumption. All analyzed samples	Fish (unspecified)	n = 1, DF 0% (LOD = 0.0002–3.7 ng/g fw for all PFAS) *n represents number of composite samples

Study	Location and Source	Seafood Type	Results
	were formed of 12 subsamples of the same food and of equal weight. The fish were cooked according to the practices reported in the survey of practices.		
Sadia et al. (2020)	Sweden (Örebro) Three fish samples from different brands were purchased from a local supermarket in February 2019.	Fish	**n = 3, DF[a] 33%, range = ND–0.0055 ng/g ww** (LOD = 0.0011; LOQ = 0.0049 ng/g)
Barbosa et al. (2018)	Belgium, France, The Netherlands, Portugal Fish were collected from different markets based on the assumption that the fish species were frequently consumed in European Union countries and the fish species contained high levels of contaminants of emerging concern. Sampling year not reported. The following fish species (origin, market country) were included: *P. platessa:* Channel, Belgium *M. australis:* South America, Portugal *M. capenis:* South Africa, Portugal *K. pelamis*: Azores, Portugal *M. edulis*: North Sea, The Netherlands; France, France	Raw and steamed fish (*P. platessa, M. australis, M. capenis, K. pelamis*, and *M. edulis*)	*P. platessa*: n = 25, DF 0% *M. australis*: n = 25, DF 0% *M. capenis*: n = 25, DF 0% *K. pelamis*: n = 25, DF 0% *M. edulis*: The Netherlands: n = 50, DF 0% France: n = 50, DF 0% (LOD = <0.01 ng/g ww for all PFCs)
Gebbink et al. (2015)	Sweden Food items were purchased at two major grocery store chains in four major Swedish cities in 1999 and 2005. In 2010, sampling was limited to Uppsala city since no systematic geographical differences in food contamination was observed in the two earlier market basket studies. The food items were selected based on Swedish food and production statistics and were not cooked before analysis. Homogenates of fish products (fresh and frozen fillets of fish, canned fish products, shellfish) were prepared for each collection year by mixing food items proportionally according to food consumption statistics. Results were not reported for the 2005 and 2010 fish product composite samples (only reported pooled with other food types).	Fish	**1999: n = 1, point = 0.0069 ng/g** (MLOQ = 0.0001 ng/g) *n represents number of composite samples
Vassiliadou et al. (2015)	Greece Samples were obtained during the winter and early spring of 2011. Finfish, squids, and shrimps were purchased from the local fish market in Kallithea, Athens, while mussels were obtained from a mariculture farm within the Saronikos Gulf, Attika. Samples were analyzed raw as well as cooked in the ways favored in Greek cuisine (pan-fried in olive oil and/or grilled).	Anchovy, bogue, hake, picarel, sand smelt, sardine, striped mullet, mussel, shrimp, squid	Anchovy (raw, fried, grilled), bogue (raw, fried, grilled), hake (raw, fried, grilled), picarel (raw, fried), sand smelt (raw, fried), sardine (raw, fried, grilled), striped mullet (raw, fried, grilled), mussel (raw, fried), shrimp (raw, fried), and squid (raw, fried, grilled): n = 4 for each, DF 0% *n represents number of composite samples (LOD = 0.18 ng/g ww; LOQ = 0.54 ng/g ww)

Study	Location and Source	Seafood Type	Results
	Quadruplicate composite samples were created for each food type, each consisting of four to six items of raw or cooked fish or shellfish.		
Carlsson et al. (2014)	Greenland (Nuuk) Seafood was purchased at the local fish market and grocery shops in June 2010. All items were originally caught in the vicinity of the Nuuk area and/or along the West coast of Greenland and represented the common food items consumed by the local Inuit population.	Salmon, halibut	Raw salmon fillet: n = 6, DF 0% Smoked salmon fillet: n = 6, DF 0% Smoked halibut fillet: n = 6, DF 0% (LOD = 0.014–0.224 ng/g for all PFAS)
Pérez et al. (2014)	Serbia (Belgrade and Novi Sad), Spain (Barcelona, Girona, and Madrid) Between September 2011 and February 2013, samples were purchased from different supermarkets and retail stores in representative cites around the world, including cities in Serbia and Spain in Europe.	Bivalves, whiting, cod, hake, salmon, herring, pangasius, trout, tuna	Spain: Bivalves (n = 28), whiting (n = 7), cod (n = 12), hake (n = 3), salmon (n = 9), herring (n = 22), pangasius (n = 9), trout (n = 19), tuna (n = 9): DF 0% Serbia: Canned tuna (n = 1), pangasius (n = 1), cod (n = 2), herring (n = 5), trout (n = 2): DF 0% (MLOD = 0.107 ng/g; MLOQ = 0.356 ng/g)
Domingo et al. (2012)	Spain (Catalonia) Foods purchased from 4 shops/stores of each of the 12 representative cities of Catalonia (Barcelona, l'Hospitalet de Llobregat, Vilanova I la Geltrú, Mataír, Sabadell, Terrassa, Girona, Tarragona, Reus, Tortosa, Lleida and Manresa) in September 2011. Shops/stores included local markets, small stores, supermarkets, and big grocery stores. For each food item, two composite samples were prepared for analysis, where each composite sample consisted of 24 individual units. Only edible parts of each food item were included in the composites.	Fish and seafood (sardine, tuna, anchovy, swordfish, salmon, hake, red mullet, sole, cuttlefish, clam, mussel, and shrimp)	**n = 2, DF NR, mean = 0.045 ng/g fw** (LOD not reported) *n represents number of composite samples
Vestergren et al. (2012)	Sweden (Malmoe, Gothenburg, Uppsala, Sundsvall) Purchasing locations of the two largest retail chains in Sweden were selected in each of four major Swedish cities. All purchases were made in spring/summer of 1999, 2005, and 2010. In 2010, the study was limited to the largest retail chains in Uppsala located in close vicinity to Stockholm. An equal amount of each food group from each of the four cities was combined into one sample pool to provide a representative sample for the Swedish urban population.	Fish products (fresh and frozen fillets of fish, canned fish products, shellfish)	**1999: n = 1, point = 0.0217 ng/g** **2005: n = 1, point = 0.0088 ng/g** **2010: n = 1, point = 0.0092 ng/g** (MDL = 0.0020 ng/g; MQL = 0.0059 ng/g) *n represents number of composite samples

Study	Location and Source	Seafood Type	Results
Noorlander et al. (2011)	The Netherlands Fish randomly purchased from several Dutch retail stores with nationwide coverage in November 2009. Fish samples were ground, homogenized, and pooled for analysis.	Fatty fish (herring, eel, mackerel, salmon), lean fish (cod, plaice, pollack, tuna), crustaceans (mussels, shrimp, crab)	**Fatty fish: n = 1, point = 0.009 ng/g** **Lean fish: n = 1, point = 0.023 ng/g** **Crustaceans: n = 1, point = 0.044 ng/g** (LOD not reported) *n represents number of composite samples
Jogsten et al. (2009)	Spain (Catalonia) Fish samples purchased from local markets, large supermarkets, and grocery stores from two different areas of Tarragona Province, Catalonia in January and February 2008. The cities of Tarragona and Reus were sampled in the northern area and L'Ametlla de Mar and Tortosa in the southern area. For each food item, two composite samples were analyzed (one composite for the northern area and one for the southern area). Each composite was formed of a minimum of six individual sub-samples of the same product.	Marinated salmon (homemade and packaged)	**Homemade: n = 2, DF NR, mean = 0.014 ng/g** Packaged: n = 2, DF NR, mean = <0.003 ng/g (LOD = 0.001 ng/g) *n represents number of composite samples *Values of ND were replaced with ½×LOD
Ericson et al. (2008a)	Spain Food samples purchased from local markets, large supermarkets, and grocery stores within Tarragona County in July 2006. Food samples were randomly purchased with origin source not specified. Each of the food samples were duplicated and combined to analyze a composite sample. Only the edible part of each food was included in the composite samples. Composite samples included the following: White fish: hake, whiting blue, sea bass, monkfish Seafood: mussel, shrimp Tinned fish: tuna, sardine, mussel Blue fish: salmon, sardine, and tuna	White fish, seafood, tinned fish, blue fish	White fish: n = 2, DF 0% Seafood: n = 2, DF 0% Tinned fish: n = 2, DF 0% Blue fish: n = 2, DF 0% (LOD = 0.001–0.65 ng/g fw) *n represents number of composite samples
Johansson et al. (2014)	Sweden Farmed rainbow trout (whole fish) were collected from fish farms along the Swedish Baltic Sea coast (brackish water). Only fish older than 12 months were sampled. Samples were collected annually from 1999 to 2010 within the Swedish National Food Agency's official food control program.	Rainbow trout	**Total: n = 36, DF[a] 58%, range = <0.011–0.04 ng/g fw** **1999: n = 10, DF[a] 90%, range = <0.011–0.034 ng/g fw** **2000: n = 3, DF[a] 100%, mean[a] (range) = 0.014 (0.012–0.017 ng/g fw** **2001: n = 4, DF[a] 75%, range = <0.011–0.040 ng/g fw** **2002: n = 1, point = 0.017 ng/g fw** 2003: n = 2, DF 0% **2004: n = 1, point = 0.020 ng/g fw**

Study	Location and Source	Seafood Type	Results
			2005: n = 4, DF[a] 25%, range = <0.011–0.020 ng/g fw **2006: n = 3, DF[a] 33%, range = <0.011–0.014 ng/g fw** **2007: n = 2, DF[a] 50%, range = <0.011–0.029 ng/g fw** 2008: n = 1, DF 0% **2009: n = 4, DF[a] 25%, range = <0.011–0.015 ng/g fw** 2010: n =1, DF 0% (MDL = 0.011 ng/g fw; MQL = 0.025 ng/g fw)
Multiple Continents			
Chiesa et al. (2019)	United States (Pacific Ocean) Wild-caught fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from USA-Pacific Ocean (Food and Agriculture Organization Area 67 and 77).	Wild-caught salmon (*Oncorhynchus kisutch* and *Oncorhynchus keta*)	*Oncorhynchus kisutch*: n = 5, DF 0% *Oncorhynchus keta*: n = 2, DF 0% (LOQ = 0.015 ng/g)
	Canada Wild-caught fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Canada (Food and Agriculture Organization Area 67).	Wild-caught salmon (*Oncorhynchus nerka*)	n = 15, DF 0% (LOQ = 0.015 ng/g)
	Norway Farmed fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Norway (Food and Agriculture Organization Area 27).	Farmed salmon (*Salmo salar*)	n = 25, DF 0% (LOQ = 0.015 ng/g)
	Scotland Wild-caught and farmed fish were collected at a wholesale fish market in Milan, Italy. Sampling year was not reported. The wild-caught salmon were from Scotland (Food and Agriculture Organization Area 27).	Wild-caught and farmed salmon (*Salmo salar*)	Wild-caught: n = 2, DF 0% Farmed: n = 17, DF 0% (LOQ = 0.015 ng/g)

Notes: DF = detection frequency; LOD = limit of detection; LOQ = limit of quantitation; MDL = method detection limit; ND = not detected; NR = not reported; ww = wet weight. Bold indicates detected levels of PFHxS in food.

[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

4.1.1.1 Other Food Sources

PFHxS was included in a suite of PFAS evaluated in FDA's 2019, 2021, and 2022 Total Diet Study Sampling (US FDA, 2022b, 2022a, 2021b, 2021a, 2020b, 2020a); however, it was not detected in any of the food samples tested. It should be noted that FDA indicated that the sample sizes used in the PFAS 2019, 2021, and 2022 Total Diet Study Sampling were limited and that the results should not be used to draw definitive conclusions about PFAS levels in the general food supply (US FDA, 2022c). PFHxS was detected in milk samples collected from a farm with groundwater known to be contaminated with PFAS; however, it was not detected in produce collected from an area near a PFAS production plant, in FDA studies of the potential exposure to the U.S. population to PFAS (US FDA, 2021c, 2018). PFHxS is not a registered pesticide under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), and the EPA does not set a 40 CFR Part 180 pesticide tolerance in food and feed commodities for PFHxS (US GPO, 2022). Maximum residue levels for PFHxS were not found in the Global Maximum Residue Level Database (Bryant Christie Inc, 2022).

Several peer-reviewed studies were identified that examined PFHxS in food sources other than seafood, including cereals, dairy, fats/other (e.g., eggs, oils, and spices), fruits and vegetables, meat, and breastmilk (Table D-5). Few U.S. studies analyzed foods from any one origin – four sampled crops grown in areas with known or suspected PFAS contamination, including biosolids-amended soils, two sampled from crops as part of greenhouse and field studies, one studied wild-caught alligator meat. Only two studies sampled from store- or market-bought meats, eggs, produce, and dairy.

Scher et al. (2018) evaluated garden produce samples from homes in Minnesota within and outside of a GCA in the vicinity of a former 3M PFAS production facility. Twenty homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. A total of 279 produce samples (232 inside GCA, 47 outside GCA) were collected between May and October 2010. PFHxS was detected in 1% of the 232 produce samples from inside the GCA (one floret sample and one leaf sample). The authors suggested that the two detections were associated with PFAS present in irrigation water that had accumulated in produce. They also noted that accumulation of PFAS was particularly high in florets. Three homes that were outside the GCA served as a reference. No PFHxS was detected in produce samples from home gardens outside the GCA. Genualdi et al. (2017) analyzed PFAS contamination in a Massachusetts cranberry bog approximately 10 miles from a military base with a history of AFFF usage. Ten cranberry samples were taken directly from trucks transporting cranberries and 32 cranberry samples were collected directly from the bog water in November 2016. PFHxS was not detected in any samples (MDL = 0.79 ng/g).

Two studies purchased food items from stores and markets for evaluation (Young et al., 2012; Schecter et al., 2010). Schecter et al. (2010) assessed PFHxS and other PFAS in food samples collected from five Dallas, Texas grocery stores in 2009. The origin or source of each food was not described. Food items included meat products (bacon, canned chili, chicken breast, ground beef, roast beef, ham, sausage, and turkey), dairy (butter, cheeses, frozen yogurt, ice cream, milk, and yogurt), eggs, grains (cereal), fruits and vegetables (apples, potatoes), and fats/other (canola oil, margarine, olive oil, peanut butter). PFHxS was not detected in any of the food samples. In Young et al. (2012), cow milk was purchased from retail markets across the continental United

States representing 17 states; the sampling year was not reported. Cow milk samples included organic milk, vitamin D added milk, and ultra-pasteurized milk. PFHxS was not detected in any of the 49 retail milk samples (MDL = 0.15 ng/g).

One study investigated PFAS levels in wild meat (Tipton et al., 2017). Tipton et al. (2017) assessed alligator tail meat that was collected during the South Carolina recreational hunting season between September to October 2015. Tail meat samples were collected from four different public hunt units – Southern Coastal, Middle Coastal, Midlands, and Pee Dee. PFHxS was detected in all samples from all hunt units. Median concentrations from Southern Coastal (n = 19), Middle Coastal (n = 17), Midlands (n = 5), and Pee Dee (n = 2) were 0.087 ng/g, 0.099 ng/g, 0.0816 ng/g, and 0.093 ng/g wet mass, respectively.

Two studies by Blaine et al. (2014; 2013) evaluated PFHxS in crops grown in greenhouse and field studies. In Blaine et al. (2014), PFAS levels were measured in celery root, pea fruit, and radish root grown in a greenhouse with control (unamended) soil, industrially impacted soil, and municipal soil (n = 3–5). PFHxS was detected in radish root from all three soils, celery shoot from the industrially impacted and municipal soil, and pea fruit from only industrially impacted soil. Mean concentrations of PFHxS in radish root for the control, industrially impacted, and municipal soil were 3.81 ng/g, 2.84 ng/g, and 4.33 ng/g, respectively. Mean concentrations of PFHxS in celery shoot for the industrially impacted and municipal soil were 3.19 ng/g and 0.38 ng/g, respectively. The mean concentration of PFHxS in pea fruit in the industrially impacted soil was 0.24 ng/g. Authors noted minor cross-contamination of the control soil due to the proximity of the unamended soil to biosolids-amended soil. In Blaine et al. (2013), authors studied the uptake of PFAS into edible crops in both field and greenhouse studies. In the field study, PFAS levels were measured in corn grain and corn stover grown with control (unamended), urban biosolids-amended, and rural biosolids-amended soil (n = 3–7). Mean PFHxS concentrations were below the LOQ in both corn grain and corn stover grown in any field study plots (<0.04 ng/g for corn grain; <0.29 ng/g for corn stover). In the greenhouse study, lettuce and tomato plants were grown in control soil, industrially impacted soil, or municipal soil (n = 3–5). Mean PFHxS concentrations were below the LOQ for lettuce and tomato grown in the control soil and for tomato grown in municipal soil; however, mean PFHxS levels were 10.44 ng/g and 5.54 ng/g for lettuce grown in industrially impacted and municipal soils, respectively, and 0.76 ng/g for tomato grown in industrially impacted soil. Sampling year was not reported.

The remaining two U.S. studies evaluated the occurrence of PFHxS in breastmilk (von Ehrenstein et al., 2009; Kuklenyik et al., 2004). von Ehrenstein et al. (2009) collected breastmilk samples between December 2004 and July 2005 from women between the ages of 18 and 38 at the time of recruitment as part of the pilot study Methods Advancement for Milk Analysis (MAMA). Women provided milk samples at two visits – the first visit was 2–7 weeks postpartum, and the second visit was 3–4 months postpartum. PFHxS was not detected in any of the samples from the first visit (n = 18) or second visit (n = 20). Similarly, PFHxS was below the LOD (0.3 ng/mL) in the samples reported by Kuklenyik et al. (2004). Kuklenyik et al. (2004) did not report information on the breastmilk donors or the sampling procedure as it was unavailable; PFHxS was not detected in either of the two samples.

Of the studies conducted in Europe examining non-breastmilk and non-seafood food items, 15 found PFHxS in at least one food type. Results for all of the identified non-U.S. studies are

presented in detail in Table D-5 and are summarized here. Across the European studies, PFHxS was found in animal products such as meat, dairy, and eggs, as well as fruits and vegetables. Similarly to studies examining seafood, most studies reported on a variety of different food types, but the majority presented results for relatively small sample sizes ($n \leq 5$). Lastly, eight of the twelve studies conducted outside the U.S. examining PFHxS in breastmilk had detectable levels.

Table D-5. Summary of PFHxS Data in Other Food

Study	Location and Source	Food Types	Results
United States			
Scher et al. (2018)	United States (Minnesota) Home garden produce samples were collected between May and October 2010 from 20 homes in 3 cities within a GCA as well as 3 homes in the Twin Cities Metro outside the GCA. Homes within the GCA were near a former 3M PFAS production facility, had previous or ongoing PFAS contamination in drinking water, and were served by the Oakdale, Minnesota public water system or were formerly or currently using a private well previously tested and shown to have detectable levels of PFOA or PFOS. 279 produce samples (232 within GCA and 47 outside GCA) consisting of mature, edible portions of plants were analyzed. Plant part categories included floret, fruit, leaf, root, seed, and stem.	Fruits and vegetables	Within GCA: **All:** n = 232, DF 1%, median (range) = ND (ND–0.066) ng/g **Floret: n = 5, DF 20%, median (range) = ND (ND–0.066) ng/g** **Leaf: n = 35, DF 3%, median (range) = ND (ND–0.046) ng/g** Garden fruit (n = 98), yard fruit (n = 13), root (n = 29), seed (n = 29), and stem (n = 23): DF 0% Outside GCA: All: n = 47, DF 0% Floret (n = 1), garden fruit (n = 15), yard fruit (n = 4), leaf (n = 12), root (n = 5), seed (n = 5), and stem (n = 5): DF 0% (MDL = 0.003 to 0.029 ng/g depending on the analyte and type of produce)
Genualdi et al. (2017)	United States (Massachusetts) Samples from cranberry bog with surface water contaminated with PFAS—likely due to proximity to a military base with a history of AFFF usage. The bog was located approximately 10 miles from the military base. Ten cranberry samples taken directly from trucks transporting cranberries (five samples each from two trucks) and 32 cranberry samples taken directly from 12 sections of the bog water. Samples collected in November 2016.	Fruits	n = 42, DF 0% (MDL = 0.79 ng/g)
Schecter et al. (2010)	United States (Texas) Food samples from five different grocery stores in Dallas, Texas were collected in 2009. Ten individual samples were collected for each food type and combined to form composite samples. The origin/source of the food samples were not reported.	Dairy; fruits and vegetables; grains; meat; fats/other	Meat Hamburger: n = 1, DF 0%, LOD = 0.04 ng/g ww Bacon: n = 1, DF 0%, LOD = 0.05 ng/g ww

Study	Location and Source	Food Types	Results
			Sliced turkey: n = 1, DF 0%, LOD = 0.02 ng/g ww Sausages: n = 1, DF 0%, LOD = 0.04 ng/g ww Ham: n = 1, DF 0%, LOD = 0.02 ng/g ww Sliced chicken breast: n = 1, DF 0%, LOD = 0.02 ng/g ww Roast beef: n = 1, DF 0%, LOD = 0.02 ng/g ww Canned chili: n = 1, DF 0%, LOD = 0.01 ng/g ww Dairy and Eggs Butter: n = 1, DF 0%, LOD = 0.09 ng/g ww American cheese: n = 1, DF 0%, LOD = 0.04 ng/g ww Other cheese: n = 1, DF 0%, LOD = 0.04 ng/g ww Whole milk: n = 1, DF 0%, LOD = 0.02 ng/g ww Ice cream: n = 1, DF 0%, LOD = 0.03 ng/g ww Frozen yogurt: n = 1, DF 0%, LOD = 0.02 ng/g ww Whole milk yogurt: n = 1, DF 0%, LOD = 0.03 ng/g ww Cream cheese: n = 1, DF 0%, LOD = 0.02 ng/g ww Eggs: n = 1, DF 0%, LOD = 0.04 ng/g ww Grains Cereals: n = 1, DF 0%, LOD = 0.04 ng/g ww Fruits and Vegetables Apples: n = 1, DF 0%, LOD = 0.02 ng/g ww Potatoes: n = 1, DF 0%, LOD = 0.04 ng/g ww Fats/Other

Study	Location and Source	Food Types	Results
			Olive oil: n = 1, DF 0%, LOD = 0.3 ng/g ww Canola oil: n = 1, DF 0%, LOD = 0.5 ng/g ww Margarine: n = 1, DF 0%, LOD = 0.03 ng/g ww Peanut butter: n = 1, DF 0%, LOD = 0.03 ng/g ww *n reflects number of composite samples, each composed of ~10 individual samples
Young et al. (2012)	United States (17 states) Retail cow's milk samples were all pasteurized whole milk, commercially available, and purchased at retail markets across the continental United States representing 17 states. Samples were organic milk, vitamin D added milk, and ultra-pasteurized milk. Sampling year not reported.	Dairy	n = 49, DF 0%, (MDL = 0.15 ng/g)
Tipton et al. (2017)	United States (South Carolina) Alligator tail meat samples were collected from a local wild game meat processer during the South Carolina recreational hunt season between September to October 2015. Samples were from four different public hunt units—Southern Coastal, Middle Coast, Midlands, and Pee Dee.	Meat	**Alligator tail:** **Southern coastal: n = 19, DF[a] 100%, median (range) = 0.087 (0.051–0.252) ng/g wet mass** **Middle coastal: n = 17, DF[a] 100%, median (range) = 0.099 (0.063–0.272) ng/g wet mass** **Midlands: n = 5, DF[a] 100%, median (range) = 0.0816 (0.054–0.158) ng/g wet mass** **Pee Dee: n = 2, DF[a] 100%, median (range) = 0.093 (0.071–0.115) ng/g wet mass** (RL not reported)
Blaine et al. (2014)	United States (Midwest) Crops grown in in greenhouse study with control (unamended), industrially impacted soil, or municipal soil. Control soil had minor cross-contamination due to proximity to biosolids-amended fields. Industrially impacted soil was amended with industrially impacted biosolids, and municipal soil was amended with municipal biosolids for over 20 years. Crops grown in the greenhouse study were grown from seed in pots, which were randomly arranged within the greenhouse. Sampling year not reported.	Fruits and vegetables	**Radish root:** **Control: n = 3–5, DF NR, mean = 3.81 ng/g** **Industrially impacted; n = 3–5, DF NR, mean = 2.84 ng/g** **Municipal: n = 3–5, DF NR, mean = 4.33 ng/g** **Celery shoot:** Control: n = 3–5, DF 0% **Industrially impacted: n = 3–5, DF NR, mean = 3.19 ng/g**

Study	Location and Source	Food Types	Results
			Municipal: n = 3–5, DF NR, mean = 0.38 ng/g **Pea fruit:** Control: n = 3–5, DF 0% **Industrially impacted: n = 3–5, DF NR, mean = 0.24 ng/g** Municipal: n = 3–5, DF 0% (LOQ = 0.03 ng/g)
Blaine et al. (2013)	United States (Midwest) Crops grown in urban and rural full-scale field study with control (unamended) and biosolids-amended soil. Three agricultural fields were amended (0.5×, 1×, or 2×) with municipal biosolids. Urban biosolids (1× and 2×) were from a WWTP receiving both domestic and industrial waste. Rural biosolids (0.5×) were from a WWTP receiving domestic waste only. Control plots were proximal to the rural and urban amended corn grain and corn stover field sites; sampling year not provided. Crops grown in greenhouse study with control (nonamended) and biosolids-amended soil. Nonamended soil obtained from a field that received commercial fertilizers and had a similar cropping system as the nearby municipal soil site. Municipal soil was obtained from a reclamation site in Illinois where municipal biosolids were applied at reclamation rates for 20 years, reaching the cumulative biosolids application rate of 1,654 Mg/ha. Industrially impacted soil was created by mixing composted biosolids from a small municipal (but impacted by PFAA manufacturing) WWTP with control soil on a 10% mass basis. Sampling year not provided.	Fruits and vegetables; grains	Field study: Corn grain: Urban nonamended: n = 3–7, DF NR, mean = <0.04 ng/g Urban 1×: n = 3–7, DF NR, mean = <0.04 ng/g Urban 2×: n = 3–7, DF NR, mean = <0.04 ng/g Rural nonamended: n = 3–7, DF NR, mean = <0.04 ng/g Rural 0.5×: n = 3–7, DF NR, mean = <0.04 ng/g Corn stover: Urban nonamended: n = 3–7, DF NR, mean = <0.29 ng/g Urban 1×: n = 3–7, DF NR, mean = <0.29 ng/g Urban 2×: n = 3–7, DF NR, mean = <0.29 ng/g Rural nonamended: n = 3–7, DF NR, mean = <0.29 ng/g Rural 0.5×: n = 3–7, DF NR, mean = <0.29 ng/g (LOQ = 0.04 ng/g for corn grain; LOQ = 0.29 ng/g for corn stover) Greenhouse study: Lettuce:

Study	Location and Source	Food Types	Results
			Nonamended: n = 3–5, DF NR, mean = <0.01 ng/g **Industrially impacted: n = 3–5, DF NR, mean = 10.44 ng/g** **Municipal: n = 3–5, DF NR, mean = 5.54 ng/g** Tomato: Nonamended: n = 3–5, DF NR, mean = <0.03 ng/g **Industrially impacted: n = 3–5, DF NR, mean = 0.76 ng/g** Municipal: n = 3–5, DF NR, mean = <0.03 ng/g (LOQ = 0.01 ng/g for lettuce; LOQ = 0.03 ng/g for tomato)
von Ehrenstein et al. (2009)	United States (North Carolina) As part of the Methods Advancement for Milk Analysis (MAMA) pilot study, 34 breastfeeding women aged 18 to 38 years at recruitment provided breastmilk samples at two visits. The first visit occurred 2–7 weeks postpartum, and the second visit occurred 3–4 months postpartum. Both visits were between December 2004 and July 2005.	Breastmilk	Visit #1: n = 18, DF 0% Visit #2: n = 20, DF 0% (LOQ = 0.30 ng/mL)
Kuklenyik et al. (2004)	United States (Georgia) Authors reported that no information was provided on the human milk donors or the sampling procedure.	Breastmilk	n = 2, DF 0% (LOD = 0.3 ng/mL)
Canada			
Kubwabo et al. (2013)	Canada (Ontario) Breastmilk samples were collected in the Kingston region of Ontario in 2003–2004.	Breastmilk	n = 13, DF 0% (MDL = 0.125 ng/mL; LOQ = 0.416 ng/mL)
Europe			
Scordo et al. (2020)	Italy Commercially available strawberry and olive fruits were purchased in two Italian supermarkets in 2018.	Fruits	**Strawberries: n = 2, DF[a] 100%, mean[a] (range) = 0.469 (0.148–0.790) ng/g dw** (MDL = 0.010 ng/g; MQL = 0.037 ng/g)

Study	Location and Source	Food Types	Results
			Olives: n = 2, DF[a] 50%, range = <0.0024–0.022 ng/g dw (MDL = 0.0024 ng/g; MQL = 0.0086 ng/g)
Sadia et al. (2020)	Sweden (Örebro) Three samples of each food type (cow milk, butter, chicken meat, beef) of different brands were purchased from a local supermarket in February 2019.	Dairy; fats/other	**Milk: n = 3, DF[a] 100%, mean[a] (range) = 0.0021 (0.0014–0.0030) ng/g ww** **Butter: n = 3, DF[a] 100%, mean[a] (range) = 0.0082 (0.0026–0.018) ng/g ww** **Beef: n = 3, DF[a] 67%, range = ND–0.0044 ng/g ww** **Chicken: n = 3, DF[a] 100%, mean[a] (range) = 0.0035 (0.0033–0.0036) ng/g ww** (LOD = 0.0011; LOQ = 0.0049 ng/g)
Sznajder-Katarzyńska et al. (2019)	Poland Milk and milk products were purchased in Polish markets in 2017. Commercially available samples of each product were obtained from five different suppliers.	Dairy	**All dairy: n = 35, DF 48.6%, sum PFHxS = 0.47 ng/g** **Milk: n = 5, DF[a] 60%, range = 0.01–0.01 ng/g** **Cottage cheese: n = 5, DF[a] 100%, range = 0.04–0.05 ng/g** **Natural yogurt: n = 5, DF[a] 60%, range = 0.01–0.02 ng/g** **Kefir/Bonny clabber: n = 5, DF[a] 60%, range = 0.01–0.02 ng/g** **Butter: n = 5, DF[a] 60%, range = 0.02–0.03 ng/g** Sour cream (n = 5), camembert-type cheese (n = 5): DF 0% (LOD = 0.003 ng/g; LOQ = 0.010 ng/g) *Range reported for detected values
Rivière et al. (2019)	France Samples collected between July 2011 and July 2012 in the center region of France. Food items were selected based on the results of a national consumption survey to obtain a representative and general view of children's (0–3 years old) food consumption. All analyzed samples were formed of 12 subsamples of the same food and of equal weight. The products purchased were prepared in a way that reflected as closely as possible what is done in the home (preparation and cooking).	Meat; dairy; fruits and vegetables; fats/other	Infant-specific foods: Milk-based beverage (n = 8), cereals (n = 5), milk-based desserts (n = 6), growing-up milk (n = 9), soups and puree (n = 11), fruit puree (n = 4), vegetable-based ready-to-eat meal (n = 20), meat/fish-based ready-to-eat meal (n = 45), infant formula (n = 28), follow-on formula (n = 33): DF 0% Common foods:

Study	Location and Source	Food Types	Results
			Non-alcoholic beverages (n = 1), dairy-based desserts (n = 1), milk (n = 1), mixed dishes (n = 1), ultra-fresh dairy products (n = 1), meat (n = 1), poultry and game (n = 1): DF 0% (LOD = 0.0002–3.7 ng/g for all PFAS) *n represents number of composite samples
Sznajder-Katarzyńska et al. (2018)	Poland Samples were purchased in Polish markets in 2017. Individual food items were selected among the most frequently consumed in Poland. Vegetables (potatoes, beetroots, carrots, white cabbage, tomatoes) and fruits (apples, cherries, strawberries) of Polish origin were bought in season when naturally ripe. Bananas, lemons, and oranges were bought after being imported to Poland. Five different samples of each fruit or vegetable were collected.	Fruits and vegetables	Apples, bananas, cherries, lemons, oranges, strawberries, beetroots, carrots, tomatoes, potatoes, and white cabbage: n = 5 for each, DF 0% (LOD = 0.006 ng/g; LOQ = 0.017 ng/g)
Surma et al. (2017)	Spain, Slovakia Spice samples were collected in powder form from Spain and Slovakia. Sampling year not reported.	Fats/other	Spain: Anise (n = 1), star anise (n = 1), white pepper (n = 1), fennel (n = 1), cardamom (n = 1), clove (n = 1), coriander (n = 1), nutmeg (n = 1), allspice (n = 1), cinnamon (n = 2), vanilla (n = 1), ginger (n = 1), peppermint (n = 1), parsley (n = 1), thyme (n = 1), laurel (n = 1), garlic (n = 1), cumin (n = 1), black pepper (n = 1), mild hot pepper (n = 1), hot hot pepper (n = 1), oregano (n = 2): DF 0% Slovakia: Anise (n = 1), star anise (n = 1), white pepper (n = 1), fennel (n = 1), cardamom (n = 1), clove (n = 1), coriander (n = 1), nutmeg (n = 1), allspice (n = 1), cinnamon (n = 1), vanilla (n = 1), ginger (n = 1): DF 0% (LOD = 0.013 ng/g; LOQ = 0.039 ng/g)
Zafeiraki et al. (2016a)	Greece, The Netherlands Home and commercially-produced eggs were collected from different regions in the Netherlands and Greece in August 2013–August 2014. Home-produced eggs were voluntarily provided, and	Fats/other	**Domestic eggs:** **The Netherlands: n = 73, DF 7%, median (range) = 1.1 (<0.05–5.2) ng/g**

Study	Location and Source	Food Types	Results
	commercial eggs were purchased from supermarkets. The yolks of the same sample of eggs were pooled, homogenized, and then analyzed.		Greece: n = 45, DF 0%, median (range) = <0.5 (<0.5) ng/g Commercial eggs: The Netherlands: n = 22, DF 0% Greece: n = 31, DF 0% (LOD = 0.15 ng/g; LOQ = 0.5 ng/g) *Median calculated only on the concentrations above LOQ
Zafeiraki et al. (2016b)	The Netherlands Samples purchased from local markets and slaughterhouses in the Netherlands in 2014. Samples included liver samples of horse, sheep, bovine, pig, and chicken.	Meat	Horse: n = 19, DF 0% Sheep: n = 18, DF 0% Bovine: n = 22, DF 0% Pig: n = 20, DF 0% Chicken: n = 20, DF 0% (LOQ = 0.5 ng/g ww)
D'Hollander et al. (2015)	Czech Republic, Belgium, Norway, Italy The Czech Republic, Belgium, Norway, and Italy were selected to represent eastern, western, northern, and southern Europe. Sampling took place between spring and summer 2010 as part of the PERFOOD study. Individual items were randomly selected in three national retail stores covering different brands or countries of origin. Of each item, three to ten single samples were combined to create a pooled sample. The food items examined were: Cereals: rice, wheat (white), oats, rye Sweets: sugar (beet), sugar (cane), honey Fruits – berries: strawberries Fruits – citrus fruit: oranges, tangerines, lemons, grapefruits Fruits – pipe and stone fruit: apples, pears, peaches, plums Fruits – others and exotic fruit: melons, grape, bananas	Grain; fruits; fats/other	Czech Republic: Cereals: wheat (white), oats, rye: n = 1 each, point = <0.010 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.004 ng/g Fruits – berries: strawberries: n = 1, point = <0.004 ng/g Fruits – citrus fruit: oranges, tangerines: n = 1, point = <0.004 ng/g Fruits – pipe and stone fruit: apples, pears, peaches: n = 1, point = <0.004 ng/g Fruits – others and exotic fruit: melons: n = 1, point = <0.004 ng/g Miscellaneous: rock salt: n = 1, point = <0.004 ng/g **Italy:** Cereals:

Study	Location and Source	Food Types	Results
	Miscellaneous: “rock” salt, “marine” salt		Rice, maize: n = 1 each, point = <0.004 ng/g Wheat (white): n = 1, point = <0.010 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.004 ng/g Fruits – citrus fruit: lemons: n = 1, point = <0.004 ng/g Fruits – pipe and stone fruit: **Apples: n = 1, point = 0.015 ng/g** Pears: n = 1, point = <0.004 ng/g **Peaches: n = 1, point = 0.016 ng/g** Plums: n = 1, point = <0.004 ng/g Fruits – others and exotic fruit: Grapes: n = 1, point = <0.004 ng/g Bananas: n = 1, point = 0.008 ng/g Miscellaneous: marine salt: n = 1, point = <0.004 ng/g **Norway:** Cereals: wheat (white): n = 1, point = <0.010 ng/g Sweets: sugar (cane), honey: n = 1 each, point = <0.004 ng/g Fruits – berries: strawberries: n = 1, point = <0.004 ng/g Fruits – citrus fruit: Oranges: n = 1, point = <0.004 ng/g **Grapefruits: n = 1, point = 0.0189 ng/g** Fruits – pipe and stone fruit: apples, pears: n = 1 each, point = <0.004 ng/g

Study	Location and Source	Food Types	Results
			Fruits – others and exotic fruit: melons: n = 1, point = 0.0038 ng/g Miscellaneous: rock salt: n = 1, point = <0.004 ng/g **Belgium:** Cereals: rice, wheat (white), wheat (dark), oats: n = 1 each, point = <0.004 ng/g Sweets: sugar (beet), honey: n = 1 each, point = <0.004 ng/g **Fruits – berries: strawberries: n = 1, point = 0.123 ng/g** Fruits – citrus fruit: oranges, lemons: n = 1 each, point = <0.004 ng/g Fruits – pipe and stone fruit: **Apples: n = 1, point = 0.197 ng/g** Pears: n = 1, point = <0.004 ng/g Plums: n = 1, point = <0.004 ng/g Fruits – others and exotic fruit: grapes: n = 1, point = <0.004 ng/g (LOD = 0.004 or 0.010 ng/g) *n represents number of composite samples
Gebbink et al. (2015)	Sweden Food items were purchased at two major grocery store chains in four major Swedish cities in 1999 and 2005. In 2010, sampling was limited to Uppsala city since no systematic geographical differences in food contamination was observed in the two earlier market basket studies. The food items were selected based on Swedish food and production statistics and were not cooked before analysis. The food items were divided into 12 groups and homogenates for each food group were prepared by mixing food items proportionally according to food consumption	Fruits and vegetables; meat; grains; fats/other	**1999:** **Dairy products: n = 1, point = 0.0007 ng/g** **Meat products: n = 1, point = 0.0059 ng/g** **Egg: n = 1, point = 0.034 ng/g** **Potatoes: n = 1, point = 0.0002 ng/g** **Soft drinks: n = 1, mean = 0.0002 ng/g** Fats: n = 1, point = <0.0001 ng/g

Study	Location and Source	Food Types	Results
	statistics. Results by food group were not reported for the 2005 and 2010 years. For all three sampling years, a homogenate was prepared by mixing proportional amounts of each food group according to consumption data for the respective year (includes fish samples).		Pastries: n = 1, point = <0.0001 ng/g Cereal products: n = 1, point = <0.0001 ng/g Vegetables: n = 1, point = <0.0001 ng/g Fruit: n = 1, point = <0.0001 ng/g Sugar and sweets: n = 1, point = <0.0001 ng/g **Year pool: n = 12, point = 0.0022 ng/g** **2005:** **Year pool: n = 12, point = 0.0010 ng/g** **2010:** **Year pool: n = 12, point = 0.0007 ng/g** (MLOQ = 0.0001 ng/g) *n represents number of composite samples
Carlsson et al. (2014)	Greenland (Nuuk) Meat was purchased at the local fish market and grocery shops in June 2010. All items were originally caught in the vicinity of the Nuuk area and/or along the West coast of Greenland and represented the common food items consumed by the local Inuit population.	Meat	**Seal beef: n = 2, DF[a] 100%, range = 0.2–0.6 ng/g ww** Narwhal: n = 6, DF NR, median = <LOD Whale beef: n = 8, DF NR, median = <LOD (LOD = 0.014–0.224 ng/g for PFAS)
Pérez et al. (2014)	Serbia (Belgrade and Novi Sad), Spain (Barcelona, Girona, and Madrid) Between September 2011 and February 2013, samples were purchased from different supermarkets and retail stores in representative cites around the world, including cities in Serbia and Spain in Europe. Foodstuffs were grouped into the following categories: cereals; pulses and starchy roots; tree-nuts, oil crops, and vegetable oils; vegetables and fruits; meat and meat products; milk, animal fats, dairy products, and eggs; and other such as candies and coffee.	Grains; fruits and vegetables; fats/other; meat; dairy	Serbia: Cereals (n = 4); pulses and starchy roots (n = 1); tree-nuts, oil crops and vegetable oils (n = 3); vegetables and fruits (n = 6); meat and meat products (n = 3); milk, animal fats, and dairy products (n = 7); and coffee (n = 1): DF 0% Spain: Cereals (n = 5); pulses and starchy roots (n = 2); tree-nuts, oil crops and vegetable oils (n = 10); vegetables and fruits (n = 9); meat and meat products (n = 6); milk, animal fats, dairy products, and eggs (n = 22); and other such as candies or coffee (n = 2): DF 0%

Study	Location and Source	Food Types	Results
			(Cereals: MLOD, MLOQ = 0.100, 0.333 ng/g; Juice MLOD, MLOQ = 0.157, 0.522 ng/g; Milk: MLOD, MLOQ = 0.118, 0.393 ng/g; Olive oil: MLOD, MLOQ = 0.051, 0.161 ng/g; Meat: MLOD, MLOQ = 0.121, 0.365 ng/g) *Artichoke was reported as one of the five cereal samples in Spain – unclear if this was a typo/error
Eschauzier et al. (2013)	The Netherlands (Amsterdam) Brewed coffee samples (n = 12) from different coffee machines were collected from all over the city. Coffee beans from four of these locations were collected to manually brew coffee. Post-mixed cola was collected (n = 4) together with corresponding tap water and an additional three cola samples from different parts of town. Sampling was conducted between February and April 2011 at various locations (cafés, universities, and supermarkets).	Fats/other	Post-mixed cola: n = 6, DF NR, mean = <0.63 ng/L Brewed coffee from coffee machines: n = 12, analyte peak areas could not be quantified due to strong matrix effects Manually brewed coffee: n = 4, analyte peak areas could not be quantified due to strong matrix effects (LOQ = 0.63 ng/L for cola; 0.95 ng/L for coffee)
Herzke et al. (2013)	Belgium, Czech Republic, Italy, Norway The Czech Republic, Belgium, Norway, and Italy were selected to represent eastern, western, northern, and southern Europe. Sampling took place between spring 2010 and 2011 as part of the PERFOOD study. Individual items were randomly selected in three national retail stores covering different brands or countries of origin. Of each item, three to ten single samples were combined to create one pooled sample per country. The following items were sampled: Root vegetables: carrots Bulb vegetables: onions Fruiting vegetables: tomatoes, courgettes, cucumbers, aubergine, peppers Brassica vegetables: cauliflower, cabbage, broccoli	Vegetables	**Belgium: n = 21, DF NR, mean = 0.00032 ng/g fw** Czech Republic: n = 16, DF 0% Italy: n = 15, DF 0% Norway: n = 17, DF 0% (MQL = 0.002–0.050 ng/g) *n represents number of composite samples *Values below the MQL were substituted with the MQL value

Study	Location and Source	Food Types	Results
	Leaf vegetables: lettuce and other salads, spinaches, chicory, pre-packed lettuce mix, pre-packed and minced frozen spinach Stem vegetables: asparagus, celery, fennel, cultivated mushrooms Starchy root tubers: potatoes, prepacked ready-to-cook pommes frites Legumes, beans, dried: peas, beans		
Hlouskova et al. (2013)	Belgium, Czech Republic, Italy, Norway Food products were randomly purchased in several nationwide supermarkets in four European regions during summer 2010. Within the sampling campaign, the collection of at least one food item per subcategory (meat, fish, hen eggs, milk and dairy products, and butter) in all four countries was acquired. Food items within each subcategory included the following: Meat: beef, canned pork meat, poultry, pork, pig/bovine liver, rabbit, and/or sheep/lamb Fish: farmed freshwater fish, farmed marine fish, and/or seafood) Hen eggs Milk and dairy products: ultra-high temperature whole cow milk, ultra-high temperature skimmed cow milk, cheese (yellow, Gouda/Edamer, etc.), and butter Samples were pooled within a respective food category but not across food groups.	Pooled milk/dairy products, meat, fish, hen eggs	**n = 50, DF 7%, mean, median (range) = 0.0335, 0.0264 (0.00485–0.0763) ng/g** (MQL = 0.002 ng/g for fish and seafood, meat, hen eggs, and cheese; 0.001 ng/mL for milk, and 0.006 ng/g for butter) *n represents number of pooled samples *Results not reported for individual food groups
Domingo et al. (2012)	Spain (Catalonia) Foods purchased from 4 shops/stores of each of the 12 representative cities of Catalonia (Barcelona, l'Hospitalet de Llobregat, Vilanova I la Geltrú, Mataŕr, Sabadell, Terrassa, Girona, Tarragona, Reus, Tortosa, Lleida and Manresa) in September 2011. Shops/stores included local markets, small stores,	Meat; fruits and vegetables; dairy; grains; fats/other	**Meat and meat products: n = 2, DF NR, 0.0032 ng/g fw** **Vegetables: n = 2, DF NR, 0.0045 ng/g fw** Tubers: n = 2, DF NR, mean = <0.0019 ng/g fw Fruits: n = 2, DF NR, mean = <0.0019 ng/g fw Eggs: n = 2, DF NR, mean = <0.002 ng/g fw Milk: n = 2, DF NR, mean = <0.0026 ng/g fw

Study	Location and Source	Food Types	Results
	supermarkets, and big grocery stores. Analyzed samples included 40 items: Meat and meat products: veal steak, loin of pork, chicken breast, steak of lamb, boiled ham, "Frankfurt"-type sausage, cured ham Vegetables and tubers: lettuce, tomato, potato, carrot Fresh fruits: apple, orange, banana Milk and dairy products: whole and semi-skimmed milk, yogurt, cheese I – low fat, cheese II – medium fat, cheese III – extra fat Cereals: French bread, pasta Pulses: lentils Industrial bakery: cookies Eggs: hen eggs Oils and fats: olive oil For each food item, two composite samples were prepared for analysis, where each composite sample consisted of 24 individual units. Only edible parts of each food item were included in the composites.		Dairy products: n = 2, DF NR, mean = <0.0011 ng/g fw Cereals: n = 2, DF NR, mean = <0.0006 ng/g fw Pulses: n = 2, DF NR, mean = <0.0013 ng/g fw Oils: n = 2, DF NR, mean = <0.0007 ng/g fw Industrial bakery: n = 2, DF NR, mean = <0.00056 ng/g fw (LOD not reported) *n represents number of composite samples
Vestergren et al. (2012)	Sweden (Malmoe, Gothenburg, Uppsala, Sundsvall) Purchasing locations of the two largest retail chains in Sweden were selected in each of four major Swedish cities. All purchases were made in spring/summer of 1999, 2005, and 2010. In 2010, the study was limited to the largest retail chains in Uppsala located in close vicinity to Stockholm. An equal amount of each food group from each of the four cities was combined into one sample pool to provide a representative sample for the Swedish urban population.	Dairy; meat; grains; fruits and vegetables; fats/other	**Meat products:** **1999: n = 1, point = 0.0085 ng/g** **2005: n = 1, point = 0.0051 ng/g** **2010: n = 1, point = 0.0045 ng/g** (MDL = 0.0019 ng/g; MQL = 0.0058 ng/g) **Egg:** **1999: n = 1, point = 0.039 ng/g** 2005: n = 1, point = <MDL **2010: n = 1, point = 0.0025 ng/g**

Study	Location and Source	Food Types	Results
			(MDL = 0.0019 ng/g; MQL = 0.0058 ng/g) Dairy products: 1999: n = 1, point = <MDL 2005: n = 1, point = <MDL 2010: n = 1, point = 0.0010 ng/g (estimated) (MDL = 0.0027 ng/g; MQL = 0.0080 ng/g) Fats: 1999: n = 1, point = <MDL 2005: n = 1, point = 0.0009 ng/g (estimated) 2010: n = 1, point = <MDL (MDL = 0.0023 ng/g; MQL = 0.0070 ng/g) Pastries: 1999: n = 1, point = 0.0012 ng/g (estimated) 2005: n = 1, point = 0.0013 ng/g (estimated) 2010: n = 1, point = <MDL (MDL = 0.0010 ng/g; MQL = 0.0030 ng/g) Vegetables: 1999: n = 1, point = 0.0012 ng/g (estimated) 2005: n = 1, point = 0.0010 ng/g (estimated) 2010: n = 1, point = 0.0012 ng/g (estimated) (MDL = 0.0010 ng/g; MQL = 0.0029 ng/g) Sugar and sweets: 1999: n = 1, point = <MDL 2005: n = 1, point = 0.0013 ng/g (estimated) 2010: n = 1, point = 0.0015 ng/g (estimated)

Study	Location and Source	Food Types	Results
			(MDL = 0.0010 ng/g; MQL = 0.0030 ng/g) Soft drinks, lemonade: 1999: n = 1, point = 0.0007 ng/g (estimated) 2005: n = 1, point = <MDL 2010: n = 1, point = <MDL (MDL = 0.0005 ng/g; MQL = 0.0015 ng/g) Cereal products: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0020 ng/g; MQL = 0.0059 ng/g) Fruit: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0010 ng/g; MQL = 0.0030 ng/g) Potatoes: 1999, 2005, 2010: n = 1 each year, point = <MDL for each year (MDL = 0.0010 ng/g; MQL = 0.0030 ng/g) *n represents number of composite samples *Paper reported that estimated concentrations are between MDL and MQL; however, there are some instances when the estimated concentrations are <MDL

Study	Location and Source	Food Types	Results
Noorlander et al. (2011)	The Netherlands Food products randomly purchased from several Dutch retail stores with nationwide coverage in November 2009. Food samples were ground, homogenized, and pooled for analysis. Food items within each subcategory included the following: Flour: whole wheat flour, flour Pork: sausage, slice of bacon, pork chop, bacon, minced meat rolled in bacon Eggs: chicken eggs Bakery products: cake, almond paste cake, biscuits, brown spiced biscuit, pie Vegetables/fruit: apple, orange, grape, banana, onion, carrot, beet, chicory or leek, tomato, cucumber, paprika, mushroom, cauliflower, broccoli, white cabbage, red cabbage, brussel sprout, spinach, endive, lettuce, French beans Cheese: gouda cheese, edammer cheese, cheese (>48% fat, less salt), cheese (>30% fat), brie cheese Beef: ground beef, beefburger, stewing steak, braising steak, minced steak Chicken/poultry: chicken leg, quarter chicken, chicken filet, chicken burger, collared chicken Butter: butter salt-free, salted, low-fat Milk: half cream milk Vegetable oil: margarine (solid/fluid), low-fat margarine, frying fat (vegetable), frying oil (vegetable), sunflower oil Industrial oil: low-fat margarine, frying fat (industrial), frying oil (industrial)	Meat; dairy; fruits and vegetables; grains; fats/other	**Butter: n = 1, point = 0.016 ng/g** **Chicken/poultry: n = 1, point = 0.003 ng/g** **Bakery products: n = 1, point = 0.006 ng/g** **Flour: point = n = 1, 0.018 ng/g** **Industrial oil: n = 1, point = 0.007 ng/g** Cheese: n = 1, point = <0.025 ng/g Milk: n = 1, point = <0.002 ng/g Eggs: n = 1, point = <0.006 ng/g Pork: n = 1, point = <0.005 ng/g Beef: n = 1, point = <0.004 ng/g Vegetables/fruit: n = 1, point = <0.012 ng/g Vegetable oil: n = 1, point = <0.002 ng/g (LOD not reported) *n represents number of composite samples

Study	Location and Source	Food Types	Results
Jogsten et al. (2009)	Spain (Catalonia) Food samples purchased from local markets, large supermarkets, and grocery stores from two different areas of Tarragona Province, Catalonia in January and February 2008. The cities of Tarragona and Reus were sampled in the northern area and L'Ametlla de Mar and Tortosa in the southern area. For each food item, two composite samples were analyzed (one composite for the northern area and one for the southern area). Each composite was formed of a minimum of six individual sub-samples of the same product.	Fruits and vegetables; meat; fats/other	Raw veal: n = 2, DF NR, mean = <0.003 ng/g Grilled veal: n = 2, DF NR, mean = <0.001 ng/g Fried veal: n = 2, DF NR, mean = <0.003 ng/g Raw pork: n = 2, DF NR, mean = <0.001 ng/g Grilled pork: n = 2, DF NR, mean = <0.001 ng/g Fried pork: n = 2, DF NR, mean = <0.002 ng/g Raw chicken: n = 2, DF NR, mean = <0.001 ng/g Grilled chicken: n = 2, DF NR, mean = <0.001 ng/g Fried chicken: n = 2, DF NR, mean = <0.001 ng/g Fried chicken nuggets (packaged): n = 2, DF NR, mean = <0.005 ng/g Black pudding: n = 2, DF NR, mean = <0.008 ng/g Lamb liver: n = 2, DF NR, mean = <0.250 ng/g Pate of pork liver (packaged): n = 2, DF NR, mean = <0.088 ng/g Foie gras of duck (packaged): n = 2, DF NR, mean = <0.043 ng/g "Frankfurt" sausages (packaged): n = 2, DF NR, mean = <0.006 ng/g Lettuce: n = 2, DF NR, mean = <0.001 ng/g Lettuce (packaged): n = 2, DF NR, mean = <0.003 ng/g Common salt (packaged): n = 2, DF NR, mean = <0.010 ng/g (LOD = 0.001 ng/g) *n represents number of composite samples *Values of ND were replaced with ½×LOD
Ericson et al. (2008a)	Spain Food samples purchased from local markets, large supermarkets, and grocery stores within Tarragona County in July 2006. Food samples were randomly purchased with origin source not specified. Each of the food samples were duplicated and combined to analyze a composite sample. Composite samples included the following:	Meat; dairy; fruits and vegetables; grains; fats/other	Vegetables (n = 2), pulses (n = 2), cereals (n = 2), pork (n = 2), chicken (n = 2), veal (n = 2), lamb (n = 2), eggs (n = 2), dairy products (n = 2), whole milk (n = 2), semi-skimmed milk (n = 2), fruits (n = 2), margarine (n = 2), oil (n = 2): DF 0% (LOD = 0.001–0.65 ng/g fw) *n represents number of composite samples

Study	Location and Source	Food Types	Results
	Vegetables: lettuce, tomato, green bean, spinach Pulses: lentils, beans, chickpeas Cereals: rice, spaghetti, bread Pork: sausage, hot dogs, steak, hamburger, ham Chicken: breast, thighs, sausage Veal: steak, hamburger Lamb: steak Dairy products: three different kinds of cheese, yogurt, "petit-Swiss" creamy yogurt, cream caramel, custard Fruits: apple, orange, pear, banana Oil: olive oil, sunflower oil, corn oil Fats: margarine Eggs		
Papadopoulou et al. (2017)	Norway Participants of the A-TEAM project collected a duplicate portion of all consumed foods and drinks, prepared as for consumption, over two consecutive weekdays. Only the samples collected in the first day were analyzed. Sampling year not reported.	Solid foods: cereals and cereal products, dairy products (not milk), fish and seafood, meat and meat products, sugar and sugar products, fats and oils, vegetables and nuts, fruits, salty snacks, eggs, potatoes; liquid foods: coffee, tea and cocoa, milk, water, alcoholic beverages, soft drinks	**Solid foods:** **n = 61, DF 66%, median (range) = 0.00088 (0–0.1) ng/g** (LOQ = 0.00011 ng/g) **Liquid foods:** **n = 61, DF 8%, median (range) = 0 (0–0.002) ng/g** (LOQ = 0.00002 ng/g) *Concentrations <LOQ were replaced by 0
Dellatte et al. (2013)	Italy (Genoa, Brescia, Ferrara, Perugia, Portici) Ready-to-eat meals were collected at nursery and primary school canteens as they were delivered to children aged 3–10 years during the spring of 2011. One canteen was selected from each city, except for Genoa which was represented by two canteens because in one, the internal school regulation	Food composite meals	n = 6, DF 0% (LOQ = 0.006 ng/g) *n represents number of composite samples

Study	Location and Source	Food Types	Results
	forbade the use of anti-stick cookware. For each canteen, lunch meals related to five school days (from Monday to Friday) were weighed, pooled, and homogenized. Beverages were not included in the composites.		
Fromme et al. (2007)	Germany (Bavaria) Thirty-one participants provided 24-hour duplicate diet samples over seven consecutive days, including one weekend, in April–October 2005. One participant only provided samples over four consecutive days. All diet samples were a normal mixed diet; no participants were on a special diet or vegetarian.	Duplicate diet samples	n = 214, DF[a] 3%, range = 0.05–3.03 ng/g fw (LOD = 0.1 ng/g fw) *Values <LOD were assigned ½ LOD
Ghelli et al. (2019)	Italy Egg samples were collected from commercial laying hen farms in 2017. Sampling was based on geographical origin of the eggs and rearing system (e.g., organic, aviary system, battery cage and barn). A total of 132 eggs were collected and eggs were boiled. Four pools (containing three homogenized yolks) were created for each of the following groups (geographical origin, rearing system), for a total of 44 samples analyzed: Group A: Pavia, barn Group B: Verona, organic Group C: Forlì-Cesena, battery cage Group D: Bologna, barn Group E: Forlì-Cesena, battery cage Group F: Ravenna, aviary system Group G: Ravenna, aviary system Group H: Bologna, organic Group I: Romagna, battery cage Group L: Romagna, organic Group M: Romagna, barn	Fats/other	Group A: n = 4, Dfa 50%, range = ND–0.4 ng/g Group B: n = 4, Dfa 25%, range = ND–traces Group C: n = 4, DF 0% Group D: n = 4, DF 0% Group E: n = 4, DF 0% Group F: n = 4, DF 0% Group G: n = 4, DF 0% Group H: n = 4, DF 0% Group I: n = 4, Dfa 25%, range = ND–traces Group L: n = 4, DF 0% Group M: n = 4, DF 0% (LOD = 0.1 ng/g for all PFAS; LOQ = 0.25 ng/g for all PFAS) *Traces defined as value between LOD and LOQ
Johansson et al. (2014)	Sweden Eggs from 20 to 25 producers were collected each year from 1999 to 2010 within the Swedish National Food Agency's official food control program. Each	Dairy; fats/other	**Hen's eggs:**

Study	Location and Source	Food Types	Results
	sample consisted of a pool of 10–12 eggs from one producer. The pooled samples comprised eggs from both conventional and organic production. Information on the number of organic eggs sampled was not available. Fresh milk was sampled from the tanks of milk transport vehicles between 1999 and 2009 as part of the food control program. The tanks generally contained milk from ten dairy farms. In 2010, milk samples were taken from the milk storage tanks on individual dairy farms. Between 10–25 milk samples were collected each year. The milk samples were extracted in two different batches.		**Total: n = 36, DF[a] 61%, range = <0.010–0.128 ng/g fw** **1999: n = 3, DF[a] 100%, mean[a] (range) = 0.038 (0.019–0.072) ng/g fw** **2000: n = 3, DF[a] 100%, mean[a] (range) = 0.038 (0.016–0.069) ng/g fw** **2001: n = 3, DF[a] 100%, mean[a] (range) = 0.033 (0.015–0.051) ng/g fw** **2002: n = 3, DF[a] 67%, range = <0.010–0.128 ng/g fw** **2003: n = 3, DF[a] 67%, range = <0.010–0.054 ng/g fw** **2004: n = 3, DF[a] 33%, range = <0.010–0.011 ng/g fw** **2005: n = 3, DF[a] 100%, mean[a] (range) = 0.015 (0.011–0.018) ng/g fw** 2006: n = 3, DF 0% **2007: n = 3, DF[a] 33%, range = <0.010–0.020 ng/g fw** **2008: n = 3, DF[a] 33%, range = <0.010–0.011 ng/g fw** **2009: n = 3, DF[a] 33%, range = <0.010–0.012 ng/g fw** **2010: n = 3, DF[a] 67%, range = <0.010–0.020 ng/g fw** (MDL = 0.010 ng/g fw; MQL = 0.033 ng/g fw) **Cow's milk (1st batch):** **Total: n = 18, DF[a] 22%, range = <0.0010–0.0011 ng/g fw** 1999: n = 1, DF 0%

Study	Location and Source	Food Types	Results
			2000: n = 2, DF[a] 50%, range = <0.0010–0.0010 ng/g fw 2001: n = 2, DF 0% **2002: n = 2, DF[a] 50%, range = <0.0010–0.0011 ng/g fw** 2003: n = 1, DF 0% 2004: n = 2, DF 0% 2005: n = 1, DF 0% **2006: n = 1, point = 0.0010 ng/g fw** 2007: n = 1, DF 0% **2008: n = 2, DF[a] 50%, range = <0.0010–0.0011 ng/g fw** 2009: n = 1, DF 0% 2010: n = 2 DF 0% (MDL = 0.0010 ng/g fw; MQL = 0.0033 ng/g fw) **Cow's milk (2nd batch):** **Total: n = 18, DF 0%** (MDL = 0.0023 ng/g fw ; MQL = 0.0063 ng/g fw)
Eriksson et al. (2013)	Denmark (Faroe Islands) Locally produced food items sampled in 2011–2012. Packaged dairy products were supplied by Faroe Islands, Meginfelag Búnaðarmanna – dairy products included samples of milk, low fat (0.5%), semi-skimmed (1.5%), yoghurt with banana and pear (3.4% fat), low fat (0.9%) plain yoghurt, and créme fraiche (18% fat). Yoghurt with banana and pear was sampled from two production batches, and the low fat plain yoghurt and créme fraiche was sampled from one production batch. Potatoes were sampled from two different farms.	Dairy; fruits and vegetables	Milk (n = 6), yogurt (n = 3), crème fraiche (n = 1), potatoes (n = 2): DF 0% (LOD = 0.0058 ng/L for milk; LOD = 0.0017 ng/g for dairy; LOD = 0.0016 ng/g for potato)
Vestergren et al. (2013)	Sweden (Kårsta)	Meat; dairy	Dairy farm cow's milk: n = 6, DF 0%

Study	Location and Source	Food Types	Results
	Study was conducted at a dairy cattle farm that was selected to represent a background contaminated agricultural area with no known point sources of PFAS in the proximity. The farm had no history of sewage sludge application to the pasture land. Milk samples were collected between November 2010 and April 2011 from a milk tank, where milk from the entire farm is stored after milking. Muscle, liver, and whole blood samples were obtained from five individual cows from the slaughterhouse on two different occasions (April and June 2011).		Cow liver: n = 5, DF 0% Cow blood: n = 5, DF 0% Cow muscle: n = 5, DF 0% (MDL not reported)
Falandysz et al. (2006)	Poland Eider duck samples were collected from the Gulf of Gdañsk in the Baltic Sea (south coast of Poland) in February 2003.	Meat	**n = 16, DF NR, mean, median (range) = 1.1, 1.1 (0.4–2.9) ng/mL** (LOD not reported) *Values reported for animal whole blood samples
Lankova et al. (2013)	Czech Republic Breastmilk samples were obtained from 50 women living in the Olomouc region from April to August 2010. The age of participating mothers ranged from 20 to 43 years. Six types of infant formula from the Czech retail market were also examined: one powdered formula for infants, two formulas for toddlers, one formula for babies with lactose intolerance, one formula for premature babies, and one soya-based formula for babies with non-milk diets. Sampling year not provided.	Fats/other; breastmilk	**Breastmilk:** **n = 50, DF (frequency of quantification) 8%, range = <0.006–0.022 ng/mL** (LOQ = 0.006 ng/mL) Infant formula: n = 6, DF (frequency of quantification) 0% (LOQ = 0.005 ng/g)
Abdallah et al. (2020)	Ireland (Dublin) Breastmilk samples obtained from mothers recruited from breastfeeding clinics at two Irish maternity hospitals. Mothers provided samples between 3 and 8 weeks postpartum. Mothers were up to 41 years of age, primiparas, in good health, and exclusively feeding one infant. Sampling year not reported.	Breastmilk	**n = 16, DF 31%, mean, median (range) = <0.04, <0.04 (<0.04–0.087 ng/mL)** (LOQ = 0.04 ng/mL) *Values <LOQ assumed to equal DF × LOQ
Nyberg et al. (2018)	Sweden (Gothenburg, Stockholm) Breastmilk samples were collected between two weeks and three months after delivery from healthy	Breastmilk	**L-PFHxS:**

Study	Location and Source	Food Types	Results
	native Swedish mothers, who were predominately non-smokers and primiparous. There were a total of 20 pooled samples analyzed from Stockholm (1972–2016), containing 9–116 individual samples per pool, and 11 pooled samples from Gothenburg (2007–2015), containing 5–11 individuals per pool. In addition, samples collected in 2012 (16 from Gothenburg and 20 from Stockholm) and in 2016 (10 from Stockholm) were analyzed individually.		**Stockholm (pooled): n = 20, DF[a] 90%, range = <0.0008–0.021 ng/mL** **Gothenburg (pooled): n = 11, DF[a] 91%, range = <0.0008–0.012 ng/mL** **Stockholm (2012, individual): n = 20, DF[a] 95%, range = <0.0008–0.025 ng/mL** **Gothenburg (2012, individual): n = 16, DF[a] 94%, range = <0.0008–0.014 ng/mL** **Stockholm (2016, individual): n = 10, DF[a] 80%, range = <0.0008–0.013 ng/mL** **Br-PFHxS:** **Stockholm (pooled): n = 20, DF[a] 25%, range = <0.0008–0.0029 ng/mL** **Gothenburg (pooled): n = 11, DF[a] 18%, range = <0.0008–0.004 ng/mL** Stockholm (2012, individual): n = 20, DF 0% **Gothenburg (2012, individual): n = 16, DF[a] 6%, range = <0.0008–0.0012 ng/mL** **Stockholm (2016, individual): n = 10, DF[a] 30%, range = <0.0008–0.004 ng/mL** (MDL = 0.0008 ng/mL)
Cariou et al. (2015)	France (Toulouse) Breastmilk samples obtained from female volunteers hospitalized between June 2010 and January 2013 for planned caesarean delivery. Samples were collected between the fourth and fifth day after delivery.	Breastmilk	**n = 61, DF 15%, mean, median (range) = 0.026, <LOD (<LOD–0.217) ng/mL** (LOD = 0.01–0.03 ng/mL; LOQ = 0.03 ng/mL) * Individual LOD values and semi-quantified values (when below LOQ) taken into account for mean calculation
Antignac et al. (2013)	France (Seine-Saint Denis, Ardèche, Isère, Loire, Savoie counties) Breastmilk samples collected from mothers participating in the ELFE pilot study. Sampling year not reported, though all mothers gave birth in October 2007. Mothers were contacted by phone one month after leaving the maternity and provided	Breastmilk	**n = 48, DF 100%, mean, median (range) = 0.049, 0.050 (0.040–0.066) ng/mL** (LOD not reported)

Study	Location and Source	Food Types	Results
	with instructions on breastmilk collection. Milk samples could be collected during several lactation sessions. On average, 15 aliquot samples of 10 mL were collected for each participant and pooled into one sample for analysis.		
Croes et al. (2012)	Belgium (Flanders) Breastfeeding mothers were recruited from 9 maternities in 24 rural communities in East and West Flanders and Flemish Brabant in May 2009 – June 2010. Breastmilk samples were collected between two and eight weeks after delivery and a subset was analyzed for perfluorinated compounds.	Breastmilk	**n = 40, DF 20%, median (10^{th}–90^{th} percentile) = <LOQ (<LOQ–0.02) ng/mL** (LOQ = 0.01 ng/mL) *For all calculations, values <LOQ were treated as ½ LOQ
Sundström et al. (2011)	Sweden (Stockholm) Breastmilk samples were collected from healthy native Swedish mothers by the Mothers' Milk Center between the second and twelfth week after delivery. The majority of mothers (76%) were nursing their first infant. Samples were collected between 1972–2008 and pooled for each year.	Breastmilk	Overall: n = 684, DF NR, range = <0.005–0.028 ng/mL 1972: n = 75, DF NR, mean = <0.005 ng/mL 1976: n = 78, DF NR, mean = <0.005 ng/mL 1980: n = 116, DF NR, mean = 0.006 ng/mL 1984/85: n = 102, DF NR, mean = 0.006 ng/mL 1988: n = 20, DF NR, mean = 0.016 ng/mL 1990: n = 20, DF NR, mean = 0.010 ng/mL 1992: n = 20, DF NR, mean = 0.011 ng/mL 1994: n = 20, DF NR, mean = 0.015 ng/mL 1995: n = 20, DF NR, mean = 0.028 ng/mL 1996: n = 20, DF NR, mean = 0.016 ng/mL 1997: n = 20, DF NR, mean = 0.016 ng/mL 1998: n = 20, DF NR, mean = 0.028 ng/mL 1999: n = 20, DF NR, mean = 0.023 ng/mL 2000: n = 20, DF NR, mean = 0.024 ng/mL 2001: n = 20, DF NR, mean = 0.017 ng/mL 2002: n = 20, DF NR, mean = 0.027 ng/mL 2003: n = 15, DF NR, mean = 0.025 ng/mL 2004: n = 20, DF NR, mean = 0.017 ng/mL 2007: n = 20, DF NR, mean = 0.017 ng/mL 2008: n = 18, DF NR, mean = 0.014 ng/mL (LOQ = 0.005 ng/mL)

Study	Location and Source	Food Types	Results
			*n represents number of pooled samples
Kärrman et al. (2010)	Spain (Catalonia) Breastmilk samples were collected from healthy primiparae mothers aged 30–39 years who lived in Tarragona County for at least the last five years. Babies were aged 41–60 days when milk samples were collected in 2007.	Breastmilk	**n = 10, DF 100%, mean, median (range) = 0.04, 0.04 (0.02–0.11) ng/mL** (LOQ = 0.01 ng/mL)
Kärrman et al. (2007)	Sweden (Uppsala, Göteborg, Lund, Lycksele) Individual breastmilk samples from 12 women in Uppsala, Sweden were collected in 2004. Composite samples were created from breastmilk samples collected from 25–90 women each year between 1996 and 2004 and pooled into an annual composite sample. Donors originated from four regions in Sweden (Uppsala 1996 -2000, 2002; Göteborg 2001; Lund 2003; Lycksele 2003-2004). All samples were collected from primiparous women (19–41 years old) during the third week after delivery.	Breastmilk	**Individual samples:** **2004: n = 12, DF[a] 100%, mean, median (range) = 0.085, 0.070 (0.031–0.172) ng/mL** **Composite samples:** **1996: n = 1, point = 0.037 ng/mL** **1997: n = 1, point = 0.030 ng/mL** **1998: n = 1, point = 0.040 ng/mL** **1999: n = 1, point = 0.044 ng/mL** **2000: n = 1, point = 0.028 ng/mL** **2001: n = 1, point = 0.028 ng/mL** **2002: n = 1, point = 0.051 ng/mL** **2003: n = 1, point = 0.025 ng/mL** **2003–2004: n = 1, point = 0.016 ng/mL** *n represents number of composite samples (DL = 0.01 ng/mL)
Beser et al. (2019)	Spain (Valencian region) Breastmilk samples were collected from 14 Spanish women (aged 30–39 years) living in the Valencian region and recruited by the perinatology group of the Health Research Institute La Fe in Valencia. Milk samples were collected at different stages after birth during 2015.	Breastmilk	n = 20, DF[a] 30% *Six samples were >MDL but <MQL – these values were not reported (MDL = 0.004 ng/mL; LOQ = 0.133 ng/mL)
Pratt et al. (2013)	Ireland	Breastmilk	n = 11, DF 0% (LOD = 0.5–5 ng/mL for all PFAS)

Study	Location and Source	Food Types	Results
	Pooled breastmilk samples were collected from 109 first-time mothers at four centers across Ireland. Sampling year not reported.		*n represents number of pooled samples

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; GCA = groundwater contamination area; LOD = limit of detection; LOQ = limit of quantitation; MAMA = Methods Advancement for Milk Analysis; MDL = method detection limit; ND = not detected; NR = not reported; RL = reporting limit; ww = wet weight; WWTP = wastewater treatment plant.

Bold indicates detected levels of PFHxS in food.

[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

D.3.2. Food Contact Materials

No studies were identified that evaluated the occurrence of PFHxS in food packaging or food contact materials (FCMs) purchased in the United States. In an analysis performed at the Department of Food Analysis and Nutrition of the University of Chemistry and Technology in Prague, Czech Republic, PFHxS was not detected in 42 samples of disposable food packaging and tableware purchased from six different European countries between May and December 2020 (LOQ = 1.7 mg/kg) (Straková et al., 2021). The five additional peer-reviewed European studies identified are summarized below and in Table D-6 (Vavrouš et al., 2016; Kotthoff et al., 2015; Surma et al., 2015; Vestergren et al., 2015; Zafeiraki et al., 2014). Three of these studies reported no detection of PFHxS in FCMs while two reported detectable levels of PFHxS in FCMs. Of these two studies, PFHxS was detected in 6% of paper-based FCM samples purchased recently in Germany (at the time of the study: 2010) and also was detected in samples purchased before 2010, but in both cases the median concentration was below the LOQ of 0.5 ng/g. For FCMs purchased in Poland, PFHxS was detected in one brand of cellulose wrapping paper (0.29 pg/cm^2) but was below the LOD (0.01 pg/cm^2) or below the LOQ (0.03 pg/cm^2) in other cellulose and polyether sulfone FCMs. Additional research is needed to evaluate PFHxS in FCMs purchased in U.S. and Canada and for FCMs with different countries of origin.

Two studies reported PFHxS in food contact materials (Kotthoff et al., 2015; Surma et al., 2015). Kotthoff et al. (2015) analyzed for PFSA and PFCA compounds in random samples of recent (purchased in 2010) individual paper-based FCMs (n = 33) from local retailers in Germany. Samples were purchased from local retailers or collected by co-workers of the institute in the first until third quarter of 2010. Baking paper purchased before 2010 (sample age ranging from a few years to decades) was collected from staff of the institutes and referred to as archived samples. PFHxS was detected in 6% of recent paper-based FCM samples with a median concentration below the LOQ (0.5 ng/g). PFHxS was also detected in archived FCM samples with a median concentration below the LOQ. In Surma et al. (2015), the authors measured levels of PFHxS in three different brands of FCMs that included wrapping papers (n = 3), breakfast bags (n = 3), baking papers (n = 3), and roasting bags (n = 3); sampling year was not reported. Items were obtained from typical, commercially available food contact products in Poland. Roasting bags were made of polyether sulfone; the remaining items were made of cellulose. PFHxS was detected in one brand of wrapping paper (brand B) at 0.29 pg/cm^2, but PFHxS was below the LOD (0.01 pg/cm^2) or below the LOQ (0.03 pg/cm^2) in all other FCMs. The authors reported that the highest content of perfluorinated compounds were reported for B brand FCM. They also reported that FCMs based on cellulose contained more PFCAs than PFASs; on the other hand, FCMs based on polyether sulfone contained more PFASs than PFCAs.

The remaining three studies did not detect PFHxS in FCMs (Vavrouš et al., 2016; Vestergren et al., 2015; Zafeiraki et al., 2014). Vavrouš et al. (2016) analyzed 15 samples of paper FCM (11 with direct food contact and 4 with indirect food contact) acquired from a market in the Czech Republic, including paper packages of wheat flour (n = 2), paper bags for bakery products (n = 2), sheets of paper for food packaging in food stores (n = 2), cardboard boxes for packaging of various foodstuffs (n = 3), coated bakery release papers for oven baking at temperatures up to 220°C (n = 3), and paper filters for coffee preparation (n = 3). PFHxS was below the LOQ (0.0030 mg/kg) in all samples. In Vestergren et al. (2015), the authors analyzed a random sample of FCMs collected in November 2012, including a baking mold (n = 1), baking cover (n = 1),

paper cup (n = 1), and paper plates (n = 2) that were purchased from major retail stores in Norway but were imported from China. PFHxS was below the MDL (0.01 $\mu g/m^2$) in all samples. Finally, Zafeiraki et al. (2014) analyzed 42 samples of FCMs made of paper, paperboard, or aluminum foil randomly obtained from retailers. All products except for microwave popcorn and rice bags were manufactured in Greece. Sampled packaging materials included unused items and used items (i.e., contained food products). Beverage and ice cream cups, wrappers, and paper boxes were collected in Athens from October to December 2012 from popular Greek fast food chain restaurants, coffee shops, and multiplex cinemas. Other FCMs (muffin cups, baking papers, and microwave popcorn and rice bags) were purchased from large supermarkets. PFHxS was below the LOD (0.18 ng/g) in all samples.

Table D-6. Studies Reporting PFHxS Occurrence in Food Contact Materials

Study	Location	Site Details	Results
Europe			
Kotthoff et al. (2015)	Germany (Schmallenberg)	Thirty-three random samples of recent individual paper-based FCMs collected in the first until the third quarter of 2010 in Germany. Individual samples were bought from local retailers or collected by coworkers of the involved institutes. Sampled products spanned all quality levels from entry level to cutting edge products. The age of the samples ranged from a few years to decades. Country of origin not reported. "Archived" older samples of FCMs (baking paper purchased before 2010) were collected from the staff of the institutes. The age of these samples ranged from a few years to decade. Country of origin not reported.	Recent samples: n = 33, DF 6%, median (range) = <LOQ (<LOQ–0.6) ng/g Archived samples: n = 3, DF NR, median (range) = <LOQ (<LOQ–0.6) ng/g (LOQ = 0.5 ng/g) *Concentrations <LOQ were considered as zero *For recent samples, Table 1 reports n = 33, the Results text reports n = 36 according to the actual sampling plan, and Table 13 in Supplemental Appendix 3 reports n = 12
Surma et al. (2015)	Poland	Three different brands of FCMs (A, B, C), including wrapping papers (n = 3), breakfast bags (n = 3), baking papers (n = 3), and roasting bags (n = 3), were obtained from typical, commercially available food contact products. Sampling year and country of origin for products not reported.	Wrapping paper: Brand A: n = 1, point = ND Brand B: n = 1, point = 0.29 pg/cm^2 Brand C: n = 1, point = ND Breakfast bag: Brand A: n = 1, point = <LOQ Brand B: n = 1, point = ND Brand C: n = 1, point = ND Baking paper: Brand A: n = 1, point = ND Brand B: n = 1, point = <LOQ Brand C: n = 1, point = <LOQ Roasting bag: Brand A: n = 1, point = ND Brand B: n = 1, point = ND Brand C: n = 1, point = ND (LOD = 0.01 pg/cm^2; LOQ = 0.03 pg/cm^2)
Vavrouš et al. (2016)	Czech Republic	Real samples of paper FCM (11 with direct food contact and 4 with indirect food contact) were acquired from a market. Samples included paper packages of wheat flour (n = 2), paper bags for bakery products (n = 2), sheets of paper for food packaging in food stores (n = 2), cardboard boxes for packaging of various foodstuffs (n = 3), coated bakery release papers for oven baking at temperatures up to 220°C (n	n = 15, DF 0% (LOQ = 0.0030 mg/kg)

Study	Location	Site Details	Results
		= 3), and paper filters for coffee preparation (n = 3). Sampling year and country of origin for products not reported.	
Vestergren et al. (2015)	Norway (Tromsø, Trondheim)	Five samples of FCMs (one baking mold, two paper plates, one baking cover, and one paper cup) were purchased from major retail stores in November 2012. Sampling campaign designed to evaluate consumer products in product categories that were previously found to contain PFAS residuals and that were representative of products imported from China in large quantities. Individual products randomly selected without prior knowledge of surface treatment with PFAS. Year of manufacture not reported.	n = 5, DF[a] 0% (MDL = 0.010 µg/m^2)
Zafeiraki et al. (2014)	Greece	Forty-two samples of FCMs made of paper, paperboard, or aluminum foil were obtained randomly from retailers. Their exact composition was not stated and there was no information about perfluorochemicals used in their manufacturing process or not. Beverage and ice cream cups, wrappers, and paper boxes were collected in Athens from October to December 2012 from popular Greek fast food chain restaurants, coffee shops, and multiplex cinemas. Other FCMs (muffin cups, baking papers, and microwave popcorn and rice bags) were purchased from large supermarkets. All products except for microwave popcorn and rice bags were manufactured in Greece. Sampled packaging materials included unused items and used items (i.e., contained food products). A microwave popcorn bag was also analyzed before and after cooking.	Beverage cups: n = 8, DF 0% Ice cream cup: n = 1, DF 0% Fast food paper boxes: n = 8, DF 0% Fast food wrappers: n = 6, DF 0% Paper materials for baking: n = 2, DF 0% Microwave bags: n = 3, DF 0% Before cooking: n = 1, point = <LOD After cooking: n = 1, point = <LOD Aluminum foil bags/wrappers: n = 14, DF 0% (LOD = 0.18 ng/g; LOQ = 0.54 ng/g)

D.3.3. Consumer Products

PFHxS has been used in laboratory applications and as a raw material or a precursor for the manufacture of PFAS/perfluoroalkyl sulfonate-based products, though production of PFHxS in the United States was phased out by its major manufacturer in 2002 (NCBI, 2022a; Sigma-Aldrich, 2014; Backe et al., 2013; Buck et al., 2011; OECD, 2011). PFHxS has also been used in firefighting foam and carpet treatment solutions, and it has been used as a stain and water repellant (NCBI, 2022a; Garcia and Harbison, 2015). PFHxS has been detected in aqueous film-forming foam, aftermarket carpet protection products, chipboards, leather, membranes for apparel, treated apparel, and photoprint ink and laser ink (NCBI, 2022a; Glüge et al., 2020; Norwegian Environment Agency, 2018; Bečanová et al., 2016; Kotthoff et al., 2015; Liu et al., 2014; Backe et al., 2013; Buck et al., 2011; Herzke et al., 2009).

Two U.S.-based studies were identified that analyzed PFHxS concentrations in a range of consumer products, including children's nap mats, household carpet/fabric-care liquids, and textiles (Zheng et al., 2020; Liu et al., 2014) (Table D-7). Few U.S. studies analyzed children's products, fabric treatments, treated fabrics, sealants, and similar products, and none of the U.S. studies reviewed sampled for PFAS in other household products and articles, such as cosmetics, cleaners, paints, upholstered furniture, etc. Of the U.S. studies, the majority of the consumer products evaluated are likely used by adults (e.g., floor waxes), can come into contact with both adults and children (e.g., treated upholstery), or the user was not specified (e.g., clothing).

Zheng et al. (2020) determined the occurrence of ionic and neutral PFAS in the childcare environment (dust and nap mats). Samples of children's nap mats were collected from seven Seattle childcare centers (n = 26; 20 polyurethane foam, 6 vinyl cover samples). PFHxS was detected in 73% of nap mat samples with a mean concentration of 0.32 ng/g. Half of the analyzed mats were purchased as new products and the other half were used. The authors reported that total PFAS levels in the new versus used mats were not significantly different. Total PFAS levels in mat foam versus mat covers were also similar. Based on these results, the authors suggested that indoor air was not the major source of PFAS in mats and that PFAS in mats could be associated with the manufacturing process.

Liu et al. (2014) analyzed the occurrence of PFAS in consumer products (including commercial carpet/fabric-care liquids, household carpet/fabric-care liquids, treated apparel, treated home textile and upholstery, treated floor waxes and stone-wood sealants, membranes for apparel, and thread-sealant tapes and pastes) purchased between March 2007 and September 2011 from local retailers and online stores in the United States. PFHxS was analyzed in a subset of these consumer products, originating from the United States, England, Dominican Republic, Vietnam, and China, and was detected in one out of two commercial carpet/fabric-care liquids samples at 194 ng/g, in two out of four household carpet/fabric-care liquids and foams samples at 88.8 ng/g and 155 ng/g, in one out of two treated children's apparel samples at 1.70 ng/g (in boy's uniform pants), in one out of two treated home textile and upholstery samples at 12.1 ng/g, in one apparel membrane sample at 7.10 ng/g, and in one out of two thread-sealant tapes and pastes samples at 60.3 ng/g. PFHxS was not detected in one treated floor wax and stone/wood sealant sample. Detection limits were not reported in the study.

Beesoon et al. (2012) detected PFHxS in all cleaning formulation-treated carpet samples (n = 9) from various rooms of a Canadian household whose members had previously been identified as having disproportionately high blood serum levels of PFHxS. The Scotchgard formulation-treated carpet samples had a mean (range) of 512 (12–2,880) ng/g of PFHxS, with a level of 17 ng/g detected in one untreated, uninstalled carpet sample taken from the same house.

Of the European studies, van der Veen et al. (2020) examined the effects of weathering on PFAS content in durable water repellent clothing collected from six suppliers in Sweden. Two pieces of each of the 13 fabrics were cut. One piece was exposed to elevated ultraviolet radiation, humidity, and temperature in an aging device for 300 hours (assumed lifespan of outdoor clothing); the other was not aged. Pieces of textile, aged and not aged, were analyzed for ionic PFAS (including PFHxS) and volatile PFAS. For 10 of 13 fabrics, PFHxS was not detected before or after aging. For one fabric, PFHxS was detected before and after aging, increasing from 0.11 to 0.68 $\mu g/m^2$. For one fabric, PFHxS was detected prior to aging at 0.89 $\mu g/m^2$ but was not detected afterward. For the remaining fabric, PFHxS was not detected prior to aging but was not analyzed after aging.

Kotthoff et al. (2015) analyzed 82 samples of consumer products collected in Germany, including outdoor textiles, carpets, cleaning agents, impregnating agents, leather samples, and ski waxes. Individual samples were bought from local retailers or collected by coworkers of the involved institutes or local clubs in Germany. The age of the samples ranged from a few years to decades. PFHxS was detected in 35% of ski wax samples (n = 13) up to 9.3 ng/g and in 96% of leather samples (n = 13) up to 10.1 $\mu g/m^2$. PFHxS was not detected in cleaning agents (n = 6), wood glue (n = 1), impregnating sprays (n = 3), outdoor textiles (n = 3), carpet (n = 6), gloves (n = 3), or awning cloth (n = 1). Favreau et al. (2016) analyzed the occurrence of 41 PFASs in a wide variety of liquid products (n = 194), including impregnating agents, lubricants, cleansers, polishes, AFFFs, and other industrial products purchased from stores and supermarkets in Switzerland. PFHxS was not detected in impregnation products (n = 60), cleansers (n = 24), or polishes (n = 18). PFHxS was detected in 4% of a miscellaneous category of products (n = 23) that included foam-suppressing agents for the chromium industry, paints, ski wax, inks, and tanning substances, with a maximum concentration of 1,700 ng/g.

The remaining two European studies did not detect PFHxS in the consumer products analyzed. Vestergren et al. (2015) analyzed furniture textile, carpet, and clothing samples (n = 40) purchased from major retail stores in Tromsø and Trondheim, Norway. All samples represented materials that had been imported from China. PFHxS was not detected in any of the samples. Schultes et al. (2018) determined levels of 39 PFAS in 31 cosmetic products collected in Sweden. The study found that 16 out of 31 samples contained measurable concentrations of at least one PFAS; however, PFHxS was not detected amongst the samples.

Of the two studies where the location of purchase was not specified, Gremmel et al. (2016) determined levels of 23 PFAS in 16 new outdoor jackets. PFHxS was detected in one jacket at a concentration of 0.01 ng/g. Bečanová et al. (2016) analyzed 126 samples of (1) household equipment (textiles, floor coverings, electrical and electronic equipment (EEE), and plastics); (2) building materials (oriented strand board, other composite wood and wood, insulation materials, mounting and sealant foam, facade materials, polystyrene, air conditioner components); (3) car interior materials; and (4) wastes of electrical and electronic equipment (WEEE) for 15 target PFAS, including PFHxS. The condition (new versus used) and production year of the samples

varied; the production year ranged from 1981 to 2010. PFHxS was detected in 42%, 22%, 30%, and 14% of household equipment, building materials, car interior, and WEEE samples, respectively. The highest level was 24.5 ng/g found in a drywall sample.

Table D-7. Summary of PFHxS Consumer Product Data

Study	Location	Site Details	Results
United States			
Zheng et al. (2020)	United States (Seattle, Washington)	Children's nap mat samples (n = 26, finely cut) from seven Seattle childcare centers, including polyurethane foam (n = 20) and vinyl cover (n = 6) samples. Sampling year not reported.	n = 26, DF 73%, mean, median (range) = 0.32, 0.30 (<ND–0.73) ng/g (MDL = 0.05 ng/g)
Liu et al. (2014)	United States (unspecified)	Consumer products commonly used indoors were purchased between March 2007 and September 2011 from local retailers and online stores in the United States. A subset of samples were analyzed for PFSAs and included commercial carpet/fabric-care liquids, household carpet/fabric-care liquids and foams, treated apparel (i.e., one girl's uniform pants and one boy's uniform pants), treated home textile and upholstery (i.e., mattress pads), treated floor waxes and stone-wood sealants, membranes for apparel, and thread-sealant tapes and pastes. The subset of products originated from the United States, England, Dominican Republic, Vietnam, and China.	Commercial carpet/fabric-care liquids: n = 2, DF[a] = 50%, range = BDL–194 ng/g Household carpet/fabric-care liquids and foams: n = 4, DF[a] = 50%, range = BDL–155 ng/g Treated apparel: n = 2, DF[a] = 50%, range = BDL–1.70 ng/g Treated home textile and upholstery: n = 2, DF[a] = 50%, range = BDL–12.1 ng/g Treated floor waxes and stone-wood sealants: n = 1, DF 0% Membranes for apparel: n = 1, point = 7.10 ng/g Thread-sealant tapes and pastes: n = 2, DF[a] = 50%, range = BDL–60.3 ng/g (DL not reported)
Canada			
Beesoon et al. (2012)	Canada (Alberta)	Carpet samples (each approximately 5 cm by 2 cm) from the finished basement, family room, dining room, attic, and from the bedrooms of the parents and children currently living in a house in the Edmonton area built in 1989. Samples were collected in September 2008 prior to a major renovation where all carpets were removed. The carpets in all rooms were identical and had been installed at the same time. The house was identified after abnormally high serum levels of PFHxS were identified in a husband and wife that were enrolled as volunteer control participants in a clinical research project. Many rooms in the house had wall-to-wall carpeting installed on top of the heated floors. Since 1991, Scotchgard has been applied to the carpets in the main floor family room and occasionally to the dining	Main floor family room: n = 1, point = 2,880 ng/g Main floor dining room: n = 1, point = 890 ng/g Parents' bedroom: n = 1, point = 122 ng/g First son's bedroom: n = 1, point = 208 ng/g Daughter's bedroom: n = 1, point = 47.0 ng/g Third son's bedroom: n = 1, point = 250 ng/g Fourth son's bedroom: n = 1, point = 147 ng/g Attic: n = 1, point = 50.0 ng/g Basement: n = 1, point = 12.0 ng/g Untreated: n = 1, point = 17.0 ng/g

Study	Location	Site Details	Results
		room. The last application was in 2007, with six intermittent applications. The house did not have a fan-forced air circulation system and there was also no fresh-air intake to the house from the outdoors. Two additional pieces of the same carpet (untreated, uninstalled) stored in the basement for many years were also evaluated.	(LOD/LOQ not reported)
Europe			
van der Veen et al. (2020)	Sweden (unspecified)	Samples of durable water repellent outdoor clothing collected from six suppliers from the outdoor textile industry in Sweden (one pair of outdoor trousers, six jackets, and six fabrics for outdoor clothes*). Each sample was cut into two pieces – one exposed to elevated UV radiation, humidity, and temperature for 300 hours (assumed lifespan of outdoor clothing) and one untreated (not aged). Sampling year not reported. Year of manufacturing not reported for nine of the 13 samples; the remaining four samples (samples 4–7) reported a manufacturing year of 2012/2013. Country of origin not reported. *The breakdown of the 13 items of outdoor clothing is reported differently in Section 2.2and Table 1 of the article. Section 2.2 reports one pair of outdoor trousers, seven jackets, four fabrics for outdoor clothes, and one outdoor overall. Table 1 shows one pair of outdoor trousers, six jackets, and six fabrics for outdoor clothes.	Values presented as not aged, aged for L-PFHxS. Samples 1, 3, 4, 6–12 (1 outdoor trousers, 4 jackets, and 5 fabrics for outdoor clothes): n = 1 (for each sample), ND, ND Sample 2 (fabric for jacket): n = 1, 0.89 µg/m^2, ND Sample 5 (men's jacket): n = 1, 0.11, 0.68 µg/m^2 Sample 13 (fabric for outdoor clothes): n = 1, ND, NA (LOD = 0.02–0.1 µg/m^2 for ionic PFAS)
Schultes et al. (2018)	Sweden (unspecified)	Thirty-one cosmetic products from five product categories (moisturizing cream, foundation, eye pencil, powder and eye shadow, shaving foam) purchased from the Swedish market in 2016–2017. Cosmetic products were selected based on (i) the 2015 KEMI survey which reported the most frequently reported PFAS in cosmetic products and (ii) a database of ingredient lists compiled by the Swedish Society for Nature Conservation. Twenty-four products listing nine different PFAS as active ingredients were purchased. In addition, seven products which did not list PFAS in their ingredients were also purchased from the same stores as control	Control: n = 7, DF 0% PFAS-containing: n = 24, DF 0% (LOD = not reported)

Study	Location	Site Details	Results
		samples. Year of manufacture and country of origin not reported.	
Favreau et al. (2016)	Switzerland (national)	Liquid consumer products, including impregnation agents, cleansers, polishes, lubricants, miscellaneous items, and commercial AFFFs purchased in 2012 and 2013 from stores and supermarkets throughout Switzerland. Products were purchased from 82 different producers and were selected based on their susceptibility to contain PFAS according to previous screenings. Miscellaneous "other" products included foam-suppressing agents for the chromium industry, paints, ski wax, inks, and tanning substances. AFFFs were divided into two sets based on the sampling source. AFFF set 1 was derived from stock solution in fire installation of industrial sites storing chemicals and petroleum products and samples may be the result of multiple AFFF fillings over the years (1990–2010 was the last documented filling date). AFFF set 2 came from commercially available AFFFs between 2012 and 2013 from six producers. Results reported for L-PFHxS.	Impregnation products: n = 60, DF 0% Cleansers: n = 24, DF 0% Polishes: n = 18, DF 0% Others: n = 23, DF 4%, mean (range) = 1,700 (1,700–1,700) ng/g AFFF set 1: n = 27, DF 81%, mean, median (range) = 293,000, 89,500 (100–1,025,000) ng/g AFFF set 2: n = 35, DF 0% (LOQ = 0.5 ng/mL) *Mean and range values only include samples where L-PFHxS was detected *ND treated as 0 for median calculations
Kotthoff et al. (2015)	Germany (Schmallenberg)	Forty-nine random samples of consumer products collected in the first until the third quarter of 2010 in Germany, including outdoor textiles, carpets, cleaning agents, impregnating agents, leather samples, and ski waxes. Individual samples were bought from local retailers or collected by coworkers of the involved institutes or local clubs (e.g., ski waxes from local skiing club). Sampled products spanned all quality levels from entry level to cutting edge products. The age of the samples ranged from a few years to decades. Country of origin not reported.	Cleaner: n = 6, DF 0% Wood glue: n = 1, DF 0% Nanosprays and impregnation sprays: n = 3, DF 0% Outdoor textiles: n = 3, DF 0% Carpet: n = 6, DF 0% Gloves: n = 3, DF 0% Ski wax: n = 13, DF 35%, median (maximum) = <LOQ (9.3 ng/g) Leather: n = 13, DF 96%, median (maximum) = 10.1 (10.1 µg/m^2) Awning cloth: n = 1, DF 0% (LOQ = 0.5 ng/g or 0.5 µg/m^2) *Concentrations <LOQ were considered as zero
Vestergren et al. (2015)	Norway (Tromsø, Trondheim)	Samples of furniture textile (samples included baby-related items such as baby mattress, baby blanket, and baby bed cover), carpet, and clothing samples were	Furniture textiles: n = 27, DF[a] 0% Carpet: n = 9, DF[a] 0%

Study	Location	Site Details	Results
		purchased from three major retail stores during November 2012–February 2013. Sampling campaign designed to evaluate consumer products in product categories that were previously found to contain PFAS residuals and that were representative of products imported from China in large quantities. Individual products randomly selected without prior knowledge of surface treatment with PFAS. Outdoor clothing was excluded. Year of manufacture not reported.	Cotton/Leather clothing: n = 4, DF[a] 0% (MDL = 0.010 µg/m^2)
Origin Unspecified			
Bečanová et al. (2016)	Not specified	One hundred twenty-six samples of (1) household equipment (textiles, floor coverings, electrical and electronic equipment (EEE), and plastics; includes children-related items such as teddy bear filling, teddy bear cover, and plush); (2) building materials (oriented strand board, other composite wood and wood, insulation materials, mounting and sealant foam, facade materials, polystyrene, air conditioner components); (3) car interior materials; and (4) wastes of electrical and electronic equipment (WEEE) purchased (for new materials) or collected from various sources (for older and used materials). Production year ranged from 1981 to 2010. Origin of production and location and year of purchase/collection not reported.	Household equipment: n = 55, DF[a] 42%, range = <MQL–5.16 ng/g Building materials: n = 54, DF[a] 22%, range = <MQL–24.5 ng/g Car interior materials: n = 10, DF[a] 30%, range = <MQL–0.479 ng/g WEEE: n = 7, DF[a] 14%, range = <MQL–0.018 ng/g (IQL = 17 pg/mL; MQL = 0.11 ng/g)
Gremmel et al. (2016)	Not specified	Sixteen outdoor jackets (15 outdoor jackets and one working jacket) purchased during August 2011 to March 2012. Besides the working jacket and two other jackets (one arrived unpacked in shop and had been on sale for four weeks while the other had been on sale since February 2010), all other jackets were new and packed in a plastic shell. Jackets were selected considering factors such as origin of production (primarily Asia, with some origins not specified), price, market, and textile. Location of purchase and year of manufacture not reported.	Jackets J0–J2, J4–J6, J8–J11, J13–J15: n = 2 (for each jacket), DF 0% Jackets J3, J7: n = 2 (for each jacket), DF NR, mean = <LOQ Jacket J12: n = 2, DF NR, mean = 0.01 ng/g (LOD = 0.05 ng/mL; LOQ = 0.01 µg/m^2)

Notes: BDL = below detection limit; DF = detection frequency; DL = detection limit; MDL = method detection limit; ND = not detected.
[a] The DF and/or mean was not reported in the study and was calculated in this synthesis. Means were calculated only when DF = 100%.

D.3.4. Indoor Dust

Several studies from the U.S. and abroad were identified that evaluated the occurrence of PFHxS and other PFAS in dust of indoor environments, primarily in homes, as well as in schools, childcare facilities, offices, and vehicles. In a Wisconsin Department of Health Services study, Knobeloch et al. (2012) examined levels of 16 perfluoroalkyl chemicals in vacuum cleaner dust from 39 Wisconsin homes across 16 counties in March and April 2008 (Table D-8). Samples from these homes built between 1890 and 2005 were collected during a pilot study to assess residential exposure to persistent contaminants found in the Great Lakes Basin. PFHxS was found in all samples at a median concentration of 16 ng/g. Mean levels of PFHxS in dust were significantly higher in homes built between 1968 and 1995 (219 ng/g vs. 57 ng/g in homes constructed in other years). Based on the results of this study, the authors suggested that perfluoroalkyl chemicals may be ubiquitous contaminants in U.S. homes. In an EPA study of 112 indoor dust samples collected from vacuum cleaner bags from homes and daycare centers in North Carolina and Ohio in 2000–2001 (EPA's CTEPP study), samples were collected from 102 homes and 10 daycare centers in North Carolina (49 homes, 5 daycare centers) and Ohio (53 homes, 5 daycare centers) (Strynar and Lindstrom, 2008). Results were not reported separately for homes and daycares. Overall, PFHxS was detected in 77.7% of all samples (n = 112) at mean and median concentrations of 874 and 45.5 ng/g, respectively. The authors concluded that the study measured perfluorinated compounds in house dust at levels that may represent an important pathway for human exposure.

Additional peer-reviewed studies were identified that evaluated the occurrence of PFHxS and other PFAS in dust of indoor environments, primarily in homes, as well as in schools, childcare facilities, offices, and vehicles (Zheng et al., 2020; Scher et al., 2019; Byrne et al., 2017; Karásková et al., 2016; Wu et al., 2014; Fraser et al., 2013; Knobeloch et al., 2012; Kato et al., 2009) (Table D-8). For those studies with results stratified for U.S. homes, PFHxS levels and detection frequencies were lowest in a study of remote Alaska Native villages (27% detection, median below 0.2 ng/g), while in other U.S. locations, PFHxS was detected in at least 40% of samples (some studies reporting 100% detection) at widely varying mean and median levels across the studies (from on the order of 10 ng/g to on the order of 200 ng/g) with one study reporting the highest mean value (219 ng/g) from homes built between 1968 and 1995. The two studies also reporting home measurements from other countries differed in how PFHxS levels in the United States ranked relative to other countries, with one study ranking the U.S. highest and the other second lowest. Few studies sampled childcare centers, vehicles, and offices, and none of the reviewed studies reported measurements in other microenvironments (e.g., public libraries, universities).

Several studies reported results from dust samples collected only from homes (Scher et al., 2019; Byrne et al., 2017); Wu et al. (2014), with one study sampling from locations near a PFAS production facility. Scher et al. (2019) evaluated indoor dust in 19 homes in Minnesota within a GCA in the vicinity of a former 3M PFAS production facility. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. In the house dust samples, collected from July to September 2010, the detection frequencies for PFHxS were 68% and 84% for entryways to the yard and interior living spaces such as the family or living rooms, respectively (n = 19 each), with median concentrations

of 8.2 ng/g and 18 ng/g, respectively. PFAS concentrations in both sampling locations were higher than corresponding soil concentrations, suggesting that interior sources were the main contributors to PFAS in house dust.

Byrne et al. (2017) assessed exposure to PFHxS and other PFAS among residents of two remote Alaska Native villages on St. Lawrence Island. PFAS concentrations were measured in dust collected from the surfaces of floors and furniture of 49 homes on St. Lawrence Island during February–April of 2013 and 2014. Residents were asked not to sweep or dust for one week prior to sampling. The authors described the overall PFAS levels in dust samples as "on the lower end of those reported worldwide in other studies." PFHxS was found in 27% of all samples (n = 49) with a median value below the LOD (0.1 ng/g–0.2 ng/g). Wu et al. (2014) measured concentrations of five PFAS in residential dust in California in 2008–2009. Dust samples were collected from the carpet or area rug in the main living area of the home. Homes of parents with young children and homes with older adults were differentiated to characterize the relationship between serum concentrations of PFAS and several other factors, including PFAS concentrations in residential dust. PFHxS was detected in 51% of samples from households with young children in Northern California (n = 82), with mean and median concentrations of 142 ng/g and 5.30 ng/g, respectively. PFHxS was detected in 52% of samples from households of older adults in central California (n = 42), with mean and median concentrations of 55 ng/g and 5.55 ng/g, respectively.

Apart from the information reported by Strynar and Lindstrom (2008), one other study included childcare centers in the locations sampled (Zheng et al., 2020). Zheng et al. (2020) collected dust samples from seven childcare centers in Seattle, Washington (n = 14) and one childcare facility in West Lafayette, Indiana (n = 6 across six rooms); the sampling year was not reported. The included childcare facilities consisted of several building types, including multiple classrooms, a former church, and a former home. Because centers were vacuumed and mopped daily, dust samples were obtained from elevated surfaces (shelving, tops of bookcases/storage cubbies) along with floor dust. PFHxS was detected in 95% of samples at mean and median concentrations of 0.34 ng/g and 0.25 ng/g, respectively.

One study evaluated PFHxS levels in vehicles and offices, in addition to homes. Fraser et al. (2013) collected dust samples between January and March 2009 from three microenvironments of 31 individuals in Boston, Massachusetts (offices (n = 31), homes (n = 30), and vehicles with sufficient dust for analysis (n = 13)). Study participants worked in separate offices located across seven buildings, which were categorized as Building A (n = 6), Building B (n = 17), or Other (n = 8). Building A was a newly constructed (approximately one year prior to study initiation) building with new carpeting and new upholstered furniture in each office. Building B was a partially renovated (approximately one year prior to study initiation) building with new carpeting throughout hallways and in about 10% of offices. The Other buildings had no known recent renovation occurred. Study offices were not vacuumed during the sampling week and participants were asked not to dust or vacuum their homes and vehicles for at least one week prior to home sampling. Because PFHxS was detected in less than 50% of samples in all three microenvironments, geometric means were not reported. The detection frequencies for PFHxS were 23%, 40%, and 46% for offices, homes, and vehicles, respectively, with the range of detected values reported as 5.24 ng/g–18.5 ng/g, 6.05 ng/g–430 ng/g, and 5.22 ng/g–108 ng/g, respectively.

Two studies evaluated dust samples collected across multiple continents (Karásková et al., 2016; Kato et al., 2009). Karásková et al. (2016) examined PFAS levels in house dust collected between April and August 2013 from the living rooms and bedrooms of 14 homes in the United States, 15 homes in Canada, and 12 homes in the Czech Republic (locations unspecified). PFHxS was detected in all U.S. samples (n = 20) at mean and median concentrations of 13.8 ng/g and 8.7 ng/g, respectively. The authors reported significant differences between countries for PFHxS concentrations, with a trend of U.S. > Canada ~ Czech Republic and suggested that the differences may be explained by differences in the market, import history, and usage of these substances. Overall, no significant differences in total PFAS concentrations were found between the bedroom and living room in the same household although significant relationships were found based on type of floors, number of residents, and age of the house. A second multicontinental study (Kato et al., 2009) measured PFC concentrations in 39 household dust samples collected in 2004 from homes in the United States (Atlanta, GA) (n = 10), United Kingdom (n = 9), Germany (n = 10), and Australia (n = 10). Across all 39 homes, PFHxS was detected in 79.5% of samples with a median concentration of 185.5 ng/g. Authors presented the median and maximum PFHxS concentrations by country in a bar chart, which showed that PFHxS was detected in all countries. The median and maximum PFHxS concentrations for the 10 United States (Atlanta, GA) house dust samples were approximately 96.4 ng/g and 231.3 ng/g, respectively. The highest median was found in Australia, followed by the United Kingdom, the United States, and Germany in decreasing order; statistical significance on the comparison of median PFHxS concentrations by country was not reported.

In general, PFHxS concentrations in dust were higher in North America than Europe, with five studies in the United States or Canada reporting maximum concentrations >1,000 ng/g in homes and daycares (Wu et al., 2014; Beesoon et al., 2012; Knobeloch et al., 2012; Strynar and Lindstrom, 2008; Kubwabo et al., 2005) compared to one study from Europe for an office storage room (Huber et al., 2011). An additional study (Kato et al., 2009) measured PFHxS concentrations in both continents and reported higher maximum concentrations in the United States compared to Germany and the United Kingdom. One European study, conducted by Haug et al. (2011), indicated that house dust concentrations are likely influenced by a number of factors related to the building (e.g., size, age, floor space, flooring type, ventilation); the residents or occupants (e.g., number of people, housekeeping practices, consumer habits such as buying new or used products); and the presence and use of certain products (e.g., carpeting, carpet or furniture stain-protective coatings, waterproofing sprays, cleaning agents, kitchen utensils, clothing, shoes, cosmetics, insecticides, electronic devices). In addition, the extent and use of the items affects the distribution patterns of PFAS compounds in dust of these buildings. Results from the remaining studies conducted in Europe are presented in Table D-8.

Table D-8. Summary of PFHxS Indoor Dust Data

Study	Location	Site Details	Results
United States			
Scher et al. (2019)	United States (Twin Cities metropolitan region, Minnesota)	Nineteen homes in three cities within a GCA near former 3M PFAS production facility as well as from three homes in the Twin Cities Metro outside the GCA. Dust samples collected from an entryway to the yard and from an interior living space (e.g., family room, living room) in each home in July–September 2010. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. Results were not reported for homes outside the GCA.	Entryway: n = 19, DF 68%, median (range) = 8.2 (<RL–94) ng/g Living room: n = 19, DF 84%, median (range) = 18 (<RL–790) ng/g (RL = 5 ng/g)
Byrne et al. (2017)	United States (St. Lawrence Island, Alaska)	Dust samples collected from the surfaces of floors and furniture from 49 homes during February–April of 2013 and 2014. Participants were asked not to sweep or dust for one week prior to sampling.	n = 49; DF 27%, median (95th percentile) = <LOD (3.13) ng/g (MDL = 0.1–0.2 ng/g for all PFAS)
Wu et al. (2014)	United States (Central Valley area, California)	Distributions of PFC dust concentrations were determined for households with young children in Northern California (n = 82) and households of older adults in central California (n = 42). Dust samples were collected in 2008–2009 from the carpet or area rug in the main living area of the homes. Homes of parents with young children and homes with older adults were differentiated to characterize the relationship between serum concentrations of PFCs and PFC concentrations measured in residential dust.	Parents of young children: n = 82, DF 51%, mean, median (range) = 142, 5.30 (ND–7,490) ng/g Older adults: n = 42, DF 52%, mean, median (range) = 55, 5.55 (ND–1,050) ng/g (LOD = 0.10 ng/ml) *Data below LOQ replaced by LOD/√2
Knobeloch et al. (2012)	United States (Great Lakes Basin, Wisconsin)	Dust samples were collected by the Wisconsin Department of Health Services from 39 Wisconsin homes across 16 counties in March–April 2008. Vacuum bags were collected or bagless vacuums were emptied into sterilized glass jars. Homes were built between 1890 and 2005.	n = 39, DF 100%, median (range) = 16 (2.1–1,000) ng/g (RL = 1 ng/g)
Zheng et al. (2020)	United States (Seattle, Washington; West Lafayette, Indiana)	Seven childcare centers in Seattle (14 samples) and one center in Lafayette (6 samples); sampling year not reported. Since all centers were vacuumed and mopped daily, dust samples from elevated surfaces (shelving, tops of bookcases/storage cubbies) were collected along with floor dust in the same sample.	n = 20; DF 95%, mean, median (range) = 0.34, 0.25 (<ND–0.89) ng/g (MDL = 0.05 ng/g)

Study	Location	Site Details	Results
Strynar and Lindstrom (2008)	United States (North Carolina; Ohio)	Dust samples from vacuum cleaner bags were obtained in 2000–2001 during EPA's Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) study from North Carolina (49 homes, 5 daycare centers) and Ohio (53 homes, 5 daycare centers). Vacuum cleaner bags were only collected if available at each site.	n = 112, DF 77.7%, mean, median (maximum) = 874, 45.5 (35,700) ng/g (LOQ = 12.9 ng/g) *Values below the LOQ assigned a value of LOQ/√2
Fraser et al. (2013)	United States (Boston, Massachusetts)	Dust samples were collected between January–March 2009 from offices (n = 31), homes (n = 30), and vehicles (n = 13) of 31 individuals. Study participants worked in separate offices located across seven buildings, which were categorized into Building A, Building B, and Other. Six samples were collected from Building A, a newly constructed (approximately one year prior to study initiation) building with new carpeting and new upholstered furniture in each office. Seventeen samples were collected from Building B, a partially renovated (approximately one year prior to study initiation) building with new carpeting throughout hallways and in about 10% of offices. Eight samples were collected from the other five remaining buildings where no known recent renovation occurred. Study offices were not vacuumed during the sampling week and homes and vehicles were not vacuumed for at least one week prior to sampling. Entire accessible floor surface areas and tops of immovable furniture were vacuumed in offices and the main living area of homes. Entire surface areas of the front and back seats of vehicles were vacuumed. Number of home dust samples was reduced to 30 because 1 participant lived in a boarding house with no main living area. Sufficient mass of dust for analysis was available from only 13 vehicles.	Homes: n = 30, DF 40%, range = 6.05–430 ng/g Offices: n = 31, DF 23%, range = 5.24–18.5 ng/g Vehicles: n = 13, DF 46%, range = 5.22–108 ng/g (LOQ = 5 ng/g) *Range of detected values reported
Canada			
Beesoon et al. (2012)	Canada (Alberta)	Dust samples collected from the vacuum bag of the central vacuum system on the same day that carpets were sampled in a house built in 1989 in the Edmonton area. Samples were collected in September 2008 prior to a major renovation where all carpets were removed. The house was identified after abnormally high serum levels of PFHxS were	2008 sampling: n = 1, point = 2,780 ng/g 2012 sampling: n = 1, point = 253 ng/g (LOD/LOQ not reported) *The 2008 value is reported as 2,780 ng/g in the text but 2,900 ng/g in Figure 2

Study	Location	Site Details	Results
		identified in a husband and wife that were enrolled as volunteer control participants in a clinical research project. Many rooms in the house had wall-to-wall carpeting installed on top of the heated floors. Since 1991, Scotchgard has been applied to the carpets in the main floor family room and occasionally to the dining room. The last application was in 2007, with six intermittent applications. The house did not have a fan-forced air circulation system and there was also no fresh-air intake to the house from the outdoors. Vacuum dust sample also collected in January 2012.	
Kubwabo et al. (2005)	Canada (Ottawa)	Sixty-seven randomly selected homes with home selection described in a previous study. Samples collected in the winter of 2002–2003 from houses with varying age and percentage of surface area covered by carpet.	n = 67, DF 85%, mean, median (range) = 391.96, 23.1 (2.28–4,305) ng/g (MDL = 4.56 ng/g) *Non-detects and values <MDL were replaced with ½ × MDL
Europe			
de la Torre et al. (2019)	Spain (unspecified), Belgium (unspecified), Italy (unspecified)	Sixty-five homes belonging to the partners of Test-Achats (Belgium), Altroconsumo (Italy), and OCU Ediciones SA (Spain). Home occupants vacuumed the entire floor of their home from September 2016 to January 2017 and vacuum bags were collected.	Total: n = 65, DF 45%, median (range) = 0.13 (ND–11.3) ng/g Spain: n = 21, DF 76%, median (range) = 0.95 (ND–7.16) ng/g Belgium: n = 22, DF 23%, median (range) = 0.13 (ND–11.3) ng/g Italy: n = 22, DF 36%, median (range) = 0.13 (ND–3.62) ng/g (LOQ = 0.18 ng/g) *Values below LOQ replaced with LOQ/(square root of 2)
Winkens et al. (2018)	Finland (Kuopio)	Sixty-three private households from the birth cohort study, LUKAS2. Floor dust samples collected in 2014/2015 from the children's bedroom (entire floor). Participants were instructed not to vacuum clean the room at least a week before sampling. For 55 rooms, dust samples were collected at the end of a 3-week air sampling period (indoor air results reported in a different study).	n = 63, DF 33.3%, median (range) = 0.68 (BDL–37.0) ng/g (MDL = 0.47 ng/g) *Results reported for l-PFHxS *Values <MDL were treated as MDL/(square root of two)

Study	Location	Site Details	Results
Padilla-Sánchez and Haug (2016)	Norway (Oslo)	Homes of staff from the Norwegian Institute of Public Health. Dust samples collected from vacuum cleaner bags provided by staff. Sampling year not provided.	n = 7, DF[a] 71%, range = ND–1 ng/g (MDL = 0.012 ng/g; MQL = 0.040 ng/g)
Jogsten et al. (2012)	Spain (Catalonia)	Dust sampling was performed in December 2009 from ten households using household vacuum cleaner dust bags. Samples were collected out of convenience and may not be representative of the entire Catalan population.	n = 10, DF[a] 100%, mean[a] (range) = 1.07 (0.17–5.3) ng/g (LOD = 0.003 ng/g)
Haug et al. (2011)	Norway (Oslo)	Forty-one homes of breastfeeding mothers recruited for a study on exposure pathways. House dust samples collected between February and May 2008 on two consecutive days while the residence was in regular use. Samples taken from elevated surfaces such as bookshelves and window sills (deposited dust) and not from the floor.	n = 41, DF (frequency of quantification)[a] 54%, mean, median (range) = 8.4, 0.60 (0.21–142) ng/g (LOQ = 0.23–1.1 ng/g) *Concentrations that were not detected or <LOQ were replaced by the LOQ divided by the square root of two
Giovanoulis et al. (2019)	Sweden (Stockholm)	Twenty preschools that had been previously sampled in 2015 and then participated in the "chemical smart preschool" initiative to reduce the presence of hazardous chemicals in the indoor environment; 2015 results are reported elsewhere. Samples for this study were collected during January to February 2018. One settled dust sample was collected from elevated surfaces (50–250 cm above the floor) from different areas of a play room at each preschool.	n = 20, DF 0% (LOD = 0.5 ng/g) *Values <LOD were replaced with ½×LOD
Harrad et al. (2019)	Ireland (Dublin, Galway, and Limerick)	Dust collected from homes (living rooms), offices, cars, and school classrooms; air samples also collected. Samples collected between August 2016 and January 2017. Sample numbers were split approximately equally from each of the three counties.	Homes: n = 32, DF 47%, mean, median (range) = 1.4, <0.1 (<0.1–9.9) ng/g Offices: n = 33, DF 44%, mean, median (range) = 2.7, <0.1 (<0.1–57) ng/g Cars: n = 31, DF 47%, mean, median (range) = 6.2, <0.1 (<0.1–49) ng/g Classrooms: n = 32, DF 38%, mean, median (range) = 5.1, <0.1 (<0.1–120) ng/g (LOD = 0.1 ng/g) *When analyte peaks are <LOD, concentrations were assumed to equal DF × LOD where DF is expressed as a fraction.

Study	Location	Site Details	Results
Huber et al. (2011)	Norway (Tromsø)	Homes and workplaces sampled in winter 2007–2008. Home samples included seven different living rooms (L-1 to L-7), one sleeping room (S, related to L-3), one sofa (a stain repellant fabric, related to L-7), and one carpet (related to L-4). Workplace samples included an office and a storage room at Fram Center; old documents and chemicals and highly contaminated samples were stored in the storage room. Samples were taken from bookshelves, commodes, TVs, electrical heaters, picture frames, window sills and sun blinds. Dust from the floor was not sampled.	All homes: n = 7, DF NR, median = 1.4 ng/g Living room: n = 7, DF[a] 86%, mean, median (range) = 1.7, 1.4 (0.8–3.1) ng/g Carpet: n = 1, point = 0.5 ng/g Sleeping room: n = 1, point = 2.1 ng/g Sofa: n = 1, point = 1.7 ng/g Office: n = 1, point = 27.8 ng/g Storage room: n = 1, point = 1,814 ng/g (LOD on column = 0.006 ng; MDL = 0.1–1.85 ng/g)
D'Hollander et al. (2010)	Belgium (Flanders)	Forty-three randomly selected homes and ten randomly selected offices throughout Flanders. Samples collected using a vacuum from bare floor, possibly covered with carpet, in 2008. In homes, the living room, bedroom, kitchen, and working area were sampled.	Homes: n = 43, DF 40%, median (95th percentile) = 0.1 (9) ng/g dw Offices: n = 10, DF NR, median (95th percentile) = 0.2 (5.1) ng/g dw (LOQ = 0.1 ng/g) *Concentration <LOQ were replaced by DF × LOQ *For homes, both Table 3 and Section 2.1 reported n = 43; however, Table 2 reported n = 45
Multiple Continents			
Karásková et al. (2016)	United States (unspecified), Canada (unspecified), Czech Republic (unspecified)	Fifty-six dust samples from 14 homes in the United States, 15 homes in Canada, and 12 homes in the Czech Republic were collected between April and August 2013. Samples were collected in living rooms and bedrooms.	United States: n = 20, DF 100%, mean, median (range) = 13.8, 8.7 (1.4–84.4) ng/g Canada: n = 20, DF 90.0%, mean, median (range) = 3.1, 1.9 (ND–11.5) ng/g Czech Republic: n = 16, DF 93.8%, mean, median (range) = 2.8, 2.0 (<MQL–9.3) ng/g (MDL = 0.05–0.24 ng/g; MQL = 0.12–0.57 ng/g; ranges represent lower bound and upper bound which were calculated by dividing the MDL/MQL by the biggest and smallest dust sample weight, respectively) *Mean calculated only from values >MQL *Median calculated by replacing values <MQL with √2/2*MQL

Study	Location	Site Details	Results
Kato et al. (2009)	United States (Atlanta, Georgia), Germany (unspecified), United Kingdom (unspecified), Australia (unspecified)	Thirty-nine household dust samples from the United States (n = 10), Germany (n = 10), United Kingdom (n = 9), and Australia (n = 10) collected in 2004 for method validation. Dust sampling procedures not described.	n = 39, DF 79.5%, median (range) = 185.5 (<LOD–43,765) ng/g (LOQ = 2.6 ng/g)

Notes: DF = detection frequency; GCA = groundwater contamination area; LOD = limit of detection; LOQ = limit of quantitation; MDL = method detection limit; MQL = method quantification limit; ND = not detected; RL = reporting limit.

D.3.5. Air

Perfluoroalkyl chemicals have been released to air from wastewater treatment plants, waste incinerators, and landfills (Ahrens et al., 2011), though there is limited information on the detection levels or frequencies of PFHxS in either indoor or ambient air. NCBI (2022a) notes that PFHxS has been detected in particulates and in the vapor phase in air and it can be transported long distances via the atmosphere; it has been detected at low concentrations in areas as remote as the Arctic and ocean waters. For example, PFHxS was detected in particle-phase air samples collected in 2007 and 2008 from the Atlantic Ocean, Southern Ocean, and Baltic Sea (NCBI, 2022a; Dreyer et al., 2009). PFHxS is not expected to be broken down directly by photolysis (NCBI, 2022a). PFHxS in the particle-phase can be removed from the atmosphere through wet and dry deposition (NCBI, 2022a). PFHxS in the vapor phase can undergo hydroxylation in the atmosphere, with a (predicted average) atmospheric hydroxylation rate of 2.16×10^{-15} cm^3/molecule – second to a (derived) rate of 1.4×10^{-13} cm^3/molecule – second at 25°C (with corresponding estimated half-life of 115 days for this reaction in air) (USEPA, 2022a; NCBI, 2022a). With a vapor pressure of 0.0046 mm Hg at 25°C (estimated), 8.10×10^{-9} mm Hg (measured average), and 8.19×10^{-9} mm Hg (predicted average), volatilization is not expected to be an important fate process for this chemical (USEPA, 2022a; NCBI, 2022a). EPA's Toxics Release Inventory reported release data for PFHxS in 2020, with total onsite disposal, offsite disposal, and other releases concentrations of 1 pound at an individual facility and 122 pounds at a second facility (USEPA, 2022b). PFHxS is not listed as a hazardous air pollutant (USEPA, 2022c).

D.3.5.1. Indoor Air

No studies from the U.S. reporting levels of PFHxS in indoor air were identified from the primary or gray literature. However, several studies from Canada and Europe were identified and are discussed briefly below and presented in Table D-9 (Harrad et al., 2019; Beesoon et al., 2012; Goosey and Harrad, 2012; Jogsten et al., 2012; Barber et al., 2007). All of these studies sampled from homes, while two studies also sampled from offices, one study also sampled a laboratory, and only one study also sampled from cars and classrooms. In studies exclusively in homes, two studies did not detect PFHxS in indoor air while the remaining three studies had PFHxS detection frequencies ranging from 21 to 100%. In one of these studies, a Canadian household with wall-to-wall Scotchgard-treated carpets on top of heated floors (and whose residents had previously been found to have disproportionately high blood serum levels of PFHxS) found 27.4 pg/m^3 PFHxS in the family room and 426 pg/m^3 PFHxS in the basement (with no clear reason for the higher levels in the basement). Additional research is needed to evaluate PFHxS in indoor air from U.S. locations and from a variety of non-home microenvironments (offices, cars, classrooms, and laboratories) in Canada and Europe.

Jogsten et al. (2012) sampled indoor air (n = 10) from selected homes in Catalonia, Spain in December 2009 and evaluated levels of 27 PFCs, though PFHxS was not detected. The remaining studies evaluated PFHxS levels in offices, vehicles, and/or schools, in addition to homes (Harrad et al., 2019; Goosey and Harrad, 2012; Barber et al., 2007). In Ireland, Harrad et al. (2019) collected air samples in homes (living rooms, n = 34), offices (n = 34), cars (n = 31), and school classrooms (n = 28) between August 2016 and January 2017. PFHxS was detected in all four indoor environments in 21%, 44%, 23%, and 25% of samples for homes, offices, cars, and classrooms, respectively. However, the median concentrations were below the LOD for all

environments. Goosey and Harrad (2012) detected PFHxS with mean (median) concentrations of 36 (23) pg/m^3 in homes (n = 20) and 94 (84) pg/m^3 in offices (n = 12) sampled in the United Kingdom between September 2008 and March 2009. In addition, Goosey and Harrad (2012) also examined seasonal variation in these same five indoor microenvironments by sampling monthly between September 2008 and August 2009. The relative standard deviation for PFHxS was between 52–106%. The observed variation could not be attributed to changes in room contents or to seasonality. The study also measured ambient air concentrations in the same location and concluded that indoor air concentrations significantly exceeded ambient air concentrations; the authors suggested that indoor emissions may influence both indoor and outdoor PFAS levels. In Norway, neutral and ionic PFAS were analyzed in indoor air samples collected from three homes and one laboratory in Tromsø between in April and June 2005 (Barber et al., 2007). Results for 15 neutral PFAS and 16 ionic PFAS (including PFHxS) were presented but PFHxS was not detected (method quantification limit [MQL] = 4.09 pg/m^3; authors considered this a high MQL and likely due to much lower sampling volume).

Table D-9. Summary of PFHxS in Indoor Air

Study	Location	Site Details	Results
Canada			
Beesoon et al. (2012)	Canada (Alberta)	Indoor air samples collected from one home in the Edmonton area, built in 1989. The house was identified after abnormally high serum levels of PFHxS were identified in a husband and wife that were enrolled as volunteer control participants in a clinical research project. Many rooms in the house had wall-to-wall carpeting installed on top of the heated floors. Since 1991, Scotchgard has been applied to the carpets in the main floor family room and occasionally to the dining room. The last application was in 2007, with six intermittent applications. The house did not have a fan-forced air circulation system and there was also no fresh-air intake to the house from the outdoors. At the time of indoor air sampling in September 2008, renovations had begun in the basement. Analysis for PFHxS conducted on Fraction 1 (particulate phase) from the main floor family room and finished basement.	Main floor family room: n = 1, point = 27.4 pg/m^3 Basement: n = 1, point = 426 pg/m^3 (LOD/LOQ not reported)
Europe			
Jogsten et al. (2012)	Spain (Catalonia)	Indoor air sampling was performed in December 2009 from ten households at approximately 1 m above the floor. Samples were collected out of convenience and may not be representative of the entire Catalan population. Both particulate and gas phases collected.	n = 10, DF 0% (LOD = 3.1–280 pg/m^3 for all ionic PFAS)
Harrad et al. (2019)	Ireland (Dublin, Galway, Limerick)	Air samples collected from homes (living rooms), offices, cars, and school classrooms; dust samples also collected. Samples collected between August 2016 and January 2017. Sample numbers were split approximately equally from each of the three counties. Gas or particulate phase not specified.	Homes: n = 34, DF 21%, mean, median (range) = <0.4, <0.4 (<0.4–0.46) pg/m^3 Offices: n = 34, DF 44%, mean, median (range) = 0.40, <0.4 (<0.4–1.4) pg/m^3 Cars: n = 31, DF 23%, mean, median (range) = 0.15, <0.4 (<0.4–0.55) pg/m^3 Classrooms: n = 28, DF 25%, mean, median (range) = <0.4, <0.4 (<0.4–2.3) pg/m^3 (LOD = 0.4 pg/m^3) *When analyte peaks are <LOD, concentrations were assumed to equal DF × LOD where DF is expressed as a fraction.

Study	Location	Site Details	Results
Goosey and Harrad (2012)	United Kingdom (Birmingham)	Samples collected from homes and offices between September 2008 and March 2009. In addition, air samples were collected monthly from one office and four living rooms between September 2008 and August 2009. Gas or particulate phase not specified.	Homes: n = 20, DF[a] 90%, mean, median (range) = 36, 23 (<1.1–220) pg/m^3 Offices: n = 12, DF[a] 83%, mean, median (range) = 94, 84 (<1.1–330) pg/m^3 Seasonal Variations: concentrations reported for each month from Sept 2008 to Aug 2009 Home 1: 12, 100, 24, 21, 22, 17, 49, 35, 4, <1.1 pg/m^3 Home 2: 110, 49, 30, 7, 30, 9, 44, 50, 110, 44, 22 pg/m^3 Home 3: <1.1, 24, 11, 12, 49, 100, 9, 16, 84, 17, 19, <1.1 pg/m^3 Home 4: <1.1, 37, 44, 5, 27, 23, 47, 25, 3, 27, 22, 37 pg/m^3 Office 1: <1.1, 8, 30, 16, 50, 33, 12, 12, 18, 9, <1.1 pg/m^3 (DL = 1.1 pg/m^3) *Where concentration <DL, ½×DL was used for calculations
Barber et al. (2007)	Norway (Tromsø)	Air samples taken from four indoor locations (three houses and one laboratory) in Tromsø in April–June 2005. PFHxS was measured in the particulate phase.	n = 4, DF[a] 0%, mean = <4.1 pg/m^3 (MQL = 4.09 pg/m^3)

D.3.5.2. Ambient Air

The EPA identified a single study reporting ambient air concentrations of PFHxS from the U.S. Kim and Kannan (2007) analyzed particle phase (n = 8) and gas phase (n = 8) concentrations of perfluorinated acids in ambient air samples collected in and around Albany, New York, in May and July 2006 to examine the relative importance of certain media pathways to the contamination of urban lakes. PFHxS was detected in all gas phase samples with mean, and median concentrations of 0.31 pg/m^3 and 0.34 pg/m^3, respectively, but was not detected in the particle phase (LOQ = 0.12 pg/m^3).

Additional studies were identified that examined PFHxS in ambient air from sampling efforts conducted in Canada and Europe. These studies are summarized below and presented in Table D-10. In Canada, Ahrens et al. (2011) did not detect PFHxS in air around a WWTP and two landfill sites. In Europe, Barber et al. (2007) collected air samples from four field sites in the United Kingdom, Ireland, and Norway for analysis of neutral and ionic PFAS. PFHxS was detected in the ambient air particle phase at two sampling events in Manchester, UK at 0.1 and 1.0 pg/m^3, at one of two sampling events in Hazelrigg, UK at 0.04 pg/m^3, at one sampling event in Norway at 0.05 pg/m^3, and at one sampling event in Ireland at 0.07 pg/m^3. Goosey and Harrad (2012) collected ambient air samples from ten different locations within a 1.5 km radius of the University of Birmingham campus in the United Kingdom. PFHxS was detected with a mean (maximum) concentration of 7.0 (30) pg/m^3. Beser et al. (2011) detected PFHxS in 5% of ambient air particulate samples from five locations in Alicante province, Spain (3 residential, 1 rural, 1 industrial), with a mean (maximum) concentration of 8.7 (15.9) pg/m^3. The highest concentration observed was at the industrial site. Harrad et al. (2020) analyzed air samples near ten Irish municipal solid waste landfills in nonindustrial areas. PFHxS was detected in more than 20% of the samples, with mean (maximum) concentrations at downwind and upwind locations of 0.34 (0.79) pg/m^3 and 0.23 (0.81) pg/m^3, respectively. One European study (Jogsten et al., 2012) did not detect PFHxS in ambient air samples collected outside homes in Catalonia, Spain.

Table D-10. Summary of the Occurrence of PFHxS in Ambient Air

Study	Location	Site Details	Results
United States			
Kim and Kannan (2007)	United States (Albany, New York)	Roof of a lakehouse building located at Washington Park Lake in May and July 2006. Both particulate and gas phases collected.	Gas: n = 8, DF NR, mean, median (range) = 0.31, 0.34 (0.13–0.44) pg/m^3 Particle: n = 8, DF 0% (LOQ = 0.12 pg/m^3) *Non-detects were set to zero; values below the LOQ were set to ½ LOQ
Canada			
Ahrens et al. (2011)	Canada (Ontario)	Samples collected on and around one municipal WWTP for 63 days between July and September 2009. Samplers were placed at the primary clarifier, aeration tank, secondary clarifier, and at four reference sites (three near [within 200 m of the treatment tanks] and one distant [~600 m from the perimeter of the WWTP]). Samples also collected at two municipal solid waste landfills between June and August 2009 for 55 days. The two landfills were 60 km apart. Samplers were located upwind and onsite of the active zone of each landfill site and one field blank was collected at each site. Both sites collected landfill gas and the active area of the landfill was kept to a minimum by covering the waste with soil and a plastic film. The passive sampling configuration used resulted in the collection of mainly PFAS in the gas phase.	WWTP: Reference sites (near): n = 3, DF 0% Primary clarifier: n = 2, DF 0% Aeration tank: n = 3, DF 0% Secondary clarifier: n = 2, DF 0% Reference site (distant): n = 1, DF 0% Landfills: Upwind: n = 2, DF 0% On site: n = 2, DF 0% (MDL = 0.04–0.87 pg/m^3 for PFCAs, PFSAs, and PFOSA)
Europe			
Harrad et al. (2020)	Ireland (multiple cities)	Samples collected from ten municipal solid waste landfills upwind and downwind at each site between November 2018 and January 2019. Location of sampling sites based on wind direction data taken from the Irish Meteorological Service, with slight modification where necessary based on local information from site operators and ease of access. Sample sites were between 150 and 500 m of the center of the landfill. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction and demolition, and	Downwind: n = 10, DF[a] 60%, mean, median (range) = 0.34, 0.23 (<0.15–0.79) pg/m^3 Upwind: n = 10, DF[a] 40%, mean, median (range) = 0.23, 0.08 (<0.15–0.81) pg/m^3 (LOD = 0.15 pg/m^3) *Non-detects replaced by ½ LOD

Study	Location	Site Details	Results
		biomedical waste. Gas or particulate phase not specified.	
Goosey and Harrad (2012)	United Kingdom (Birmingham)	Ten different locations within a 1.5 km radius of the University of Birmingham campus. Samples collected in March 2009. In addition, air samples were collected monthly from three outdoor locations between September 2008 and August 2009 where two of the locations were among the ten sampled in March 2009 and the third location was Harwell, Oxfordshire in Southeast England, a semi-rural site. Gas or particulate phase not specified.	n = 10, DF[a] 50%, mean, median (range) = 7.0, 2.7 (<1.1–30) pg/m^3 (DL assumed to be 1.1 pg/m^3) Seasonal Variations: concentrations reported for each month from Sept 2008 to Aug 2009 Birmingham site 1: 12, 11, 3.8, 2.0, 420, <0.8, <0.8, <0.8, 1.3, 1.4 pg/m^3 Birmingham site 7: 1.5, 8.3, <0.8, 3.8, 1.9, 5.3, 5.6, <0.8, 12, <0.8, 20 pg/m^3 Harwell site: <0.8, 1.6, 4.9, <0.8, 1.3, <0.8, 1.0, <0.8, 4.9, 1.5, 20, 8.8 pg/m^3 (DL assumed to be 0.8 pg/m^3) *For concentration < DL, ½×DL was used for calculations
Jogsten et al. (2012)	Spain (Catalonia)	Outdoor air sampling conducted in December 2009 for the purposes of comparison to indoor air and dust samples. Number of sites not specified but assumed to be ten because indoor air was sampled from ten homes. Samples were collected out of convenience and may not be representative of the entire Catalan population. Both particulate and gas phases collected.	n = 10, DF 0% (LOD = 3.1–280 pg/m^3 for all ionic PFAS)
Beser et al. (2011)	Spain (Alicante province)	Samples collected from April to July 2010 from five stations. Two stations were placed in Elche (one in a residential area and the other in an industrial area). The third station was placed in a residential area of Alicante City. The fourth station was in a rural area of Pinoso and the last station was in a residential area of Alcoy. Concentrations reported for PM2.5-bound PFHxS.	Elche (residential): n = 11, DF[a] 0% Elche (industrial): n = 13, DF[a] 7.7%, mean = 15.9 pg/m^3 Alicante City: n = 11, DF[a] 0% Pinoso: n = 3, DF[a] 0% Alcoy: n = 3, DF[a] 33%, mean = 1.5 pg/m^3 (MQL = 1.4 pg/m^3) *Mean calculated from values >MQL
Barber et al. (2007)	United Kingdom (Hazelrigg, Manchester); Ireland (Mace Head); Norway (Kjeller)	Samples collected from four field sites in Europe: Hazelrigg (semirural) and Manchester (urban) were sampled in two sampling events in February–March 2005 and November 2005–January 2006; Mace Head (rural) was sampled in March 2006; and Kjeller (rural) was sampled in November–December 2005. PFHxS was measured in the particulate phase.	Hazelrigg first sampling event: n = 2, DF NR, mean = <5.9 pg/m^3 (MQL = 5.93 pg/m^3) Feb/Mar 2005 *The glass-fibre filters were analyzed in a batch of samples that showed contamination problems, so the high associated blank value

Study	Location	Site Details	Results
			used to calculate the MQL put most analytes <MQL. Hazelrigg second sampling event: n = 10, DF[a] = 100%, mean (range) = 0.04 (0.01–0.06) pg/m^3 (MQL = 0.002 pg/m^3) Manchester first sampling event: n = 2, DF[a] 100%, mean (range) = 1.0 (0.9–1.0) pg/m^3 (MQL = 0.45 pg/m^3) Manchester second sampling event: n = 1, point = 0.1 pg/m^3 (MQL = 0.002 pg/m^3) Mace Head: n = 4, DF[a] 100%, mean (range) = 0.07 (0.05–0.11) pg/m^3 (MQL = 0.004 pg/m^3) Kjeller: n = 2, DF[a] 100%, mean (range) = 0.05 (0.05) pg/m^3 (MQL = 0.01 pg/m^3) *Means calculated from values >MQL

D.3.6. Soil

The production of PFHxS and its use as a raw material or precursor for manufacturing PFAS-based products, as well as its previous use in firefighting foam and carpet treatment solutions, could result in its release to soils through various waste streams (NCBI, 2022a). When released to soil, PFHxS is expected to have very high mobility (NCBI, 2022a). The maximum concentration of PFHxS in soil samples collected from a PFAS production facility in Minnesota was 3,470 ng/g, with PFHxS detected in 90% of the samples collected (NCBI, 2022a; ATSDR, 2021; 3M, 2007). The maximum concentration of PFHxS in soil samples collected at a fire training area at a PFAS production facility in Minnesota was 62.2 ng/g, with PFHxS detected in 100% of samples (ATSDR, 2021; 3M, 2007). PFHxS was also detected in soil samples collected from a former sludge incorporation area at a PFAS production facility in Decatur, Alabama, with average levels ranging from 3.56 ng/g to 270 ng/g, with PFHxS detected in 86% of the samples collected (ATSDR, 2021; 3M, 2012). PFHxS has been found to accumulate in the roots of maize plants grown in soil containing PFHxS and other PFAS (ATSDR, 2021; Krippner et al., 2015).

Several additional peer-reviewed studies were identified that evaluated the occurrence of PFHxS and other PFAS in soil collected in the U.S. Two studies by Scher et al. (2019; 2018) evaluated soils collected in 2010 from the garden planting area of 20 homes in Minnesota within a GCA impacted by the former 3M PFAS production facility. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. Both studies reported similar median PFHxS levels of 0.08 ng/g and 0.057 ng/g (n = 20–34) from the 2019 and 2018 publications, respectively. Scher et al. (2018) also reported a median PFHxS concentration of 0.078 ng/g from six samples collected outside the GCA.

Three studies analyzed soils potentially impacted by AFFF use (Nickerson et al., 2020; Eberle et al., 2017; Anderson et al., 2016). Anderson et al. (2016) examined surface and subsurface soil from 40 sites across 10 active Air Force installations throughout the continental United States and Alaska between March and September 2014. Installations were included if there was known historic AFFF release in the period 1970–1990. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. The selected sites were not related to former fire training areas and were characterized according to volume of AFFF release – low, medium, and high. Across all sites, the PFHxS detection frequency was 76.92% in 100 surface soil samples (median concentration of detects was 5.7 ng/g) and 59.62% in 112 subsurface soil samples (median concentration of detects was 4.4 ng/g). PFHxS was detected more frequently at high-volume release sites (82.5% in 32 surface soil samples with mean concentration of 19.7 ng/g; 87.5% in 31 subsurface soil samples with mean concentration of 9.3 ng/g) than at low-volume sites (75.0% in 12 surface soil samples with mean concentration of 13.9 ng/g; 58.8% in 17 subsurface soil samples with mean concentration of 57.9 ng/g) and medium-volume sites (59.2% in 56 surface soil samples with mean concentration of 39.4 ng/g; 71.4% in 64 subsurface soil samples with mean concentration of 55.4 ng/g). Nickerson et al. (2020) developed a method to quantify anionic, cationic, and zwitterionic PFAS from AFFF-impacted soils. The method was applied to two soil cores collected from two different AFFF-impacted former fire training areas; the sampling year and geographic location were not provided. Eleven soil samples, corresponding to 11 depths ranging from 0.46 m to 15.1 m, were evaluated from Core E, and 12 soil samples, at depths ranging from

0.30 m to 14.2 m, were evaluated from Core F. PFHxS was detected at all depths analyzed for both cores, with concentrations ranging from 1.17 ng/g dw to 160.6 ng/g dw for Core E and 0.66 ng/g dw to 296.4 ng/g dw for Core F. Eberle et al. (2017) investigated the effects of an in situ chemical oxidation treatment for remediation of chlorinated volatile organic compounds and PFAAs co-contaminants. Soil samples were collected in 2012–2013 before and after a pilot scale field test at a former fire training site at Joint Base Langley-Eustis, Virginia. Monthly fire training activities were conducted at the site from 1968 to 1980 and irregular fire training activities continued until 1990. Impacted soil was excavated in 1982 but details were not provided. PFHxS was detected in 4 of 5 pre-treatment samples and in all 14 post-treatment samples. In the available three paired pre- and post-treatment soil samples, two pairings showed a decrease in PFHxS concentration after treatment, from 6.7 ng/g to 1.40 ng/g in one pairing and from 12 ng/g to 1.44 ng/g in the other pairing. In the third pairing, PFHxS was not detected pre-treatment and was found at 0.31 ng/g post-treatment.

Of the remaining two studies conducted in the United States, Venkatesan and Halden (2014) conducted outdoor mesocosm studies to examine the fate of PFAS in biosolids-amended soil collected during 2005–2008. Biosolids were obtained from a WWTP in Baltimore that primarily treated wastewater from domestic sources with only minor contribution (1.9%) from industry. PFHxS was not detected in the control (nonamended) soil and not consistently detected in the biosolids-amended soil (MDL = 0.03–0.14 ng/g dw). In a field and greenhouse study, Blaine et al. (2013) studied the uptake of PFAS into edible crops grown in control and biosolids-amended soil. In the field study, urban biosolids were obtained from a WWTP receiving both domestic and industrial waste while rural solids were obtained from a WWTP receiving domestic waste only. Mean PFHxS concentrations were below the LOQ (0.01 ng/g) in the urban control and biosolids-amended soils and in the rural control soil. In the rural biosolids-amended soil, the mean PFHxS concentration was 0.016 ng/g. In the greenhouse study, three soils (nonamended control, industrially impacted, and municipal) were investigated. Industrially impacted soils contained composted biosolids from a small municipal WWTP that was impacted by PFAA manufacturing while municipal soils were obtained from a reclamation site in Illinois where municipal biosolids were applied for 20 years. PFHxS was detected in all three soils at an average concentration of 0.44 ng/g, 1.38 ng/g, and 5.11 ng/g in control, industrially impacted, and municipal soil, respectively. Authors noted that the trace levels of PFAS detected in the control soil may be due to minor cross-contamination from plowing, planting, or atmospheric deposition from the surrounding area where biosolids have been applied.

Studies conducted in Europe and Canada were also identified and are summarized below and presented in Table D-11. Of the European studies, a study in Ireland (Harrad et al., 2020) examined soil samples collected upwind and downwind from ten municipal solid waste landfills and found PFHxS levels to be lower at downwind locations. Based on the overall study findings, however, the authors concluded there was no discernible impact of the landfills on concentrations of PFAS in soil surrounding these facilities. In the Maltese islands, Sammut et al. (2019) sampled soil from small urban fields and found PFHxS concentrations to range from non-detectable levels to 0.12 ng/g. Grønnestad et al. (2019) investigated soils from a skiing area in Norway to elucidate exposure routes of PFAS into the environment from ski products, such as ski waxes. PFHxS was below the limit of quantification in soil samples from both the Granåsen skiing area and the Jonsvatnet reference area. One study performed in Belgium (Groffen et al., 2019) evaluated soils collected at a 3M fluorochemical plant in Antwerp and at four sites located

at increasing distances from the plant. PFHxS levels were elevated at the plant site but not detected at other locations. Four studies were conducted at locations near firefighting training sites, which found varying results from non-detected levels to 2,344 ng/g dw (Dauchy et al., 2019; Skaar et al., 2019; Hale et al., 2017; Filipovic et al., 2015).

Table D-11. Summary of PFHxS Data in Soil

Study	Location	Site Details	Results
United States			
Scher et al. (2019)	United States (Twin Cities metropolitan region, Minnesota)	Area near former 3M PFAS production facility. Soil composite samples (200–500 g) collected in 2010 from the garden planting area of 20 homes in 3 cities within a GCA as well as from 3 homes in the Twin Cities Metro outside the GCA. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or a private well previously tested and shown to have detectable levels of PFOA or PFOS. Results were not reported for homes outside the GCA.	n = 20, DF 85%, median (90th percentile) = 0.08 (0.16) ng/g (RL = 0.0008–0.033 ng/g for all PFAS)
Scher et al. (2018)	United States (Twin Cities metropolitan region, Minnesota)	Area near former 3M PFAS production facility. Soil samples collected in 2010 at a sample depth of 0–6 inches from the garden planting area of 20 homes in 3 cities within a GCA as well as from 3 homes in the Twin Cities Metro outside the GCA. Homes within the GCA had previous or ongoing PFAS contamination in drinking water and were served by the Oakdale, Minnesota public water system or was formerly or currently using a private well previously tested and shown to have detectable levels of PFOA or PFOS. At 14 homes, soil samples were collected on two separate days.	Within GCA: n = 34, DF 71%, median (range) = 0.057 (ND–0.24) ng/g Outside GCA: n = 6, DF 100%, median (range) = 0.078 (0.028–0.11) ng/g (MDL = 0.008–0.033 ng/g for all PFAS) *Values below the method reporting limit but above the lowest calibration limit (estimated values) were included in all analyses as quantitative results *Values below the lowest calibration limit were replaced with ½ of the limit
Anderson et al. (2016)	United States (national)	Forty AFFF-impacted sites from ten active U.S. Air Force installations with historic AFFF release between 1970 and 1990 that were not related to former fire training areas. It is assumed that the measured PFAS profiles at these sites reflect the net effect of several decades of all applicable environmental processes. AFFF-impacted sites included emergency response locations, hangers and buildings, and testing and maintenance related to regular maintenance and equipment performance testing of emergency vehicles and performance testing of AFFF solution. Previous remedial activities for co-occurring contaminants were not specifically controlled for in the site selection process; active remedies had not been applied at any of the sites selected. Approximately ten samples were	Surface soil: Overall: n = 100, DF 76.92%, median (maximum) = 5.7 (1,300) ng/g *Breakdown by site:* Emergency Response (low-volume release): n = 12, DF 75.0%, mean (range) = 13.9 (0.38–64) ng/g Hangars and Buildings (medium-volume release): n = 56, DF 59.2%, mean (range) = 39.4 (0.34–1,300) ng/g Testing and Maintenance (high-volume release):

Study	Location	Site Details	Results
		collected between March and September 2014 at each site for surface and subsurface soil; sites were grouped according to volume of AFFF release—low-volume typically had a single AFFF release, medium-volume had one to five releases, and high-volume had multiple releases.	n = 32, DF 82.5%, mean (range) = 19.7 (0.29–180) ng/g (RL = 0.29 ng/g) Subsurface soil: Overall: n = 112, DF 59.62%, median (maximum) = 4.4 (520) ng/g *Breakdown by site:* Emergency Response (low-volume release): n = 17, DF 58.8%, mean (range) = 57.9 (0.37–520) ng/g Hangars and Buildings (medium-volume release): n = 64, DF 71.4%, mean (range) = 55.4 (0.35–1,300) ng/g Testing and Maintenance (high-volume release): n = 31, DF 87.5%, mean (range) = 9.3 (1.1–40) ng/g (RL = 0.31 ng/g) *Median calculated using quantified detections *Non-detects were substituted with ½ the reporting limit
Nickerson et al. (2020)	United States (unspecified)	Soil cores E and F from two different AFFF-impacted fire training areas; sampling year and geographic location not provided. Soil core E contained 11- 0.3 m increment samples from 0.3–15.2 m below ground surface and was collected in an area where the surficial soils were likely disturbed due to regrading and other soil redistribution activities. Soil core F contained 12- 0.61 m increment samples from 0–14.2 m below ground surface and was collected in an area where the surficial soils were highly permeable only within the upper 0.5 to 1 m, and the underlying impermeable clay layer exhibited a relatively high cation exchange capacity and organic carbon content. The water table was relatively shallow (depth <3 m) at both sites.	Core E: 0.46 m = 1.44 ng/g dw 2.9 m = 2.12 ng/g dw 3.66 m = 4.17 ng/g dw 3.96 m = 15.21 ng/g dw 4.27 m = 28.68 ng/g dw 4.57 m = 4.13 ng/g dw 4.88 m = 5.73 ng/g dw 7.01 m = 13.86 ng/g dw 8.38 m = 160.6 ng/g dw 10.5 m = 139.0 ng/g dw 15.1 m = 1.17 ng/g dw Core F: 0.30 m = 11.07 ng/g dw 1.22 m = 296.4 ng/g dw 1.83 m = 276.2 ng/g dw

Study	Location	Site Details	Results
			2.44 m = 106.2 ng/g dw 3.05 m = 42.69 ng/g dw 4.11 m = 28.78 ng/g dw 7.62 m = 14.19 ng/g dw 8.84 m = 6.26 ng/g dw 9.45 m = 3.25 ng/g dw 10.5 m = 0.66 ng/g dw 11.9 m = 3.06 ng/g dw 14.2 m = 7.96 ng/g dw (LOD/LOQ not reported)
Eberle et al. (2017)	United States (Joint Base Langley-Eustis, Virginia)	Pilot testing area in former fire training area (Training Site 15) at Joint Base Langley-Eustis where monthly fire training activities were conducted from 1968 to 1980 in a zigzag pattern burn pit. Facility was abandoned in 1980 but irregular fire training activities using an above-ground germed burn pit continued until 1990. Impacted soil was removed in 1982 but additional details of the excavation are not well known. Soil samples collected for pre- (April and September 2012) and post- (December 2013) in situ chemical oxidation treatment using a peroxone activated persulfate (OxyZone) technology. Treatment was conducted in Test Cell 1 over 113 days (April–August 2013). Soil samples were collected adjacent to wells; wells outside Test Cell 1 were used as sentry wells. Well IDs for pre- and post-sampling were not provided but the following three pairings were assumed based on Table 2 in the paper: U-20 with SB-106; U-16 with SB-112; and I-1 with SB-109.	Pre-treatment: I-1 (1.2–4.3 m) = 12 ng/g I-2 (1.2–4.3 m) = 83 ng/g U-12 (2.1 m) = 1.2 ng/g U-16 (3.0 m) = 6.7 ng/g U-20 (1.8 m) = ND (LOQ = 0.68–0.72 ng/g) Post-treatment: SB-101 (4.3 m) = 8.08 ng/g SB-105 (1.8 m) = 0.83 ng/g SB-106/U-20 (1.8 m) = 0.31 ng/g SB-106 (4.3 m) = 5.08 ng/g SB-107 (1.8 m) = 2.11 ng/g SB-107 (4.3 m) = 3.99 ng/g SB-108 (1.8 m) = 1.48 ng/g SB-108 (4.3 m) = 4.83 ng/g SB-109/I-1 (3 m) = 1.44 ng/g SB-111 (4.3 m) = 11.85 ng/g SB-112 (1.8 m) = 2.57 ng/g SB-112/U-16 (3 m) = 1.4 ng/g SB-114 (1.8 m) = 3.63 ng/g SB-114 (4.3 m) = 16.17 ng/g (LOQ = 0.12 ng/g)
Venkatesan and Halden (2014)	United States (Baltimore, Maryland)	Archived agricultural soil (nonamended) collected during 2005–2008 at a depth of 0–20 cm from the United States Department of Agriculture-Agricultural Research Service Beltsville Agricultural Research Center; number of sampling sites and number of samples not provided.	Nonamended: n = NR, DF 0% Amended: n = NR, authors reported that PFHxS was not consistently detected in biosolids-amended mesocosms (MDL = 0.03–0.14 ng/g dw for all PFAS)

Study	Location	Site Details	Results
		Biosolids-amended soil obtained by mixing biosolids and soil at a volumetric ratio of 1:2. Biosolids were from Back River WWTP in Baltimore, a full-scale activated sludge treatment plant. Raw wastewater was primarily from domestic sources with only minor contribution (1.9%) from industry.	
Blaine et al. (2013)	United States (Midwest)	Urban and rural full-scale field study with control (nonamended) and biosolids-amended plots. Three agricultural fields were amended (0.5×, 1×, or 2×) with municipal biosolids. Urban biosolids (1× and 2×) were from a WWTP receiving both domestic and industrial waste. Rural biosolids (0.5×) were from a WWTP receiving domestic waste only. Control plots were proximal to the rural and urban amended corn grain and corn stover field sites; sampling year not provided. Greenhouse study with control (nonamended) and biosolids-amended soil. Nonamended soil obtained from a field that received commercial fertilizers and had a similar cropping system as the nearby municipal soil site. Municipal soil was obtained from a reclamation site in Illinois where municipal biosolids were applied at reclamation rates for 20 years, reaching the cumulative biosolids application rate of 1,654 Mg/ha. Industrially impacted soil was created by mixing composted biosolids from a small municipal (but impacted by PFAA manufacturing) WWTP with control soil on a 10% mass basis. Sampling year not provided.	Field study: Urban non-amended: n = 3–7, DF NR, mean < 0.01 ng/g Urban 1×: n = 3–7, DF NR, mean < 0.01 ng/g Urban 2×: n = 3–7, DF NR, mean < 0.01 ng/g Rural non-amended: n = 3–7, DF NR, mean < 0.01 ng/g Rural 0.5×: n = 3–7, DF NR, mean = 0.16 ng/g (LOQ = 0.01 ng/g) Greenhouse study: Nonamended: n = 3–5, DF NR, mean = 0.44 ng/g Industrially impacted: n = 3–5, DF NR, mean = 1.38 ng/g Municipal: n = 3–5, DF NR, mean = 5.11 ng/g (LOQ not reported)
Canada			
Cabrerizo et al. (2018)	Canada (Melville and Cornwallis Islands)	Catchment areas of lakes in the Cape Bounty Arctic Watershed Observatory on southern Melville Island (West, East, and Headwater lakes) during summer (late July-early August) 2015 and 2016, representing an environment largely unimpacted by direct human activity; data for 19 sampling sites available (S6, S11–S28). Catchment areas of lakes on Cornwallis Island (Resolute, North, Small, Meretta, 9 Mile, and Amituk lakes) near the community of Resolute Bay during summer (late July-early August) 2015 and 2016.	Melville Island lakes: n = 19, DF[a] 95%, range = <LOD–0.0070 ng/g dw Cornwallis Island lakes: n = 8, DF[a] 100%, mean[a] (range) = 0.0267 (0.0003–0.1062) ng/g dw (LOD = 0.0001–0.018 ng/g for all PFAS)

Study	Location	Site Details	Results
		Resolute Bay has a military and civilian airport which discharged its wastewaters into the upper area of the catchment until 1997, three old solid waste landfills 1.5–2 km west of the airport used until the mid-1990s, and Arctic research and military training facilities close to the airport that support activities such as vehicle use, firefighting, and construction/demolition; eight sampling sites (S29–S36).	
Mejia-Avendaño et al. (2017)	Canada (Lac-Mégantic, Quebec)	Site of July 2013 Lac-Mégantic train accident where 63 out of 72 train cars carrying 8 million liters of crude oil derailed and a major oil fire ignited. Seven types of AFFFs and approximately 33,000 L of AFFF concentrates were used. Samples were collected in July 2013 weeks after the accident from the western shores of Chaudière River, at the point where the oil and AFFF runoff reached the river, approximately 500 m from the edge of the derailment site; in July 2015 from the fire burn site and adjacent area in downtown Lac-Mégantic where the soil was continuously excavated for remediation (the site was the closest to the accident site among the areas open to sampling); and from a background, nonimpacted area next to Chaudière River, about 5 km from the accident site, on the east shore of the river and on the opposite side of the accident.	Background: n = 3, DF NR, mean = 0.015 ng/g dw 2013: n = 12 (from 12 sites), DF[a] 92%, range = <LOD–0.747 ng/g dw 2015: n = 11 (from 9 sites), DF[a] 73%, range = <LOD–0.234 ng/g dw (LOD = 0.01 ng/mL; LOQ = 0.02 ng/mL)
Dreyer et al. (2012)	Canada (Ottawa, Ontario)	Two ombrotrophic Mer Bleue bog peat core samples collected in October 2009 and cut into 5-cm segments (nine segments for the first core, eight segments for the second core); Mer Bleue selected because it is undisturbed and well investigated and is located in a meltwater channel of the postglacial Ottawa River. Peat cores sampled to determine their suitability for determining historic atmospheric contamination; contaminants present due to atmospheric deposition only. The year for each segment was estimated through dating of Mer Bleue peat cores collected in the same year for a different study.	First core (first parallel; second parallel): 2009: 0.131; 0.040 ng/g 2006: 0.028; 0.033 ng/g 2001: 0.017; 0.005 ng/g 1992: 0.006; 0.010 ng/g 1983: 0.010; 0.012 ng/g 1973: 0.014; 0.029 ng/g 1962: 0.025; 0.021 ng/g 1945: 0.014; 0.022 ng/g 1927: 0.019; 0.015 ng/g 1912: 0.019; 0.017 ng/g Second core (first parallel; second parallel): 2009: ND; 0.157 ng/g 2006: 0.046 ng/g; ND 2001: 0.006 ng/g; ND 1992: ND; 0.014 ng/g

Study	Location	Site Details	Results
			1983: 0.010; 0.018 ng/g 1973: 0.011; 0.007 ng/g 1962: 0.017; 0.015 ng/g 1945: 0.012; 0.008 ng/g 1927: ND; ND (IDL = 0.0022 ng/g) *Authors estimated the year for each core segment using cores for a different study that underwent dating
Europe			
Groffen et al. (2019)	Belgium (Antwerp)	3M perfluorochemical plant and four sites with increasing distance from plant were selected based on prior biomonitoring studies in the vicinity of the plant. The four sites are: Vlietbos (1 km SE from 3M), Rot-Middenvijver (2.3 km ESE from 3M), Burchtse Weel (3 km SE from 3M), and Fort 4 (11 km SE from 3M). Samples collected in June 2016.	Plant: n = 13, DF 31%, mean, median (range) = 6.88, <LOQ (<LOQ–32) ng/g dw 1 km from plant: n = 10, DF 0% 2.3 km from plant: n = 10, DF 0% 3 km from plant: n = 10, DF 0% 11 km from plant: n = 14, DF 0% (LOQ = 4.91 ng/g dw)
Dauchy et al. (2019)	France (unspecified)	Site where fluorosurfactant-based foams have been used extensively. From 1969 to 1984, the site was an oil refinery, with the exact location of the firefighting training area, frequency of training sessions, and history of firefighting training activities unknown. From 1987 to date, it has been a large training area for firefighters. Samples collected in six areas from two sampling campaigns. First sampling campaign collected 30 soil cores between 2 m and 4 m in June 2015 from areas 1–5 (composite soil samples collected every 25 cm in the topmost meter and then every 50 cm). Second sampling campaign collected 14 soil cores between 4 m and 15 m from areas 1–6 (thickness of composite soil samples ranged from 25 cm to 100 cm) in October 2016. Area 1 stored raw oil products when the oil refinery was operating; a preliminary survey showed hydrocarbon traces in the area, suggesting that an incident had occurred and that fluorinated surfactants could have been used. Area 2 is one of the main areas used for firefighting activities since 1987; training sessions held directly on the ground before 10-cm	Area 1: SC-71, SC-72, SC-73, SC-74, SC-75 = <2 ng/g dw (1–2 m) SC-76, SC-77, SC-78 = <2 ng/g dw (0–1 m) Area 2: SC-58 = 2,344, 322, 277, 101, 95, 147 ng/g dw (0–0.25, 0.5–0.75, 1–1.5, 2–2.5, 3–3.5, 3.5–4 m) SC-59 = 79, 37, 113, 61, 25 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 3–3.5 m) SC-60 = 37, <20, 6, 5, 9, 6, <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 2–2.5, 2.5–3 m) SC-61 = 88, <20, 11, 13, <20, 30, 58 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m) SC-62 = 231, <20, 25, 59, 46 ng/g dw (0–0.25, 0.5–0.75, 1–1.5, 2–2.5, 3.5–4 m) SC-63 = 112, 54, 126, 53, 32, 12, 11, 10, 6 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5 m)

Study	Location	Site Details	Results
		thick concrete slab was built in the 1990s. Area 3 was used for firefighting activities since 1987 and is situated on a 1-meter thick concrete slab on the foundations of the former oil refinery. Area 4 corresponds to the site's WWTP where sludge and sediment from a lagoon were stored directly on the ground; influents of the WWTP are highly contaminated by PFAS. Area 5 was used for firefighting training exercises by the former oil refinery. Area 6 is used for firefighting exercises with tank trucks.	SC-64 = 6, 15, 29, 49, 14, 6, <4 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m) SC-65 = 12, 30, 25, <4, 17, 18, 21 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 3–3.5, 3.5–4 m) SC-66 = 8, <4, <4, 8, 3, 2, <2, <2, <2, 8 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5, 3.5–4 m) SC-58b = 17, 2, <10, 4 ng/g dw (4–5, 5–6, 9–10, 14–15 m) SC-59b = 14, 39, 34, 11, 63 ng/g dw (3–4, 4–5, 6–7, 9–10, 14–15 m) SC-65b = 3, 15, 5, 5 ng/g dw (4–5, 7–9, 9–11, 14–15 m) SC-67 = 4, 5, 5, 12 ng/g dw (0–1, 1.3–2, 2–3, 4–5 m) Area 3: SC-40 = <4 ng/g dw (0–1 m) SC-43 = <2 ng/g dw (1–2 m) SC-45 = <2 ng/g dw (0–1 m) SC-47 = 18 ng/g dw (0–1 m) SC-48 = <4 ng/g dw (0–1 m) SC-41 = 5, <10 ng/g dw (0–0.25, 1–2 m) SC-42 = <10 ng/g dw (0–0.25, 1–2, 3–4 m) Area 4: SC-33 = <20 ng/g dw (0–1 m) SC-34 = 13 ng/g dw (1–2 m) SC-35 = <2 ng/g dw (0–1m) SC-36 = <4 ng/g dw (0.3–1 m) SC-37 = 43 ng/g dw (0.1–1.1 m) SC-37b = 11, 6, <10 ng/g dw (0–0.25, 1–1.5, 3–4 m) SC-38 = 35, 2 ng/g dw (0.25–1, 2–3 m) Area 5: SC-10 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5, 2.5–3, 3–3.5 m) SC-11 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75, 0.75–1, 1–1.5, 1.5–2, 2–2.5 m)

Study	Location	Site Details	Results
			SC-12 = <2 ng/g dw (0–0.25, 0.25–0.5, 0.5–0.75 m) Area 6: SC-21 = 148, 290, 273, 278, 98 ng/g dw (0–0.25, 0.25–1, 2–3, 8–9, 13–15 m) SC-22 = 8, 11, 6, 7 ng/g dw (0–0.25, 0.25–1, 1–2, 3–4 m) SC-23 = 59, 29, 29, 16 ng/g dw (0–0.25, 0.25–1, 1–2, 3–4 m) SC-24 = 85, 31, 43 ng/g dw (0–0.25, 1–2, 3–4 m) SC-25 = 7, 26 ng/g dw (0–0.25, 1.5–2 m) SC-26 = 4, 3 ng/g dw (2–3, 4–5 m) (LOQ = 2 ng/g dw)
Skaar et al. (2019)	Norway (Ny-Ålesund)	Samples collected in June 2016 in and around the international research facilities (Ny-Ålesund) near local firefighting training site. Background soil samples were collected at representative locations.	Background: n = 8, DF 0% Contaminated: n = 2, DF[a] 100%, mean[a] (range) = 29.42 (13.82–45.02) ng/g dw (IDL = 0.003 ng; LOD = 0.001 ng/g dw; LOQ = 0.002 ng/g dw) *Values reported for sum of branched and linear PFHxS isomers *Table 1 and Table S2 reported a total of nine samples across background and contaminated sites; however, Tables S11 and S13 report a total of ten samples (two contaminated sites from Table S11 and eight background sites from Table S13
Hale et al. (2017)	Norway (Gardermoen)	Samples collected in June 2015 from six locations around a firefighting training facility west of the Oslo airport site. Samples were taken at 0–1 m, 1–2 m, 2–3 m, and 3 to groundwater table level (which was in all cases above 4 m). Facility was established in 1989 and AFFF was used extensively. AFFF containing PFOS was banned at the facility in 2007 and a complete ban on organofluorine AFFF was enforced in 2011. The soil is known to be contaminated with a range of perfluorinated compounds.	n = 22 (across all depths), DF 36%, range = 3.0–25.3 ng/g (LOD = 1 ng/g) *Range reported for detects *The DF and range extracted are reported in the results (Section 3.1); however, Table S2 of the individual sample data show all concentrations ranging from <1.8 to <2.5 ng/g
Filipovic et al. (2015)	Sweden (Stockholm)	Five locations at a closed air force base F18 in Tullinge Riksten, 19 km south of Stockholm city	Intermediate soil depot: n = 10, DF 80%, range = <0.02–3.1 ng/g dw

Study	Location	Site Details	Results
		center, where AFFFs were used. Samples collected in two sampling campaigns in December 2011 and May 2012. The air force base was formally demobilized in 1986 but continued to be used as an air force school for combat command and air surveillance until 1994. Of note, the air force base encountered numerous accidents and incidents during the transfer from propeller era to the jet engine era, including planes crashing upon takeoff and landing, fire incidents, accidental dispersion of jet engine starting fuel. The area was sold to a land developer in 1996 and is in the process of being transformed into a municipal area. Of the five sites samples, the intermediate soil depot site (sampling depth 0–3 m) and the soil depot site (sampling depth 0–3 m) were assembled by the land developing corporation; the J34 Hawker Hunter site was the collision site of two J34 Hawker Hunter aircrafts in the 1960s, resulting in huge amounts of firefighting foam being dispersed (sampling depth 0.5–1.0 m); the main firefighting training facility was intensively used between 1970–1985 and sparsely used between 1985–1994 (sampling depth 0.5–1.0 m); and the old fire station was used to store AFFF stock solution (sampling depth 0.5–1.0 m).	J34 Hawker Hunter site: n = 5, DF 0% Main firefighting training facility: n = 15, DF 100%, mean[a] (range) = 3.9 (0.1–21.3) ng/g dw Old fire station: n = 5, DF 100%, mean[a] (range) = 3.7 (1.6–6) ng/g dw Soil depot: n = 10, DF 40%, range = <0.02–0.33 ng/g dw (MDL = <0.02–<0.1 ng/g dw)
Harrad et al. (2020)	Ireland (multiple cities)	Samples collected from ten municipal solid waste landfills upwind and downwind at each site between November 2018 and January 2019. At each upwind/downwind location, nine sub-samples of soil were taken in a "W" formation. Samples were collected from the same areas as air samples and were taken within the boundaries of the landfill operational facility. Soil used as capping on landfill cells was not sampled to ensure soil samples were not collected from soil placed after landfill operations ceased and that farming activities would not influence concentrations found. Waste accepted by the landfills included: municipal solid waste, industrial (non-hazardous) waste, construction & demolition, and biomedical waste.	Downwind: n = 9, DF[a] 11%, mean, median (range) = 0.00077, <0.001 (<0.001–0.0029) ng/g dw Upwind: n = 7, DF[a] 57%, mean, median (range) = 0.0018, 0.0023 (<0.001–0.0037) ng/g dw (LOD = <0.001 ng/g dw) *Non-detects replaced by ½ LOD *Soil samples from three upwind locations and one downwind location destroyed in transit from field to laboratory
Grønnestad et al. (2019)	Norway (Granåsen, Jonsvatnet)	Upper layer soil samples (3–10 cm in depth) collected in June 2017 and 2018 from Granåsen (skiing area)	Reference area: n = 10, DF 0% Skiing area: n = 10, DF 0%

Study	Location	Site Details	Results
		and Jonsvatnet (reference site). Five samples per year were analyzed for each site. Located 10 km from Trondheim city center, Granåsen is the main arena for winter sports in Trondheim and hosts an annual ski jumping World Cup event and regional, national, and international competitions in cross-country skiing. Located 15 km away from Trondheim city center, Jonsvatnet is a natural forest area not used for ski-sports and is in the vicinity of an ecological farm next to Lake Jonsvatnet. The two study areas have similar vegetation.	(LOQ = 0.032 ng/g dw) *For the reference area, Table S2 reported a DF = 10% but a range, mean, and standard deviation of <LOQ
Sammut et al. (2019)	Malta	Six surface soil samples (#10, 14, 15, 18, 20, and 22) collected between June and August 2015 from random small urban fields	n = 6, DF[a] 17%, range = ND–0.12 ng/g (LOD = 0.40 ng/g; LOQ = 0.50 ng/g) *The study reported two samples <LOQ, two samples ND, one sample <LOD, and one sample with a detectable value

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; dw = dry weight; GCA = groundwater contamination area; LOD = limit of detection; LOQ = limit of quantitation; MDL = method detection limit; 0.5×, 1×, or 2× = ½, 1, or 2 times the agronomic rate of biosolids application to meet nitrogen requirements of the crop; m = meter; cm = centimeter; ND = not detected; NR = not reported; RL = reporting limit; WWTP = wastewater treatment plant.

[a] Venkatesan and Halden (2014) reported that PFHxS was not consistently detected in biosolids-amended mesocosms.

D.3.7. Sediment

When released into water, PFHxS is not expected to adsorb to suspended solids and sediments based on its physico-chemical properties (NCBI, 2022a). However, PFHxS was detected in 28% of sediment samples collected along the Mississippi River shoreline in the vicinity of a PFAS production facility in Minnesota, at a maximum concentration of 11.5 ng/g (NCBI, 2022a; ATSDR, 2021; 3M, 2007). PFHxS was also detected in 96% of sediment samples collected from two coves of the Mississippi River (East and West coves) located at the eastern and western ends of the PFAS production facility, at a maximum concentration of 126 ng/g (ATSDR, 2021; 3M, 2007). PFHxS was not detected in any Mississippi River transect sediment samples (collected at points crossing the river near the PFAS facility) (ATSDR, 2021; 3M, 2007). PFHxS has been detected in sediment core samples collected from three Canadian Arctic lakes in 2003 and 2005 at concentrations ranging from approximately 1 ng/g to 10 ng/g (NCBI, 2022a; Stock et al., 2007).

D.4. Recommended RSC

The EPA followed the Exposure Decision Tree approach to determine the RSC for PFHxS (USEPA, 2000b). The EPA first identified the U.S. general population as the population of concern (Box 1; see Section 2.4.2). Second, the EPA identified several relevant PFHxS exposures and pathways (Box 2), including dietary consumption, incidental oral consumption via exposure to dust, consumer products, and soil, dermal exposure via soil, consumer products, and dust, and respiration via ambient air. Several of these may be potentially significant exposure sources. Third, the EPA determined that there was not adequate quantitative data to describe the central tendencies and high-end estimates for all of the potentially significant sources (Box 3). For example, studies from Canada and Europe indicate that indoor air may be a significant source of exposure to PFHxS. At the time of the literature search, the EPA was unable to identify studies assessing PFHxS concentrations in indoor air samples from the U.S. and therefore, the agency does not have adequate quantitative data to describe the central tendency and high-end estimate of exposure for this potentially significant source in the U.S. population. However, the agency determined there were sufficient data, physical/chemical property information, fate and transport information, and/or generalized information available to characterize the likelihood of exposure to relevant sources (Box 4). Notably, based on the studies summarized in the sections above, there are significant known or potential uses/sources of PFHxS other than drinking water (Box 6), though there is not information available on each source to make a characterization of exposure (Box 8A). For example, there are several studies from the U.S. indicating that PFHxS may occur in multiple food products (e.g., eggs, seafood, meats, vegetables, fruit) and consumer products (e.g., building materials, clothing, furniture). However, the majority of studies examined very few samples (i.e., n=1-5) of each type of media. Therefore, it is not possible to determine which source, if any, can be considered major or minor contributors to total PFHxS exposure. Given these considerations, following recommendations of the Exposure Decision Tree (USEPA, 2000b), the EPA recommends an RSC of 20% (0.20) for PFHxS.