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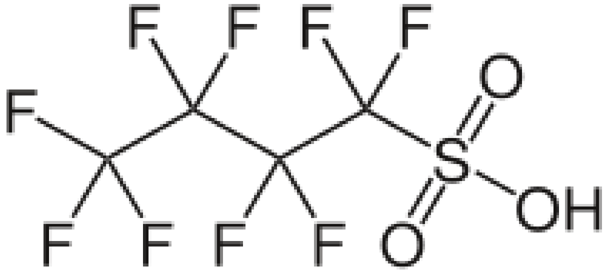
DRAFT Human Health Ambient Water Quality Criteria:

Perfluorobutane Sulfonic Acid (PFBS) and Related Salts



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1 Introduction: Background and Scope

The U.S. Environmental Protection Agency's national recommended ambient water quality criteria (AWQC) for human health are scientifically derived numeric values that define ambient water concentrations that are expected to protect human health from the adverse effects of individual pollutants in ambient water.

Section 304(a)(1) of the Clean Water Act (CWA) requires the EPA to develop and publish, and from time-to-time revise, recommended criteria for the protection of water quality that accurately reflect the latest scientific knowledge. Water quality criteria for human health developed under section 304(a) are based solely on data and scientific judgments about the relationship between pollutant concentrations and human health effects. Section 304(a) criteria do not reflect consideration of economic impacts or the technological feasibility of meeting pollutant concentrations in ambient water.

The EPA's recommended section 304(a) criteria provide technical information for states and authorized Tribes[a] to consider and use in adopting water quality standards that ultimately provide the basis for assessing water body health and controlling discharges of pollutants into waters of the United States. Under the CWA and its implementing regulations, states and authorized Tribes are required to adopt water quality criteria to protect the designated uses of waters (e.g., public water supply, aquatic life, recreational use, industrial use). The EPA's recommended water quality criteria do not substitute for the CWA or regulations, nor are they regulations themselves. Thus, the EPA's recommended criteria do not establish legal rights or obligations or impose legally binding requirements and are not final agency actions. States and authorized Tribes may adopt, where appropriate, other scientifically defensible water quality criteria that differ from these recommendations. The EPA's water quality standards regulation at 40 CFR 131.20(a) requires states and authorized Tribes to consider any new or updated national section 304(a) recommended criteria as part of their triennial review process, and, if the state or authorized Tribe does not adopt new or revised criteria for parameters that correspond to those new or revised 304(a) criteria, to provide an explanation when it submits its triennial review to EPA. This requirement is to ensure that state or Tribal water quality standards reflect the current science and protect applicable designated uses.

The water quality criteria that are the subject of this document are draft national AWQC recommendations for human health issued under CWA section 304(a). Unless expressly indicated otherwise, all references to "human health criteria," "criteria," "water quality criteria," "ambient water quality criteria recommendations," or similar variants thereof are references to draft national AWQC recommendations for human health.

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[a] Throughout this document, the term *states* means the 50 states, the District of Columbia, the Commonwealth of Puerto Rico, the Virgin Islands, Guam, American Samoa, and the Commonwealth of the Northern Mariana Islands. The term *authorized Tribe* or *Tribe* means an Indian Tribe authorized for treatment in a manner similar to a state under CWA section 518 for the purposes of section 303(c) water quality standards.

Perfluorobutane sulfonic acid (PFBS) is a member of the per- and polyfluoroalkyl substances (PFAS) class. PFAS are a large class of thousands of synthetic chemicals that have been in use in the United States and around the world since the 1940s (EPA, 2018). The ability for PFAS to withstand heat and repel water and stains makes them useful in a wide variety of consumer, commercial, and industrial products, and in the manufacturing of other products and chemicals. The current scientific evidence has shown the potential for harmful health effects after human exposure to certain PFAS. The persistence and resistance to hydrolysis, photolysis, metabolism, and microbial degradation of PFAS raise additional concerns about long-term exposure and human health effects.

The EPA developed the draft human health criteria (HHC) for PFBS to reflect the latest scientific information for input values, including exposure factors (i.e., body weight [BW], drinking water intake [DWI] rate, and fish consumption rate [FCR]), bioaccumulation factors (BAFs), human health toxicity values (i.e., reference dose [RfD]), and relative source contribution (RSC). The draft criteria are based on the EPA's current *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (2000a), which is referred to as the "2000 Methodology" in this document (EPA, 2000a).

2 Problem Formulation

Problem formulation provides a strategic framework for ambient water quality criteria development to systematically identify the major factors and chemical-specific scientific issues to be considered in the assessment (EPA, 2014a). The structure of this draft criteria document is intended to be consistent with general concepts of health assessments as described in the EPA's *Framework for Human Health Risk Assessment to Inform Decision Making* (EPA, 2014a).

In developing AWQC, the EPA follows the assessment method outlined in the 2000 Methodology (EPA, 2000a). The 2000 Methodology describes different approaches for addressing water and nonwater exposure pathways to derive human health AWQC depending on the toxicological endpoint of concern, the toxicological effect (noncarcinogenic or carcinogenic), and whether toxicity is considered a linear or threshold effect. Water sources of human exposure include both consuming drinking water and eating fish or shellfish from inland and nearshore water bodies that have been contaminated with pollutants. For pollutants that exhibit a threshold of exposure below which deleterious human health effects are unlikely to occur, as is the case for noncarcinogens and nonlinear carcinogens, the EPA applies an RSC. The RSC is the percentage of the total exposure to a contaminant that is attributed to the combination of drinking water and eating freshwater and estuarine fish and shellfish, where the remainder of exposure is allocated to other sources of oral exposure and other routes of exposure. The RSC is calculated by examining the data for other sources (e.g., air, food, soil) and pathways of exposure following the exposure decision tree for calculation of an RSC described in the 2000 Methodology (EPA, 2000a).

For carcinogenic substances for which the cancer slope factor was quantified using linear low-dose extrapolation, only the exposures from drinking water and fish ingestion are reflected in the human health AWQC; nonwater sources are not explicitly included, and no RSC is applied

(EPA, 2000a). This is because in these situations, AWQC are derived with respect to the *incremental* lifetime cancer risk posed by the presence of a substance in ambient water, rather than an individual's total risk from all exposure sources. Therefore, the resulting AWQC represents the ambient water concentration that is expected to increase an individual's lifetime risk of cancer from exposure to the pollutant by no more than one chance in one million (10^{-6}) for the general population (male and female adults, 21 years and older; referred to as "general population" herein), regardless of the additional lifetime cancer risk due to exposure, if any, to that substance from other sources. The EPA calculates AWQC at a 10^{-6} cancer risk level for the general population (EPA, 2000a). The 2000 Methodology recommends that states set human health criteria cancer risk levels for the target general population at either 10^{-5} or 10^{-6} and also notes that states and authorized Tribes can choose a more stringent risk level, such as 10^{-7}.

For substances that are carcinogenic, the EPA takes an integrated approach by considering both cancer and noncancer effects when deriving AWQC (EPA, 2000a,b). Where sufficient data are available, the EPA first derives separate AWQC for both carcinogenic and noncarcinogenic toxicity endpoints and then selects the lower (more health protective) of the two values for the recommended AWQC.

PFBS may exist in multiple forms, such as isomers or associated salts and each form may have a separate Chemical Abstracts Service registry number (CASRN) or no CASRN at all. Additionally, these compounds have various names under different classification systems. PFBS and its related salts are members of the group of PFAS known as short-chain perfluoroalkane sulfonates. PFBS is an acid that is generally present as the sulfonate anion at typical environmental pH values. Therefore, the conclusions in this document apply to all isomers of PFBS, as well as nonmetal salts of PFBS that would be expected to dissociate in aqueous solutions of pH ranging from 4 to 9. For purposes of this assessment, "PFBS" will signify the ion, acid, or any nonmetal salt of PFBS.

2.1 Uses and Sources of PFBS

PFAS are manufactured chemicals that have been widely used in industrial and consumer processes and products over the past several decades in the United States due to their repellant and surfactant properties. PFAS are persistent chemicals based on their physicochemical properties. Concerns about persistence of PFAS stem from the resistance of these compounds to hydrolysis, photolysis, metabolism, and microbial degradation.

PFBS has been used as 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; EPA, 2007). Prior to its use as a PFOS replacement, PFBS had been produced 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 of PFOS and other long-chained PFAS in humans resulted in their replacement with shorter-chain PFAS, such as PFBS, in consumer products and applications (EPA, 2021a,b). PFBS and other shorter-chain PFAS possess the desired chemical properties of longer-chain PFAS, but have shorter half-lives in humans (EPA, 2021a,b).

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 (EPA, 2021a,b). PFBS has also been used as a mist suppressant for chrome electroplating and has been found associated with the use of aqueous film-forming foam (AFFF) (EPA, 2021a,b). PFBS has been detected in dust, carpeting and carpet cleaners, floor wax, and food packaging (ATSDR, 2021; EPA, 2021a,b).

2.2 Environmental Fate and Transport in the Environment

The European Chemical Agency (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). The persistence of PFBS has been attributed to the strong carbon-fluorine (C-F) bond. PFBS has a high solubility in water (52.6 grams per liter [g/L] at 22.5–24 degrees Celsius (°C) for the potassium salt) and high mobility in the environment[b] (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 (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.

2.3 Occurrence and Detection in Sources Relevant to Ambient Water Quality Criteria

PFBS has been detected in a variety of environmental matrices. Studies describing the occurrence and detection of PFBS in sources relevant to ambient water quality criteria, including ambient water, fish, and shellfish, were identified through systematic literature searches of the peer reviewed and gray literature (see Section 6.2 below and Appendix B of EPA, 2024a for additional detail) and are described below. Additional occurrence information for sources other than ambient water (e.g., air, food, soil) is summarized in Section 6.2 as part of the determination of the RSC.

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[b] A measure of mobility is the sediment or soil organic carbon-water partition coefficient (K_{oc}) with units of liters per kilogram (L/kg) and commonly expressed as log K_{oc}, which is unitless.

2.3.1 Occurrence in Surface Water

Studies evaluating the occurrence of PFBS in surface water in North America or Europe are summarized in Table A-1. 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 nondetect (ND) to 6.2 nanograms per liter (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 concentrations ranging from ND to 47 ng/L. Several other studies from North America (four from the United States and two from Canada) evaluated surface waters from sites for which authors did not indicate whether sites were associated with any specific, known PFAS releases (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 detection frequency (DF) of PFBS in the Nakayama et al. (2010) study was 43% with median and maximum levels of 0.71 ng/L and 84.1 ng/L, respectively. As reported in EPA (2024b), Pan et al. (2018) sampled surface water sites in the Delaware River with 100% DF, though PFBS levels were relatively low (0.52 ng/L 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 ng/L 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 wastewater treatment plants (WWTPs) (Galloway et al., 2020; Zhang et al., 2016). Among these sites, PFBS DFs (0% to 100%) and PFBS levels (ND to 336 ng/L) varied. In general, PFBS DFs that ranged from 0% to 3% were associated with samples collected upstream of PFAS point sources, and higher PFBS 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 ng/L 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 in ambient water 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 greater mean PFBS concentration for the contaminated lake versus a control lake. In addition to AFFF, Nakayama et al. (2007) identified industrial sources, including metal-plating facilities and textile and paper production, as contributing to the total PFAS contamination in North Carolina's Cape Fear River Basin. Nakayama et al. (2007) reported a PFBS DF of 17% and PFBS concentrations ranging from ND to 9.41 ng/L at these sites.

The EPA identified additional studies evaluating surface water samples from sites in Europe with known or suspected PFAS releases associated with AFFF use (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, PFBS DFs varied (0 to 100%) and PFBS levels were less than 250 ng/L overall.

Ten studies conducted in Europe evaluated sites with no known fluorochemical source of contamination (Ahrens et al., 2009a,b; Barreca et al., 2020; Ericson et al., 2008; 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). While 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 ng/L 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).

2.3.2 Occurrence in Freshwater and Estuarine Fish and Shellfish

Based on the available data collected to date, PFBS has been rarely detected in freshwater and estuarine fish and shellfish in the U.S. Several large-scale sampling efforts have been conducted by the EPA and other agencies to determine PFAS levels in fish. In the EPA's 2013–2014 National Rivers and Streams Assessment (NRSA), PFBS was detected at concentrations above the method detection limit (MDL) (0.1 ng/g) but below the quantitation limit (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 (EPA, 2020). However, in the 2008–2009 NRSA, PFBS was not detected in any fish species sampled (Stahl et al., 2014). In the EPA's *2015 Great Lakes Human Health Fish Fillet Tissue Study*, PFBS was detected at a concentration of 0.36 ng/g in a smallmouth bass fillet composite collected from Lake Erie, New York (EPA, 2021c). In the National Rivers and Streams Assessment 2018–2019 (EPA, 2023) PFBS was a target chemical but was not detected in any of the fish samples analyzed. Note that PFBS was not a target chemical in the EPA's National Lake Fish Tissue Study (EPA, 2009) or the EPA's National Lakes Assessment 2017 (EPA, 2022a). PFBS was a target chemical for the National Lakes Assessment 2022 (EPA, 2024c), but was not detected in any of the fish samples analyzed (MDL 0.090 wet weight [ww]). More recently, PFBS has been detected in several estuarine species, including 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, 2024).

3 Criteria Formulas: Analysis Plan

Human health AWQC for toxic pollutants may be necessary to protect designated uses of water bodies related to ingestion of water (i.e., public water supply or source water protection) and ingestion of freshwater/estuarine fish and shellfish. *See* CWA 303(c)(2)(A)–(B). Although the AWQC are based on chronic health effects data (both cancer and noncancer effects), the criteria are intended to also be protective against adverse effects that may reasonably be expected to occur as a result of elevated acute or short-term exposures (EPA, 2000a). Human health AWQC are expected to provide adequate protection not only for the general population over a lifetime of exposure, but also for sensitive life stages and subpopulations who, because of high water- or fish intake rates, or because of biological sensitivities, have an increased risk of receiving a dose that would elicit adverse effect (EPA, 2000a).

The derivation of human health AWQC requires information about both the toxicological endpoints of concern from exposure to water pollutants and human exposure pathways for those pollutants. The EPA considers two primary pathways of human exposure to pollutants present in a particular water body when deriving human health 304(a) AWQC: (1) direct ingestion of drinking water obtained from the water body; and (2) consumption of fish and shellfish obtained from the water body.

The equations for deriving human health AWQC are presented as Equations (Eqs.) 1 and 2 for noncancer and non-linear carcinogenic effects, and Eqs. 3 and 4 for linear carcinogenic effects. The EPA derives two separate recommended human health AWQC based on 1) the consumption of both water and aquatic organisms (Eq. 1), called "water + organism"; and 2) the consumption of freshwater/estuarine fish and shellfish alone (Eq. 2), called "organism only." The use of one criterion over the other depends on the designated use of a particular water body or water bodies (i.e., drinking water source and/or fishable waters). The EPA recommends applying organism only AWQC (Eq. 2) to a water body where the designated use includes supporting fishable uses under section 101(a) of the CWA but the water body is not a drinking water supply source (e.g., non-potable estuarine waters that support fish or shellfish for human consumption) (EPA, 2000a).

The EPA recommends including the drinking water exposure pathway for ambient surface waters where drinking water is a designated use for the following reasons: (1) drinking water is a designated use for surface waters under the CWA, and therefore, criteria are needed to ensure that this designated use can be protected and maintained; (2) although they are rare, some public water supplies provide drinking water from surface water sources without treatment; (3) even among the majority of water supplies that do treat surface waters, existing treatments might not be effective for reducing levels of particular contaminants; and (4) in consideration of the agency's goals of pollution prevention, ambient waters should not be contaminated to a level where the burden of achieving health objectives is shifted away from those responsible for pollutant discharges and placed on downstream users that must bear the costs of upgraded or supplemental water treatment (EPA, 2000a).

The equations for deriving the criteria values are as follows (EPA, 2000a):

Equations for Noncancer and Nonlinear Carcinogen HHC:

Consumption of water and organisms:

$$\mathrm{AWQC} = \frac{\mathrm{RfD} \times \mathrm{RSC} \times \mathrm{BW} \times 1{,}000^{c}}{\mathrm{DWI} + \sum_{i=2}^{4} (\mathrm{FCR}_i \times \mathrm{BAF}_i)} \qquad \text{(Eq. 1)}$$

For consumption of organisms only:

$$\mathrm{AWQC} = \frac{\mathrm{RfD} \times \mathrm{RSC} \times \mathrm{BW} \times 1{,}000^{c}}{\sum_{i=2}^{4} (\mathrm{FCR}_i \times \mathrm{BAF}_i)} \qquad \text{(Eq. 2)}$$

Where:
AWQC = ambient water quality criteria, expressed in micrograms per liter (µg/L)
RfD = reference dose, expressed in milligrams per kilogram-day (mg/kg-d)
RSC = relative source contribution, unitless
BW = body weight, expressed in kg

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[c] 1,000 µg/mg is used to convert the units of mass from milligrams to micrograms.

DWI = drinking water intake, expressed in L/d

$\sum_{i=2}^{4}$ = summation of values for aquatic trophic levels (TLs), where the letter *i* stands for the TLs to be considered, starting with TL 2 and proceeding to TL 4

FCR_i = fish consumption rate for aquatic TLs (i) 2, 3, and 4, expressed in kg/d

BAF_i = bioaccumulation factor for aquatic TLs (i) 2, 3, and 4, expressed in L/kg

Equations for Linear Carcinogens HHC:

Consumption of water and organisms:

$$AWQC = \frac{RSD \times BW \times 1{,}000^{d}}{DWI + \sum_{i=2}^{4} (FCR_i \times BAF_i)} \qquad \text{(Eq. 3)}$$

For consumption of organisms only:

$$AWQC = \frac{RSD \times BW \times 1{,}000^{d}}{\sum_{i=2}^{4} (FCR_i \times BAF_i)} \qquad \text{(Eq. 4)}$$

Where:

AWQC = ambient water quality criteria, expressed in micrograms per liter (µg/L)

RSD = RSD = risk specific dose; the cancer risk level (i.e., a target risk for the population; 1 in 1 million or 10^{-6}) divided by the cancer slope factor (i.e., incidence of cancer relative to dose in units of $[mg/kg/day]^{-1}$), expressed in milligrams per kilogram-day (mg/kg-d)

BW = body weight, expressed in kg

DWI = drinking water intake, expressed in L/d

$\sum_{i=2}^{4}$ = summation of values for aquatic trophic levels (TLs), where the letter *i* stands for the TLs to be considered, starting with TL 2 and proceeding to TL 4

FCR_i = fish consumption rate for aquatic TLs (i) 2, 3, and 4, expressed in kg/d

BAF_i = bioaccumulation factor for aquatic TLs (i) 2, 3, and 4, expressed in L/kg

The EPA rounds AWQC to the number of significant figures in the least precise parameter as described in the 2000 Methodology (EPA, 2000a, Section 2.7.3). The EPA used a rounding procedure that is consistent with the 2000 Methodology (EPA, 2000a) and the 2015 HHC update (https://www.epa.gov/wqc/human-health-water-quality-criteria-and-methods-toxics).

4 AWQC Input Parameters

4.1 Exposure Factor Inputs

National recommended HHC establish ambient concentrations of pollutants in waters of the United States which, if not exceeded, will protect the general population from adverse health impacts from those pollutants due to consumption of aquatic organisms (i.e., freshwater and estuarine fish and shellfish) and water (EPA, 2000a). It is the EPA's longstanding practice to set national recommended HHC at a level intended to be adequately protective of human exposure over a lifetime (EPA, 2000a). To accomplish this, the EPA uses a combination of median values,

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[d] 1,000 µg/mg is used to convert the units of mass from milligrams to micrograms.

mean values, and percentile estimates for the HHC inputs consistent with the EPA's 2000 Methodology (EPA, 2000a). The EPA's assumptions afford an overall level of protection targeted at the high end of the general adult population (i.e., the target population or the criteria-basis population) (EPA, 2000a). This approach is reasonably conservative and appropriate to meet the goals of the CWA and the 304(a) criteria program (EPA, 2000a). If the EPA determines that another population or life stage (e.g., pregnant women and their fetuses, young children) is the target population, then exposure parameters for that target population or life stage could be considered in the derivation of the criteria (EPA, 2000a). Potentially sensitive life stages for PFBS are explored further in a comparative analysis in Appendix D.

4.1.1 Body Weight

The BW for the general adult population including males and females, ages 21 years and older, was selected for the PFBS HHC, consistent with the population selected in the agency's most recent major update to existing 304(a) HHC (EPA, 2015) and the EPA's 2000 Methodology (EPA, 2000a). The EPA used the mean weight for adults ages 21 and older of 80.0 kg, based on National Health and Nutrition Examination Survey (NHANES) data from 1999 to 2006 as reported in Table 8.1 of the EPA's *Exposure Factors Handbook* (EPA, 2011), the EPA's most recent publication of body weight exposure factors.

4.1.2 Drinking Water Intake Rate

For adults ages 21 and older, the EPA used an updated DWI of 2.3 L/d, rounded from 2.345 L/d. This DWI was estimated using the Food Commodity Intake Database consumption calculator (http://fcid.foodrisk.org[e]) which is based on NHANES 2005–2010 data used to develop the EPA's *Exposure Factors Handbook Update to Chapter 3, Ingestion of Water and Other Select Liquids* (EPA, 2019, Section 3.3.1.1). This rate represents the per capita estimate of combined direct and indirect community water[f] ingestion at the 90th percentile for adults, males and females, ages 21 and older. The EPA selected the per capita rate for the updated DWI because it represents the average daily dose estimates; that is, it includes both people who drank water during the survey period and those who did not, which is appropriate for a national-scale assessment such as the development of CWA section 304(a) national human health criteria development (EPA, 2019, Section 3.2.1). The updated DWI of 2.3 L/d reflects the latest scientific knowledge in accordance with CWA 304(a)(1).

The EPA's selection of the DWI of 2.3 L/d is consistent with the 2000 Methodology's selection of a rate based on per capita community water ingestion at the 86th percentile for adults

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[e] The FCID Consumption Calculator is an application that uses National Health and Nutrition Examination Survey/What We Eat in America (NHANES/WWEIA) food intake and FCID recipes to estimate food commodity consumption for the purposes of pesticide dietary exposure assessment, as well as consumption estimates for EPA's *Exposure Factors Handbook* (EFH) users (University of Maryland, 2024).

[f] *Community water* includes direct and indirect use of tap water for household uses and excludes bottled water and other sources (EPA, 2019, Section 3.3.1.1). *Direct ingestion* is defined as direct consumption of water as a beverage, while *indirect ingestion* includes water added during food preparation (e.g., cooking, rehydration of beverages) but not water intrinsic to purchased foods (EPA, 2019, Section 3.1).

surveyed in the U.S. Department of Agriculture's *1994–1996 Continuing Survey of Food Intake by Individuals (CSFII)* analysis (EPA, 2000a, Section 4.3.2.1).

4.1.3 Fish Consumption Rate

The FCR used for adults ages 21 years and older is 22.0 g/d, or 0.0220 kg/d (EPA, 2014b, Table 9a). This FCR represents the 90th percentile per capita consumption rate of fish from inland and nearshore waters for U.S. adults ages 21 years and older based on NHANES data from 2003–2010. The 95% confidence interval (CI) of the 90th percentile per capita FCR is 19.1 g/d and 25.4 g/d, respectively.

As recommended in the 2000 Methodology, the EPA used TL-specific FCRs to better represent human dietary consumption of fish. An organism's trophic position in the aquatic food web can have an important effect on the magnitude of bioaccumulation of certain chemicals. The TL-specific FCRs are numbered 2, 3, and 4, and they account for different categories of fish and shellfish species based on their position in the aquatic food web: TL 2 accounts for benthic filter feeders; TL 3 accounts for forage fish; and TL 4 accounts for predatory fish (EPA, 2000a).

The EPA used the following TL-specific FCRs to derive the AWQC: TL 2 = 7.6 g/d (0.0076 kg/d) (95% CI [6.4, 9.1] g/d); TL 3 = 8.6 g/d (0.0086 kg/d) (95% CI [7.2, 10.2] g/d); and TL 4 = 5.1 g/d (0.0051 kg/d) (95% CI [4.0, 6.4] g/d). Each TL-specific FCR represents the 90th percentile per capita consumption rate of fish and shellfish from inland and nearshore waters from that particular TL for U.S. adults ages 21 years and older (EPA, 2014b, Tables 16a, 17a, and 18a). The sum of these three TL-specific FCRs is 21.3 g/d, which is within the 95% CI of the overall FCR of 22.0 g/d. The EPA recommends using the TL-specific FCRs when deriving AWQC; however, the overall FCR (22.0 g/d) may be used if a simplified approach is preferred.

4.2 Bioaccumulation Factor (BAF)

4.2.1 Approach

Several attributes of the bioaccumulation process are important to understand when deriving national BAFs for use in developing national recommended section 304(a) AWQC. First, the term *bioaccumulation* refers to the uptake and retention of a chemical by an aquatic organism from all surrounding media, such as water, food, and sediment. The term *bioconcentration* refers to the uptake and retention of a chemical by an aquatic organism from water only. In some cases, experiments conducted in a lab that measure *bioconcentration* can be used to estimate the degree of *bioaccumulation* expected in natural conditions. However, for many chemicals, particularly those that are highly persistent and hydrophobic, the magnitude of bioaccumulation by aquatic organisms can be substantially greater than the magnitude of bioconcentration. In these cases, an assessment of bioconcentration alone underestimates the extent of accumulation in aquatic biota. Accordingly, the EPA guidelines presented in the 2000 Methodology (EPA, 2000a) emphasize using, when possible, measures of *bioaccumulation* as opposed to measures of *bioconcentration* (EPA, 2000a).

The EPA estimated BAFs for this draft PFBS AWQC using the 2000 Methodology (EPA, 2000a) and the associated *Technical Support Document, Volume 2: Development of National Bioaccumulation Factors* (Technical Support Document, Volume 2) (EPA, 2003). Specifically, these documents provide a framework for identifying alternative procedures to derive national TL-specific BAFs for a chemical based on the chemical's properties (e.g., ionization and hydrophobicity), metabolism, and biomagnification potential (EPA, 2000a, 2003). As described in the 2000 Methodology, the purpose of the EPA's national BAF is to represent the long-term, average bioaccumulation potential of a chemical in aquatic organisms that are commonly consumed by humans throughout the United States (EPA, 2000a). The EPA evaluated results from field BAF and laboratory bioconcentration factor (BCF) studies on aquatic organisms commonly consumed by humans in the United States for use in developing national trophic-level BAFs. National BAFs are not intended to reflect fluctuations in bioaccumulation over short periods (e.g., a few days) because human health AWQC are generally designed to protect humans from long-term (lifetime) exposures to waterborne chemicals (EPA, 2003).

The EPA followed the approach described in Figure 3-1 of the Technical Support Document, Volume 2 (EPA, 2003). The EPA used the best available data to classify each chemical according to this framework, and to derive the most appropriate BAFs following the 2000 Methodology (EPA, 2000a) and Technical Support Document, Volume 2 (EPA, 2003). Best available data consisted of peer-reviewed literature sources, government reports, and professional society proceedings, when sufficient information was provided to indicate the quality and usability of the data.

The framework provides six procedures to calculate a national BAF based on the pollutant's physical and chemical properties (see Figure 1, Procedures 1–6). Each procedure contains a hierarchy of the BAF derivation methods (listed below); however, this hierarchy should not be considered inflexible (EPA, 2000a). The four methods are:

1. **BAF Method.** This method calculates national TL-specific BAFs using water and fish and shellfish tissue concentration data obtained from field studies. Field-measured BAFs are calculated by dividing the concentration of a contaminant in an organism by the concentration of that contaminant in the surrounding water.

   For nonionic organic chemicals, BAFs are normalized to allow a common basis for averaging BAFs from several studies by adjusting for the water-dissolved portions of the chemical.

   In order to calculate representative TL-specific national BAFs used to calculate national recommended 304(a) criteria, the EPA averaged multiple field BAFs using a geometric mean of the normalized BAFs, first by species and then by TL, to calculate the TL baseline BAFs.

2. **BSAF Method.** This method uses biota-sediment accumulation factors (BSAFs) to estimate bioaccumulation. While BAFs are calculated by dividing the concentration of a contaminant in an organism by the concentration of the contaminant in water, BSAFs divide the concentration in the organism by the concentration in surrounding sediments. BSAFs are useful when calculating site-specific criteria for compounds that are highly hydrophobic—these compounds have the potential to cause bioaccumulation in aquatic organisms even when concentrations in the water column are below detection limits.

3. **BCF Method.** This method estimates BAFs from laboratory-measured BCFs. Experiments designed to calculate BCFs aim to measure bioconcentration resulting from an organism's exposure to contaminated water. Unlike BAFs measured in the field, BCF experiments do not capture bioaccumulation from other routes of exposure or biomagnification (the increase in bioaccumulation at higher levels of the food chain). However, BCFs may be used to estimate bioaccumulation if a contaminant's chemical and physical properties indicate that the compound is likely to primarily accumulate in the organism via the water exposure route, and there is no evidence that the contaminant biomagnifies in the food chain. If insufficient field-collected data are available to calculate a national BAF, then the EPA may also estimate bioaccumulation using laboratory measured BCFs and a food chain multiplier term, which accounts for biomagnification.

   A similar process to the one described in the BAF method description (above) for normalizing of water-dissolved portions of the chemical and particulate organic carbon content is used for calculating national BAFs from laboratory-measured BCF data. Ionic organic chemicals are normalized, then multiplied by the food chain multiplier if biomagnification is likely to occur. All available BCFs are averaged using a geometric mean across species and then across TL to compute baseline BAFs.

4. **K_{ow} Method.** This method predicts BAFs based on a chemical's octanol-water partition coefficient (K_{ow}), with or without adjustment using a food chain multiplier, as described in Section 5.4 of the Technical Support Document, Volume 2 (EPA, 2003).

4.2.2 Data Selection and Evaluation

The EPA conducted a systematic literature search in June 2023 of publicly available literature sources to determine whether they contained information relevant to calculating national BAFs for human health AWQC, using the 2000 Methodology and Technical Support Document, Volume 2 (EPA, 2000a, 2003). Initially, bioaccumulation data published in Burkhard (2021) was reviewed for inclusion. Burkhard (2021) evaluated bioaccumulation literature through mid-2020. To supplement this literature, a second literature search was conducted to identify additional bioaccumulation data published from 2020 through June 2023. The literature search for reporting the bioaccumulation of PFBS was implemented by developing a series of chemical-based search terms (see Appendix B) consistent with the process for derivation of BAFs used in the development of the EPA's Final Aquatic Life Criteria for perfluorooctanoic acid (PFOA) (EPA, 2024d) and PFOS (EPA, 2024e) and described in Burkhard (2021). These terms included chemical names and Chemical Abstracts Service Registry Numbers (CASRNs or CAS), synonyms, tradenames, and other relevant chemical forms (i.e., related compounds) (see Section 8). Databases searched were Current Contents, ProQuest CSA, Dissertation Abstracts, Science Direct, Agricola, Scopus, PubMed, Google Scholar, TOXNET, and UNIFY (database internal to the EPA's ECOTOX database). The supplemental literature search yielded > 10,000 results and the citation list that were further refined by excluding citations on analytical methods, human health, terrestrial organisms, bacteria, and where PFBS was not a chemical of study. The citations meeting the search criteria were reviewed for reported BAFs and/or reported concentrations in which BAFs could be calculated. Data from papers that met the inclusion and data quality screening criteria described below were extracted into the chemical dataset for PFBS.

Specifically, studies were evaluated for inclusion in the dataset used for calculating national BAFs for PFBS using the following evaluation criteria:

- Only BAF studies that included units for tissue, water, and/or BAFs were included.
- Mesocosm, microcosm, and model ecosystem studies were not selected for use in calculating BAFs.
- BAF studies in which concentrations in tissue and/or water were below the minimum level of detection were excluded.
- Only studies performed using freshwater or brackish water were included; high salinity values were excluded.
- Studies of organisms (e.g., damselfly, goby) and tissues (e.g., fish bladder) not commonly consumed by humans or not used as surrogate species for those commonly consumed by humans were excluded. Information on the ecology, physiology, and biology of the organism was used to determine whether an organism is a reasonable surrogate of a commonly consumed organisms.
- Studies in which the BAFs were not found to be at steady state were excluded.
- Initially, for pooled samples, averaging BAF data from multiple locations was only considered acceptable if corresponding tissue and water concentrations were available from matching locations (e.g., a BAF would not have been calculated using water and tissue samples collected from eight separate locations with tissue concentrations collected from only six of these corresponding locations). After further review, water samples averaged from samples collected between tissue sampling sites, were considered acceptable as these water samples were determined to be from the same overall spatial area of the study.

In addition to the evaluation criteria listed above, PFBS bioaccumulation data were also evaluated using five study quality criteria outlined in Burkhard (2021) to further evaluate BAF literature for inclusion in the national BAF calculation (Table 1).

As noted in Burkhard (2021), study quality determinations based on temporal and spatial coordination were subjective. In the absence of adequate quantifiable information regarding sample location (site coordinates for both water and tissue collection locations) or temporal coordination (specific dates of sample collection), additional follow-up with study authors was used to determine final quality values.

Table 1. Bioaccumulation factor (BAF) study quality criteria based on suggested criteria in Burkhard (2021).

Criteria	1	2	3
Number of water samples collected	> 3 samples	2–3 samples	1 sample
Number of organism samples collected	> 3 samples	2–3 samples	1 sample
Temporal coordination of water and biota samples	Concurrent collection of samples	Collected within a 1-year time frame	Collected > 1-year time frame
Spatial coordination of water and biota samples	Collected from same locations	Collected from reasonably close locations (1 kilometer [km]–2 km)	Significantly different sampling locations
General experimental design	Assigned a default value of zero for studies in which tissues from individual species were identified and analyzed		Assigned a value of 3 for studies in which tissues were from mixed species or reported as a taxonomic group.

Notes: The scores for each BAF were totaled and used to determine the overall confidence ranking for each individual BAF. The sum of quality values for the five criteria listed in Table 1 were classified as high quality (total score of 4 or 5), medium quality (total score of 5 or 6) or low quality (total score ≥ 7). Only high and medium quality data were included in final national BAFs calculations.

4.2.3 BAFs for PFBS

Following the decision framework presented in Figure 1, the EPA selected one of the four methods to develop a national-level BAF for this chemical. Because PFBS is an organic chemical that predominantly exist in an anionic form in water (ATSDR, 2021; EPA, 2021a,b, 2024a), the BSAF and K_{ow} methods would not be applicable. The EPA selected the BAF estimate using the BAF method (i.e., based on a field measured BAF) because sufficient field measured BAF data were available for PFBS.

The national-level BAF equation adjusts the TL baseline BAFs for nonionic organic chemicals by national default values for lipid content, as well as dissolved and particulate organic carbon content. The partitioning of PFBS is related to protein binding properties (ATSDR, 2021; ECHA, 2019); therefore, the EPA did not normalize measured BAF values for PFBS using lipid content when calculating baseline and national BAFs. The EPA selected the recommended 50th percentile dissolved and particulate organic carbon content for the national-level default values which is consistent with the goal of national BAFs (i.e., as central tendency estimates), as described in Section 6.3 of the Technical Support Document, Volume 2 (EPA, 2003). Adjustment for water-dissolved portions of PFBS is applied to TL baseline BAFs (EPA, 2000a) (see Appendix B).

The EPA followed the framework described in the Technical Support Document, Volume 2 (EPA, 2003), also presented in Figure 1, to select a procedure for estimating national BAFs for PFBS.

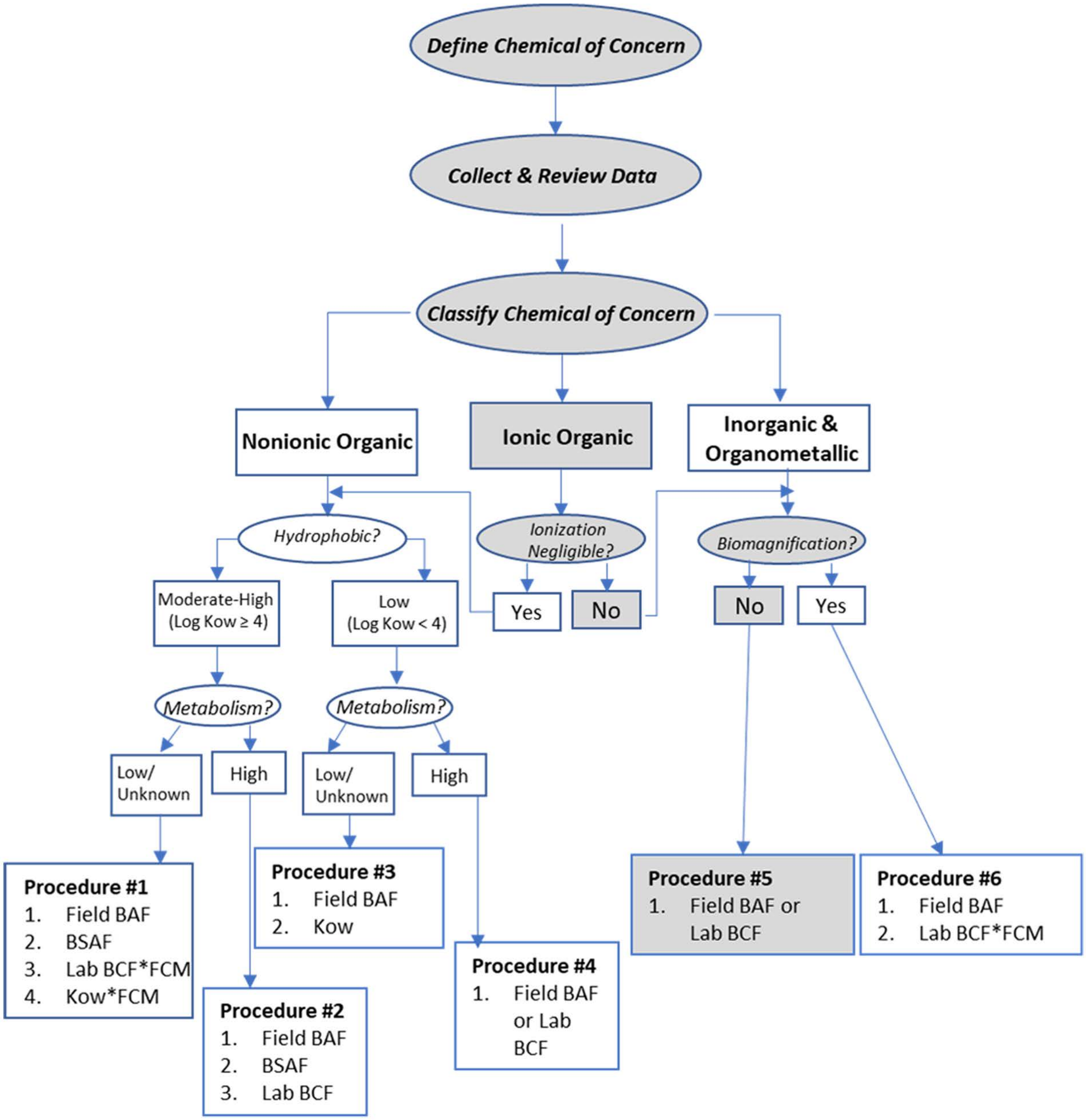

Figure 1. Application of the BAF framework for PFBS; gray boxes indicate steps followed based on available information for PFBS (EPA, 2000a).

Based on the characteristics for PFBS, the EPA selected Procedure 5 for deriving a national BAF value. PFBS has the following characteristics:

- Ionic organic chemicals, with ionization not negligible (ATSDR, 2021; EPA, 2021a,b, 2024a).
- Biomagnification unlikely (Loi et al., 2011).

The EPA was able to locate peer-reviewed, field-measured BAFs for TLs 2, 3, and 4 from the sources evaluated for which sufficient information was provided to indicate the quality and usability of the data; therefore, the EPA included only field BAF studies. The EPA used the BAF method to derive the national BAF values for PFBS:

- TL 2 = 360 L/kg
- TL 3 = 290 L/kg
- TL 4 = 870 L/kg

5 Selection of Toxicity Value

5.1 Approach

The EPA considered all available final toxicity values for both noncarcinogenic and carcinogenic toxicological effects after chronic oral exposure to develop AWQC for PFBS. As described in the 2000 Methodology (EPA, 2000a), where data are available, the EPA derives AWQC for both noncarcinogenic and carcinogenic effects and selects the more protective value for the recommended AWQC. (See Section 7, Criteria Derivation: Analysis.)

For noncarcinogenic toxicological effects, the EPA uses a chronic-duration oral reference values (RfVs; RfDs or equivalent) to derive human health AWQC. An RfV is 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 (EPA, 2002). An RfV may be derived from an animal toxicological study or a human epidemiological study, from which a point of departure (POD; i.e., a no-observed-adverse-effect level [NOAEL], lowest-observed-adverse-effect level [LOAEL], or benchmark dose [BMD]) can be derived. To derive the RfV, uncertainty factors are applied to the POD to reflect the limitations of the data in accordance with the EPA human health risk assessment methodology (EPA, 2014a, 2021b, 2024a).

For carcinogenic toxicological effects, the EPA uses an oral CSF, when applicable and available, to derive human health AWQC. The oral CSF is an upper bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to a stressor. This value may also be derived from animal toxicological studies or human epidemiological studies.

In developing AWQC, the EPA conducts a systematic search of peer-reviewed, publicly available final toxicity assessments to obtain the toxicity value(s) (RfV and/or CSF) for use in developing AWQC. The EPA identified toxicological assessments by systematically searching websites of the following EPA program offices, other national and international programs, and state programs in January 2024:

- EPA, Office of Research and Development
    - Integrated Risk Information System (IRIS) program (EPA, 2024f)
    - Provisional Peer-Reviewed Toxicity Values (PPRTV) (EPA, 2024g)
    - ORD Human Health Toxicity Values (EPA, 2024h)

- EPA, Office of Pesticide Programs (EPA, 2024i)
- EPA, Office of Pollution Prevention and Toxics (EPA, 2024j)
- EPA, Office of Water (EPA, 2024k)
    - Drinking Water Health Effects Support Documents
    - Toxicity Assessments
- U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry (ATSDR, 2024)
- Health Canada (HC, 2023)
- California Environmental Protection Agency, Office of Environmental Health Hazard Assessment (CalEPA, 2024)

After identifying and documenting all available final toxicity values, the EPA followed a systematic process to consider the identified toxicity values and select the toxicity value(s) to derive the AWQC for noncarcinogenic and carcinogenic effects. The EPA selected IRIS toxicity values to derive the draft AWQC if *any* of the following conditions were met:

1. The EPA's IRIS toxicological assessment was the only available source of a toxicity value.
2. The EPA's IRIS toxicological assessment was the most current source of a toxicity value.
3. The toxicity value from a more current toxicological assessment from a source other than the EPA's IRIS program was based on the same principal study and was numerically the same as an older toxicity value from the EPA IRIS program.
4. A more current toxicological assessment from a source other than the EPA's IRIS program was available, but it did not include the relevant toxicity value (chronic-duration oral RfD or CSF).
5. A more current toxicological assessment from a source other than the EPA's IRIS program was available, but it did not introduce new science (e.g., the toxicity value was not based on a newer principal study) or use a more current modeling approach compared to an older toxicological assessment from the EPA's IRIS program.

The EPA selected the toxicity value from a peer-reviewed, publicly available source other than the EPA IRIS program to derive the draft AWQC if *any* of the following conditions were met:

1. The chemical is currently used as a pesticide, and the EPA Office of Pesticide Programs had a toxicity value that was used in pesticide registration decision-making.
2. A toxicological assessment from a source other than the EPA's IRIS program was the only available source of a toxicity value.
3. A more current toxicological assessment from a source other than the EPA's IRIS program introduced new science (e.g., the toxicity value was based on a newer principal study) or used a more current modeling approach compared to an older toxicological assessment from the EPA's IRIS program.

5.2 Toxicity Value for PFBS

5.2.1 Reference Dose

After following the approach outlined in Section 5.1, the EPA identified the *Provisional Peer-Reviewed Toxicity Values for Perfluorobutane Sulfonic Acid (PFBS) and Related Compound Potassium Perfluorobutane Sulfonate* (EPA, 2021a), developed by the EPA's PPRTV program. The EPA identified a second human health assessment, *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)* (EPA, 2021b). These documents are identical and were identified as the most recent toxicity assessment(s) for PFBS, which use the best available science in the evaluation of noncancer risk. The EPA did not identify any other assessments that presented newer scientific information (i.e., unique RfVs) for PFBS.

The EPA's final human health toxicity assessment for PFBS (EPA, 2021a,b) considered all publicly available human epidemiology, animal toxicology, and mechanistic studies that evaluated health effects after PFBS exposure. The assessment identified associations between PFBS exposure and thyroid, developmental, and kidney health effects based on toxicology studies in animals. Limited evidence from human epidemiology studies was identified; findings for thyroid or kidney health effects was equivocal, and no studies evaluating developmental effects were identified. Human epidemiology and animal toxicology studies evaluated other health effects following PFBS exposure including effects on the reproductive system, liver, and lipid and lipoprotein homeostasis, but the evidence did not support clear associations between exposure and effect (EPA, 2021a,b). The most sensitive noncancer effect observed from sufficient quality studies was an adverse developmental effect on thyroid activity, specifically decreased serum total thyroxine, in newborn mice (postnatal day [PND] 1) born to mothers that had been orally exposed to K^+PFBS throughout gestation (Feng et al., 2017; EPA, 2021a,b).

To develop the chronic RfD for PFBS[g], the EPA derived a human equivalent dose (HED) of 0.095 mg/kg-d from BMD modeling of the critical effect in mice. The EPA then applied a composite uncertainty factor (UF) of 300 (i.e., 10× for intraspecies variability [UF_H], 3× for interspecies differences [UF_A], and 10× for database deficiencies [UF_D] to yield the chronic oral **RfD of 3×10^{-4} mg/kg-d** (EPA, 2021a,b).

5.2.2 Cancer Slope Factor

Under the 2005 EPA *Guidelines for Carcinogen Risk Assessment* (EPA, 2005), the PFBS toxicity assessment determined that there is *Inadequate Information to Assess Carcinogenic Potential* for PFBS (EPA, 2021a,b). Therefore, these most recent assessment did not derive a CSF for PFBS (EPA, 2021a,b).

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[g] Data for K^+PFBS were used to derive the chronic RfD for the free acid (PFBS), resulting in the same value (3×10^{-4}), after adjusting for differences in molecular weight between K^+PFBS (338.19) and PFBS (300.10) (EPA, 2021a,b).

6 Relative Source Contribution (RSC) Derivation

6.1 Approach

The EPA applies an RSC to the RfD when calculating an AWQC based on noncancer effects or for carcinogens that are known to act through a nonlinear mode of action to account for the fraction of an individual's total exposure allocated to AWQC-related sources (EPA, 2000a). The purpose of the RSC is to ensure that the level of a chemical allowed by a criterion (e.g., the AWQC), when combined with other identified sources of exposure (e.g., diet excluding freshwater and estuarine fish and shellfish, ambient and indoor air) common to the population of concern, will not result in exposures that exceed the RfD. In other words, the RSC is the portion of total daily exposure equal to the RfD that is attributed to consumption of ambient water (directly or indirectly in beverages like coffee tea or soup, as well as from transfer to dietary items prepared with ambient water) and fish and shellfish from inland and nearshore waters relative to other exposure sources; the remainder of the exposure equal to the RfD is allocated to other potential exposure sources. The EPA considers any potentially significant exposure source and route when deriving the RSC.

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

An RSC determination first requires "data for the chemical in question… representative of each source/medium of exposure and… relevant to the identified population(s)" (EPA, 2000a). The term "data" in this context is defined as ambient sampling measurements in the media of exposure, not internal human biomonitoring metrics. More specifically, the data must adequately characterize exposure distributions including the central tendency and high-end exposure levels for each source and 95% confidence intervals for these terms (EPA, 2000a). The EPA's approach recommends a "ceiling" RSC of 80% and a "floor" RSC of 20% to account for uncertainties including unknown sources of exposure, changes to exposure characteristics over time, and data inadequacies.

The EPA's Exposure Decision Tree approach states that when there are insufficient environmental monitoring and/or exposure intake data to permit quantitative derivation of the RSC, the recommended RSC is 20%. In the case of AWQC development, this means that 20% of the exposure equal to the RfD is allocated to the consumption of ambient water and fish and shellfish from inland and nearshore waters and the remaining 80% is reserved for other potential sources, such as diet (excluding fish and shellfish from inland and nearshore waters), air, consumer products, etc. This 20% RSC can be replaced if sufficient data are available to develop a scientifically defensible alternative value. If scientific data demonstrating that sources and routes of exposure other than drinking water are not anticipated for a specific pollutant, the RSC can be raised as high as 80% based on the available data, allowing the remaining 20% for other potential sources (EPA, 2000a). Applying a lower RSC (e.g., 20%) is a more health protective approach to public health and results in a lower AWQC.

To derive an RSC for PFBS, the EPA evaluated the exposure information identified through conducting prior systematic literature searches performed as part of the EPA's *Maximum Contaminant Level Goals (MCLGs) for Three Individual Per- and Polyfluoroalkyl Substances (PFAS) and a Mixture of Four PFAS* (EPA, 2024a), which included available information on all exposure sources and routes for PFBS. To identify information on PFBS exposure routes and sources to inform RSC determination, the EPA considered primary literature published between 2003–2020 that was collected by the EPA's Office of Research and Development as part of an effort to evaluate evidence for pathways of human exposure to eight PFAS, including PFBS. The full description of methods for peer-reviewed journal articles is available in the EPA's *Maximum Contaminant Level Goals (MCLGs) for Three Individual Per- and Polyfluoroalkyl Substances (PFAS) and a Mixture of Four PFAS* (EPA, 2024a). In order to consider more recently published information on PFBS exposure, the EPA incorporated the results of a date-unlimited gray literature search that was conducted in February 2022 and 2024 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. The following description in Section 6.2 is a summary of the information provided in the Appendix of the final MCLG document three individual PFAS, including PFBS (EPA, 2024a).

6.2 Summary of Potential Exposure Sources of PFBS Other Than Water and Freshwater and Estuarine Fish/Shellfish

6.2.1 Dietary Sources

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 Total Diet Study (FDA, 2020a,b, 2021a,b; EPA, 2024a); however, it was not detected in any of the food samples tested. It should be noted that the 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 (FDA, 2023). 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 (FDA 2018, 2021c). Maximum residue levels for PFBS were not found in the Global MRL Database (Bryant Christie, Inc., 2024).

In addition to efforts by the 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 C-1 in the Appendix). 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 C-1)—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; Kärrman et al., 2007, 2010; Lankova et al., 2013; Nyberg et al., 2018). Some PFBS dietary studies use the term “seafood” to indicate fish and shellfish from ocean, freshwater, or estuarine water bodies. Information about the water bodies assessed in individual studies is reported in the articles.

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

Several peer-reviewed publications that examined PFBS concentrations in marine fish and shellfish are also available. Schecter et al. (2010) detected PFBS in cod samples, averaging 0.12 ng/g ww. 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). 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.

The European Food Safety Authority 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 ng/kg bw-d (minimum) to 0.10–3.72 ng/kg bw-d (maximum) (EFSA, 2012). Of 27 studies conducted in Europe, 12 found PFBS in at least one food type (Table C-1). Eight of the 12 studies included food samples obtained solely from markets (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 marine fish and shellfish; 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.

In summary, in Europe and North America, PFBS has been detected in multiple food types, including fruits, vegetables, meats, marine fish and shellfish, and other fats. 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.

6.2.2 Food Contact Materials

PFBS is not authorized for use in food packaging in the United States; however, PFBS has been detected in food packaging materials in the few available studies that investigate this potential route of exposure (ATSDR, 2021; EPA, 2021a,b). In one report from the United States, PFBS was detected in fast-food packaging (7/20 samples) although the concentrations detected were not reported (Schaider et al., 2017).

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

The remaining three studies did not detect PFBS in FCMs. Zafeiraki et al. (2014) analyzed FCMs made of paper, paperboard, or aluminum foil collected from a Greek market. PFBS was not detected in any of the samples of beverage cups (n = 8), ice cream cups (n = 1), fast-food paper boxes (n = 8), fast-food wrappers (n = 6), paper materials for baking (n = 2), microwave bags (n = 3), or aluminum foil bags/wrappers (n = 14). 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 that PFBS may be present in food packaging materials; however, further research is needed to understand which packaging materials generally contain PFBS at the highest concentrations and with the greatest frequency. There are also uncertainties related to data gaps on topics that may influence whether food packaging is a significant PFBS exposure source in humans, including differences in transfer efficiency from different packaging types directly to humans or indirectly through foodstuffs.

6.2.3 Consumer Product Uses

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.

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 nonwoven 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 ng/g 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 (one pair of outdoor trousers, seven jackets, four fabrics for outdoor clothes, and one 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 levels of 5-fold or more. For eight 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 [$\mu g/m^2$], 45 to 350 $\mu g/m^2$, and 9.6 to 130 $\mu g/m^2$ respectively for the three 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 $\mu g/m^2$, 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 $\mu g/m^2$), ski wax samples (up to 3.1 micrograms per kilogram [µg/kg]), leather samples (up to 120 µg/kg), and gloves (up to 2 µg/kg). Favreau et al. (2016) analyzed the occurrence of 41 PFAS in a wide variety of liquid products (n = 132 consumer products, 194 total products), including impregnating agents, lubricants, cleansers, polishes, AFFFs, and other industrial products purchased from stores and supermarkets in Switzerland. PFBS was not detected in impregnation products (n = 60), cleansers (n = 24), or polishes (n = 18). PFBS was detected in 13% of a miscellaneous category of products (n = 23) that included foam-suppressing agents for the chromium industry, paints, ski wax, inks, and tanning substances, with mean and maximum concentrations of 998 and 2,992 parts per million (ppm), respectively (median = ND).

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

Of the two studies for which purchase location(s) were not specified, Gremmel et al. (2016) determined levels of 23 PFAS in 16 new outdoor jackets since it has been shown that outdoor jackets emit PFAS to the air as well as into water during washing. The jackets were selected based on factors such as fabric and origin of production (primarily Asia, with some origins not specified). PFBS (concentration of 0.51 $\mu g/m^2$) was only detected in one large hardshell jacket made of 100% polyester that was polyurethane-coated and finished with Teflon® (production origin unknown). Bečanová et al. (2016) analyzed 126 samples of (1) household equipment (textiles, floor coverings, electrical and electronic equipment [EEE], and plastics); (2) building materials (oriented strand board, other composite wood and wood, insulation materials, mounting and sealant foam, facade materials, polystyrene, air conditioner components); (3) car interior materials; and (4) wastes of electrical and electronic equipment (WEEE) for 15 target PFAS, including PFBS. The condition (new versus used) and production year of the samples varied; the production year ranged from 1981 to 2010. The origin(s) of production were not specified. PFBS was detected in 31/55, 9/54, 7/10, and 6/7 household equipment, building materials, car interior, and WEEE samples, respectively. The highest level was 11.4 µg/kg found in a used 1999 screen associated with WEEE.

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

6.2.4 Indoor Dust

Dust ingestion may be an important exposure source of PFAS including PFBS (ATSDR, 2021), though it should be noted that dust exposure may also occur via inhalation and dermal routes. The EPA identified several studies conducted in the United States, Canada, various countries in Europe, and across multiple continents analyzed PFBS in dust of indoor environments (primarily in homes, but also schools, childcare facilities, offices, and vehicles; see Table C-2). 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 greater than 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.

6.2.5 Air

PFAS have been released to air from WWTPs, waste incinerators, and landfills (EPA, 2016). 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, the EPA's Toxic Release Inventory did not report release data for PFBS in 2020 (EPA, 2024l). In addition, PFBS is not listed as a hazardous air pollutant (EPA, 2024m).

6.2.5.1 Indoor Air

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

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

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

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

6.2.5.2 Ambient Air

Similar to studies on indoor PFBS air concentrations, no studies from the U.S. reporting levels of PFBS in ambient air were identified from the peer-reviewed or gray literature. Four studies conducted across Europe (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 local PFAS-related activities (e.g., AFFF usage, PFAS manufacturing or use). PFBS was detected in the particle phase of outdoor air samples during one of the two sampling events in Manchester at 2.2 pg/m^3 and one of the two sampling events in Hazelrigg at 2.6 pg/m^3. PFBS was not detected above the method quantification limit at the Ireland and Norway sites. Harrad et al. (2020) measured PFBS in air near 10 Irish municipal solid waste landfills located in nonindustrial areas. Air samples were collected upwind and downwind of each landfill. PFBS was detected in more than 20% of the samples, with mean concentrations (ranges) at downwind and upwind locations of 0.50 (< 0.15–1.4) pg/m^3 and 0.34 (< 0.15–1.2) pg/m^3, respectively. Beser et al. (2011) and Jogsten et al. (2012) did not detect PFBS in ambient air samples in Spain. Beser et al. (2011) analyzed fine airborne particulate matter (PM 2.5) in air samples collected from five stations located in Alicante province, Spain (three residential, one rural, one 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 MDL.

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

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

6.2.6 Soil

PFBS can be released into soil from manufacturing facilities, industrial uses, fire/crash training sites, and biosolids containing PFBS (ATSDR, 2021; EPA, 2021a,b). The 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 C-3). 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-Avendaño et al., 2017). In these four studies, DFs ranged from 35% to 100%. PFBS concentrations in the study of the U.S. Air Force installations ranged from ND–79 ng/g, and PFBS concentrations ranged from ND–58.44 ng/g at one fire training site

(Nickerson et al., 2020). The study of the other fire training site measured PFBS pretreatment (0.61–6.4 ng/g) and posttreatment (0.07–0.83 ng/g) (Eberle et al., 2017). The DFs and range of PFBS concentrations measured in soils at the site of the train derailment were 75% DF and ND–3.15 ng/g, respectively, for the AFFF run-off area (measured in 2013, the year of accident) and 36% DF and ND–1.25 ng/g, respectively, at the burn site and adjacent area (measured in 2015) (Mejia-Avendaño et al., 2017).

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

6.2.7 Summary and Recommended RSC for PFBS

As mentioned above, the scope of exposure sources considered for the draft recommended human health AWQC is limited to surface water used for drinking water and the consumption of freshwater/estuarine fish and shellfish (EPA, 2000a), consistent with previous human health AWQC (EPA, 2015). The EPA followed the Exposure Decision Tree approach to determine the RSC for PFBS (EPA, 2000a; see Figure 2).

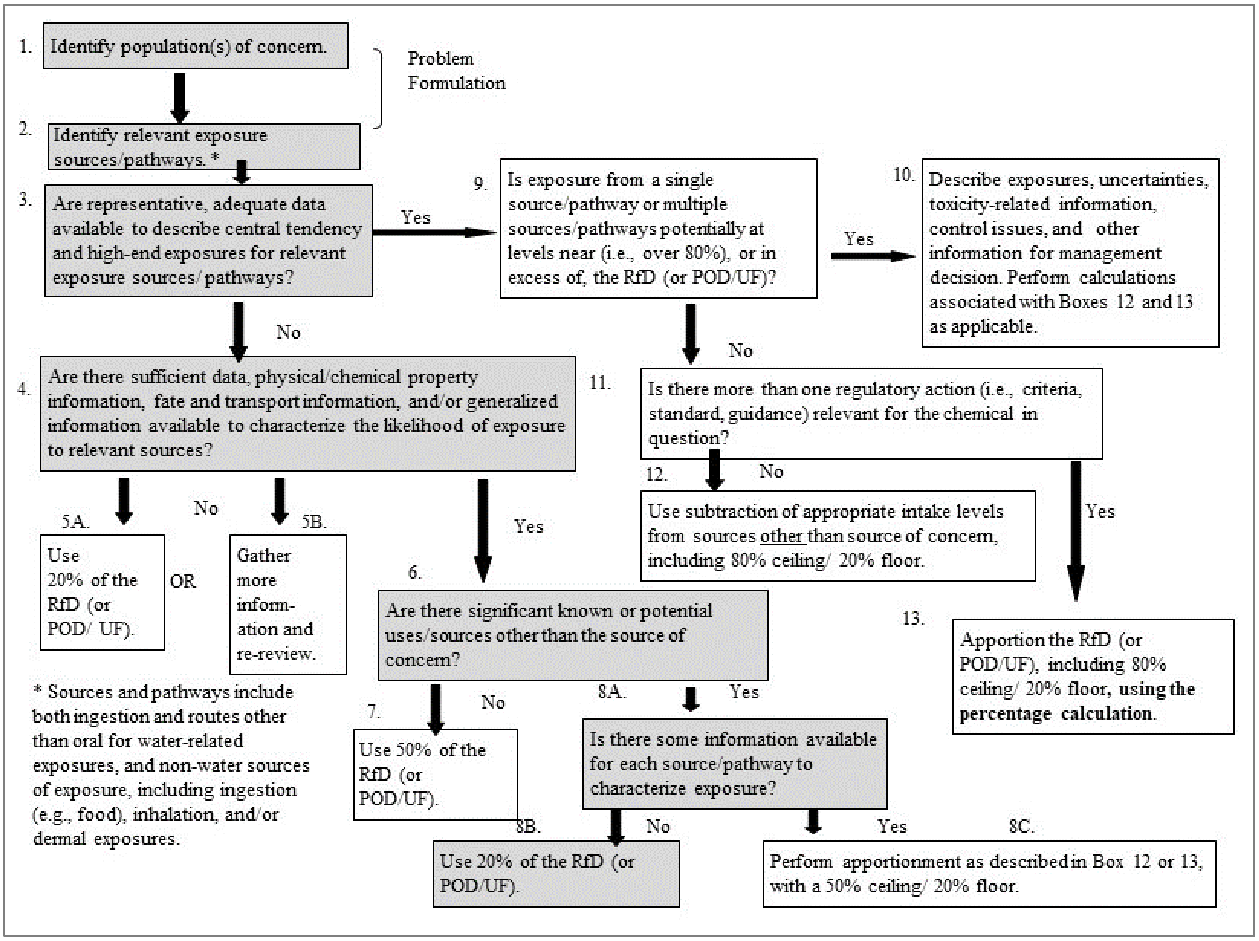

Figure 2. RSC exposure decision tree framework for PFBS; figure adapted from EPA (2000a) with gray boxes indicating key decision points for this chemical.

To identify the population(s) of concern (Box 1, Figure 2), the EPA first identified potentially sensitive subpopulations or life stages based on the PFBS exposure interval in the critical study from which the critical effect (adverse developmental effect on thyroid activity) was selected for RfD derivation in the PFBS toxicity assessment (EPA, 2021a,b). Since the critical effect is the most sensitive adverse health effect that was identified from the available data of sufficient quality, then the exposure interval may be a sensitive window of exposure. The exposure interval of the critical study in rodents corresponds to the following two potentially sensitive human life stages: women of childbearing age who may be or become pregnant; and pregnant women and their developing fetus. However, limited information was available regarding specific PFBS exposure in these two life stages from different environmental sources. Therefore, the EPA considered exposures in the general U.S. population, ages 21 years of age and older, which includes these two life stages.

Second, the EPA identified PFBS relevant exposure sources/pathways (Box 2, Figure 2), including nonfish (except marine) dietary consumption, incidental oral consumption via dust, consumer products, and soil or dermal exposure via soil, consumer products, and dust, and inhalation exposure via indoor or ambient air. Several of these may be potentially significant exposure sources.

Third, the EPA evaluated whether adequate data were available to describe the central tendencies and high-end exposures for all potentially significant exposure sources and pathways (Box 3, Figure 2). The EPA determined that there were inadequate quantitative data to describe the central tendencies and high-end estimates for all of the potentially significant sources. For example, studies from Canada and Europe indicate that indoor and ambient air may be a significant source of exposure to PFBS. At the time of the literature search, the EPA was unable to identify studies assessing PFBS concentrations in indoor or ambient air samples from the United States and therefore, the agency does not have adequate quantitative data to describe the central tendency and high-end estimate of exposure for this potentially significant source in the U.S. population.

Fourth, the agency determined whether there were sufficient data, physical/chemical property information, fate and transport information, and/or generalized information available to characterize the likelihood of exposure to relevant sources (Box 4, Figure 2). Sufficient information on PFBS was available to characterize the likelihood of exposure. The agency relied on the studies summarized above to determine if there are potential uses/sources of PFBS other than AWQC-related sources (Box 6, Figure 2). There are significant known or potential uses/sources of PFBS other than AWQC-related sources. Based on this information, the next step was to determine if adequate information was available on PFBS to characterize each source/pathway of exposure (Box 8a, Figure 2). The EPA determined there is not enough information available on each source to make a quantitative characterization of exposure among exposure sources. For example, there are several studies from the U.S. indicating that PFBS may occur in dust sampled from various microenvironments (e.g., homes, offices, daycare centers, vehicles). However, the majority of studies sampled in only one location and few studies examined dust samples outside of the home (e.g., one study from the U.S. assessed PFBS occurrence in dust sampled from vehicles). Additionally, though several studies from around the U.S. measured PFBS concentrations in dust from houses, the detection frequencies in these studies varied widely (from 3% to 59%) and may be a result of uncertainties including home characteristics, behaviors of the residents, and the presence or absence of PFBS-containing materials or products (Haug et al., 2011). Therefore, it is not possible to determine whether dust, food sources other than freshwater/estuarine fish and shellfish, or consumer products, may be major or minor contributors to total PFBS exposure. Therefore, the data are insufficient to allow for quantitative characterization of the different exposure sources. The EPA's Exposure Decision Tree approach states that when there is insufficient environmental and/or exposure data to permit quantitative derivation of the RSC, the recommended RSC for the general population is 20% (EPA, 2000a). Thus, the EPA recommends an RSC of **20% (0.20)** for PFBS for AWQC for both the water plus organism AWQC as well as the organism only AWQC (Box 8b, Figure 2).

7 Criteria Derivation: Analysis

Table 2 summarizes the input parameters used to derive the draft recommended human health AWQC that are protective of exposure to PFBS from consuming drinking water and/or eating fish and shellfish (organisms) from inland and nearshore waters. The criteria calculations are presented below. These criteria recommendations are based on the 2000 Methodology (EPA, 2000a) and the toxicity and exposure assumptions described above (see Section 4, AWQC Input Parameters; Section 5, Selection of Toxicity Value; and Section 6, Relative Source Contribution Derivation).

Table 2. Input parameters for the human health AWQC for PFBS.

Input Parameter		Value
RfD		0.0003 mg/kg-d
CSF		No data
RSC		0.20
BW		80.0 kg
DWI		2.3 L/d
FCR	TL 2	0.0076 kg/d
	TL 3	0.0086 kg/d
	TL 4	0.0051 kg/d
BAF	TL 2	360 L/kg
	TL 3	290 L/kg
	TL 4	870 L/kg

Notes: RfD = reference dose; CSF = cancer slope factor; RSC = relative source contribution; BW = bodyweight; DWI = drinking water intake; FCR = fish consumption rate; TL = trophic level; BAF = bioaccumulation factor.

7.1 AWQC for Noncarcinogenic Toxicological Effects

For consumption of water and organisms:

$$\text{AWQC}\ (\mu g/L) = \frac{\text{RfD}\ (mg/kg\text{-}d) \times \text{RSC} \times \text{BW}\ (kg) \times 1{,}000\ (\mu g/mg)}{\text{DWI}\ (L/d) + \sum_{i=2}^{4} (\text{FCR}_i\ (kg/d) \times \text{BAF}_i\ (L/kg))}$$

$$= \frac{0.0003\ mg/kg\text{-}d \times 0.20 \times 80.0\ kg \times 1{,}000\ \mu g/mg}{2.3\ L/d + ((0.0076\ kg/d \times 360\ L/kg) + 0.0086\ kg/d \times 290\ L/kg) + (0.0051\ kg/d \times 870\ L/kg))}$$

$$= 0.4011\ \mu g/L$$

$$= 0.4\ \mu g/L\ \text{(rounded)}$$

For consumption of organisms only:

$$\text{AWQC } (\mu g/L) = \frac{\text{RfD (mg/kg-d)} \times \text{RSC} \times \text{BW (kg)} \times 1{,}000\ (\mu g/mg)}{\sum_{i=2}^{4} (\text{FCR}_i \text{ (kg/d)} \times \text{BAF}_i \text{ (L/kg)})}$$

$$= \frac{0.0003 \text{ mg/kg-d} \times 0.20 \times 80.0 \text{ kg} \times 1{,}000\ \mu g/mg}{(0.0076 \text{ kg/d} \times 360 \text{ L/kg}) + (0.0086 \text{ kg/d} \times 290 \text{ L/kg}) + (0.0051 \text{ kg/d} \times 870 \text{ L/kg})}$$

$$= 0.4965\ \mu g/L$$

$$= 0.5\ \mu g/L \text{ (rounded)}$$

7.2 AWQC for Carcinogenic Toxicological Effects

The PFBS toxicity assessments determined that there is *Inadequate Information to Assess Carcinogenic Potential* for PFBS (EPA, 2021a,b; see Section 5, Selection of Toxicity Value). The EPA derives cancer-based HHC for contaminants that have been determined to be *Carcinogenic to Humans* or *Likely to Be Carcinogenic to Humans* (EPA, 2000a,c). Therefore, the EPA did not derive AWQC for carcinogenic toxicological effects.

7.3 AWQC Summary for PFBS

The EPA derived the draft recommended AWQC for PFBS using a noncarcinogenic toxicity endpoint. The human health AWQC for noncarcinogenic effects for PFBS are **0.4 µg/L** (400 ng/L) for consumption of water and organisms and **0.5 µg/L** (500 ng/L) for consumption of organisms only (Table 3). The EPA evaluated the use of exposure factors relevant to sensitive subpopulations based on the critical effect(s) used to derive the RfD (Appendix D). Based on the results of this evaluation, the criteria based on exposure factors for the general adult (≥ 21 years of age) population are the most health protective.

Under the EPA's recently finalized Method 1633 (EPA, 2024n) for aqueous samples, the level of quantification (LOQ) representing the observed LOQs in the multi-laboratory validation study, range from 1 to 4 ng/L for PFBS. The pooled MDL for PFBS is 0.37 ng/L. The pooled MDL value is derived from the multi-laboratory validation study using MDL data from eight laboratories and represents the sensitivity that should be achievable in a well-prepared laboratory but may not represent the actual MDL used for data reporting or data quality assessments (EPA, 2024n). The MDLs and ranges presented here provide a reference for comparison of analytical concentrations and recommended criteria.

Table 3. Summary of the EPA's recommended human health AWQC for PFBS chemicals.

	Human Health AWQC
Water and Organism	0.4 µg/L (400 ng/L)
Organism Only	0.5 µg/L (500 ng/L)

8 Consideration of Noncancer Health Risks from PFAS Mixtures

The EPA recently released its final *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* (referred to here as the PFAS mixtures framework; EPA, 2024o). The PFAS mixtures framework describes three flexible, data-driven approaches that facilitate practical component-based mixtures evaluation of two or more PFAS based on dose additivity, consistent with the EPA*'s Guidelines for the Health Risk Assessment of Chemical Mixtures* (EPA, 1986) and *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (EPA, 2000d). The approaches described in the EPA PFAS mixtures framework may support interested federal, state, and Tribal partners, as well as public health experts and other stakeholders to assess the potential noncancer human health hazards and risks associated with PFAS mixtures. The EPA is providing an illustration of one approach which could be applied to PFAS mixture HHC derivation. The PFAS mixtures framework underwent peer review by the EPA Science Advisory Board (EPA, 2022b) and public review and the EPA responded to comments (EPA, 2024p). The public comment period ended on May 30, 2023. The public docket can be accessed at www.regulations.gov under Docket ID: EPA-HQ-OW-2022-0114.

Dose additivity means that the combined effect of the component chemicals in a mixture is equal to the sum of the individual doses or concentrations scaled for potency. As noted in the PFAS mixtures framework, exposure to a number of individual PFAS has been shown to elicit the same or similar profiles of adverse effects in various organs and systems. Many toxicological studies of PFAS as well as other classes of chemicals support the health-protective conclusion that chemicals that elicit the same or similar observed adverse effects following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise (EPA, 2024o). Importantly, few studies have examined the toxicity of PFAS mixtures, particularly with component chemical membership and proportions that are representative of the diverse PFAS mixtures that occur in the environment. Mixtures assessments for chemicals that share similar adverse health effects, and therefore assume dose additivity, typically apply component-based assessment approaches.

The Hazard Index (HI) approach is one of the component-based mixtures assessment approaches described in the PFAS mixtures framework. In order to support states and Tribes interested in addressing potential noncancer risks of PFAS mixtures, the application of the HI approach for deriving HHC for mixtures is described below. States and authorized Tribes may choose to adopt this approach to derive HHC for PFAS mixtures. Use of the HI approach to assess risks associated with PFAS mixtures was supported by the EPA Science Advisory Board (EPA, 2022b).

In the HI approach (see PFAS mixtures framework; EPA, 2024o), 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) (EPA, 1986). The HQs for the component chemicals are then summed to derive a mixture-specific HI (for the specified exposure route/medium). Since the HI is unitless, the E and the RfV inputs to the HI formula must be expressed in the same dose units (e.g., mg/L) (Eq. 5). For example, in the context of the human health criteria, HQs for each individual PFAS are calculated by dividing the measured ambient water concentration of each component PFAS (e.g., expressed as µg/L) by its corresponding human health criterion (e.g., expressed as µg/L), and the resulting component PFAS HQs are summed to yield the PFAS mixture HI (Eqs. 5-7). Either water-plus-organism or organism-only HHC can be used in the PFAS mixtures HI approach; however, the type of HHC selected for HI calculation should be consistent. Because cancer data are lacking for most PFAS, the HI approach is currently recommended for PFAS HHC based on noncancer effects.

A hypothetical example is included below to illustrate using the HI approach to derive an HHC for a mixture of three PFAS. A PFAS mixture HI exceeding 1 indicates that co-occurrence of two or more PFAS in a mixture in ambient water exceeds the health-protective level(s), indicating potential health risks. Some individual PFAS have HHC below the analytical MDLs (e.g., PFOA, PFOS). If one such PFAS is included as a component PFAS in the HI approach, then any detectable level of that component PFAS in surface water will result in a component HQ greater than 1, and thus, an HI greater than 1 for the PFAS mixture.

$$\mathrm{HI} = \sum_{\mathrm{i}=1}^{\mathrm{n}} \mathrm{HQ_i} = \sum_{\mathrm{i}=1}^{\mathrm{n}} \frac{\mathrm{E_i}}{\mathrm{HHC_i}} \qquad \text{(Eq. 5)}$$

$$\mathrm{HI} = \mathrm{HQ_{PFBS}} + \mathrm{HQ_{PFAS_X}} + \mathrm{HQ_{PFAS_Y}} \qquad \text{(Eq. 6)}$$

$$\mathrm{HI} = \left(\frac{[\mathrm{PFBS_{ambient\ water}}]}{[\mathrm{PFBS_{HHC}}]}\right) + \left(\frac{\mathrm{PFAS_{x,ambient\ water}}}{[\mathrm{PFAS_{x,HHC}}]}\right) + \left(\frac{[\mathrm{PFAS_{Y,ambient\ water}}]}{[\mathrm{PFAS_{Y,HHC}}]}\right) \qquad \text{(Eq. 7)}$$

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
HHC_i = human health criterion for component PFAS (i)
HQ_{PFAS} = hazard quotient for a given individual PFAS
$PFAS_X$ = Hypothetical PFAS
$PFAS_Y$ = Hypothetical PFAS
[$PFAS_{\text{ambient water}}$] = concentration of a given PFAS in ambient water
[$PFAS_{HHC}$] = water-plus-organism HHC or organism-only HHC for a given PFAS

9 Chemical Name and Synonyms

- Perfluorobutane Sulfonic Acid (CASRN 375-73-5)
- Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3)
- PFBS
- K^+PFBS
- 1,1,2,2,3,3,4,4,4-Nonafluoro-1-butanesulfonic acid
- 1-Perfluorobutanesulfonic acid

- Nonafluoro-1-butanesulfonic acid
- Nonafluorobutanesulfonic acid
- Perfluorobutanesulfonic acid
- 1,1,2,2,3,3,4,4,4-Nonafluorobutane-1-sulphonic acid
- Perfluorobutanesulfonate
- Perfluorobutane sulfonate
- 1-Butanesulfonic acid, 1,1,2,2,3,3,4,4,4-nonafluoro-
- 1-Butanesulfonic acid, nonafluoro-
- Perfluoro-1-butanesulfonate
- Perfluorobutylsulfonate

10 References

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Appendix A: Summary of Supporting Literature for Surface Water Occurrence

Table A-1. Compilation of studies describing PFBS occurrence in surface water.

Study	Location	Site Details	PFBS Results
North America			
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
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)
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
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
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
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

Study	Location	Site Details	PFBS Results
Lindstrom et al. (2011)	United States (Alabama)	32 surface water samples (ponds and streams) from areas with historical land application of fluorochemical industry-impacted biosolids	DF[a] 63%, range = ND–208 ng/L
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
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
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
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
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
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
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
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
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%
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
Europe			
Ahrens et al. (2009a)	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. (2009b)	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
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%
Barreca et al. (2020)	Italy (Lombardia Region)	Rivers and streams with no known fluorochemical sources	DF[a] 39%, range = ND–16,000 ng/L
Boiteux et al. (2012)	France (national)	Rivers; some locations may have upstream industrial sources	DF 1%, range = ND–5 ng/L
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%
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

Study	Location	Site Details	PFBS Results
Ericson et al. (2008)	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%
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%
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
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
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
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
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

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%
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
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
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
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
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

Study	Location	Site Details	PFBS Results
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
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
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
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
Multiple Continents			
Pan et al. (2018)	United States (Delaware River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 2.19 (0.52–4.20) ng/L
	United Kingdom (Thames River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 5.06 (3.26–6.75) ng/L
	Germany and the Netherlands (Rhine River)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 21.9 (0.46–146) ng/L
	Sweden (Mälaren Lake)	Sampling sites were not proximate to known point sources of any fluorochemical facilities	DF[a] 100%, mean (range) = 1.43 (0.75–1.92) ng/L

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

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

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

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

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Appendix B: Bioaccumulation Factor (BAF) Supporting Information

Search Strings used for literature review of PFBS bioaccumulation data:

(“375-73-5” OR “29420-49-3” OR “45187-15-3” OR “Perfluorobutane Sulfonic Acid” OR PFBS OR “Potassium Perfluorobutane Sulfonate” OR “K+PFBS” OR “nonafluorobutane-1-sulfonic acid” OR “1,1,2,2,3,3,4,4,4-Nonafluoro-1-butanesulfonic acid” OR “Nonafluorobutane sulfonic Acid” OR “Perfluro-1-butanesulfonate” OR “1-Perfluorobutanesulfonic acid” OR “Nonafluoro-1-butanesulfonic acid” OR “Nonafluorobutanesulfonyl fluoride” OR “Nonafluorobutane-1-sulfonatato” OR “Nonafluorobutanesulfonic acid” OR “Perfluorobutanesulfonic acid” OR “Perfluorobutane sulfonic acid” OR “1,1,2,2,3,3,4,4,4-Nonafluorobutane-1-sulphonic acid” OR “1,1,2,2,3,3,4,4,4-Nonafluoro-1-butanesulfonyl fluoride” OR “1,1,2,2,3,3,4,4,4-nonafluorobutane-1-sulfonyl fluoride” OR “Perfluorobutanesulfonate” OR “Perfluorobutane Sulfonate” OR “Perfluorobutanesulfonyl fluoride” OR “1-Butanesulfonic acid, 1,1,2,2,3,3,4,4,4-nonafluoro-” OR “1-butanesulfonic acid, 1,1,2,2,3,3,4,4,4-nonafluror, potassium salt” OR “1-Butanesulfonic acid, nonafluoro-” OR “Perfluoro-1-butanesulfonate” OR “Perfluorobutylsulfonate” OR “Perfluoro-1-butanesulfonyl fluoride” OR “Potassium;1,1,2,2,3,3,4,4,4-nonafluorobutane-1-sulfonate” OR “Potassium nonafluoro-1-butanesulfonate” OR “Ammonium nonafluorobutane-1-sulfonate” OR “Ammonium perfluorobutanesulfonate” OR “Potassium perfluorobutanesulfonate” OR “Potassium perfluorobutane sulfonate” OR “Potassium nonafluorobutane-1-sulfonate” OR “Potassium nonafluoro-1-butanesulfonate” OR “Potassium PFBS” OR “PFBuS” OR “C539348” OR “1FV02N6NVO” OR “DTXSID5030030” OR “FC-98” OR “EFTOP FBSA” OR “UNII-1FV02N6NVO” OR “SCHEMBL23932” OR “CHEMBL1198521” OR “HSDB 8294” OR “CHEBI:132446” OR “CS-B0899” OR “MFCD01320794” OR “AKOS015852768” OR “NCI60_006096” OR “FT-0676348” OR “FT-0676859” OR “N0709” OR “D77221” OR “Q410426”) AND (”Bioaccumulation Factor” OR ”Bioconcentration Factor” OR bcf OR baf OR bioaccumulation OR bioconcentration OR uptake OR depuration OR accumulation)

BAF Calculation Description for PFBS

The EPA used the decision framework presented in the *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000), Technical Support Document, Volume 2: Development of National Bioaccumulation Factors* (Technical Support Document, Volume 2) (EPA, 2003) to identify procedures to derive national trophic level-specific BAFs for PFBS based on that chemical's properties (e.g., ionization, hydrophobicity), metabolism, and biomagnification potential (see Figure 1). The EPA followed the guidelines provided in Section 5.5 of EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000)* (EPA's 2000 Methodology) (EPA, 2000), to assess the occurrence of cationic and anionic forms of PFBS at typical environmental pH ranges. PFBS is a nonionic organic chemical (with ionization significant at typical environmental pH ranges) (EPA, 2021a,b).

As explained in Section 5.5 of EPA's 2000 Methodology (EPA, 2000), when a significant fraction of the total chemical concentration is expected to be present as the ionized species in water, procedures for deriving the national BAF rely on empirical (measured) methods (i.e., Procedures 5 and 6). The EPA followed the guidelines in Sections 3.2.1 and 3.2.2 of the Technical Support Document, Volume 2, to evaluate the biomagnification potential of PFBS. Based on the information in Loi et al. (2011), it was determined that biomagnification was unlikely. Based on the characteristics of PFBS, the EPA selected Procedure 5 for deriving national BAF values for this chemical.

As described in Section 4.2.1, for a given procedure, the EPA selected the method that provided BAF estimates for all three TLs (TL 2–TL 4) in the following priority:

- BAF estimates using the BAF method (i.e., based on field-measured BAFs) if possible.
- BAF estimates using the BCF method if (a) the BAF method did not produce estimates for all three TLs and (b) the BCF method produced national-level BAF estimates for all three TLs.

The EPA was able to locate field-measured BAFs for TLs 2, 3, and 4 for PFBS from the peer-reviewed literature sources for which sufficient information was provided to determine the quality and usability of the data. Therefore, the EPA used the BAF method (EPA, 2003) to derive the national BAF values for this chemical.

Calculating Baseline BAFs

As described in Section 4.2.3, the national-level BAF equation adjusts the TL baseline BAFs for nonionic organic chemicals by national default values for lipid content, as well as dissolved and particulate organic carbon content. However, the partitioning of PFBS is related to protein binding properties (ATSDR, 2021; ECHA, 2019). The EPA considered protein-normalizing the measured BAF values in the baseline BAF equation; however, insufficient data were available from the scientific literature on protein content of aquatic organisms and on the binding efficiencies of PFBS to various proteins in aquatic organisms. Because of this lack of data on the relationship between protein content and PFBS bioaccumulation, attempts to normalize BAFs based on protein content would likely introduce greater uncertainty into BAF averages.

Consistent with the EPA's 2000 Methodology (EPA, 2000), a procedure analogous to the one used to adjust for the water-dissolved portions of a nonionic organic chemical is applied to the measured BAFs for PFBS. As described in EPA's (2003) Technical Support Document, Volume 2, the K_{poc} (the equilibrium partition coefficient of the chemical between the particulate organic carbon [POC] phase and the freely dissolved phase of water) is approximately equal to the K_{ow} of a hydrophobic organic chemical. It is further described in the EPA's (2003) Technical Support Document, Volume 2, that K_{doc} (the equilibrium partition coefficient of the chemical between the dissolved organic carbon [DOC] phase and the freely dissolved phase of water) is directly proportional to the K_{ow} of a hydrophobic organic chemical, and that K_{doc} is less than the K_{ow}. The log K_{oc} for PFBS provided in ATSDR (2021) is 2.06 (as determined from a groundwater aquifer study) and was used in the national BAF calculations to adjust for the water-dissolved portions of a nonionic organic chemical. The EPA determined that the K_{oc} values were applicable to POC but there is no indication that they would be applicable to DOC. Thus, the amount of PFBS partitioned to DOC was presumed to be part of the aqueous fraction of the f_{fd} equation, resulting in the following formula (Eq. 1):

$$f_{fd}=\frac{1}{[1+(\mathrm{POC}\cdot K_{OC})]} \qquad \text{(Eq. 1)}$$

Where:

- f_{fd} = fraction of the total concentration of chemical in water that is freely dissolved.
- POC = national default value of 0.5 mg/L (refer to page 5-44 of the EPA's 2000 Methodology [EPA, 2000]) is used in baseline BAF calculations, unless this value is reported in the BAF source.
- K_{oc} = PFBS log K_{oc}; log k_{oc} = 2.06 (ATSDR, 2021).

Because the measured BAFs for PFBS are not adjusted for lipid or protein content, the baseline BAF equation (refer to Eq. 5-10 on pages 5-24 and 5-25 of the EPA's 2000 Methodology [EPA, 2000]) is adjusted (as shown below in Eq. 2) to determine the freely dissolved PFBS in water:

$$\text{Baseline BAF}=\frac{\text{Measured BAF}}{f_{fd}}-1 \qquad \text{(Eq. 2)}$$

The EPA used this equation to calculate baseline BAFs from field measured BAFs based on total concentrations.

Dissolved PFBS Baseline BAFs

The EPA included results from several field BAF studies for PFBS reported as dissolved (i.e., filtered) concentrations in its baseline BAF calculations. Because these dissolved PFBS data are presumed to represent the freely-dissolved (non-particulate) fraction, the ffd term in Eq. 2 is set to 1. Also, as described above, the measured BAFs for PFBS are not being adjusted for lipid or protein content to calculate baseline BAFs for PFBS. Thus, Eq. 3 is used to calculate the freely dissolved concentration of PFBS for "baseline BAFs" using field-measured dissolved PFBS BAFs:

$$\text{Baseline BAF}=\text{Measured (dissolved) BAF}-1 \qquad \text{(Eq. 3)}$$

Calculating the National BAFs

Final baseline BAFs were used to compute national BAFs for PFBS. Eq. 4 (an equation analogous to the equation used for nonionic organic chemicals in the EPA's 2015 Updated Human Health criteria for calculating national BAFs (see Eq. 5-28 on Page 5-42 of the EPA's 2000 Methodology [EPA, 2000]) is used to convert the baseline BAF to a national BAF for each trophic level:

$$\text{National BAF}_{(\text{TL n})} = \left[\left(\text{Final Baseline BAF}^{\text{fd}}\right)_{\text{TL n}} + 1\right] \cdot (\text{f}_{\text{fd}}) \qquad \text{(Eq. 4)}$$

Where:

- National BAF = national BAF (L/kg-tissue).
- (Final Baseline BAF)$_{\text{TL n}}$ = mean baseline BAF for TL "n" (L/kg-lipid).
- f_{fd} = fraction of the total concentration of chemical in water that is freely dissolved.

In summary, for PFBS, the baseline BAFs are calculated using Equation 2 (for field measured BAFs calculated from total water concentrations) and Equation. 3 (for field BAFs calculated from dissolved water concentrations) for each TL. National BAFs are then calculated from TL baseline BAFs using Equation 4 as shown below.

National Trophic level BAF calculations:

$$\text{National BAF PFBS}_{(\text{TL 2})} = [(355.9)_{\text{TL 2}} + 1] \times (0.9999)$$
= 356.9 L/kg
= 360 L/kg (rounded)

$$\text{National BAF PFBS}_{(\text{TL 3})} = [(285.6)_{\text{TL 3}} + 1] \times (0.9999)$$
= 286.6 L/kg
= 290 L/kg (rounded)

$$\text{National BAF PFBS}_{(\text{TL 4})} = [(866.9)_{\text{TL 4}} + 1] \times (0.9999)$$
= 867.8 L/kg
= 870 L/kg (rounded)

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Appendix C: Supporting Literature for Deriving the Relative Source Contribution

Table C-1. Compilation of studies describing PFBS occurrence in food.

Study	Location and Source	Food Types	Results
North America			
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
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
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
Schecter et al. (2010)	United States (Texas) Grocery stores	Dairy, fruits and vegetables, grain, meat, seafood[h], 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
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
Young et al. (2012)	United States (17 states) Retail markets	Dairy	ND in retail cow's milk
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

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[h] Some PFBS dietary studies use the term "seafood" to indicate fish and shellfish from ocean, freshwater, or estuarine water bodies. Information about the water bodies assessed in individual studies is reported in the articles.

Study	Location and Source	Food Types	Results
Europe			
Barbosa et al. (2018)	Belgium, France, the Netherlands, Portugal Various markets	Seafood	ND in raw and steamed fish (*P. platessa, M. australis, M. capenis, K. pelamis*, and *M. edulis*)
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
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
Ericson et al. (2008)	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
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** **Dairy (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
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**
Falandysz et al. (2006)	Poland Gulf of Gdañsk, Baltic Sea south coast	Meat, seafood	ND in eider duck, cod

Study	Location and Source	Food Types	Results
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
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**
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
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
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
Lankova et al. (2013)	Czech Republic Retail market	Fats/other	ND in infant formula
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

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)
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)**
Rivière et al. (2019)	France Based on results of national consumption survey	Seafood, fats/other	ND in infant food, vegetables, nonalcoholic beverages, dairy-based desserts, milk, mixed dishes, fish, ultra-fresh dairy products, meat, poultry and game
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
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
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

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)
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
Zafeiraki et al. (2016a)	Greece, the Netherlands Home and commercially produced	Fats/other	ND in chicken eggs
Zafeiraki et al. (2016b)	The Netherlands Local markets and slaughterhouses	Meat	ND for horse, sheep, cow, pig, and chicken liver
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 C-2. Compilation of studies describing PFBS occurrence in indoor dust.

Study	Location	Site Details	Results
North America			
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
Kubwabo et al. (2005)	Canada (Ottawa)	Homes (67)	DF 0%
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)
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
Zheng et al. (2020)	United States (Seattle, Washington and West Lafayette, Indiana)	Childcare facilities (20 samples from 7 facilities in Seattle and 1 in West Lafayette)	DF 90%, mean (range) = 0.34 (ND–0.86) ng/g
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
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
Giovanoulis et al. (2019)	Sweden (Stockholm)	Preschools (20)	DF 0%

Study	Location	Site Details	Results
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
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
Jogsten et al. (2012)	Spain (Catalonia)	Homes (10)	DF 60%, range = ND–6.5 ng/g
Padilla-Sánchez and Haug (2016)	Norway (Oslo)	Homes (7)	DF 14% (single detection), range = ND–3 ng/g
Winkens et al. (2018)	Finland (Kuopio)	Homes (63 children's bedrooms)	DF 12.7%, median (range) = ND (ND–13.5) ng/g
Multiple Continents			
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
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

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 C-3. Compilation of studies describing PFBS occurrence in soil.

Study	Location	Site Details	Results
North America			
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
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
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
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
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
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
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)
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
Venkatesan and Halden (2014)	United States (Baltimore, Maryland)	Control (nonamended) soil from Beltsville Agricultural Research Center	DF 0%

Study	Location	Site Details	Results
Europe			
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
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%
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)
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
Hale et al. (2017)	Norway (Gardermoen)	Firefighting training site	DF 0%
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

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.

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Appendix D: Comparative Analysis for Potentially Sensitive Populations for PFBS

The EPA evaluated several exposure scenarios for PFBS to determine whether the national recommended criteria for the general population, male and female adults ≥ 21 years old, are sufficiently protective of potentially sensitive subpopulations. To accomplish this, the EPA considered three additional exposure scenarios, as supported by data from the EPA *Exposure Factors Handbook* (EFH; EPA, 2011) and the Human Health Methodology (EPA, 2000). Specifically, the EPA evaluated exposure parameters for "all ages" as well as two potentially sensitive life stages associated with the critical effect used to derive the PFBS chronic RfD, i.e., adverse developmental effect on thyroid activity, specifically decreased serum total thyroxine, in newborn mice (postnatal day [PND] 1) born to mothers that had been orally exposed to K+PFBS throughout gestation (EPA, 2021a,b). Based on this exposure interval in the critical study, potentially sensitive subpopulations in humans include women of childbearing age who may be or become pregnant and pregnant women (Table D-1).

For the body weight exposure parameter, a mean bodyweight of 75 kg for pregnant women (all trimesters) was identified in the EFH (2011, Ch. 8, Table 8-29). A representative body weights for the "all ages" scenario was not specifically presented in the EFH (EPA, 2011). To address this data limitation, for this exercise, the EPA assumed that the average body weight for "all ages" was 71.6 kg based on the sum of the time-weighted averages of the mean male and female combined body weights from 1 year up to 80 years old from the NHANES (1999–2006) (Table 8-3; EPA, 2011). A body weight average of 67 kg for women of childbearing age was identified in the Human Health Methodology (EPA, 2000); however, this average is based on an older NHANES dataset (NHANES III; WESTAT 2000). More recent NHANES data (1999–2006) suggest that the mean body weight for women of childbearing age ranges from 65.9 kg for 16 to < 21-year-olds to 77.1 kg for 40 to < 50-year-olds (Table 8-5; EPA, 2011). Using these data, the EPA assumed a time-weighted average body weight of 73.4 kg for women of childbearing age (Table 8-5; EPA, 2011).

Drinking water intake values were available for all populations (Table D-1).

The EPA encountered several data limitations for trophic level specific fish consumption rates for some of these potentially sensitive populations. The EPA's national criteria are typically derived using trophic-level specific fish consumption rates (FCRs), paired with trophic-level specific bioaccumulation factors (BAFs) to account for the potential bioaccumulation of some chemicals in aquatic food webs and the broad physiological differences between trophic levels which may influence bioaccumulation (EPA, 2000). Trophic level specific FCRs for women of childbearing age were identified (Table D-1). However, trophic level specific FCRs are not available for two of the potentially sensitive life stages: all ages and pregnant women. Therefore, criteria could not be calculated for these life stages. However, in all cases with available data, the total FCR for the alternative scenarios is lower than the FCR for the general population. Because bodyweights are similar for all of the considered populations (see above and Table D-1), the FCR is likely to be the main determinant of the criteria value, with a larger FCR resulting in a lower, more health protective criterion. Therefore, criteria based on the general population are expected to be protective of the identified potentially sensitive life

Table D-1. Comparison of noncancer-based HHC values for different candidate sensitive populations identified from the critical effect and study.

Population	Bodyweight (kg)	Drinking Water Intake (L/day)	Fish Consumption Rate (g/day)				Criteria (µg/L)	
			Total	TL 2	TL 3	TL 4	W + O	OO
General, adult (≥ 21 years)	**80[a]**	**2.3[b]**	**22[c]**	**7.6[c]**	**8.6[c]**	**5.1[c]**	**0.4**	**0.5**
Women of childbearing Age (13–49 years)	73.4[d]	2.1[e]	15.8[c]	5.6[c]	6.0[c]	2.9[c]	0.6	0.8
All Ages (Birth to 80 years)	71.6[f]	2.0[b]	19.3[g]	NA	NA	NA	ND	ND
Pregnant Women	75[h]	2.1[e]	10[i]	NA	NA	NA	ND	ND

Notes: g/day = grams of fish consumed per day; L/day = liters of water per day; NA = not available; ND = not determined; OO = organism only; W + O = water plus organism.

Bold values indicate draft national recommended criteria.

Gray highlighting indicates most health protective HHC based on noncancer effects.

[a] EPA, 2011, *Exposure Factors Handbook*, Ch. 8, Table 8-1, NHANES 1999–2006.Recommended mean bodyweight for adults.

[b] Estimated using the FCID calculator (University of Maryland, 2024; https://fcid.foodrisk.org/), NHANES 2005–2010, community water, 90th percentile per capita rate.

[c] EPA, 2014; NHANES 2003–2010 survey data, 90th percentile per capita rate, freshwater and estuarine fish and shellfish edible portion, adults ≥ 21 years.

[d] Time weighted average of combined bodyweights for women ages 16 to < 50 years, NHANES 1999–2006 (EPA, 2011; Table 8-5).

[e] EPA, 2019, *Exposure Factors Handbook*; Update Ch. 3., Table 3-62, Community water, 90th percentile, per capita rate.

[f] Time weighted average of mean male and female combined body weights from 1 year up to 80 years, NHANES 1999–2006 (EPA, 2011; Table 8-3).

[g] Estimated using the FCID calculator (University of Maryland, 2024; https://fcid.foodrisk.org/), NHANES 2005–2010; freshwater and estuarine fish and shellfish combined, 90th percentile per capita rate; male and female, all ages included.

[h] EPA, 2011, Exposures Factors Handbook, Ch 8, mean, NHANES 1999–2006, Table 8-29

i Estimated using the FCID calculator (University of Maryland, 2024; https://fcid.foodrisk.org/), NHANES 2005–2010; freshwater and estuarine fish and shellfish combined, 90th percentile per capita rate pregnant females only.

stages (Table D-1). Separately, paired bodyweight adjusted FCRs are not available for specific trophic levels which precludes the use of body-weight adjusted DWI rates to derive ambient water quality criteria.

For illustrative purposes, the EPA calculated criteria based on the exposure parameters for women of childbearing age. As demonstrated in Table D-1, criteria based on the exposure inputs for the general population result in more health protective criteria and thus are protective of the potentially susceptible life stage of women of childbearing age (Table D-1). Overall, when bodyweight averages are similar, the resulting criteria are driven predominantly by the FCR; thus, a higher FCR results in a more health protective criteria.

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