

Drinking Water Health Advisory:
Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and
Related Compound Potassium Perfluorobutane Sulfonate
(CASRN 29420-49-3)

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Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)**

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Abbreviations and Acronyms

6:2 FTS	6:2 fluorotelomer sulfonic acid	EGLE	Michigan Department of Environment, Great Lakes, and Energy
ADAF	age-dependent adjustment factor	EPA	United States Environmental Protection Agency
AFFF	aqueous film-forming foam	Eq	equation
ANSI	American National Standards Institute	EU	European Union
ASTSWMO	Association of State and Territorial Solid Waste Management Officials	FCID	Food Commodity Intake Database
ATSDR	Agency for Toxic Substances and Disease Registry	FCM	food contact material
BMD	benchmark dose	FDA	United States Food and Drug Administration
BMDL	benchmark dose lower limit	fw	fresh weight
bw or BW	body weight	GCA	groundwater contamination area
CASRN	Chemical Abstracts Service Registry Number	GenX chemicals	hexafluoropropylene oxide dimer acid and its ammonium salt
CDPHE	Colorado Department of Public Health and Environment	HA	Health Advisory
CPHEA	Center for Public Health and Environmental Assessment	HED	human equivalent dose
CSF	cancer slope factor	HI	hazard index
DF	detection frequency	HIDOH	Hawai'i Department of Health
DOH	Department of Health	HQ	hazard quotient
DQO	data quality objective	IBWA	International Bottled Water Association
dw	dry weight	IDEM	Indiana Department of Environmental Management
DWI	drinking water intake	iHA	Interim Health Advisory
DWI-BW	body weight-adjusted drinking water intake	Illinois EPA	Illinois Environmental Protection Agency
DWTP	drinking water treatment plant	ITRC	Interstate Technology and Regulatory Council
ECHA	European Chemicals Agency	K _{aw}	air-water partition coefficient
EEE	electronic equipment	km ²	square kilometers
EF	exposure factor	K ⁺ PFBS	potassium perfluorobutane sulfonate
EFH	Exposure Factors Handbook	L	liters
EFSA	European Food Safety Authority		

L/kg bw-day	liters per kilogram body weight per day	NOAEL	no observed adverse effect level
L/m ² ·hr	liters per square meter per hour	NRSA	National Rivers and Streams Assessment
LC/MS/MS	liquid chromatography/tandem mass spectrometry	ODH OEHHA	Ohio Department of Health Office of Environmental Health Hazard Assessment
LOAEL	lowest observed adverse effect level	Ohio EPA	Ohio Environmental Protection Agency
LOQ	limit of quantification	ORD	Office of Research and Development
Maine DEP	Maine Department of Environmental Protection	OST	Office of Science and Technology
MCLG	Maximum Contaminant Level Goal	OW	Office of Water
MDH	Minnesota Department of Health	PECO	populations, exposures, comparators, and outcomes
mg/kg bw-day	milligrams per kilogram body weight per day	PFAA PFAS	perfluoroalkylated acid per- and polyfluoroalkyl substances
MPa	megapascal		
MRL	minimum reporting level	PFBA	perfluorobutanoic acid
MS/MS	tandem mass spectrometry	PFBS	perfluorobutane sulfonic acid
MW	molecular weight		
NCHS	National Center for Health Statistics	PFC PFCA	perfluorinated chemical perfluoroalkyl carboxylic acid
NCOD	National Contaminant Occurrence Database	PFDA	perfluorodecanoic acid
ND	non-detect	PFDoDA	perfluorododecanoic acid
NDEP	Nevada Division of Environmental Protection	PFDoS	perfluorododecane sulfonic acid
NF	nanofiltration	PFDS	perfluorodecane sulfonic acid
ng/g	nanograms per gram		
ng/kg bw-day	nanograms per kilogram body weight per day	PFHpA PFHpS	perfluoroheptanoic acid perfluoroheptane sulfonic acid
ng/L	nanograms per liter		
ng/mL	nanograms per milliliter	PFHxA	perfluorohexanoic acid
NHANES	National Health and Nutrition Examination Survey	PFHxS	perfluorohexane sulfonic acid
NHIS	Human Nutrition Information Service	PFNA PFNS	perfluorononanoic acid perfluorononane sulfonic acid
NOAA	National Oceanic and Atmospheric Administration	PFOA PFOS	perfluorooctanoic acid perfluorooctanesulfonic acid

PFOSA	perfluorooctanesulfonamide	UF _A	interspecies uncertainty factor
PFPA	perfluoropentanoic acid		
PFPeA	perfluoropentanoic acid	UF _C	composite uncertainty factor
PFPS	perfluoropentane sulfonic acid		
		UF _D	database uncertainty factor
PFSA	perfluoroalkane sulfonate	UF _H	intraspecies uncertainty factor
PFT _{Tr} DA	perfluorotridecanoic acid		
PFT _{Tr} S	perfluorotridecane sulfonic acid	UF _L	LOAEL to NOAEL extrapolation uncertainty factor
PFUnDA	perfluoroundecanoic acid		
PFUnS	perfluoroundecane sulfonate	UF _S	subchronic to chronic exposure duration extrapolation uncertainty factor
pg/cm ²	picograms per square centimeter		
pg/m ³	picograms per cubic meter	WEEE	wastes of electrical and electronic equipment
PM	particulate matter	Wisconsin DHS	Wisconsin Department of Health Services
PMN	pre-manufacture notice		
PND	post-natal day	WTP	water treatment plant
POD	point of departure	ww	wet weight
ppm	parts per million	WWTP	wastewater treatment plant
ppt	parts per trillion	µg/kg	micrograms per kilogram
POE	point-of-entry	µg/m ²	micrograms per square meter
POU	point-of-use		
PWS	public water system		
QC	quality control		
REACH	Registration, Evaluation, Authorization and Restriction of Chemicals		
RfD	reference dose		
RO	reverse osmosis		
RPF	relative potency factor		
RSC	relative source contribution		
SDWA	Safe Drinking Water Act		
T ₄	thyroxine		
TCEQ	Texas Commission on Environmental Quality		
TSCA	Toxic Substances Control Act		
UCMR	Unregulated Contaminant Monitoring Rule		
UF	uncertainty factor(s)		

Executive Summary

Perfluorobutane sulfonic acid (PFBS; CASRN 375-73-5) and its related compound potassium perfluorobutane sulfonate (K^+ PFBS; CASRN 29420-49-3) are shorter-chain members of a group of substances known as per- and polyfluoroalkyl substances (PFAS). In water, K^+ PFBS fully dissociates to the deprotonated anionic form of PFBS (PFBS $^-$; CASRN 45187-15-3) and the K^+ cation at environmental pH levels (pH 4–9). Herein, these three PFBS chemical forms are referred to collectively as PFBS.

PFBS is a replacement chemical for the longer-chain perfluorooctane sulfonic acid (PFOS), a PFAS that was voluntarily phased out (with some exceptions) by its primary U.S. manufacturer (3M Company) between 2000 and 2002 (U.S. EPA, 2007; 3M, 2002). Prior to its use as a replacement for PFOS, PFBS was produced as a byproduct during production of perfluorooctane sulfonyl fluoride-based chemicals and was present in consumer products as an impurity (AECOM, 2019). PFBS is used in the manufacture of paints, cleaning agents, and water- and stain-repellent products and coatings (U.S. EPA, 2021a). PFBS has been detected in drinking water, groundwater, and surface water and has been found in dust, carpeting and carpet cleaners, floor wax, foods including seafood (fish and shellfish) and vegetables, food packaging, indoor and outdoor air, soil, biosolids, and some consumer products (ATSDR, 2021; U.S. EPA, 2021a; see Section 3.3.1). 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 land application of contaminated biosolids (ATSDR, 2021; U.S. EPA, 2021a). PFBS is water soluble (52.6 g/L at 22.5–24 °C for the potassium salt) and volatilization from water surfaces is not expected to be an important fate process (ATSDR, 2021; U.S. EPA, 2021a). PFBS has been detected in the serum of humans in the general population (U.S. EPA, 2021a).

The U.S. Environmental Protection Agency (EPA) is issuing a lifetime noncancer drinking water Health Advisory (HA) for PFBS of 2,000 nanograms per liter (ng/L) or 2,000 parts per trillion (ppt). This is the first HA for PFBS and its finalization fulfills a commitment described in EPA's PFAS Strategic Roadmap (U.S. EPA, 2021b). The final PFBS toxicity assessment titled *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)* (U.S. EPA, 2021a) serves as the basis of the toxicity information used to derive the lifetime noncancer HA for PFBS. The critical adverse effect is thyroid effects in mice (specifically, decreased serum levels of the thyroid hormone thyroxine [T_4]) observed at post-natal day (PND) 1, after 20-day gestational exposure to PFBS (Feng et al., 2017). Based on this critical effect, a chronic reference dose (RfD) of 3×10^{-4} milligrams per kilogram body weight per day (mg/kg bw-day) for PFBS was derived.

In accordance with EPA's *Recommended Use of Body Weight^{3/4} as the Default Method in Derivation of the Oral Reference Dose* (U.S. EPA, 2011), serum PFBS half-lives were used to scale a toxicologically equivalent dose of orally administered PFBS from animals to humans. Following EPA's *Benchmark Dose Technical Guidance* (U.S. EPA, 2012b), benchmark dose (BMD) modeling of thyroid effects in offspring after gestational exposure to PFBS resulted in a benchmark dose lower confidence limit (BMDL) for 0.5 SD change from the control (BMDL_{0.5SD}) human equivalent dose (HED) of 0.095 mg/kg bw-day. This HED point of departure (POD) based on decreased levels of T_4 in newborn offspring was divided by a composite uncertainty factor (UF_C) of 300 to derive the chronic RfD.

Sensitive populations or life stages within the general population indicated by the critical study used to derive the chronic RfD for PFBS are the developing embryo and fetus. Therefore, drinking water exposure to pregnant women as well as women of childbearing age, who may be or become pregnant, were identified as two sensitive populations or life stages. EPA selected the body weight-adjusted drinking water intake (DWI-BW) exposure factor (EF) of 0.0354 liters per kilogram body weight per day (L/kg bw-day) for women of childbearing age because it is more health protective than the DWI-BW for pregnant women. However, PFBS HA values, when rounded to one significant figure, were the same when calculated using EFs for either women of childbearing age, pregnant women, or the general population (all ages).

The physical/chemical properties and available exposure information for PFBS suggest multiple potentially significant exposure sources (seafood, other foods, indoor air, and some consumer products) other than drinking water ingestion. However, information is not available to quantitatively characterize the relative exposure contributions from non-drinking water exposure sources. Therefore, following the Exposure Decision Tree approach within EPA's 2000 *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (U.S. EPA, 2000a), EPA recommends a relative source contribution (RSC) of 20 percent (0.20) for use in PFBS HA derivation.

There is insufficient toxicity information available to derive a one-day HA for PFBS. Derivation of a 10-day HA was considered because the subchronic and chronic RfDs are both based on a 20-day exposure study, which may be used to derive a 10-day HA. However, EPA did not derive a 10-day HA because the critical health effect on which the chronic RfD used to calculate the lifetime HA is based (i.e., decreased serum levels of T₄ in newborn mice) resulted from PFBS exposure during a developmental life stage (Feng et al., 2017). EPA's risk assessment guidelines for developmental toxicity indicate that adverse effects can result from even brief exposure during a critical period of development (U.S. EPA, 1991). The critical study observed persistent health effects into adulthood, suggesting the potential for long-term health consequences of gestational-only PFBS exposure and that gestation is at least one critical exposure window for PFBS. Therefore, the lifetime HA for PFBS of 2000 ng/L and the chronic RfD from which it is derived are considered applicable to short-term PFBS exposure (including during pregnancy) as well as lifetime exposure via drinking water. This lifetime HA applies to PFBS (CASRN 375-73-5), K⁺PFBS (CASRN 29420-49-3), and PFBS⁻ (CASRN 45187-15-3).

No studies evaluating the carcinogenicity of PFBS in humans or animals were identified (U.S. EPA, 2021a). In accordance with EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b), EPA concluded that there is "*Inadequate Information to Assess Carcinogenic Potential*" for PFBS by any route of exposure (U.S. EPA, 2021a). Therefore, a 10⁻⁶ cancer risk concentration cannot be derived for PFBS at this time.

EPA developed two analytical methods to quantitatively assess drinking water for targeted PFAS that include PFBS: EPA Method 533 (U.S. EPA, 2019b), which has a quantitation limit of 3.5 ng/L for PFBS, and EPA Method 537.1, Version 2.0 (U.S. EPA, 2020b), which has a quantitation limit of 6.3 ng/L for PFBS. These analytical methods can both effectively and accurately measure PFBS in drinking water at levels significantly lower than the lifetime HA of 2,000 ng/L. EPA finished drinking water sampling results have not identified PFBS levels that approached the lifetime HA of 2,000 ng/L. However, treatment technologies, including reverse osmosis (RO), nanofiltration (NF), and sorption-based processes such as activated carbon and

ion exchange are available and have been shown to remove PFBS in drinking water; however, sorption has less efficacy with PFBS than similar longer-chained PFAS.

1.0 Introduction and Background

The Safe Drinking Water Act (SDWA) (42 U.S.C. § § 300f - 300j-27) authorizes the U.S. Environmental Protection Agency (EPA) to develop drinking water Health Advisories (HAs).¹ HAs are national non-enforceable, non-regulatory drinking water concentration levels of a specific contaminant at or below which exposure for a specific duration is not anticipated to lead to adverse human health effects.² HAs are intended to provide information that tribal, state, and local government officials and managers of public water systems (PWSs) can use to determine whether actions are needed to address the presence of a contaminant in drinking water. HA documents reflect the best available science and include HA values as well as information on health effects, analytical methodologies for measuring contaminant levels, and treatment technologies for removing contaminants from drinking water. EPA's lifetime HAs identify levels to protect all Americans, including sensitive populations and life stages, from adverse health effects resulting from exposure throughout their lives to contaminants in drinking water.

In April 2021, EPA published a final toxicity assessment for two per- and polyfluoroalkyl substances (PFAS): perfluorobutane sulfonic acid (PFBS) and its related compound potassium perfluorobutane sulfonate (K⁺PFBS) (U.S. EPA, 2021a). K⁺PFBS differs from PFBS by being associated with a potassium ion. In water, K⁺PFBS fully dissociates to the deprotonated anionic form of PFBS (PFBS⁻; CASRN 45187-15-3) and the K⁺ cation at environmental pH levels (pH 4–9). Herein, these three PFBS chemical forms are referred to collectively as PFBS. Completing the toxicity assessment was an essential step to better understanding the potential human health effects of exposure to PFBS. The chronic noncancer reference dose (RfD) calculated in the toxicity assessment allows EPA to develop a final lifetime HA that will help communities make informed decisions to better protect human health. The final PFBS HA satisfies a commitment described in EPA's PFAS Strategic Roadmap (U.S. EPA, 2021b).

1.1 History under SDWA

PFBS is not currently regulated under SDWA. The 1996 amendments to SDWA require that EPA issue a new list of unregulated contaminants (once every five years) to be monitored by PWSs.³ Under the Unregulated Contaminant Monitoring Rule (UCMR), EPA samples drinking water systems to collect data for contaminants that are known or suspected to be found in drinking water and do not have health-based standards under SDWA. The first four UCMRs required monitoring of all large public drinking water systems (>10,000 people), and a subset of smaller systems serving <10,000 people. PFBS was one of six PFAS monitored in drinking water under the third UCMR (UCMR 3) between 2013 and 2015 (U.S. EPA, 2012a). It is also one of 29 PFAS that will be monitored under the fifth UCMR (UCMR 5) between 2023 and 2025 (U.S. EPA, 2021c). The collection of drinking water occurrence data supports EPA's future

¹ SDWA § 1412(b)(1)(F) authorizes EPA to “publish health advisories (which are not regulations) or take other appropriate actions for contaminants not subject to any national primary drinking water regulation.” www.epa.gov/sites/default/files/2020-05/documents/safe_drinking_water_act-title_xiv_of_public_health_service_act.pdf

² This document is not a regulation and does not impose legally binding requirements on EPA, states, tribes, or the regulated community. This document is not enforceable against any person and does not have the force and effect of law. No part of this document, nor the document as a whole, constitutes final agency action that affects the rights and obligations of any person. EPA may change any aspects of this document in the future.

³ SDWA § 1445(a)(1)(D)(2)(B) — “Not later than 3 years after the date of enactment of the Safe Drinking Water Act Amendments of 1996 and every 5 years thereafter, the Administrator shall issue a list pursuant to subparagraph (A) of not more than 30 unregulated contaminants to be monitored by public water systems and to be included in the national drinking water occurrence data base maintained pursuant to subsection (g).”

regulatory determinations and may support additional actions to protect public health (U.S. EPA, 2021c).

1.2 Current Advisories and Guidelines

Table 1 provides final drinking water guideline values for PFBS that have been developed by states. The state values range from 100 to 667,000 parts per trillion (ppt) or nanograms per liter (ng/L); this broad range of values may in part reflect differences in the type of value derived, state guidance/methodology for deriving values, or data included in the evaluation (see references for more details).

Table 1. State Guideline Values for PFBS

State ^{a,b}	PFBS Level (ppt [ng/L])	Standard/Guidance	Type of Medium	Reference
California	500	Notification level	Drinking water	California OEHHA (2021)
Colorado	400,000	Translation level	Groundwater; Surface water	CDPHE (2020b)
Hawai'i	600	Environmental action levels	Groundwater	HIDOH (2021)
Illinois	2,100	Health-based guidance level	Drinking water; Groundwater	Illinois EPA (2021a)
Indiana	> 2,100	Action level	Drinking water	IDEM (2022)
Maine	400,000	Remedial action guideline	Groundwater	Maine DEP (2018)
Michigan	420	Maximum contaminant level	Drinking water; Groundwater	EGLE (2020)
Minnesota	100	Health-based value	Drinking water; Groundwater	MDH (2022)
Nevada	667,000	Basic comparison level	Drinking water	NDEP (2020)
Ohio	2,100	Action level	Drinking water	Ohio EPA and ODH (2022)
Pennsylvania	10,000	Medium-specific concentration	Groundwater; Residential use	Environmental Quality Board (2021)
	29,000	Medium-specific concentration	Groundwater; Non-residential use	

State ^{a,b}	PFBS Level (ppt [ng/L])	Standard/Guidance	Type of Medium	Reference
Texas	34,000	Tier 1 protective concentration level	Groundwater	TCEQ (2021)
Washington	345	State action level	Drinking water	Washington DOH (2021)
Wisconsin	450,000	Recommended enforcement standard	Groundwater	Wisconsin DHS (2020)
	90,000	Recommended preventive action limit	Groundwater	

Notes:

^a The information was compiled from two sources: 1) EPA regional office outreach by EPA’s Office of Science and Technology (OST) in March 2022; and 2) information from the Interstate Technology and Regulatory Council’s (ITRC) *Standards and guidance values for PFAS in groundwater, drinking water, and surface water/effluent (wastewater)* PFAS Water and Soil Values Table, last updated in April 2022 (available for download here: <https://pfas-1.itrcweb.org/fact-sheets/>).

^b Only states with final guidelines are included; other states may be developing guidelines for PFBS.

In 2020, the European Chemicals Agency (ECHA) adopted an agreement that identified PFBS as a “Substance of Very High Concern” (ECHA, 2020) based on a “very high potential for irreversible” human and environmental health effects, and properties including moderate bioaccumulation in humans, high persistence and mobility in the environment, high potential for long-range transport, and difficulty of remediating and purifying water.

Table 2 provides drinking water guideline values for PFBS that were developed by international agencies. The international guideline values range from 90 to 15,000 ppt or ng/L.

Table 2. International Guideline Values for PFBS

Country ^{a,b}	PFBS Level (ppt [ng/L])	Standard/Guidance	Type of Medium	Reference
Canada	15,000	Screening value	Drinking water	Health Canada (2016)
European Union (EU)	100 ng/L ^{c,d}	Parametric value	Water intended for human consumption	EU (2020)
	500 ng/L ^{c,e}	Parametric value	Water intended for human consumption	
Denmark	100 ^f	Health based	Groundwater	Danish EPA (2021)
Germany	6,000	Significance threshold	Groundwater	Von der Trenck et al. (2018)
Italy	3,000	Environmental quality standard	Drinking water	Valsecchi et al. (2017)

Country ^{a,b}	PFBS Level (ppt [ng/L])	Standard/Guidance	Type of Medium	Reference
Sweden	90 ^g	Administrative	Drinking water	Concawe (2016)

Notes:

^a The information was collected from the Interstate Technology and Regulatory Council’s (ITRC) *Standards and guidance values for PFAS in groundwater, drinking water, and surface water/effluent (wastewater)* PFAS Water and Soil Values Table, last updated in April 2022 (available for download here: <https://pfas-1.itrcweb.org/fact-sheets/>).

^b Only countries with guideline values provided in the ITRC table are included; other countries may be developing guidelines for PFBS.

^c Parametric values from Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption. By January 12, 2026, Member States shall take measures necessary to ensure that water intended for human consumption complies with the parametric values set out in Part B of Annex I in the EU Directive 2020/2184 (EU, 2020).

^d Pertains to a sum of a subset of 20 individual PFAS that includes PFBS: PFBA, PFPA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFBS, PFPS, PFHxS, PFHpS, PFOS, PFNS, PFDS, PFUnS, PFDoS, PFTrS.

^e Total PFAS

^f Applies to the individual results for PFOA, PFOS, PFNA, PFBA, PFBS, PFHxS, PFHxA, PFHpA, PFOSA, PFDA, 6:2 FTS, PFPS, PFHpS, PFNS, PFDS, PFUnS, PFDoS, PFTrS, PFPA, PFUnDA, PFDoDA, PFTrDA as well as the sum of concentrations of these 22 PFAS.

^g This limit also applies to the sum of PFOS, PFHxS, PFBS, PFOA, PFHpA, PFHxA and PFPeA.

1.3 Uses and Sources of PFBS

PFBS is a replacement chemical for perfluorooctane sulfonic acid (PFOS), a chemical that was voluntarily phased out (with some exceptions) by its primary U.S. manufacturer, 3M Company, by 2002 (3M, 2002; U.S. EPA, 2007). PFBS and its potassium salt were listed on the original EPA Toxic Substances Control Act (TSCA) Chemical Substance Inventory⁴ as existing chemicals that were already in commerce when TSCA was enacted in 1976 (15 U.S.C. § 2601 *et seq.*). Therefore, PFBS and its potassium salt were not subject to the pre-manufacture notice (PMN) reporting process. They are listed as “active” on the inventory but have not been reviewed under the TSCA New Chemicals program.⁵ EPA also evaluates existing chemicals under amended TSCA;⁶ however, to date, PFBS has not been designated as a high priority substance for risk evaluation. PFBS and its potassium salt are subject to Section 8 Chemical Data Reporting.⁷ While there has not been recent reporting on PFBS, in 2020 there was a report on the potassium salt (K⁺PFBS) for one industrial processing and use scenario⁸ but not for consumer/commercial uses.

Prior to its use as a replacement chemical, PFBS had been produced solely as a byproduct and was present in consumer products as an impurity (AECOM, 2019). Concerns arising in the early 2000s about the environmental persistence, bioaccumulation potential, and long half-lives in humans of longer-chain PFAS resulted in the use of shorter-chain PFAS such as PFBS as

⁴ TSCA Inventory. Available at <https://www.epa.gov/tsc-a-inventory/how-access-tsc-a-inventory>

⁵ Mandated by section 5 of TSCA, EPA’s New Chemicals program helps manage the potential risk to human health and the environment from chemicals new to the marketplace. Section 5 of TSCA is available at <https://www.epa.gov/assessing-and-managing-chemicals-under-tsc-a/15-usc-ch-53-toxic-substances-control-act>

⁶ On June 22, 2016, President Obama signed the Frank R. Lautenberg Chemical Safety for the 21st Century Act, which updates TSCA. Available at <https://www.congress.gov/114/plaws/publ182/PLAW-114publ182.pdf>

⁷ Basic information about Chemical Data Reporting available here <https://www.epa.gov/chemical-data-reporting/basic-information-about-chemical-data-reporting>

⁸ Section 8 reporting: Processing—incorporation into formulation, mixture, or reaction product; Sector: Electrical Equipment, Appliance, and Component Manufacturing; Function Category: Flame retardant

replacements for longer-chain PFAS in consumer products and applications (U.S. EPA, 2021a). PFBS and other shorter-chain PFAS possess the desired chemical properties of longer-chain PFAS, but have shorter half-lives in humans (U.S. EPA, 2021a).

Environmental releases of PFBS may result directly from the production and use of PFBS itself, production and use of PFBS-related substances for various applications, and/or from the degradation of PFBS precursors (i.e., substances that may form PFBS during use, as a waste, or in the environment). PFBS is used in the manufacture of paints, cleaning agents, and water- and stain-repellent products and coatings (U.S. EPA, 2021a). PFBS has also been used as a mist suppressant for chrome electroplating and has been detected in association with the use of aqueous film-forming foam (AFFF) (U.S. EPA, 2021a). PFBS has been detected in dust, carpeting and carpet cleaners, floor wax, and food packaging (ATSDR, 2021; U.S. EPA, 2021a).

1.4 Environmental Fate, Occurrence in Water, and Exposure to Humans

1.4.1 *Environmental Fate and Transport in the Environment*

The ionic nature of PFAS, including PFBS, influences physicochemical properties such as water or lipid solubility and bioaccumulative potential, which impacts environmental fate and transport and potential human health and ecological effects after exposure (U.S. EPA, 2021a). ECHA reports that PFBS is stable to hydrolysis, oxidation, and photodegradation in the atmosphere, and there have been no reports of abiotic degradation under environmental conditions (ECHA, 2019). 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 K_{oc} 1.2 to 2.7) (ECHA, 2019).

The Norwegian Environment Agency conducted a literature review of physicochemical properties and environmental monitoring data for PFBS to assist an evaluation under Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) (Arp and Slinde, 2018). No studies were identified that observed degradation of PFBS under environmental conditions, including atmospheric photolysis. The review determined that the air-water partition coefficient (K_{aw}) for PFBS is too low to measure and 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. ECHA (2019) modeled photodegradation of PFBS in air and concluded that PFBS has the potential for long-range transport.

1.4.2 *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 (ATSDR, 2021; U.S. EPA, 2021a). PFBS has been found in rain as well as in snow/ice in the Arctic and Antarctic (Arp and Slinde, 2018). EPA collected information about PFBS occurrence in water (described below and in Appendix B, Tables B-1 to B-3). 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. 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

sources of PFAS contamination. However, the maximum reported PFBS concentrations were measured at sites with known PFAS contamination from manufacturing facilities (drinking water) (Pitter et al., 2020) or AFFF usage (groundwater and surface water) (Anderson et al., 2016).

1.4.2.1 Drinking Water

EPA required the most nationally representative sampling for PFBS in drinking water to date under the UCMR 3. Sampling for the UCMR 3 was conducted between 2013 and 2015. PFBS was detected above the minimum reporting level (MRL)⁹ of 90 ng/L in eight PWSs (across four U.S. states and one U.S. territory) out of a total of 4,920 PWSs with results (U.S. EPA, 2017). PFBS concentrations ranged from 90 (the MRL) to 370 ng/L. Results are available in EPA's National Contaminant Occurrence Database (NCOD).¹⁰ EPA included PFBS among the analytes that will be monitored under the UCMR 5 and will use EPA analytical Method 533, which was demonstrated through multilab validation of the method to support a lower UCMR 5 defined MRL of 3 ng/L.

Some states have monitored for PFBS in drinking water since the UCMR 3 using improved EPA analytical methods 533 and 537.1 (see Section 5.0). PFBS has been detected in the finished drinking water from at least 17 U.S. states (ADEM, 2020; CADDW, 2021; CDPHE, 2020a; Illinois EPA, 2021b; KYDEP, 2019; MA EEA, 2020; Maine DEP, 2020; MDE, 2021; Michigan EGLE, 2021; NCDEQ, 2021; NHDES, 2021; NJDEP, 2021; NMED, 2021; Ohio DOH, 2021; PADEP, 2021; SCDHEC, 2020; VTDEC, 2021). State drinking water PFBS monitoring studies often focus on investigating areas known to be affected by PFBS. In states where samples were collected using random sampling site selection (AL, CO, IL, KY, MA, MI, NH, ND, NJ, OH, SC, and VT), PFBS concentrations ranged from non-detect (ND) to 310 ng/L (ppt). Where monitoring was targeted to areas known or suspected to have sources of PFBS (CA, ME, MD, NC, and PA), concentrations were higher and the percentage of samples with PFBS concentrations above the reporting limit often exceeded 20%. Based on the available finished drinking water sampling from states, no finished drinking water samples from any state had PFBS at concentrations exceeding 310 ng/L.

Peer-reviewed studies on PFBS occurrence in drinking water (including bottled water, tap water, and well water intended for consumption) reporting results from North America and/or Europe were reviewed (see literature search methods in Appendix A and study details in Appendix B, Table B-1).

Seven studies analyzed drinking water in areas of North America where study authors did not indicate whether sampling sites were associated with known or suspected sources of PFAS release (Appleman et al., 2014; Boone et al., 2014, 2019; Bradley et al., 2020; Dasu et al., 2017; Hu et al., 2019; Kaboré et al., 2018; Subedi et al., 2015). Three of these seven studies (Appleman et al., 2014; Boone et al., 2019; Bradley et al., 2020) evaluated finished or treated water from drinking water treatment plants (DWTPs). Appleman et al. (2014) detected PFBS in 100% of finished water samples taken from DWTPs that used surface water, groundwater, or blended

⁹ The MRL refers to the quantitation level selected by EPA to ensure reliable and consistent results. It is the minimum quantitation level that can be achieved with 95 percent confidence by capable analysts at 75 percent or more of the laboratories using a specified analytical method (EPA, 2021g).

¹⁰ EPA's NCOD is available at <https://www.epa.gov/sdwa/national-contaminant-occurrence-database-ncod>

water as source water, some of which were reportedly known to have been impacted by upstream wastewater effluent discharge. PFBS levels ranged from 0.43 – 37 ng/L across 11 sites with finished water samples. Boone et al. (2019) also reported that some sampling locations in their study had known or suspected sources of wastewater in the source water but did not identify which ones; PFBS levels in this study ranged from ND to 11.9 ng/L. Bradley et al. (2020) reported PFBS concentrations of ND–0.5 ng/L in treated pre-distribution tap water from four sites. Six studies analyzed tap water from homes (Boone et al., 2014; Bradley et al., 2020; Dasu et al., 2017; Hu et al., 2019; Kaboré et al., 2018; Subedi et al., 2015). Across these six studies, PFBS was detected in at least one sample per study (DFs 5–100%) at concentrations ranging from ND to 14.15 ng/L; in three of the six studies, the maximum PFBS concentration was < 1 ng/L. In Boone et al. (2014), tap water (for which Mississippi River water was the source) was tested at one private home during both low and high river stages, and PFBS concentrations were 14.15 ng/L and 2.12 ng/L, respectively. In Hu et al. (2019), the tested water samples were archived samples from 1989–1990 (PFBS concentrations in these samples ranged from ND–2.97 ng/L).

Three studies conducted in North America examined PFBS levels in drinking water from areas with known or suspected PFAS releases (Boone et al., 2014; Lindstrom et al., 2011; Scher et al., 2018) and two of the three studies detected PFBS. Boone et al. (2014) analyzed samples from three drinking-water wells at sites impacted by AFFF. PFBS was found in all three wells (mean PFBS concentrations 9.09–29 ng/L). Lindstrom et al. (2011) sampled six drinking-water wells in areas impacted by up to 12 years of field applications of biosolids contaminated by a fluoropolymer manufacturer. PFBS was detected in four of the six wells, and concentrations were as high as 56.5 ng/L (mean PFBS concentration was 19.7 ng/L). Scher et al. (2018) found no PFBS in tap water from exterior taps of 23 homes near a former 3M PFAS production facility, 20 of which had been identified as being located within the groundwater contamination area (GCA).

Of the available studies conducted in Europe, 17 analyzed drinking water samples at sites for which authors did not indicate whether there were any known associations with PFAS sources or releases. Fourteen of these 17 studies analyzed tap water from private and/or public sources (cafes, homes, offices, public fountains); of these 14 studies, 12 detected PFBS in at least one sample. Across these 12 studies, mean PFBS concentrations ranged from 0.015 in Sweden (Filipovic and Berger, 2015) to 13.2 ng/L in the Netherlands (Ullah et al., 2011) and the maximum PFBS concentration was 69.43 ng/L (Barcelona; Ericson et al., 2009). Four of the 17 studies (Boiteux et al., 2012; Eriksson et al., 2013; Eschauzier et al., 2012, 2013) analyzed finished or treated water at DWTPs, and PFBS levels in these studies ranged from ND in the Faroe Islands (Eriksson et al., 2013) to 24 ng/L in the Netherlands (Eschauzier et al., 2012).

Nine European studies analyzed drinking water samples from areas near fluoropolymer manufacturing facilities, AFFF-contaminated military airfields, or fire training sites that may use AFFF. Six of the nine studies detected PFBS, with maximum concentrations ranging from 11 to 765 ng/L (Brandsma et al., 2019; Gebbink et al., 2017; Gyllenhammar et al., 2015; Li et al., 2018; Pitter et al., 2020; Weiss et al., 2012). The other three studies (all performed in France) found no detectable levels of PFBS in treated water from DWTPs located downstream of fluorochemical manufacturing facilities or a wastewater treatment plant (WWTP) that processes raw sewage from a fluorochemical manufacturing facility (Bach et al., 2017; Boiteux et al.,

2017; Dauchy et al., 2012). Among the six studies that detected PFBS, the highest measured PFBS concentration (765 ng/L) was detected in municipal water in Veneto, Italy, sampled from areas near a fluoropolymer manufacturing facility (Pitter et al., 2020). The study authors reported that the facility was the only likely source of PFAS and estimated a groundwater contamination plume with an area of 190 square kilometers (km²) affecting public and private drinking water sources (Pitter et al., 2020). In the studies that analyzed water samples from areas near AFFF-contaminated military airfields or fire training sites (Gyllenhammar et al., 2015; Li et al., 2018; Weiss et al., 2012), PFBS DFs ranged from 0 to 100%, PFBS concentrations ranged from ND to 130 ng/L, and maximum PFBS concentrations ranged from 11 to 130 ng/L.

1.4.2.2 Bottled Water

The United States does not have standards for PFAS in bottled water. The Standard of Quality set by the International Bottled Water Association (IBWA) for PFAS in bottled water is 5 ng/L for one PFAS and 10 ng/L for more than one PFAS (IBWA, 2022). One available study analyzed bottled water in the United States (101 samples representing 66 brands) and reported a PFBS DF of 17% and PFBS concentrations ranging from ND to 1.44 ng/L (Chow et al., 2021). Of eight available studies that analyzed bottled water in Canada (one study) or Europe (seven studies), the study in Canada detected PFBS in 9% of samples at a maximum PFBS concentration of 0.23 ng/L (Kaboré et al., 2018). Four of seven studies that analyzed bottled water in different European countries detected PFBS at concentrations ranging from ND to 51 ng/L (DF 0–29%); however, most of the studies did not specify the origin of the bottled water (Gellrich et al., 2013; Harrad et al., 2019; Le Coadou et al., 2017; Ünlü Endirlik et al., 2019). The other three European studies did not detect PFBS in bottled water.

1.4.2.3 Groundwater

In addition to the studies described in Section 1.4.2.1 that reported groundwater PFBS concentrations in well water intended for direct consumption, several other studies evaluated the occurrence of PFBS in raw groundwater in the United States or Europe (see Table B-2). 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.

Four U.S. studies assessed PFBS concentrations in groundwater at sites known to be contaminated with PFAS from the use of AFFF (Anderson et al., 2016; Eberle et al., 2017; Moody et al., 2003; Steele et al., 2018). Of the three studies that reported PFBS detections, two reported DFs of 78.26% and 100% (Anderson et al., 2016; Eberle et al., 2017); 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 (ASTSWMO) 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 (3M, 2007; ATSDR, 2021). 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 (MRL = 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., 2012, 2017, 2019; Gobelius et al., 2018; Gyllenhammar et al., 2015; Høisæter et al., 2019; Wagner et al., 2013). 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.

1.4.2.4 Surface Water

Studies evaluating the occurrence of PFBS in surface water are available from North America, Europe, and across multiple continents (see Table B-3). 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 – 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 (Nakayama et al., 2010; Pan et al., 2018; Subedi et al., 2015;

Veillette et al., 2012; Yeung et al., 2017). 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 (Lasier et al., 2011; Procopio et al., 2017; Zhang et al., 2016). 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 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; Nakayama et al., 2007; Post et al., 2013). 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.

Seven studies evaluated surface water samples from sites in Europe with known or suspected PFAS releases associated with AFFF use (Dauchy et al., 2017; Gobelius et al., 2018; Mussabek et al., 2019) 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) (Boiteux et al., 2012; Eschauzier et al., 2012; Lorenzo et al., 2015; Rostkowski et al., 2009; Zhao et al., 2015) and/or wastewater effluent discharges (Labadie and Chevreuil, 2011; Lorenzo et al., 2015; Möller et al., 2010; Wilkinson et al., 2017). 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 source of contamination (Ahrens et al., 2009a, 2009b; Barreca et al., 2020; Ericson et al., 2008b; Eriksson et al., 2013; Loos et al., 2017; Munoz et al., 2016; Pan et al., 2018; Shafique et al., 2017; Wagner et al., 2013). 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).

1.4.3 Exposure in Humans

As described in EPA's final PFBS toxicity assessment, PFBS has been detected in the serum of humans in the general population (U.S. EPA, 2021a). In American Red Cross plasma samples collected in 2015, 8.4% of samples had a quantifiable serum PFBS concentration, ranging from the lower limit of quantitation (LOQ) to 4.2 nanograms per milliliter (ng/mL) (Olsen et al., 2017). Results for the majority of serum samples were below the lower LOQ for PFBS, and the 95th percentile concentration was 0.02 ng/mL (Olsen et al., 2017). Data from the 2013–2014 National Health and Nutrition Examination Survey (NHANES) reported a 95th percentile concentration for PFBS in serum that was at or below the level of detection (0.1 ng/mL) (Olsen et al., 2017). Another study studied temporal trends of PFBS in blood serum from primiparous nursing women in Sweden ~2000–2002 around the time of increased manufacturing of PFBS after it was introduced as a replacement for PFOS (Glynn et al., 2012). An increase in PFBS blood serum levels was observed between 1996 and 2010, and regression analysis suggested that PFBS levels doubled on average every six years (Glynn et al., 2012).

Studies in animals show that PFBS is well absorbed following oral administration and distributes to all tissues of the body (Bogdanska et al., 2014). Distribution is predominantly extracellular (Olsen et al., 2009) and based on its resistance to metabolic degradation, the majority of PFBS is eliminated unchanged in urine and feces. Two studies that measured PFBS half-life in humans found overlapping ranges of 21.6–87.2 days (Xu et al., 2020) and 13.1–45.7 days (Olsen et al., 2009). The relatively rapid rate of elimination (days to weeks) of PFBS, compared with longer-chain PFAS (years), could lead to a lack of detection in biomonitoring detects which should not be interpreted as a lack of occurrence or exposure potential (U.S. EPA, 2021a). For more information, see U.S. EPA (2021a).

2.0 Problem Formulation and Scope

2.1 Conceptual Model

A conceptual model provides useful information to characterize and communicate the potential health risks related to PFBS exposure from drinking water and to outline the scope of the HA. The sources of PFBS, the routes of exposure for biological receptors of concern (e.g., various human activities related to tap water ingestion such as drinking, food preparation, and consumption), the potential health effects, and exposed populations including sensitive populations and life stages are depicted in the conceptual diagram below (Figure 1).

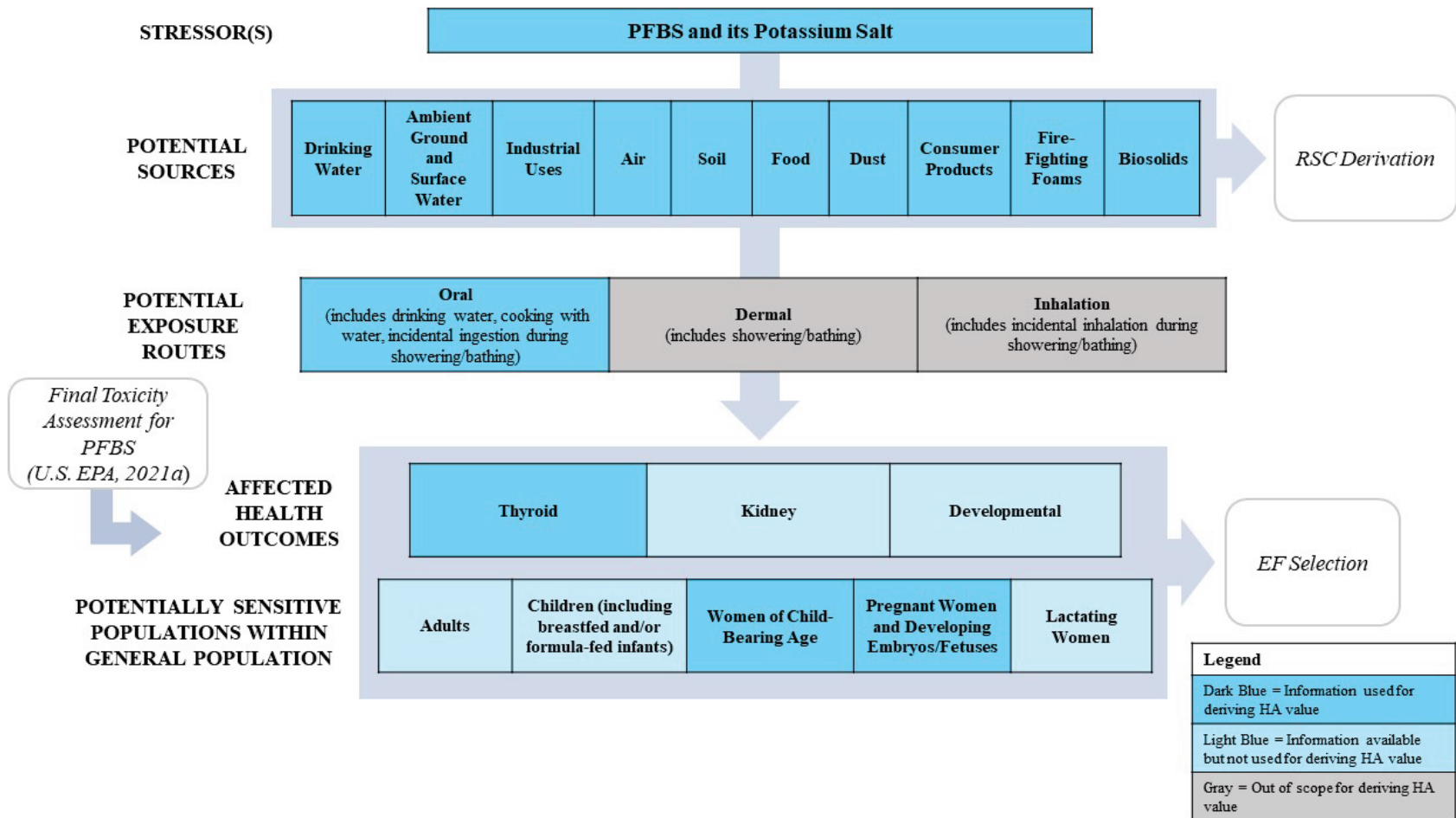


Figure 1. Conceptual Model for the Development of the Drinking Water Health Advisory for PFBS

The conceptual model is intended to explore potential links between exposure to a contaminant or stressor and the adverse health outcomes, and to outline the information sources used to identify or derive the input values used for the HA derivation, which are the RfD, relative source contribution (RSC), and exposure factor (EF). The conceptual model also illustrates the scope of the PFBS HA, which considers the following factors:

Stressors: The scope of this drinking water HA includes PFBS, its potassium salt (K^+ PFBS), and PFBS⁻ since K^+ PFBS fully dissociates in water to the deprotonated anionic form of PFBS (PFBS⁻; CASRN 45187-15-3) and the K^+ cation at environmental pH levels (pH 4–9), consistent with the scope of the PFBS toxicity assessment (EPA, 2021a).

Potential Sources of Exposure: The scope of the HA derivation is limited to drinking water from public water facilities or private wells. Sources of PFBS exposure include both ground and surface waters used for drinking. To develop the RSC, information about non-drinking water sources was identified to determine the portion of the RfD attributable to drinking water. Potential non-drinking water sources of PFBS include but are not limited to foods, indoor dust, indoor and outdoor air, soil, biosolids, and consumer products (see Figure 1).

Potential Exposure Routes: Oral exposure to PFBS from contaminated drinking water sources (e.g., via drinking water, cooking with water, and incidental ingestion from showering) is the focus of the HA. The drinking water HA value does not apply to other exposure routes. However, information on other potential routes of exposure including dermal exposure (contact of exposed parts of the body with water containing PFBS during bathing or showering, dishwashing); and inhalation exposure (during bathing or showering or using a humidifier or vaporizer) was considered to develop the RSC.

Affected Health Outcomes: The PFBS final toxicity assessment (U.S. EPA, 2021a) 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. As part of the PFBS final toxicity assessment, human and animal studies of other health effects after PFBS exposure included the evaluation of effects on the reproductive system, liver, and lipid and lipoprotein homeostasis but the evidence did not support clear associations between exposure and effect. No cancer studies were identified for PFBS (U.S. EPA, 2021a).

Potentially Sensitive Populations or Life Stages: The receptors are humans in the general population who could be exposed to PFBS from oral exposure to tap water through ingestion at their homes, workplaces, schools, and daycare centers. Within all ages of the general population, there are potentially sensitive populations or life stages that may be more susceptible due to increased exposure and/or response. Potentially sensitive populations include the developing embryo and fetus (exposed to PFBS via the pregnant woman) and women of childbearing age who may be or become pregnant.

2.2 Analysis Plan

2.2.1 Health Advisory Guidelines

Assessment endpoints for HA guidelines or values can be developed, depending on the available data, for both short-term (one-day and ten-day) and lifetime exposure using information on the noncarcinogenic and carcinogenic toxicological endpoints of concern. Where data are available,

HAs can reflect sensitive populations or life stages that may be more susceptible and/or more highly exposed.

One-Day HA is protective of noncancer effects for up to 1 day of exposure and is typically based on an *in vivo* toxicity study with a duration of 7 days or less. It is typically calculated for an infant.

Ten-Day HA is protective of noncancer effects for up to 10 days of exposure and is typically based on an *in vivo* toxicity study with a duration of 7 to 30 days. It is typically calculated for an infant.

Lifetime HA is designed to be protective of noncancer effects over a lifetime of exposure and is typically based on a chronic *in vivo* experimental animal toxicity study and/or human epidemiological data.

10⁻⁶ Cancer Risk Concentration is the concentration of a carcinogen in water at which the population is expected to have a one in a million (10⁻⁶) excess cancer risk above background after exposure to the contaminant over a lifetime. It is calculated for carcinogens classified as known or likely human carcinogens (U.S. EPA, 1986, 2005b). Cancer risk concentrations are not derived for substances for which there is suggestive evidence of carcinogenic potential unless the cancer risk has been quantified.

2.2.2 Sources of Toxicity Information for Health Advisory Development

The final toxicity assessment for PFBS, entitled *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)*, published in April 2021 by EPA's Office of Research and Development (ORD) Center for Public Health and Environmental Assessment (CPHEA) (U.S. EPA, 2021a), serves as the basis of the toxicity information and chronic RfD used to derive the lifetime noncancer HA for PFBS. It also synthesizes and describes other information on PFBS including physicochemical properties and toxicokinetics. The PFBS toxicity assessment was published after rigorous scientific review, including internal and external review, and public comment.

To develop the final toxicity assessment for PFBS, EPA reviewed and analyzed the available toxicokinetics and toxicity data for PFBS. Briefly, online scientific databases (PubMed, Web of Science, TOXLINE, and TSCATS via TOXLINE) were searched using search terms focused on chemical name and synonyms with no limitations on publication type, evidence stream (i.e., human, animal, in vitro, and in silico), or health outcomes. The identified studies were screened using Populations, Exposures, Comparators, and Outcomes (PECO) criteria and relevant studies underwent study quality evaluation. Dose-response studies were identified for dose-response modeling and a point-of-departure (POD) and uncertainty factors (UFs) were selected for RfD derivation. For more information, please see Section 2.3 in U.S. EPA (2021a).

2.2.3 Approach and Scope for Health Advisory Derivation

2.2.3.1 Approach for Deriving Noncancer HAs

The following equations (Eqs. 1–3) are used to derive the HAs.¹¹ Lifetime HAs and 10⁻⁶ cancer risk concentrations are only derived for chemicals without an existing National Primary Drinking Water Regulation.

$$\text{One-Day HA} = \left(\frac{\text{POD}}{\text{UF}_C * \text{DWI-BW}} \right)$$

POD is typically derived from a toxicity study of duration 7 days or less

(Eq. 1)

$$\text{Ten-Day HA} = \left(\frac{\text{POD}}{\text{UF}_C * \text{DWI-BW}} \right)$$

POD is typically derived from a toxicity study of duration 7–30 days

(Eq. 2)

$$\text{Lifetime HA} = \left(\frac{\text{RfD}}{\text{DWI-BW}} \right) * \text{RSC}$$

RfD is typically derived from a chronic study

(Eq. 3)

Where:

POD is the point of departure, typically a lowest observed adverse effect level (LOAEL), a no observed adverse effect level (NOAEL), or a BMDL from the critical study.

UF_C is the composite UF or total UF value after multiplying individual UFs. UFs are established in accordance with EPA best practices (U.S. EPA, 2002) and consider uncertainties related to the following: variation in sensitivity among the members of the human population (i.e., inter-individual variability), extrapolation from animal data to humans (i.e., interspecies uncertainty), extrapolation from data obtained in a study with less-than-lifetime exposure to lifetime exposure (i.e., extrapolating from subchronic to chronic exposure), extrapolation from a LOAEL rather than from a NOAEL, and extrapolation when the database is incomplete. For PFBS, the value of UF_C was determined in the final PFBS toxicity assessment (U.S. EPA, 2021a).

DWI-BW is the 90th percentile drinking water intake (DWI), adjusted for body weight (bw), for the selected population in units of liter per kilogram body weight per day (L/kg bw-day). The DWI-BW considers direct and indirect consumption of tap water (indirect water consumption encompasses water added in the preparation of foods or beverages, such as tea and coffee). For PFBS, the value of this parameter is based on the critical study identified in the PFBS final toxicity assessment (U.S. EPA, 2021a), and is identified in Chapter 3 of EPA's *Exposure Factors Handbook* (U.S. EPA, 2019a).

RfD is the reference dose—an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure of the human population to a substance that is likely to be without an appreciable risk of deleterious effects during a lifetime. The value of this parameter

¹¹ <https://www.epa.gov/system/files/documents/2022-01/dwtable2018.pdf>

was derived in the final PFBS toxicity assessment and is based on the critical effect and study identified in that assessment (U.S. EPA, 2021a).

RSC is the relative source contribution—the percentage of the total oral exposure attributed to drinking water sources (U.S. EPA, 2000a) where the remainder of the exposure is allocated to other routes or sources. The RSC is calculated by examining other sources of exposure (e.g., air, food, soil) and pathways of exposure in addition to drinking water using the methodology described for calculation of an RSC described in U.S. EPA (2000a) and Section 3.3.

2.2.3.2 *Scope of Noncancer Health Advisory Values*

Adequate data are available to derive a lifetime HA for PFBS. Neither one-day nor ten-day HA values were derived for PFBS. U.S. EPA (2021a) derived subchronic and chronic RfDs but did not derive an RfD for exposure durations of 7 days or less on which to base a one-day HA for PFBS. Derivation of a 10-day HA was considered because the subchronic and chronic RfDs are both based on a 20-day exposure study, which may be used to derive a ten-day HA. However, the critical health effect on which the chronic RfD used to calculate the lifetime HA is based (i.e., decreased serum levels of the thyroid hormone thyroxine [T₄] in newborn mice) resulted from PFBS exposure during a developmental life stage. EPA’s risk assessment guidelines for developmental toxicity indicate that adverse effects can result from even brief exposure during a critical period of development (U.S. EPA, 1991). The critical study for the subchronic and chronic RfDs for PFBS observed persistent health effects into adulthood suggesting the potential for long-term health consequences of gestational-only PFBS exposure and that gestation is at least one critical exposure window for PFBS. Therefore, the lifetime HA (calculated in Section 4.0) and the chronic RfD from which it is derived (see Table 4) are considered applicable to short-term PFBS exposure scenarios (including during pregnancy) via drinking water.

2.2.3.3 *Approach and Scope for Deriving Cancer Risk Concentrations*

The following equations (Eqs. 4-5) are used to derive cancer risk concentrations.

Calculated for non-mutagenic carcinogens¹² only:

$$\mathbf{10^{-6} \text{ Cancer Risk Concentration}} = \frac{1 \times 10^{-6}}{\text{CSF} * \text{DWI-BW}}$$

(Eq. 4)

Calculated for mutagenic carcinogens only:

$$\mathbf{10^{-6} \text{ Cancer Risk Concentration}} = \frac{1 \times 10^{-6}}{\text{CSF}} * \sum_i \left(\frac{F_i * \text{ADAF}_i}{\text{DWI-BW}_i} \right)$$

(Eq. 5)

Where:

CSF is the cancer slope factor—an upper bound, approximating a 95 percent confidence limit of the increased cancer risk from a lifetime of oral exposure to a stressor. The value for this parameter is derived in the final toxicity assessment when data are available.

¹² <https://www.epa.gov/system/files/documents/2022-01/dwtable2018.pdf>

DWI-BW_i is the 90th percentile bw-adjusted DWI in units of L/kg bw-day for each age group (i), considered when calculating cancer risk concentrations for mutagenic carcinogens.

ADAF_i is the age-dependent adjustment factor for each age group (i), used when calculating cancer risk concentrations for carcinogens that act via a mutagenic mode of action (U.S. EPA, 2005a,b).

F_i the fraction of life spent in each age group (i), used when calculating cancer risk concentrations for mutagens (U.S. EPA, 2005a).

2.2.3.4 *Scope of Cancer Risk Concentration Derivation*

As described in the toxicity assessment for PFBS, a CSF was not derived because no studies evaluating the carcinogenicity of PFBS in humans or animals had been identified (U.S. EPA, 2021a). In accordance with the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b), EPA concluded that there is “*Inadequate Information to Assess Carcinogenic Potential*” for PFBS by any route of exposure (U.S. EPA, 2021a). Therefore, a 10⁻⁶ cancer risk concentration cannot be derived for PFBS at this time.

2.2.4 *Exposure Factors for Deriving Health Advisory*

2.2.4.1 *Exposure Factor Selection*

An EF, such as body weight-adjusted drinking water intake (DWI-BW), is one of the input values for deriving a drinking water HA. EFs are factors related to human activity patterns, behavior, and characteristics that help determine an individual’s exposure to a contaminant. EPA’s *Exposure Factors Handbook* (EFH)¹³ is a resource for conducting exposure assessments and provides EFs based on information from publicly available, peer-reviewed studies. Chapter 3 of the EFH presents EFs in the form of DWI and DWI-BW for various populations or life stages within the general population (U.S. EPA, 2019a). The use of EFs in HA calculations is intended to protect sensitive populations and life stages within the general population from adverse effects resulting from exposure to a contaminant.

When developing HAs, the goal is to protect all ages of the general population including potentially sensitive populations or life stages such as children. The approach to select the EF for the drinking water HA includes a step to identify potentially sensitive population(s) or life stage(s) (i.e., populations or life stages that may be more susceptible or sensitive to a chemical exposure) by considering the available data for the contaminant. Although data gaps can prevent identification of the most sensitive population (e.g., not all windows of exposure or health outcomes have been assessed for PFBS), the critical effect and POD that form the basis for the RfD 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 interval, may identify a particularly sensitive population or life stage (e.g., pregnant women, formula-fed infants, lactating women). In those cases, EPA can select the corresponding DWI-BW for that sensitive population or life stage from the EFH (U.S. EPA, 2019a) for use in HA derivation. When multiple potentially sensitive populations or life stages are identified based on the critical effect or other health effects data (from animal or human studies), EPA selects the population or life stage with the greatest DWI-BW because it is

¹³ EPA’s EFH is available at <https://www.epa.gov/expobox/about-exposure-factors-handbook>

the most health protective. For deriving lifetime HAs, the RSC corresponding to the selected sensitive life stage is also determined when data are available (see Section 3.3). In the absence of information indicating a potentially sensitive population or life stage, the EF corresponding to all ages of the general population may be selected.

To derive chronic HAs, EPA typically uses DWI normalized to body weight (i.e., DWI-BW in liter [L] of water consumed/kg bw-day) for all ages of the general population or for a sensitive population or life stage, when identified. The Joint Institute for Food Safety and Applied Nutrition’s Food Commodity Intake Database (FCID) Consumption Calculator Tool¹⁴ includes the EFs from EPA’s EFH and can also be used to estimate DWI-BW for specific populations, life stages, or age ranges. EPA uses the 90th percentile DWI-BW to ensure that the HA is protective of the general population as well as sensitive populations or life stages (U.S. EPA, 2000a, 2016a). In 2019, EPA updated its EFs for DWI-BW based on newly available science (EPA, 2019a).

Table 3 shows EPA EFs for some sensitive populations or life stages. Other populations or life stages may also be considered depending on the available information regarding sensitivity to health effects after exposure to a contaminant.

Table 3. EPA Exposure Factors for Drinking Water Intake

Populations or Life Stages	DWI-BW (L/kg bw-day)	Description of Exposure Metric	Source
General population, all ages	0.0338	90th percentile direct and indirect consumption of community water, consumer-only two-day average, all ages.	2019 Exposure Factors Handbook Chapter 3, Table 3-21, NHANES 2005–2010 (U.S. EPA, 2019a)
Children	0.143	90th percentile direct and indirect consumption of community water, consumer-only two-day average, birth to < 1 year.	2019 Exposure Factors Handbook Chapter 3, Table 3-21, NHANES 2005–2010 (U.S. EPA, 2019a)
Formula-fed infants	0.249	90th percentile direct and indirect consumption of community water, formula-consumers only, 1 to < 3 months. Includes water used to reconstitute formula, plus all other community water ingested.	Kahn et al. (2013) Estimates of Water Ingestion in Formula by Infants and Children Based on CSFII 1994–1996 and 1998 ^{a,b}

¹⁴ Joint Institute for Food Safety and Applied Nutrition’s FCID, Commodity Consumption Calculator is available at <https://fcid.foodrisk.org/percentiles>

Populations or Life Stages	DWI-BW (L/kg bw-day)	Description of Exposure Metric	Source
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 (U.S. EPA, 2019a)
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 (U.S. EPA, 2019a)
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 ^c (U.S. EPA, 2019a)

Notes: CSFII = continuing survey of food intake by individuals; L/kg bw-day = liter per kilogram body weight per day.

^a The sample size does not meet the minimum reporting requirements as described in the Third Report on Nutrition Monitoring in the United States (LSRO, 1995).

^b Chapter 3.2.3 in U.S. EPA (2019a) cites Kahn et al. (2013) as the source of drinking water ingestion rates for formula-fed infants. While U.S. EPA (2019a) provides the 95th percentile total direct and indirect water intake values, Office of Water/Office of Science and Technology (OW/OST) policy is to utilize the 90th percentile DWI-BW. OW/OST was able to identify the 90th percentile DWI-BW in Kahn et al. (2013) and report the value in this table.

^c 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: Human Nutrition Information Service (HNIS)/National Center for Health Statistics (NCHS) Analytical Working Group Recommendations (NCHS, 1993).

2.2.4.2 Determining Proportion of RfD Attributable to Drinking Water

To account for aggregate risk from exposures and exposure pathways other than oral ingestion of drinking water, EPA applies an RSC when calculating HAs to ensure that total human exposure to a contaminant does not exceed the daily exposure associated with the RfD. 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 transfer to dietary items prepared with drinking water) relative to other exposure pathways. The remainder of the exposure equal to the RfD is allocated to other potential exposure sources (U.S. EPA, 2000a). The purpose of the RSC is to ensure that the level of a contaminant (e.g., HA value), when combined with other identified sources of exposure common to the population of concern, will not result in exposures that exceed the RfD (U.S. EPA, 2000a).

To determine the RSC, EPA follows the Exposure Decision Tree for Defining Proposed RfD (or POD/UF) Apportionment in EPA’s guidance, *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (U.S. EPA, 2000a). EPA considers whether there are significant known or potential uses/sources other than drinking water, the adequacy of data and strength of evidence available for each relevant exposure medium and pathway, and whether adequate information on each source is available to quantitatively characterize the exposure

profile. The RSC is developed to reflect the exposure to the general population or a sensitive population within the general population.

Per EPA’s guidance, in the absence of adequate data to quantitatively characterize exposure to a contaminant, EPA typically recommends an RSC of 20%. When 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, thereby allocating the remaining 20% to other potential exposure sources (U.S. EPA, 2000a).

To inform the RSC determination, available information on all exposure sources and routes for PFBS was identified using the literature search and screening method described in Appendix A. To identify information on PFBS exposure routes and sources to inform RSC determination, EPA considered primary literature published between 2003–2020 and collected by EPA ORD as part of an effort to evaluate evidence for pathways of human exposure to eight PFAS, including PFBS. In order to consider more recently published information on PFBS exposure, EPA incorporated the results of a date-unlimited gray literature search that was conducted in February 2022 as well as an ad hoc process to identify relevant and more recently published peer-reviewed scientific literature. The literature resulting from the search and screening process included only final (not draft) documents and articles that were then reviewed to inform the PFBS RSC.

3.0 Health Advisory Input Values

3.1 Toxicity Assessment Values

Table 4 summarizes the peer-reviewed chronic noncancer toxicity values from EPA’s *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)* (U.S. EPA, 2021a).

Table 4. Chronic Noncancer Toxicity Information for PFBS for Deriving the Lifetime HA

Health Assessment	PFBS Exposure in Critical Study	RfD (mg/kg bw-day)	Critical Effect	Principal Study
Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)	Days 1–20 of gestation	K ⁺ PFBS: 3 x 10 ⁻⁴ PFBS: 3 x 10 ⁻⁴	Decreased serum total T ₄ in newborn (PND 1) mice	Oral gestational exposure study in mice (Feng et al., 2017)

Notes: mg/kg bw-day = milligram per kilogram body weight per day; PND = post-natal day.

Source: U.S. EPA, 2021a

As stated in U.S. EPA (2021a), the thyroid effect of decreased thyroid hormones, specifically serum total T₄, in newborn (PND1) mice exposed to K⁺PFBS throughout gestation was selected as the critical effect (Feng et al., 2017). This critical effect and study were used to derive the chronic RfDs for K⁺PFBS and PFBS of 3 × 10⁻⁴ milligrams per kilogram body weight per day (mg/kg bw-day).

Based on EPA's *Recommended Use of Body Weight^{3/4} as the Default Method in Derivation of the Oral Reference Dose* (U.S. EPA, 2011), serum half-lives were used to scale a toxicologically equivalent dose of orally administered K⁺PFBS from animals to humans. Following EPA's *Benchmark Dose Technical Guidance Document* (U.S. EPA, 2012b), benchmark dose (BMD) modeling of thyroid effects following gestational exposure to K⁺PFBS resulted in a benchmark dose lower confidence limit for 0.5 standard deviation change from the control (BMDL_{0.5SD}) human equivalent dose (HED) of 0.095 mg/kg bw-day.

This POD (HED) served as the critical effect and was divided by a composite UF (UF_C) of 300. The UF_C is based on an animal-to-human UF (UF_A) of 3 to account for extrapolation from mice to humans; an intrahuman UF (UF_H) of 10 to account for interindividual differences in human susceptibility; and a database UF (UF_D) of 10 to account for deficiencies in the toxicity database. A value of 1 was applied for the extrapolation from subchronic to a chronic exposure duration UF (UF_S) because extrapolation from subchronic to chronic was not needed, and UF_L because a LOAEL to NOAEL approach was not used. Data for K⁺PFBS were used to derive the chronic RfD for the free acid (PFBS), resulting in the same value (3×10^{-4} mg/kg bw-day), after adjusting for differences in molecular weight (MW) between K⁺PFBS (338.19) and PFBS (300.10) (see Section 6.0 in U.S. EPA [2021a] for more details). This chronic RfD for PFBS was used to derive the lifetime HA.

No studies evaluating the carcinogenicity of PFBS in humans or animals were identified (U.S. EPA, 2021a). In accordance with EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005b), EPA concluded that there is "*Inadequate Information to Assess Carcinogenic Potential*" for PFBS by any route of exposure (U.S. EPA, 2021a) and did not derive a 10^{-6} cancer risk concentration.

3.2 Exposure Factors

To identify potentially sensitive populations or life stages, EPA considered the PFBS exposure interval used in the critical study selected for chronic RfD derivation in the final PFBS toxicity assessment (U.S. EPA, 2021a). In the critical study pregnant mice were orally exposed to K⁺PFBS throughout all of gestation (days 1–20 of gestation) (Feng et al., 2017; U.S. EPA, 2021a), identifying the developing fetus (exposed via the pregnant mother) as a population that may be particularly susceptible to PFBS exposure. The critical study did not permit a more precise identification of the most sensitive or critical PFBS exposure window during prenatal development since exposure was throughout all of gestation. The critical effect of thyroid development in the developing mouse embryo and fetus is relevant to humans. Human thyroid development occurs in three phases during gestation, and while there are some timing differences in thyroid development between humans and rodents (see Section 6.1.1.3 in U.S. EPA, 2021a), two phases of thyroid development occur during gestation in both the mouse and human.

The gestational exposure in the critical study is relevant to two potentially sensitive populations or life stages—women of childbearing age (13 to < 50 years) who may be or become pregnant, and pregnant women and their developing embryo and fetus (Table 5). EPA selected women of childbearing age as the sensitive life stage for HA derivation because the DWI-BW is greater (0.0354 L/kg bw-day) than for pregnant women (0.0333 L/kg bw-day). EPA addresses exposure to the sensitive developing embryo and fetus because they are exposed to drinking water via the pregnant mother. Additional support for the women of childbearing age population including

pregnant women (and their developing embryo and fetus) includes the high rate of unintended pregnancies reported in the United States (30.6%) (United Health Foundation, 2021). To derive the HA value, EPA used the DWI-BW of 0.0354 L/kg bw-day representing the consumers-only two-day average of direct and indirect community water consumption at the 90th percentile for women of childbearing age (13 to < 50 years) (Table 5, in bold).

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

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 (U.S. EPA, 2019a)
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 (U.S. EPA, 2019a)

Notes: L/kg bw-day = liters of water consumed per kilogram body weight per day. The DWI-BW used to calculate the PFBS lifetime HA is in bold.

3.3 Relative Source Contribution

As stated in the analysis plan, EPA collected and evaluated information about PFBS exposure routes and sources to inform RSC determination. Results from the literature search are described below.

3.3.1 Non-Drinking Water Sources and Routes

EPA presents information below from studies performed in the United States as well as studies published globally for this emerging contaminant to be as comprehensive as possible, given that the overall information is limited. While the studies from non-U.S. countries inform an understanding global exposure sources and trends, the RSC determination is based on the available data for the United States.

3.3.1.1 Dietary Sources

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 (U.S. FDA, 2020a,b, 2021a,b, 2022a,b); 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 (U.S. 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 (U.S. FDA 2018, 2021c). Maximum residue levels for PFBS were not found in the Global MRL Database (Bryant Christie Inc., 2022).

In addition to efforts by FDA, 34 peer-reviewed studies conducted in North America (n = 7), Europe (n = 26), and across multiple continents (n = 1) 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; Beser et al., 2019; Cariou et al., 2015; Kärman et al., 2007, 2010; Lankova et al., 2013; Nyberg et al., 2018).

Of eight studies conducted in North America, four U.S. studies (Blaine et al., 2014; Byrne et al., 2017; Schechter et al., 2010; Scher et al., 2018) found PFBS in at least one food item. Locations and food sources varied in these studies. In Schechter 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 to date, seafood (including fish and shellfish) has been found to contain the highest concentrations of PFBS out of all food types examined. Several large-scale sampling efforts have been conducted by EPA and other agencies to determine PFAS levels in fish. In 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 (U.S. EPA, 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 EPA's 2015 Great Lakes Human Health Fish Fillet Tissue Study (U.S. EPA, 2021d). 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 EPA's National Lake Fish Tissue Study (U.S. EPA, 2009a).

Several peer-reviewed publications that examined PFBS concentrations in fish and shellfish are also available. As mentioned previously, Schechter 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 27 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 where no particular source of PFAS contamination was identified (D'Hollander et al., 2015; Domingo et al., 2012; Eschauzier et al., 2013; Hlouskova et al., 2013; Pérez et al., 2014; Scordo et al., 2020; Surma et al., 2017; Sznajder-Katarzyńska et al., 2019). 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 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.

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 (ATSDR, 2021; U.S. EPA, 2021a). In one report from the United States, PFBS was detected in fast-food packaging (7/20 samples) although the concentrations detected were not reported (Schaidler et al., 2017).

Five 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/cm²), 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/cm².

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. Vavrous 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 few available studies conducted in the United States and Europe indicate 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.

3.3.1.2 Consumer Products

Consumer products could also be a source of PFBS exposure as noted in Section 1.3. Several studies examined a range of consumer products and found multiple PFAS, including PFBS, at various levels (Bečanová et al., 2016; Favreau et al., 2016; Gremmel et al., 2016; Kotthoff et al., 2015; Liu et al., 2014; Schultes et al., 2018; van der Veen et al., 2020; Vestergren et al., 2015; Zheng et al., 2020). 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. Additionally, two European studies analyzed commercially available AFFF products which have been formulated with PFAS and are associated with elevated levels of these chemicals in environmental media (Favreau et al., 2016; Høisæter et al., 2019).

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 [$\mu\text{g}/\text{m}^2$], 45 to 350 $\mu\text{g}/\text{m}^2$, and 9.6 to 130 $\mu\text{g}/\text{m}^2$ 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 $\mu\text{g}/\text{m}^2$, 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 $\mu\text{g}/\text{m}^2$), ski wax samples (up to 3.1 micrograms per kilogram [$\mu\text{g}/\text{kg}$]), leather samples (up to 120 $\mu\text{g}/\text{kg}$), and gloves (up to 2 $\mu\text{g}/\text{kg}$). 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 $\mu\text{g}/\text{m}^2$) 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 $\mu\text{g}/\text{kg}$ found in a used 1999 screen associated with WEEE.

PFBS was also evaluated in AFFFs in Switzerland (Favreau et al., 2016) and Norway (Høisæter et al., 2019). In currently commercially available AFFFs from Switzerland, PFBS was detected in 11% of samples (n = 35) with a maximum concentration of 0.1 ppm (Favreau et al., 2016). In AFFFs used at a firefighting training facility in Norway, PFAS concentrations in 1:100 diluted AFFF were predominately PFOS (88.7%). PFBS contributed to 1.2% of the concentration of the 23 total PFAS tested in the diluted foam, with a concentration of 1,400,000 ng/L (Høisæter et al., 2019).

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.

3.3.1.3 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. Eighteen studies conducted in the United States, Canada, various countries in Europe, and across multiple continents 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.

3.3.1.4 Air

PFAS have been released to air from WWTPs, waste incinerators, and landfills (U.S. EPA, 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 (U.S. EPA, 2022a). In addition, PFBS is not listed as a hazardous air pollutant (U.S. EPA, 2022b).

Indoor Air

Three studies in Europe, conducted in Norway (Barber et al., 2007), Spain (Jogsten et al., 2012), and Ireland (Harrad et al., 2019), analyzed the occurrence of PFBS in indoor air samples.

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; PFOS and PFBA were the only detected PFCs in these indoor air samples.

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.

Ambient Air

Four studies conducted across Europe (Barber et al., 2007; Beser et al., 2011; Harrad et al., 2020; Jogsten et al., 2012) and one study conducted in Canada (Ahrens et al., 2011) analyzed ambient air samples for PFBS. Two of the studies (Barber et al., 2007; Harrad et al., 2020) 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. Authors did not indicate whether any of the sites had a history of PFAS impact. PFBS was detected in the particle phase of outdoor air samples during one of the two sampling events in Manchester at $2.2 \text{ pg}/\text{m}^3$ and one of the two sampling events in Hazelrigg at $2.6 \text{ pg}/\text{m}^3$. 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 $0.50 (< 0.15\text{--}1.4) \text{ pg}/\text{m}^3$ and $0.34 (< 0.15\text{--}1.2) \text{ pg}/\text{m}^3$, 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 ($\text{PM}_{2.5}$) 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 Arctic air in one study, with a DF of 66% and mean concentration of 0.1 pg/m³ (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.

3.3.1.5 Soil

PFBS can be released into soil from manufacturing facilities, industrial uses, fire/crash training sites, and biosolids containing PFBS (ATSDR, 2021, U.S. EPA, 2021a). EPA identified 16 studies that evaluated the occurrence of PFBS and other PFAS in soil, with studies conducted in the United States, Canada, and Europe (see Table B-6). Two U.S. studies and two Canadian studies (Blaine et al., 2013; Cabrerizo et al., 2018; Dreyer et al., 2012; Venkatesan and Halden, 2014) 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. (2018, 2019) 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 (Eberle et al., 2017; Nickerson et al., 2020), and another at the site of a train derailment and fire in Canada (Mejia-Avenidaño 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 (0.61–0.64 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-Avenidaño 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; $\geq 70\%$) 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; Hale et al., 2017; Skaar et al., 2019).

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., 2013, 2014; 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.

3.3.2 RSC Determination

In summary, based on the physical properties, detected levels, and available exposure information for PFBS, multiple non-drinking water sources (seafood [including fish and shellfish]) and other foods including vegetables, indoor air, and some consumer products) are potentially significant exposure sources. Following the Exposure Decision Tree within EPA's 2000 *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (U.S. EPA, 2000a), significant potential sources other than drinking water ingestion were identified (Box 8A in the Decision Tree). However, information is not available to quantitatively characterize the relative exposure contributions from the non-drinking water sources (Box 8B in the Decision Tree, U.S. EPA, 2000a).

EPA also considered the exposure information specifically for the identified sensitive population. The identified sensitive lifestage, based on the critical study and effect, is women of childbearing age (13 to <50 years) who may be or become pregnant. However, the literature search did not identify non-drinking water exposure information specific to women of childbearing age that could be used quantitatively to derive an RSC. Since neither the available data for the general population (all ages) nor the sensitive population enabled quantitative characterization of relative exposure sources and routes, EPA relied on an RSC of 20% (see Section 2.2.4.2 above; U.S. EPA, 2000a), which means that 20% of the exposure equal to the RfD is allocated to drinking water and the remaining 80% is reserved for other potential exposure sources such as food, indoor air, and some consumer products.

4.0 Lifetime Noncancer Health Advisory Derivation

The lifetime noncancer HA for PFBS is calculated as follows:

$$\text{Lifetime HA} = \left(\frac{\text{RfD}}{\text{DWI-BW}} \right) * \text{RSC}$$

(Eq. 3)

$$\text{Lifetime HA} = \left(\frac{0.0003 \frac{\text{mg}}{\text{kg bw-day}}}{0.0354 \frac{\text{L}}{\text{kg bw-day}}} \right) * 0.2$$

$$\begin{aligned} \text{Lifetime HA} &= 0.0017 \frac{\text{mg}}{\text{L}} \left(\text{rounded to } 0.002 \frac{\text{mg}}{\text{L}} \right) \\ &= 2 \frac{\mu\text{g}}{\text{L}} \\ &= 2,000 \frac{\text{ng}}{\text{L}} \end{aligned}$$

EPA is issuing a lifetime noncancer drinking water HA for PFBS of 2,000 ng/L (ppt). The critical health effect on which the chronic RfD used to calculate the lifetime HA is based (i.e., decreased serum levels of the T₄ in newborn mice) resulted from PFBS exposure during a developmental life stage. In Feng et al. (2017), developmental effects occurred at PND 1 and were sustained through pubertal (PND 30) and adult periods (PND 60). This is consistent with the potential for long-term health consequences of gestational-only PFBS exposure and suggests that gestation is at least one critical window for PFBS. EPA's risk assessment guidelines for developmental toxicity indicate that adverse effects can result from even brief exposure during a critical period of development (U.S. EPA, 1991). Therefore, the lifetime HA for PFBS of 2000 ng/L and the chronic RfD from which it is derived are considered applicable to short-term PFBS exposure scenarios (including during pregnancy) as well as lifetime exposure scenarios via drinking water. This lifetime HA applies to PFBS (CASRN 375-73-5), K⁺PFBS (CASRN 29420-49-3), and PFBS⁻ (CASRN 45187-15-3).

5.0 Analytical Methods

EPA developed two liquid chromatography/tandem mass spectrometry (LC/MS/MS) analytical methods to quantitatively monitor drinking water for targeted PFAS that include PFBS: EPA Method 533 (U.S. EPA, 2019b) and EPA Method 537.1, Version 2.0 (U.S. EPA, 2020b). The methods discussed below can be used to accurately and reasonably quantitate PFBS at ng/L levels that are three orders of magnitude below the PFBS lifetime HA of 2000 ng/L.

EPA Method 533 monitors for 25 select PFAS with published measurement accuracy and precision data for PFBS in reagent water, finished groundwater, and finished surface water and a single laboratory-derived MRL or approximate quantitation limit for PFBS at 3.5 ng/L (0.0035 µg/L). For further details about the procedures for this analytical method, please see *Method*

533: *Determination of Per- and Polyfluoroalkyl Substances in Drinking Water by Isotope Dilution Anion Exchange Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry* (U.S. EPA, 2019b).

EPA Method 537.1 (representing an update to EPA Method 537 [U.S. EPA, 2009b]) monitors for 18 select PFAS with published measurement accuracy and precision data for PFBS in reagent water, finished groundwater, and finished surface water and a single laboratory-derived MRL or approximate quantitation limit for PFBS at 6.3 ng/L (0.0063 µg/L). For further details about the procedures for this analytical method, please see *Method 537.1: Determination of Selected Per- and Polyfluorinated Alkyl Substances in Drinking Water by Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry (LC/MS/MS)* (U.S. EPA, 2020b).

Drinking water analytical laboratories have different performance capabilities dependent upon their instrumentation (manufacturer, age, usage, routine maintenance, operating configuration, etc.) and analyst experience. Some laboratories will effectively generate accurate, precise, quantifiable results at lower concentrations than others. Organizations leading efforts that include the collection of data need to establish data quality objectives (DQOs) to meet the needs of their program. These DQOs should consider establishing reasonable quantitation limits that laboratories can routinely meet, without recurring quality control (QC) failures that will necessitate repeating sample analyses, increase costs, and potentially reduce laboratory capacity. Establishing a quantitation limit that is too high may result in important lower-concentration results being overlooked.

EPA's approach to establishing DQOs within the UCMR program serves as an example. EPA established MRLs for UCMR 5,¹⁵ and requires laboratories approved to analyze UCMR samples to demonstrate that they can make quality measurements at or below the established MRLs. EPA calculated the UCMR 5 MRLs using quantitation-limit data from multiple laboratories participating in an MRL-setting study. The laboratories' quantitation limits represent their lowest concentration for which future recovery is expected, with 99% confidence, to be between 50 and 150%. The UCMR 5-derived and promulgated MRL for PFBS is 0.003 µg/L (3 ng/L).

6.0 Treatment Technologies

This section summarizes available drinking water treatment technologies that have been demonstrated to remove PFBS from drinking water, but it is not meant to provide specific operational guidance or design criteria. High-pressure membrane processes such as nanofiltration (NF) and reverse osmosis (RO) are generally effective at removing organic solutes and dissolved ions and have been shown to successfully reduce or remove PFBS from drinking water (Appleman et al., 2014). NF generally removes 20–70% of PFBS (Jin et al., 2021), although 93% (Appleman et al., 2013) and 99.8% (U.S. EPA, 2021e) removal have been reported with NF. The amount of contaminant removed by membranes is referred to as a rejection rate; RO tends to have a higher rejection rate than NF. Direct filtration NF and RO membranes have been successful in removing PFBS at full-scale water treatment works to below the 3 ppt EPA UCMR 5 reporting limit (Appleman et al., 2014; Konradt et al., 2021; Liu et al., 2021; Quiñones and Snyder, 2009; Thompson et al., 2011). Absorption-based NF and RO membranes have had

¹⁵ Information about UCMR 5 is available at <https://www.epa.gov/dwucmr/fifth-unregulated-contaminant-monitoring-rule>

success with PFBS treatment at laboratory scale (Zhang et al., 2019). Hybrid membrane processes, such as applying direct-current electrical fields or photocatalysts across lower pressure membranes, have had success with other short-chain sulfonates at laboratory scale (Tsai et al., 2010; Urtiaga, 2021). For more information about hybrid membrane processes, see Soriano et al. (2020) or (2017). Installing high-pressure membranes may have additional benefits on finished water quality by removing other contaminants and disinfection byproduct precursors. Sorption-based processes such as activated carbon and ion exchange have been shown to remove PFBS in drinking water to below the EPA method reporting limit of 3 ppt for UCMR 5; however, the media usage rate is higher than for other PFAS with longer carbon backbones (McCleaf et al., 2017; Murray et al., 2021). Information about PFBS treatment efficacy with sorption-based processes is still emerging; more information about the suitability of these technologies is expected to be available in the future. Most other treatment processes are viewed as not sufficiently effective or cost efficient to reduce PFBS concentrations in drinking water. For example, coagulation, flocculation, sedimentation, and biologically active carbon filtration are generally ineffective at removing PFBS (Quiñones and Snyder, 2009; Sun et al. 2016). Ozonation has increased concentrations of PFBS at full-scale water treatment plants (WTPs), possibly due to PFAS precursor compound oxidation (Sun et al., 2016). Boiling water will concentrate PFBS and should not be considered as an emergency action.

Non-treatment PFBS management practices such as changing source waters, source water protection, or consolidation are also viable options for reducing PFBS concentrations in finished drinking water. One resource for protecting source water from PFAS, including PFBS, is the *PFAS – Source water Protection Guide and Toolkit* (ASDWA, 2020), which shares effective strategies for addressing PFAS contamination risk in source waters. Source water protection is particularly important since natural attenuation is not a valid PFBS management strategy. PFBS will not degrade by abiotic reaction mechanisms such as hydrolysis and photolysis under environmental conditions (Lassen et al., 2013; NICNAS, 2005). Likewise, Quinete et al. (2010) studied biotic PFBS degradability using the manometric respirometry test (OECD, 1992b) and the closed-bottle test (OECD, 1992a) with River Rhine water as inoculum; PFBS did not show signs of biodegradation in either test.

NF and RO are high pressure processes where water is forced across a membrane. The water that transverses the membrane is known as permeate or produce, and has few solutes left in it; the remaining water is known as concentrate, brine, retentate, or reject water and forms a waste stream with concentrated solutes. The main PFBS removal mechanisms in NF and RO are steric exclusion, solution-diffusion, and electrostatic interaction (Jin et al., 2021). NF has a less dense active layer than RO, which enables lower operating pressures but also makes it less effective at removing contaminants. Higher operating pressures and initial flux generally enhance removal. Temperature and pH are also significant parameters affecting performance. In general, organic NF membranes have lower operating costs and easier processing than inorganic membranes while maintaining appropriate robustness for PFBS treatment (Jin et al., 2021). NF and RO tend to have high operating expenses, use significant amounts of energy, and generate concentrate waste streams which require disposal. Generally, NF and RO require pre- and posttreatment processes.

PFBS removal fluxes are generally around 40 liters per square meter per hour ($L/[m^2 \cdot hr]$) at about 0.7 megapascal (MPa) operating pressure (Wang et al., 2018). Temperature can

dramatically impact flux; it is common to normalize flux to a specific reference temperature for operational purposes (U.S. EPA, 2005c). It is also common to normalize flux to pressure ratios to identify productivity changes attributable to fouling (U.S. EPA, 2005c). It is important to note that water may traverse the membranes from outside-in or inside-out; different system configurations operating at the same flux produce differing quantities of finished water. This means that membrane systems with differing configurations cannot be directly compared based on flux. Total flow per module and cost per module are more important decision support indicators for capital planning.

High-pressure membranes may have effects when added onto a well-functioning treatment train. For instance, high-pressure membranes may remove beneficial minerals and increase corrosivity. Increased water corrosivity may need to be addressed through corrosion treatment modifications and water may require mineralization. For more information, see AWWA (2007).

6.1 Point-of-Use Devices for Individual Household PFBS Removal

Although the focus of this section is the different available options for removal of PFBS at DWTPs, centralized treatment technologies can also be used in a decentralized fashion as point-of-entry (POE) (where the distribution system meets a service connection) or point-of-use (POU) (at a specific tap or application) treatment in cases where centralized treatment is impractical or individual consumers wish to further reduce their individual household risks. Many home drinking water treatment units are certified by independent third-party accreditation organizations using American National Standards Institute (ANSI) standards to verify contaminant removal claims. NSF International has developed a protocol for NSF/ANSI Standard 58 (RO) that establishes minimum requirements for materials, design, construction, and performance of POU systems (NSF/ANSI, 2021). Currently, these standards provide certification procedures for PFOA and PFOS removal in drinking water to below EPA's 2016 PFOA and PFOS HA level of 70 ppt. When properly maintained, these systems may reduce other PFAS, including PFBS, although removal should not be automatically inferred for PFAS not specified within the protocol. PFBS removal by faucet filters has reportedly averaged 94%, whereas pitcher filters had an average of 65% removal, refrigerator filters 29%, single-stage under-sink filters 84%, two-stage filters > 92%, and RO filters 94% (Herkert et al., 2020). PFBS specific certification procedures may be developed in the future by voluntary consensus standards organizations. Individuals interested in POU or POE treatment should check with the manufacturers of these devices as to whether they have been independently certified for the reduction of PFBS levels in drinking water.

6.2 Treatment Technologies Summary

Non-treatment PFBS management options, such as changing source waters, source water protection, or consolidation, are viable strategies for reducing PFBS concentrations in finished drinking water. Should treatment be necessary, NF along with RO are the best means for removing PFBS from drinking water and can be used in central treatment plants or in POU/POE applications. Sorption processes such as activated carbon or ion exchange may successfully remove PFBS, but with lower efficacy than PFAS with a longer carbon backbone such as PFOS. PFBS treatment technologies often require pre- as well as post-treatment and may help remove other unwanted contaminants and disinfection byproduct precursors. These treatment processes are separation technologies and produce waste streams with PFBS on or in them.

7.0 Consideration of Noncancer Health Risks from PFAS Mixtures

EPA recently released a *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* (U.S. EPA, 2021f) that is currently undergoing Science Advisory Board (SAB) review. That draft document describes a flexible, data-driven framework that facilitates practical component-based mixtures evaluation of two or more PFAS based on current, available EPA chemical mixtures approaches and methods (U.S. EPA, 2000b). Examples are presented for three approaches—Hazard Index (HI), Relative Potency Factor (RPF), and Mixture BMD—to demonstrate application to PFAS mixtures. To use these approaches, specific input values and information for each PFAS are needed or can be developed. These approaches may help to inform PFAS evaluation(s) by federal, state, and tribal partners, as well as public health experts, drinking water utility personnel, and other stakeholders interested in assessing the potential noncancer human health hazards and risks associated with PFAS mixtures.

The HI approach, for example, could be used to assess the potential noncancer risk of a mixture of four component PFAS for which HAs, either final or interim (iHA), are available from EPA (PFOA, PFOS, GenX chemicals [hexafluoropropylene oxide dimer acid and its ammonium salt], and PFBS). In the HI approach described in the draft framework (U.S. EPA 2021f), a hazard quotient (HQ) is calculated as the ratio of human exposure (E) to a human health-based toxicity value (e.g., reference value [RfV]) for each mixture component chemical (i) (U.S. EPA, 1986). The HI is dimensionless, so in the HI formula, E and the RfV must be in the same units (Eq. 6). In the context of PFAS in drinking water, a mixture PFAS HI can be calculated when health-based water concentrations (e.g., HAs, Maximum Contaminant Level Goals [MCLGs]) for a set of PFAS are available or can be calculated. In this example, HQs are calculated by dividing the measured component PFAS concentration in water (e.g., expressed as ng/L) by the relevant HA (e.g., expressed as ng/L) (Eqs. 7, 8). The component chemical HQs are then summed across the PFAS mixture to yield the mixture PFAS HIs based on interim and final HAs.

$$HI = \sum_{i=1}^n HQ_i = \sum_{i=1}^n \frac{E_i}{RfV_i} \quad (\text{Eq. 6})$$

$$HI = HQ_{PFOA} + HQ_{PFOS} + HQ_{GenX} + HQ_{PFBS} \quad (\text{Eq. 7})$$

$$HI = \left(\frac{[PFOA_{water}]}{[PFOA_{iHA}]} \right) + \left(\frac{[PFOS_{water}]}{[PFOS_{iHA}]} \right) + \left(\frac{[GenX_{water}]}{[GenX_{HA}]} \right) + \left(\frac{[PFBS_{water}]}{[PFBS_{HA}]} \right) \quad (\text{Eq. 8})$$

Where:

HI = hazard index

n = the number of component (i) PFAS

HQ_i = hazard quotient for component (i) PFAS

E_i = human exposure for component (i) PFAS

RfV = human health-based toxicity value for component (i) PFAS

HQ_{PFAS} = hazard quotient for a given PFAS

[PFAS_{water}] = concentration of a given PFAS in water

[PFAS_{HA}] = HA value, interim or final, for a given PFAS

In cases when the mixture PFAS HI is greater than 1, this indicates an exceedance of the health protective level and indicates potential human health risk for noncancer effects from the PFAS mixture in water. When component health-based water concentrations (in this case, HAs) are below the analytical method detection limit, as is the case for PFOA and PFOS, such individual component HQs exceed 1, meaning that any detectable level of those component PFAS will result in an HI greater than 1 for the whole mixture. Further analysis could provide a refined assessment of the potential for health effects associated with the individual PFAS and their contributions to the potential joint toxicity associated with the mixture. For more details of the approach and illustrative examples of the RPF approach and Mixture BMD approaches please see U.S. EPA (2021f).

8.0 Health Advisory Characterization

EPA is issuing a lifetime noncancer drinking water HA for PFBS of 2,000 ng/L or 2,000 ppt based on the best available science. This is the first HA for PFBS. The PFBS HA is considered applicable to both short-term and chronic risk assessment scenarios because the critical effect identified for PFBS can result from developmental exposure and leads to long-term adverse health effects (Feng et al., 2017). The input values for the HA include 1) the chronic RfD which was developed in the toxicity assessment for PFBS (U.S. EPA 2021a); 2) the RSC based on exposure information collected from a literature search and following EPA's Exposure Decision Tree (U.S. EPA, 2000a) and presented herein; and 3) the DWI-BW, described herein, selected for the sensitive population or lifestage. The PFBS toxicity assessment was published after rigorous scientific review, including internal and external review, and public comment.

Some of the uncertainties associated with the PFBS lifetime noncancer HA are due to data gaps. The PFBS toxicity assessment, which was the basis for the chronic RfD used to derive the HA, performed a systematic literature search and identified a limited number of studies examining health effects after PFBS exposure (U.S. EPA, 2021a). The toxicity assessment literature search did not identify available chronic studies or cancer studies for PFBS. Only a small number of human studies per health outcome category were identified. The identified animal studies of repeated-dose PFBS exposure used K⁺PFBS as the tested substance and only examined noncancer effects. Further, since neurodevelopmental effects are of particular concern when perturbations in thyroid hormone occur during development, studies evaluating neurodevelopmental effects following PFBS exposure during development are needed (U.S. EPA, 2021a). Mechanistic studies were assessed as part of the systematic literature review but mechanism(s) of toxicity for PFBS for the various health outcomes have not been established.

Based on the data gaps and limitations described above, there is some uncertainty about whether the most sensitive population or life stage for PFBS exposure has been identified. Results of the literature search for information that could inform RSC determination for PFBS indicate that there is significant exposure from media other than drinking water, but the available data do not allow for quantitative characterization of the contributions of non-drinking water exposures. This

final HA is based on a recent toxicity assessment and recent literature searches of the publicly available scientific information regarding health effects, exposure, analytical methods, and treatment technologies for PFBS.

8.1 Comparative Analysis of Exposure Factors for Different Populations

The exposure duration in the critical study identified for PFBS in the toxicity assessment (U.S. EPA, 2021a) is throughout gestation which suggests that pregnant women and their developing embryo and fetus represent a sensitive life stage. In addition to drinking water exposure to pregnant women (and their developing embryo and fetus), the gestational exposure window is relevant to drinking water exposure to women of childbearing age (13 to < 50 years) who may be or become pregnant (Table 5).

EPA compared the impact of using the DWI-BW for the 90th percentile for the general population (all ages) with the DWI-BWs for the potentially sensitive populations identified, women of childbearing age and pregnant women on the HA value (Table 6). All three HA values are the same when rounded to one significant figure (i.e., all are 0.002 ppm). This indicates that the lifetime noncancer HA developed for PFBS based on the selected DWI-BW for women of childbearing age is protective of the 90th percentile of all ages of the general population.

Table 6. Comparison of HA Values Using EPA Exposure Factors for Drinking Water Intake for Different Candidate Populations

Population	DWI-BW (L/kg bw-day)	HA calculated/HA rounded to one significant figure	Description of Exposure Metric	Source
Pregnant women	0.0333	0.00180/ 0.002 ppm	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 (U.S. EPA, 2019a)
Women of childbearing age	0.0354	0.00169/ 0.002 ppm	90th percentile direct and indirect consumption of community water, consumer-only two-day average, 13 to < 50 years.	
General population, all ages	0.0338	0.00177/ 0.002 ppm	90th percentile direct and indirect consumption of community water, consumer-only two-day average, all ages.	2019 Exposure Factors Handbook Chapter 3, Table 3-21, NHANES 2005–2010 (U.S. EPA, 2019a)

Notes: L/kg bw-day = liters of water consumed per kilogram body weight per day. The DWI-BW used to calculate the PFBS lifetime HA is in bold.

9.0 References

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Appendix A: Relative Source Contribution – Literature Search and Screening Methodology

Information on all exposure sources and routes for perfluorobutane sulfonic acid (PFBS) was gathered through a literature search in a manner consistent with the Office of Science and Technology's (OST's) process the collection of information for relative source contribution (RSC) derivation. In this process, a literature search of both the peer reviewed and gray literature for the chemical of interest was conducted. All of the primary studies that were identified from the search are final documents or articles.

In 2020, U.S. Environmental Protection Agency's (EPA's) Office of Research and Development (ORD) conducted a broad literature search to evaluate evidence for pathways of human exposure to eight per- and polyfluoroalkyl substances (PFAS): perfluorooctanoic acid (PFOA), perfluorooctanesulfonic acid (PFOS), perfluorobutanoic acid (PFBA), perfluorobutane sulfonic acid (PFBS), perfluorodecanoic acid (PFDA), perfluorohexanoic acid (PFHxA), perfluorohexane sulfonic acid (PFHxS), and perfluorononanoic acid (PFNA). This search was not date limited and spanned the information collected across the Web of Science, PubMed, and ToxNet/ToxLine (now ProQuest) databases. The results of the PFBS literature search of publicly available sources are available through 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 non-human (i.e., those not identified as human) studies. 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) plus 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. 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.

After this 2020 literature search was conducted, 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 keyword¹⁶ search for water terms.

Following the Populations, Exposures, Comparators, and Outcomes (PECO) inclusion criteria outlined in Table A-1, 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 made a final decision. The title and abstract screening of and of this synthesis resulted in 191 unique studies being tagged as relevant

¹⁶ Keyword list: water, aquifer, direct water, freshwater, fresh water, groundwater, groundwater, 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

(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 ORD, 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 the ORD literature search, 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.

Table A-1. Populations, Exposures, Comparators, and Outcomes (PECO) Criteria

PECO Element	Inclusion Criteria
Population	Adults and/or children in the general and impacted populations 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)

Note:

^a Surface water and groundwater were not included as relevant media in ORD’s literature search. Studies were re-screened for these two media in this synthesis.

The evidence database 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 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 A-1. Based on full-text review, four studies were identified as relevant.

Using the screening results from the evidence database and this synthesis, a total of 151 studies were identified as relevant and are summarized below.

To supplement the primary literature database, EPA also searched the following 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:

- U.S. EPA. 2021a. *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*

- Centers for Disease Control’s national reports on human exposures to environmental chemicals
- EPA’s CompTox Chemicals Dashboard
- EPA’s fish tissue studies
- EPA’s Toxics Release Inventory
- EPA’s Unregulated Contaminant Monitoring Rule (UCMR) data
- Relevant documents submitted under Toxic Substances Control Act (TSCA) and relevant reports from U.S. EPA’s Office of Chemical Safety and Pollution Prevention
- 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 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 RSC 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*
- Additional sources, as needed

EPA has included available information from these gray literature sources for PFBS relevant to its uses, chemical and physical properties, and for occurrence in drinking water (directly or indirectly in beverages like coffee, tea, commercial beverages, or soup), ambient air, foods (including fish and shellfish), incidental soil/dust ingestion, and consumer products. EPA has also included available information specific to PFBS on any regulations that may restrict PFBS levels in media (e.g., water quality standards, air quality standards, food tolerance levels).

Appendix B: Compilation of Data on PFBS Occurrence in Environmental Media Collected from Primary Literature

This appendix includes tables resulting from the efforts to identify and screen primary literature (i.e., peer-reviewed journal articles), described in Appendix A, as well as extract data that may be relevant to informing the RSC derivation for PFBS.

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

Study	Location	Site Details	Results
North America			
Bradley et al. (2020)	United States (Chicago, Illinois; East Chicago, Indiana)	Residential tap water (45 sites); treated, pre-distribution tap water from water filtration plants (4 sites)	Residential tap water: DF ^a 47%, range = ND–0.8 ng/L Pre-distribution tap water = DF ^a 75%, range = ND–0.5 ng/L
Hu et al. (2019)	United States (national)	Archived tap water samples (collected 1989–1990) from 225 homes of Nurses' Health Study participants (across 22 states)	DF 5%, median (range) = 0.20 (ND–2.97) ng/L
Boone et al. (2019)	United States (national)	Treated water from 25 DWTPs; some locations reportedly had known or suspected sources of wastewater in the source water, but the study did not identify which	DF 96%, median (range) = 1.17 (ND–11.9) ng/L
Dasu et al. (2017)	United States (Ohio, Kentucky)	Tap water collected in 2003–2006 from 25 homes of Health Outcomes and Measures of the Environment study participants	DF 16%, range = ND–11.7 ng/L
Subedi et al. (2015)	United States (New York)	Tap water (from outdoor taps; 27 samples) from 4 homes around Skaneateles Lake that use an enhanced treatment unit for onsite wastewater treatment	DF 7%, mean (range) = 0.44 (ND–0.48) ng/L

Study	Location	Site Details	Results
Appleman et al. (2014)	United States (Wisconsin, Oklahoma, Alaska, California, Alabama, Colorado, Ohio, Nevada, Minnesota, New Jersey)	Finished water from DWTPs where source waters were impacted by upstream wastewater effluent discharge	DF 100% (n=19), mean ^a (range) = 4.27 (0.43 - 37) ng/L
Scher et al. (2018)	United States (Twin Cities metropolitan region, Minnesota)	Tap water from exterior taps of homes near former 3M PFAS production facility; 20 homes within and 3 homes outside of the GCA (GCA defined by well monitoring conducted by Minnesota Department of Health and the Minnesota Pollution Control Agency)	Within GCA: DF 0% Outside GCA: DF 0%
Boone et al. (2014)	United States (New Orleans, Louisiana)	Tap water from one home when the river source water was at a low stage (2.95 ft) or a high stage (8.32 ft); well water samples from wells on a firefighting training site that used AFFF (3 wells sampled before carbon adsorption treatment and 1 well sampled after; number of samples collected per well not reported)	Tap water (low river stage): DF 100%, mean of primary and duplicate = 14.15 ng/L Tap water (high river stage): DF 100%, mean of 4 replicates = 2.12 ng/L Well 1: DF NR, mean = 11.9 ng/L Well 1 (after carbon adsorption treatment): DF NR, mean = 9.09 ng/L Well 2: DF NR, mean = 9.265 ng/L Well 3: DF NR, mean = 29 ng/L
Lindstrom et al. (2011)	United States (Alabama)	Samples from 6 wells used for drinking water located in areas with historical land application of fluorochemical industry-impacted biosolids	DF ^a 66%, mean (range) = 19.7 (ND–56.5) ng/L

Study	Location	Site Details	Results
Chow et al. (2021)	United States (Baltimore, Maryland metropolitan area)	101 different non-carbonated bottled water products representing 66 brands, purchased from 19 different retail food and beverage chains	DF 17%, median (range) = 0.25 (ND–1.44) ng/L
Europe			
Harrad et al. (2019)	Ireland (Dublin, Galway, and Limerick counties)	Bottled water (31) from Galway city shops; tap water (private supply) from 25 homes with private water supplies; tap water (main public supply) from 34 homes and 32 offices (combined)	Bottled water: DF 29%, mean (range) = 3.7 (ND–51) ng/L Tap water (private supply): DF 0% Tap water (main public supply): DF 8%, mean (range) = 0.52 (ND–15.06) ng/L
Ünlü Endirlik et al. (2019)	Turkey (33 provinces)	Bottled water (26 samples representing 18 different brands, both plastic- and glass-bottled); municipal tap water (94 samples)	Bottled water: DF 8%, mean (range) = 0.20 (ND–0.21) ng/L Tap water: DF 87%, mean (range) = 0.29 (ND–0.85) ng/L
Ciofi et al. (2018)	Italy (Tuscany)	8 drinking water samples from various rural, urban, and industrial districts of Tuscany (origins not further described, but latitudinal and longitudinal coordinates for sampling locations were provided)	DF 0%
Le Coadou et al. (2017)	France (national)	Bottled water (25 samples of natural mineral water and 15 samples of spring water)	DF 2.5% (only one detection); single detection value (range) = 1.4 (ND–1.4) ng/L
Shafique et al. (2017)	Germany (Leipzig)	Tap water (2 samples) from one location (authors' research institute)	DF NR, mean = 1.3 ng/L

Study	Location	Site Details	Results
Filipovic and Berger (2015)	Sweden (Bollebygd, Bromma, Umeå)	Tap water from four WWTPs (4 or 5 samples from each)	Bollebygd: DF 75%, mean = 0.015 ng/L Norrvatten, Bromma: DF 100%, mean = 1.33 ng/L Stockholm Vatten, Bromma: DF 100%, mean = 1.55 ng/L Umeå: DF 100%, mean = 0.035 ng/L
Zafeiraki et al. (2015)	Greece, the Netherlands	Bottled water (5 samples each from Greece and the Netherlands); tap water samples (37 samples from the Netherlands and 43 samples from Greece)	Tap water: Greece: DF 2.3% (only one detection); single detection value (range) = 0.7 (ND–0.7) ng/L The Netherlands: DF 35%, median (range) = 7.6 (ND–13.7) ng/L Bottled water: Greece: DF 0% The Netherlands: DF 0%
Eschauzier et al. (2013)	The Netherlands (Amsterdam)	Hot water and tap water from two different locations (A and B), where A and B originated from different DWTPs; additional tap water samples (n=4) from cafes, universities, and supermarkets	Hot water A: point = 3.3 ng/L Tap water A: point = 3.2 ng/L Hot water B: point = 19 ng/L Tap water B = 16 ng/L Tap water (n=4): DF NR, mean (range) = 16 (14–17) ng/L
Gellrich et al. (2013)	Germany (Hesse, Saxony Anhalt); Switzerland; Czech Republic	Bottled water; spring water; tap water from homes	Bottled mineral water: DF 16%, median (range) = 2.6 (ND–13.3) ng/L Spring water: DF 6%, median (range) = 3.2 (ND–3.2) ng/L Tap water: DF 42%, median (range) = 2.7 (ND–5.8) ng/L
Eriksson et al. (2013)	Denmark (Faroe Islands)	Treated water from DWTPs (source water from Havnardal Lake or Kornvatn Lake)	Havnardal Lake: DF 0% Kornvatn Lake: DF 0%

Study	Location	Site Details	Results
Boiteux et al. (2012)	France (national)	Treated water from DWTPs across two sampling campaigns (41 samples in first campaign, 69 samples in second campaign)	<p>First campaign (treated water originating from surface water): DF 46%, median = < 1 ng/L, maximum = 3 ng/L</p> <p>First campaign (treated water originating from groundwater): DF 40%, median = < 1 ng/L, maximum = 3 ng/L</p> <p>Second campaign (treated water originating from surface water): DF NR, range = ND–< 10 ng/L</p> <p>Second campaign (treated water originating from groundwater): DF NR, range = ND–13 ng/L</p>
Eschauzier et al. (2012)	The Netherlands (Amsterdam)	Finished water from DWTP (n=5); tap water from 1 home	<p>Finished water from DWTP: DF NR, mean (range) = 20 (17–24) ng/L</p> <p>Tap water: point = 19 ng/L</p>
Llorca et al. (2012)	Germany, Spain	Mineral bottled water (2 samples from Germany, 4 samples from Spain); tap water (84 samples from Spain, 5 samples from Germany); well water (2 samples from Spain, 0 samples from Germany)	<p>Bottled water (both Germany and Spain): DF 0%</p> <p>Tap water:</p> <p>Germany: DF 0%</p> <p>Spain: DF 35%, mean (range) = 8.3 (ND–36 ng/L)</p> <p>Well water (Spain): DF 0%</p>
Ullah et al. (2011)	Belgium (Antwerp); Germany (Schmallenberg); Italy (Ispra); the Netherlands (Amsterdam); Norway (Tromsø); Sweden (Stockholm)	Tap water from seven research institutes in six European countries	<p>Belgium: point = 2.94 ng/L</p> <p>Germany: point = 0.092 ng/L</p> <p>Italy: point = 0.502 ng/L</p> <p>The Netherlands: DF^a 100%, mean^a (range) = 13.2 (7.61–18.8) ng/L</p> <p>Sweden: point = 0.955 ng/L</p> <p>Norway: point = ND</p>

Study	Location	Site Details	Results
Hölzer et al. (2011)	Germany (Sauerland)	Tap water (56 samples) treated from Lake Möhne, which became contaminated by perfluorocompounds through application of polluted soil conditioner to agricultural fields	DF 43%, mean (range) = 11 (ND–36) ng/L
Ericson et al. (2009)	Spain (5 regions of Catalonia)	Tap water from 40 locations identified as important supply areas	Overall: DF 73%, mean (range) = 4.52 (ND–69.43) ng/L Barcelona: DF 86%, mean (range) = 11.99 (ND–69.43) ng/L Girona: DF 57%, mean (range) = 1.13 (ND–4.91) ng/L Lleida: DF 43%, mean (range) = 0.07 (ND–0.16) ng/L Tarragona: DF 86%, mean (range) = 0.32 (ND–0.55) ng/L Terres de l'Ebre: DF 80%, mean (range) = 0.45 (ND–1.28) ng/L
Ericson et al. (2008b)	Spain (Tarragona Province)	Bottled water; municipal tap water from public fountains of most populated towns in the province	Bottled water: DF 0% Tap water: DF 0%
Pitter et al. (2020)	Italy (Veneto region)	Treated water from DWTP where its source water was contaminated by PFAS manufacturing plant	DF 89.5%, median (range) = 91.5 (ND–765.0) ng/L
Brandsma et al. (2019)	The Netherlands (Dordrecht)	Tap water from homes within 50 km of fluorochemical manufacturing plant	DF ^a 100%, range = 2.5–11 ng/L
Li et al. (2018)	Sweden (Ronneby)	Finished water from Brantafors DWTP, near AFFF-contaminated military airfield; finished water from Kärragården DWTP	Brantafors: point = 130 ng/L Kärragården: DF 0%

Study	Location	Site Details	Results
Boiteux et al. (2017)	France (northern)	Treated water from DWTPs located 15–39 km downstream of industrial WWTP that processes raw sewage from fluorochemical manufacturing facility	DF 0%
Bach et al. (2017)	France (southern region)	Treated water from two DWTPs downstream of a fluoropolymer manufacturing facility	DF 0%
Gebbink et al. (2017)	The Netherlands (Zwijndrecht, Dordrecht, Papendrecht, Sliedrecht, Utrecht, Wageningen)	Drinking water collected from city halls in municipalities close to PFAS production plant (D1–D4), at residential home in Utrecht (D5), and at the RIKILT institute in Wageningen (D6)	D1: point = 3.4 ng/L D2: point = 3.4 ng/L D3: point = 19 ng/L D4: point = 2.3 ng/L D5: point = 1.0 ng/L D6: point = 0.54 ng/L
Gyllenhammar et al. (2015)	Sweden (Uppsala)	Finished water from DWTPs; private well (Klastorp) downstream of a military airport using AFFF	Bäcklösa: DF ^a 9%, range = ND–11 ng/L Gränby: DF 0% Private well: DF 0%
Dauchy et al. (2012)	France (unspecified)	Treated water from DWTPs located 15 km downstream of fluorochemical manufacturing facility	DF 0%
Weiss et al. (2012)	Germany (Cologne)	Private well water 950 m (Well A) and 2,000 m (Well B) downstream of a fire training area; Well A is inside the contamination plume.	Well A: DF 100%, mean ^a (range) = 50 (20–100) ng/L Well B: DF ^a 86%, range = ND–20 ng/L
Multiple Continents			
Kaboré et al. (2018)	Canada (Great Lakes, St. Lawrence River)	Tap water from homes (8 sites)	DF 100%, mean (range) = 0.5 (0.3–0.8) ng/L
	Canada (rest of Canada)	Tap water from homes (11 sites); bottled water (11 brands)	Tap water: DF 73%, mean (range) = 0.1 (ND–0.5) ng/L Bottled water: DF ^a 9%, range = ND–0.23 ng/L

Study	Location	Site Details	Results
	United States (Illinois, California)	Tap water from homes (2 sites)	DF ^a 50%; ND and 0.28 ng/L
	Norway (Oslo)	Tap water from a home (1 site)	Point = 0.72 ng/L
	France (Le Mans, Paris, Guadeloupe in French West Indies)	Tap water from homes (3 sites)	DF ^a 67%, range = ND–0.32 ng/L

Notes: AFFF = aqueous film-forming foam; DF = detection frequency; DWTP = drinking water treatment plant; ft = feet; GCA = groundwater contamination area; km = kilometer; m = meter; ND = not detected; ng/L = nanogram per liter; NR = not reported; 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%.

Table B-2. 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
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)

Study	Location	Site Details	Results
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%
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

Study	Location	Site Details	Results
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
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%.

Table B-3. 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

Study	Location	Site Details	PFBS Results
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
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

Study	Location	Site Details	PFBS Results
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
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

Study	Location	Site Details	PFBS Results
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
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 mS/cm.

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

Study	Location and Source	Food Types	Results
North America			
Schechter 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 Gdańsk, 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 (<i>P. platessa</i> , <i>M. australis</i> , <i>M. capenis</i> , <i>K. pelamis</i> , and <i>M. edulis</i>)

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%.

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 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

Study	Location	Site Details	Results
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
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

Study	Location	Site Details	Results
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
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%.

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
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

Study	Location	Site Details	Results
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)
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.