



RMP
REGIONAL MONITORING
PROGRAM FOR WATER QUALITY
IN SAN FRANCISCO BAY

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PFAS in San Francisco Bay: Synthesis and Monitoring Strategy

2026 Revision

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Acknowledgements

Land acknowledgement text from Alex Tavizon with the California Indian Environmental Alliance:

The following land acknowledgment evolved out of the collaborative efforts of the original Native Peoples of the San Francisco Bay Region and has been adapted to apply to the broader San Francisco Estuary. While merely a symbolic statement, it is intended to foster a recognition of the tragic history of Native Peoples, to affirm their continued presence and contributions, and to remind non-Natives that the land upon which they live, work, and recreate is stolen from the original stewards of the land.

We acknowledge that the San Francisco Bay is the unceded ancestral homeland of many indigenous people, including the Him-R^n (HIM-rin) Ohlone (oh-LOW-nee) Jalquin (HAWL-keen), Saclan (SAHK-lan) Tribe, the Villages of Lisjan (Lih-SHAWN), the Karkin (CAR-kin), Muwekma (Moo-WECK-mah), Ramaytush (RAH-my-toosh), Tamien (THA-mee-in), and Yokuts (YOH-cuts) Ohlone, Coast and Bay Miwok (MEE-wok), Patwin (PAT-win), and the Amah (AH-mah) Mutsun (MOOT-soon) Tribal Band. The broader San Francisco Estuary is also the homeland of the Plains Miwok, Wappo (WAH-po), Wintun (WIN-tun), and Nisenan (NEE-seh-nahn) people. We recognize that we benefit from living and working in their traditional homeland. We wish to pay our respects by acknowledging the ancestors, elders, and relatives of these Tribal Communities and by affirming their sovereign rights as First Peoples of these Nations.

Please consider that although this statement acknowledges the ancestral and unceded territories of the original Native Peoples of the Estuary, additional steps are required to move towards meaningful restorative justice. This can be working with Estuary Tribes toward ecological restoration and incorporating Traditional Ecological Knowledge, building beneficial and constructive relationships, supporting Tribes in restoring their role as land stewards, and elevating and integrating Native communities in the planning and decision-making for a healthy and resilient Estuary. Tribal communities have been stewarding our natural environment since time immemorial; there is much we need to learn from the Estuary's tribal communities about stewardship.

It is vitally important that we recognize that the land on which we reside is unceded Tribal territory, and also acknowledge and support the Native Peoples who continue to form a crucial part of our Estuary community today.

PFAS Analytes

PFAS Class	PFAS Abbreviation	PFAS Name (Conjugate Base of Acids in parentheses)
Perfluoroalkyl Acids (PFAAs)		
Perfluoroalkyl Carboxylic Acids/ Carboxylates (PFCAs)	Ultra-Short Chain	
	TFA	Trifluoroacetic acid (Trifluoroacetate)
	PFPrA	Pentafluoropropionic acid (Pentrafluoropropionate)
	Short-Chain	
	PFBA	Perfluorobutanoic acid (Perfluorobutanoate)
	PFPeA	Perfluoropentanoic acid (Perfluoropentanoate)
	PFHxA	Perfluorohexanoic acid (Perfluorohexanoate)
	PFHpA	Perfluoroheptanoic acid (Perfluoroheptanoate)
	Long-Chain	
	PFOA	Perfluorooctanoic acid (Perfluorooctanoate)
	PFNA	Perfluorononanoic acid (Perfluorononanoate)
	PFDA	Perfluorodecanoic acid (Perfluorodecanoate)
	PFUnDA	Perfluoroundecanoic acid (Perfluoroundecanoate)
	PFDoDA	Perfluorododecanoic acid (Perfluorododecanoate)
	PFTTrDA	Perfluorotridecanoic acid (Perfluorotridecanoate)
	PFTeDA	Perfluorotetradecanoic acid (Perfluorotetradecanoate)
Perfluoroalkyl Sulfonic Acids/ Sulfonates (PFSAAs)	Ultra-Short Chain	
	PFMeS	Trifluoromethanesulfonic acid (Trifluoromethanesulfonate)
	PFEtS	Pentafluoroethanesulfonic acid (Perfluoroethanesulfonate)
	PFPrS	Heptafluoropropanesulfonic acid (Heptafluoropropanesulfonate)
	Short-Chain	
	PFBS	Perfluorobutanesulfonic acid (Perfluorobutanesulfonate)
	PFPeS	Perfluoropentanesulfonic acid (Perfluoropentanesulfonate)
	Long-Chain	
	PFHxS	Perfluorohexanesulfonic acid (Perfluorohexanesulfonate)
	PFHpS	Perfluoroheptanesulfonic acid (Perfluoroheptanesulfonate)
	PFOS	Perfluorooctanesulfonic acid (Perfluorooctanesulfonate)
	PFNS	Perfluorononanesulfonic acid (Perfluorononanesulfonate)
	PFDS	Perfluorodecanesulfonic acid (Perfluorodecanesulfonate)
	PFDoS	Perfluorododecanesulfonic acid (Perfluorododecanesulfonate)

PFAS Class	PFAS Abbreviation	PFAS Name (Conjugate Base of Acids in parentheses)
Known PFCA Precursors		
Fluorotelomer Sulfonic Acids/ Sulfonates (FTSs)	4:2 FTS	1H, 1H, 2H, 2H-perfluorohexane sulfonic acid (1H, 1H, 2H, 2H-perfluorohexane sulfonate)
	6:2 FTS	1H, 1H, 2H, 2H-perfluorooctane sulfonic acid (1H, 1H, 2H, 2H-perfluorooctane sulfonate)
	8:2 FTS	1H, 1H, 2H, 2H-perfluorodecane sulfonic acid (1H, 1H, 2H, 2H-perfluorodecane sulfonate)
Fluorotelomer Carboxylic Acids/ Carbonates (FTCAs)	3:3 FTCA	2H, 2H, 3H, 3H-perfluorohexanoic acid (2H, 2H, 3H, 3H-perfluorohexanoate)
	5:3 FTCA	2H, 2H, 3H, 3H-perfluorooctanoic acid (2H, 2H, 3H, 3H-perfluorooctanoate)
	7:3 FTCA	2H, 2H, 3H, 3H-perfluorodecanoic acid (7:3 FTCA, 2H, 2H, 3H, 3H-perfluorodecanoate)
Fluorotelomer Alcohols (FTOHs)	4:2 FTOH	1,1,2,2-Tetrahydroperfluoro-1-Hexanol
	6:2 FTOH	1,1,2,2-Tetrahydroperfluoro-1-Octanol
	7:2 sFTOH	1-Perfluoroheptyl ethanol
	8:2 FTOH	1,1,2,2-Tetrahydroperfluoro-1-Decanol
	10:2 FTOH	1,1,2,2-Tetrahydroperfluoro-1-Dodecanol
	12:2 FTOH	1,1,2,2-Tetrahydroperfluoro-1-Tetradecanol
Polyfluoroalkyl phosphate esters (PAPs)	6:2 monoPAP	1H,1H,2H,2H-perfluorooctylphosphate
	8:2 monoPAP	1H,1H,2H,2H-perfluorodecylphosphate
	6:2 diPAP	Bis(1H,1H,2H,2H-perfluorooctyl)phosphate
	8:2 diPAP	Bis(1H,1H,2H,2H-perfluorodecyl)phosphate
Known PFSA Precursors <i>(*may produce some PFCAs as a secondary product)</i>		
Perfluoroalkyl sulfonamides (FASAs)	PFOSA	Perfluorooctanesulfonamide
	N-MeFOSA	N-Methylperfluorooctanesulfonamide
	N-EtFOSA	N-Ethylperfluorooctanesulfonamide
Perfluorooctane Sulfonamido Acetic Acids/ Acetates (FOSAAAs)	N-MeFOSAA	N-Methylperfluoro-1-octanesulfonamidoacetic acid (N-Methylperfluoro-1-octanesulfonamidoacetate)
	N-EtFOSAA	N-Ethylperfluoro-1-octanesulfonamidoacetic acid (N-Ethylperfluoro-1-octanesulfonamidoacetate)
Perfluoroalkane sulfonamido ethanols (FASEs)	N-MeFOSE	N-Methylperfluoro-1-octanesulfonamidoethanol
	N-EtFOSE	N-Ethylperfluoro-1-octanesulfonamidoethanol

PFAS Class	PFAS Abbreviation	PFAS Name (Conjugate Base of Acids in parentheses)
<i>Ether PFAS</i>		
Per- and Polyfluoroether Carboxylic Acids/ Carboxylates (PFECAs)	HFPO-DA (GenX)	2,3,3,3-Tetrafluoro-2-(1,1,2,2,3,3,3-heptafluoropropoxy)propionic acid
	ADONA	Decafluoro-3H-4,8-dioxanonoic acid (Decafluoro-3H-4,8-dioxanonoate)
	NFDHA	Perfluoro-3,6-dioxaheptanoic acid (Perfluoro-3,6-dioxaheptanoate)
	PFMBA	Perfluoro-4-methoxybutanoic acid (Perfluoro-4-methoxybutanoate)
	PFMPA	Perfluoro-3-methoxypropanoic acid (Perfluoro-3-methoxypropanoate)
Perfluoroalkylether Sulfonic Acids/Sulfonates (PFESAs)	9Cl-PF3ONS	9-chlorohexadecafluoro-3-oxanonane-1-sulfonic acid (9-chlorohexadecafluoro-3-oxanonane-1-sulfonate)
	11Cl-PF3OUdS	11-chloroeicosafluoro-3-oxaundecane-1-sulfonic acid (11-chloroeicosafluoro-3-oxaundecane-1-sulfonate)
	PFEESA	Perfluoro(2-ethoxyethane)sulfonic acid (Perfluoro(2-ethoxyethane)sulfonate)

Other Acronyms

Acronym	Definition
AFFF	Aqueous Film Forming Foam
AOF	Adsorbable Organic Fluorine
BACWA	Bay Area Clean Water Agencies
CEC	Contaminant of Emerging Concern
CIC	Combustion Ion Chromatography
dTOP	Direct Total Oxidizable Precursor Assay
DTSC	California Department of Toxic Substances Control
EOF	Extractable Organic Fluorine
ESL	Environmental Screening Level
¹⁹ F-NMR	Fluorine-19 Nuclear Magnetic Resonance
GC-MS	Gas Chromatography-Mass Spectrometry
LC-MS	Liquid Chromatography Mass Spectrometry
MDL	Method Detection Limit
NTA	Nontarget Analysis
PIGE	Particle-Induced Gamma-ray Emission spectroscopy
POTW	Publicly-Owned Treatment Works
QA/QC	Quality Assurance and Quality Control
TF	Total Fluorine
TOF	Total Organic Fluorine
TOP	Total Oxidizable Precursor Assay
USEPA	United States Environmental Protection Agency
WY	Water Year

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Executive Summary

Per- and polyfluoroalkyl substances (PFAS) are a broad class of thousands of fluorine-rich specialty chemicals. PFAS have been widely used in consumer products and industrial applications since the 1950s, due to their surfactant properties, thermal stability, and non-reactivity. PFAS are stable during use and resistant to degradation in the environment, earning them the moniker of “forever chemicals.” Some PFAS are highly persistent or, in the case of precursors, degrade to substances that are highly persistent. As a result, PFAS are widespread in the global environment, including surface water, groundwater, sediment, precipitation, wildlife, and humans.

Historically, the most widely produced PFAS were perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS). In the US, phaseouts of these long-chain PFAS began in the early 2000s. Following these ‘legacy’ PFAS phaseouts, production in North America and Europe shifted toward replacement PFAS, resulting in a wide range of alternatives including short-chain chemicals.

The Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) has monitored PFAS since 2004. Due to the RMP’s visionary monitoring program for emerging contaminants, the RMP has the most comprehensive long-term multi-matrix PFAS dataset in California. This monitoring to date has mainly focused on standardized methods that quantify a limited set (~40 analytes or less) of targeted long-chain PFAS, short-chain PFAS, and a limited number of PFAS precursors with available analytical standards.

Growing scientific understanding about their widespread environmental occurrence and concerns for impacts to ecosystems and human health have made PFAS a high priority for regulatory agencies. To inform management information needs, this report synthesizes monitoring data for San Francisco Bay from 2012–2025. The data inform understanding of the linkage between PFAS in industrial and consumer products and potential water quality impacts to the Bay.

Wastewater and urban stormwater runoff are both important pathways for PFAS transport to the Bay from upstream product use. The dominant PFAS detected in both wastewater and stormwater are PFHxA, PFPeA, PFOA, and PFOS. Median concentrations of each of the analytes are in the low nanograms/liter (4–16 ng/L), with the median sum of PFAS quantified in wastewater and stormwater both around 50 ng/L. Wastewater monitoring shows significant levels of unknown PFAS precursors in influent and effluent, which highlight that standardized methods may quantify only a small fraction of PFAS in environmental samples. Unlike wastewater effluent concentrations from various treatment plants, which were within an order of magnitude of each other, PFAS concentrations in urban stormwater runoff at different sites around the Bay varied by up to two orders of magnitude. PFAS is a focus contaminant class for RMP stormwater monitoring, which is quickly evolving as the RMP explores various approaches and methods for stormwater sample collection, analysis, and modeling.

PFAS concentrations in the Bay are similar to levels observed around the world and indicate potential risk to wildlife and humans that consume contaminated fish. PFOS is the dominant PFAS detected in Bay sediment (median 0.3 ng/g dw), prey fish (median 2 ng/g ww), sport fish (median 1 ng/g ww), bird eggs (median 70 ng/g ww), and harbor seal serum (median 52 ng/g ww). In contrast, Bay water PFAS concentrations are less dominated by PFOS and include more short-chain PFAS, such as PFHxA and PFPeA. The sum of PFAS concentrations are higher in all matrices in South and Lower South Bay compared to the rest of the Bay. This is partially due to limited hydraulic flushing relative to other San Francisco Bay regions, which increases the influence of PFAS discharges from wastewater and stormwater runoff.

Temporal trends in PFOS are emerging from the RMP's datasets in South Bay. Average concentrations of PFOS in sediment, bird eggs, and sport fish in South and Lower South Bay show statistically significant declines over the last ten years. Mean sediment concentrations of PFOS in South and Lower South Bay have declined from 1.5 ng/g dw in 2012 to 0.4 ng/g dw in 2023, representing a 73% decline. PFOS concentrations in bird eggs in Lower South Bay have decreased by 79% from peak median concentrations of 638 ng/g ww in 2016 to 135 ng/g in 2024. PFOS in shiner surfperch, an important sport fish species, declined by a more modest 15% in South Bay between 2014 (median 4.0 ng/g ww) and 2024 (3.4 ng/g ww). Trends for sediment, bird eggs, and sport fish are not as pronounced in other subembayments, which also tend to have lower concentrations compared to South Bay. Wastewater effluent results have shown a 63% decrease in PFOA between 2014 (median 16 ng PFOA/L) and 2020/2022 (6 ng PFOA/L). These decreases are consistent with observed trends in studies from other regions, and may be attributed to phaseout of long-chain PFAS in primary sources. However, these trends are not yet clear in other subembayments, and for other short-chain and long-chain PFAS besides PFOS.

PFAS are classified as a High Concern in the Bay within the RMP's tiered risk-based framework for emerging contaminants. This evaluation is driven primarily by long-chain PFAS (PFOS, PFOA, PFNA) in sport fish. California has not yet established human health or regulatory thresholds for fish in California, although the Office of Environmental Health Hazard Assessment (OEHHA) is developing a PFOS fish consumption advisory. In the absence of a California-specific value, the RMP uses the most protective human health value from another state. The state of Massachusetts has developed the most stringent fish consumption advisories in the US. For Bay-wide samples in 2024, 87% of Bay fish samples exceeded the one meal per day general population threshold for PFOS (≤ 0.50 ng/g), and 9% of samples exceeded the one meal per week general population threshold (≤ 3.52 ng/g). Levels in PFOS also indicate potential impacts to wildlife that eat Bay fish based on a Canadian Federal Environmental Quality Guideline for PFOS.

There is evidence that many individual PFAS have additive toxicity in wildlife and humans, and assessing compound by compound may underestimate cumulative PFAS risks. To address this concern, the European Union has an environmental quality standard for the weighted sum of 25 PFAS. San Francisco Bay water PFAS

concentrations exceed this threshold, providing another line of evidence that these 25 PFAS may pose risks to humans and wildlife that consume Bay fish.

The PFAS monitoring strategy for the next five years will evolve continuously based on ongoing RMP study findings, rapidly developing science, and urgent information needs to manage sources, pathways, and potential risks from this complex and broad contaminant class. The strategy consists of: 1) expanding analytical tools to improve monitoring of PFAS as a class, 2) evaluating temporal trends in legacy and contemporary PFAS subclasses, 3) investigating and evaluating major sources, pathways, and loadings, and 4) developing conceptual and quantitative models to synthesize monitoring datasets.

The monitoring strategy continues application of EPA Method 1633 across monitored matrices to build upon past datasets and track temporal and geographic trends and inform risks from PFOS and other PFAS. At the same time, the RMP will explore and pilot new methods that target additional PFAS subgroups, as well as broader methods to more comprehensively characterize PFAS as a class. These newer methods will improve PFAS measurements in the Bay.

To improve evaluation of trends, we recommend increasing the frequency of sport fish monitoring in the Bay from the current five year monitoring cycle. We also recommend a prey fish study to assess temporal trends combined with an updated monitoring design for S&T sport fish and prey fish. Monitoring of sport and prey fish may be coordinated with water and sediment monitoring to inform our understanding of exposure pathways that lead to bioaccumulation of PFOS and other PFAS in fish.

The monitoring strategy also emphasizes investigating various pathways for PFAS entering the Bay. Additional ongoing and planned efforts to monitor and evaluate wastewater and stormwater runoff are integrated within the ongoing USEPA-funded PFAS Sources to Solutions project. After completion of this project in 2029, we will update the monitoring strategy for these major pathways.

Finally, we will develop conceptual and quantitative models to synthesize available datasets. Several models are outlined for near future development, including a Bay mass-balance model, a conceptual model for bioaccumulation, and a stormwater watershed model to estimate loadings. These models will provide an important framework for informing the management levers to protect water quality in the Bay.

1 Introduction

The Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) published its first per- and polyfluoroalkyl substances (PFAS) Synthesis and Strategy in 2018 (Sedlak et al., 2018). This report represents a significant update. The purpose of this document is threefold.

- First, this report provides a comprehensive updated synthesis of the most relevant monitoring data for PFAS in San Francisco Bay (“the Bay”) collected by the RMP and other scientists between 2015–2025, focusing on the most relevant recent datasets and incorporating older datasets for trends evaluations.
- Second, this report provides an updated risk screening evaluation based on the synthesized data.
- Third, this report identifies the priority information needed to support monitoring and management, and outlines an updated strategy for RMP monitoring of PFAS to inform those priority information needs.

1.1 Introduction to PFAS

Definition, Subgroups, and Chemical Properties

PFAS are a broad class of thousands of fluorine-rich specialty chemicals. They have been widely used in consumer products and industrial applications since the 1950s due to their surfactant properties, thermal stability, and non-reactivity. The synthetic carbon-fluorine bonds unique to PFAS make these compounds stable during use and resistant to degradation in the environment, earning them the moniker of “forever chemicals.” Some PFAS are highly persistent or, in the case of precursors, degrade to substances that are highly persistent. As a result, PFAS are also widespread in the global environment, including surface waters, groundwater, sediments, precipitation, and wildlife.

PFAS have been linked to a variety of adverse health impacts in wildlife and humans. These include but are not limited to metabolic dysfunction, immune deficiencies, and cancers (ATSDR, 2021). In response to growing evidence of their health impacts, ‘legacy’ PFAS such as PFOA and PFOS ([see PFAS Analyte Acronyms](#)) were phased out of production in the US beginning in the early- to mid-2000s. A wide diversity of ‘alternative’ or novel PFAS have since been produced as replacement compounds. Many data gaps remain on PFAS sources and environmental fate, as thousands of individual PFAS are registered for use, but fewer than 50 have been the subject of significant monitoring in environmental media (Evich et al., 2022; Y. Wang et al., 2019).

PFAS Definitions

Definitions of PFAS as a chemical class vary. The Organisation for Economic Cooperation and Development definition includes any chemical with at least a perfluorinated methyl group (–CF₃) or a perfluorinated methylene group (–CF₂–) (OECD, 2021), whereas the United States Environmental Protection Agency (USEPA)

[definition](#)¹ includes only compounds with at least two adjacent carbon atoms where one carbon is fully fluorinated and the other is at least partially fluorinated. The broadest existing definition, which laws in California and other states use, includes all organic chemicals containing “at least one fully fluorinated carbon atom.” This definition includes fluorinated pharmaceuticals and pesticides, in addition to compounds that contain chains of fully or highly fluorinated carbons.

PFAS include both perfluoroalkyl substances, which are fully fluorinated (i.e., only fluorine atoms bonded to the carbon alkyl backbone), and polyfluoroalkyl substances, (i.e., carbon backbone is not fully fluorinated and includes at least one carbon-hydrogen bond or one unsaturated carbon bond). Historically, the most-produced and best studied class of PFAS were the perfluoroalkyl acids (PFAAs), which are “terminal acids” because many precursors transform into PFAAs and they do not break down under environmental conditions. PFAAs are composed of a fully fluorinated carbon chain ‘tail’ and functional group ‘head’.

The two major subclasses of PFAAs, perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonic acids (PFSAs), can be divided into long-chain and short-chain groups based on the number of fluorinated carbons in their ‘tail’ group to differentiate environmental fate properties. Long-chain compounds are PFCAs with eight or more carbons (seven or more fully fluorinated carbons), like PFOA, and PFSAs with six or more carbons, such as PFOS. Short-chain compounds refer to PFCAs with four to seven carbons, such as PFBA, and PFSAs with four to five carbons, such as PFBS. PFCAs and PFSAs with fewer than three carbons such as TFA and TFMS are considered ultra-short chain compounds. Long-chain PFCAs and PFSAs are the most widely monitored PFAS in the environment.

PFAS that can degrade or transform into terminal PFAAs are referred to as PFAA “precursors.” PFAA precursors include linear fluorotelomer-based substances (e.g., FTOHs, FTSS, FTCAs) and precursors formed through electrochemical fluorination that can have linear or branched carbon chains (e.g., FOSAAs, FASEs). In recent decades, a growing number of novel PFAS chemistries have also been developed, some of which may also be PFAA precursors, such as perfluoroalkyl phosphate esters. Only a handful of known precursors are included within standard analytical methods, with a large presence of unknown precursors (both legacy and novel) anticipated in environmental matrices.

PFAS can also be categorized into nonpolymeric and polymeric compounds. The vast majority of environmentally monitored PFAS are non-polymeric PFAS, which include PFAAs. Polymeric PFAS include a repeating sequence of one or more types of monomer unit, and can be further categorized into fluoropolymers, polymeric perfluoropolyethers, and side-chain fluoropolymers (Buck et al., 2011). Because of their larger size and molecular weight compared to nonpolymeric PFAS, polymeric PFAS are

¹PFAS that contain one of three chemical structures: 1) $R-(CF_2)_n-CF(R')R''$, where both the CF_2 and CF moieties are saturated carbons, and none of the R groups can be hydrogen; 2) $R-CF_2OCF_2-R'$, where both the CF_2 moieties are saturated carbons, and none of the R groups can be hydrogen; 3) $CF_3C(CF_3)RR'$, where all the carbons are saturated, and none of the R groups can be hydrogen.

thought to be less mobile, bioavailable, and toxic. However, nonpolymeric PFAS are often involved in and released during the production of polymeric PFAS (Lohmann & Letcher, 2023). Side chain fluoropolymers, which consist of a non-fluorinated polymer backbone with fluorinated side-chains, are of particular concern because they can degrade into non-polymeric PFAS such as PFAAs or other PFAA precursors.

PFAS Use and Production Shifts

Until recent decades, the most widely and abundantly produced PFAS were PFOS and PFOA, which were manufactured as various salts or produced indirectly from the degradation of precursors and production feedstocks (Z. Wang et al., 2014, 2017). In the US, industry-led voluntary production phaseouts of these compounds and some precursors, as well as PFHxS, other long-chain PFAAs, and a few related precursors, began in the early 2000s. These restrictions were formalized through government-led stewardship programs by 2015. This federal action was part of a broader international collaboration across North America and Europe to reduce human and ecological risks associated with exposure to these compounds, including the listing of these PFAS in the Stockholm Convention (2009–2025). However, PFAA production (predominantly from fluoropolymer production; Prevedouros et al., 2006) and use in products continues in some countries (predominantly China; Simon et al., 2026), and some of these products are imported into the US.

Following these ‘legacy’ PFAS phaseouts, production in North America and Europe shifted toward replacement PFAS. A wide range of alternative chemistries also emerged, such as cyclic PFAS and halogen-substituted PFAS. In some cases, novel precursors and polyfluorinated PFAS were introduced with the intention of increasing environmental degradability. Many were later found to be similarly persistent or to transform to other highly persistent terminal PFAS. Production patterns also shifted towards short-chain PFAS and their precursors, which are generally less bioaccumulative than their long-chain counterparts, although some (e.g., short-chain FASA) have been found to be as bioaccumulative and toxic as the legacy PFAS they were designed to replace (Britton et al., 2024; Pickard et al., 2024). However, short-chain PFAS can still reach observable levels in tissues at high exposure concentrations, and recent studies suggest that many of these alternative PFAS can and do still bioaccumulate in wildlife and humans. Many of these replacement PFAS also remain highly persistent and have little to no available hazard information for human or environmental risk.

Physical-chemical Properties and Environmental Behavior

Non-polymeric PFAS are characterized by their extraordinary chemical and thermal stability. The most widely produced PFAS subclass, PFAAs, historically had surfactant-like properties that made them both bioaccumulative and water soluble. Long-chain PFAS are generally more bioaccumulative and more likely to partition to organic matter and sediments than short-chain PFAS (Ehsan, 2026; Khan et al., 2023; Z. Wang et al., 2015).

In addition to the length of the fluorinated carbon-chain tail, functional group substitutions also play an important role in driving environmental PFAS behavior. For example, PFASs are generally more bioaccumulative than PFCAs of the same fluorinated carbon chain length (Conder et al., 2008; Fischer et al., 2024). Some PFAS precursors, such as FASAs, contain more neutrally charged head groups, which increases their hydrophobicity and bioaccumulative potential compared to ionic PFAS of the same fluorinated carbon chain length (Pickard et al., 2024). Neutrally charged PFAS are also more volatile and can undergo rapid long-range atmospheric transport. More functional group substitutions also increase susceptibility to transformation, though often to other persistent terminal PFAS. For example, neutrally charged precursors such as FASAs, FASEs, and FTOHs can volatilize and transport globally before being abiotically transformed to intermediate precursors or terminal acids, which are then deposited from the atmosphere. Many of these precursors can also be biotransformed, forming stable intermediates (e.g., FASAs, FTCAs) or terminal acids (e.g., PFASs, PFCAs).

Many PFAS also exhibit unique bioaccumulation behaviors. In contrast to neutrally charged hydrophobic persistent organic pollutants, which often concentrate in the fatty tissues of animals, PFAS more commonly bind to proteins and phospholipids and tend to accumulate in blood, liver, and eggs (Fair & Houde, 2018; Houde et al., 2011; J. M. Sun et al., 2022). In aquatic ecosystems, exposure to ionizable PFAS (e.g., PFAAs) can often occur predominantly through aqueous exposure due to their mobility in water, although sediment exposure in benthic-linked ecosystems can become increasingly important with increasing PFAS hydrophobicity (e.g., long chain lengths, neutral functional groups). Some PFAS, particularly long-chain compounds, can also exhibit trophic magnification. Apex predators such as polar bears, killer whales, sea otters, and seals have shown some of the highest reported concentrations in environmental tissue samples (Fair & Houde, 2018; Houde et al., 2011; Khan et al., 2023).

Short-chain PFAS are generally much less bioaccumulative in animals and do not biomagnify, although they can still be observed in animals at high exposure concentrations. They are, however, known to be enriched in plants relative to long-chain PFAS, due to their high mobility in soils and water. While most alternative or novel PFAS have received less study than PFOS and PFOA, many are still highly persistent. Given the similarity in structure and studies conducted to date, many of these compounds will exhibit similar physical and chemical characteristics, environmental persistence, and toxicological effects (Brendel et al., 2018; Y. Wang et al., 2019).

Concerns relating to the persistence of PFAS, the high toxicity of well-studied members of this class, and the general pattern of regrettable substitution observed in industry have led scientific and regulatory bodies to recommend broad, class-based monitoring and management approaches. The RMP has followed that recommendation. Initially, long-chain PFAS like PFOS and PFOA were classified as Moderate Concern within the RMP's tiered risk-based framework for contaminants of emerging concern (CECs), while all other PFAS were classified as Possible Concern (Sedlak et al., 2018). The 2020 RMP Emerging Contaminants Workgroup meeting reclassified all PFAS as Moderate Concern, consistent with the rapidly evolving scientific and regulatory response to PFAS as a broad class of priority compounds for management actions (Miller et al., 2020).

Due to their Moderate Concern classification, the 2022 RMP routine monitoring (Status and Trends) redesign added PFAS monitoring to every matrix; previously, PFAS were only monitored in bird eggs and sport fish as part of Status and Trends. PFAS were subsequently classified as High Concern in the revised RMP CEC Strategy based on an updated framework and risk evaluation (Sutton, Miller, et al., 2024).

1.2 Management Questions and Drivers

The guiding management questions for the RMP's Emerging Contaminants Workgroup, in which PFAS are a priority focus area, are:

1. Which CECs have the potential to adversely impact beneficial uses in the Bay?
2. What are the sources, pathways, loadings, and processes leading to the presence of individual CECs or groups of CECs in the Bay?
3. What are the physical, chemical, and biological processes that may affect the transport and fate of individual CECs or groups of CECs in the Bay?
4. Have levels of individual CECs or groups of CECs changed over time in the Bay or pathways? What are potential drivers contributing to change?
5. Are the concentrations of individual CECs or groups of CECs predicted to increase or decrease in the future?
6. What are the effects of management actions?

RMP monitoring of PFAS in the Bay and Bay pathways can help link PFAS use in products to transport and potential impacts in the Bay to inform management decisions. Management of PFAS has evolved as a high priority for regulatory agencies to reflect growing scientific understanding about environmental occurrence and impacts to ecosystems and human health.

In California, the Department of Toxic Substances Control (DTSC) Safer Consumer Products Program has established a clear rationale for management actions directed at the entire PFAS class (Balan et al., 2021), and has begun to apply this approach, starting with carpets and rugs made or sold in California. Ongoing work includes PFAS in converted textiles, leathers, artificial turf, and floor maintenance products. Their 2024–2026 workplan also describes potential evaluations of PFAS in paints, cleaning products, motor vehicle parts, food contact materials, electronics, and pet care products. The RMP seeks to implement monitoring that directly informs and supports the DTSC Safer Consumer Products Program.

Other recent state regulations on PFAS in consumer products include PFAS bans in Aqueous Film Forming Foam (AFFF) (SB 1044), food packaging (AB 1200), products intended for infants and children (AB 652), apparel and specified household textiles (AB1817), and cosmetics, shampoos, and lotions (AB 2771). All of these laws take a class-based approach to banning PFAS in response to human exposure and associated health concerns. Twenty-nine other states have also adopted policies to restrict PFAS in a variety of consumer and industrial uses (*Safer States: Bill Tracker for PFAS*, 2026).

Since 2019, the State Water Resources Control Board (SWRCB) and the nine Regional Water Boards throughout California have been implementing PFAS investigations under

the authority of California statutes, with a priority on protecting drinking water sources and identifying and cleaning up pollutant sources affecting public supply wells. Together they lead various investigation orders to determine whether there have been significant releases of PFAS at sites in each region likely to have discharged PFAS, including airports, chrome platers, petroleum refineries, bulk fuel terminals, landfills, and wastewater treatment plants. These efforts typically require investigation of PFAS in groundwater, soil, and stormwater runoff samples, with a focus on standardized PFAS methods such as EPA 1633 and are part of a statewide PFAS action plan that also includes sampling specific PFAS in drinking water and wastewater. Moving forward, the Water Board's strategy includes industrial sampling outwards from the identified impacted public water supply wells, investigations at potential source sites, and data analysis and visualization to inform the public and decision makers. The Water Board's PFAS Team is a group of technical and support staff within the State Water Board Divisions of Drinking Water and Water Quality, and the Regional Water Quality Control Boards. This team is working on developing a statewide strategy to understand PFAS sources and evaluate statewide management strategies to protect water quality. The San Francisco Bay Regional Water Quality Control Board Groundwater General Permit, effective as of July 2026, includes minimum monitoring requirements for PFAS and PFAS effluent limitations for certain types of discharges.

Bay Area Clean Water Agencies (BACWA), in coordination with the SWRCB and the San Francisco Bay Regional Water Quality Control Board, took a different approach to monitoring wastewater from the SWRCB's 13267 and 133383 Investigation Orders (Water Code Sections 13267 and 13383 Order for the Determination of the Presence of Per- and Polyfluoroalkyl Substances at Publicly Owned Treatment Works, 2020). BACWA partnered with SFEI to conduct monitoring of wastewater influent, effluent, and biosolids at a subset of publicly-owned treatment works (POTWs) in the region that was representative of the diverse POTWs in the region; Phase 2 of the study investigated PFAS sewershed sources (Lin et al., 2023, 2024). Currently, BACWA has also funded a Phase 3 study to continue improving understanding of PFAS in wastewater to inform management actions. PFAS monitoring provisions are currently omitted from current Bay POTW NPDES permits based on the understanding that the utilities are voluntarily funding and conducting wastewater monitoring to support management decisions.

Municipalities, as part of their Municipal Regional Stormwater permit issued by the San Francisco Bay Regional Water Quality Control Board, are evaluating PFAS removal from stormwater runoff from low impact development facilities. Urban stormwater influent and effluent flowing into eight these bioretention facilities in the region are being monitored for PFAS.

Bay monitoring can inform future Clean Water Act §305(b) Integrated Report evaluation of San Francisco Bay by the SF Bay Regional Water Quality Control Board and the SWRCB. Evaluations would be based on target analytes with regulatory thresholds. No thresholds are currently in place in California, although the California Office of Environmental Health Hazard Assessment is in the process of developing an advisory fish tissue threshold for PFOS, which could provide a basis for a §305(b) assessment of Bay impairment in the future. If the Bay is determined to be impaired by PFAS, a Total

Maximum Daily Load cleanup plan may be needed, which would require information on loads, trends in concentrations in the Bay, and projected rates of recovery.

The USEPA has developed and implemented several federal regulatory actions to tackle broader issues related to PFAS with some adjustments in recent years. The agency's PFAS Strategic Roadmap, released in 2021, describes an approach centered on five principles: 1) consider the lifecycle of PFAS; 2) get upstream of the problem; 3) hold polluters accountable; 4) ensure science-based decision-making; and 5) prioritize protection of disadvantaged communities. The USEPA established the first federal drinking water standards for PFAS in 2024, including Maximum Contaminant Levels for PFOS and PFOA. On December 18, 2024, the agency published studies of PFAS industrial sources and discharges as part of its Preliminary Effluent Guidelines Program Plan 16. This included the POTW Influent Study, an effort to collect and analyze nationwide data on industrial discharges of PFAS to POTWs and PFAS in POTW influent and effluent. The report also provides updates on the agency's Effluent Limitations Guidelines rulemaking efforts regarding PFAS discharges from PFAS manufacturers, metal finishers and electroplaters, and landfills. In addition, the agency has finalized a new rule to designate PFOS and PFOA as hazardous substances under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and designate sites contaminated with PFAS as Superfund sites. Several PFAS are now listed in the Toxics Release Inventory, which requires manufacturing facilities to report environmental discharges annually, improving information on releases across the US. The agency is also promoting development of improved and standardized methods to analyze PFAS across environmental matrices.

To inform source control, in 2023 the USEPA established a final rule under the Toxic Substances Control Act (TSCA) to require all manufacturers and importers of PFAS and PFAS-containing articles in any year since 2011 to report information on PFAS uses, production volumes, disposal, exposures, and hazards. Subsequent TSCA reporting rule updates extended the reporting start date to 2026 and proposed some exemptions to categories deemed unnecessary. Required reporting will result in the largest-ever dataset of US manufacturing and use of PFAS. Meanwhile, in January 2024, the EPA finalized a Significant New Use Rule that prevents companies from starting or resuming the manufacture or processing of 329 PFAS that have not been made or used for many years without a complete EPA review and risk determination. Without this rule, companies could have resumed use of these PFAS at any time.

1.3 Analytical Methods

Since monitoring for PFAS in the Bay began in the early 2000s, analytical methods have continued to evolve. The RMP now uses established analytical methods for monitoring PFAS in the Bay's matrices. The RMP is also piloting newer methods that have more recently become available. The variety of methods used are important to contextualize results over time and understand the key data gaps across matrices.

RMP studies have focused on **targeted methods**, which quantify levels of specific PFAS in various matrices with a high degree of precision and sensitivity. Targeted methods typically incorporate a chromatographic instrument paired with mass

spectrometry to separate, identify, and quantify compounds within a sample. Common instruments used include variations of liquid chromatography coupled with mass spectrometry (LC-MS) and gas chromatography coupled with mass spectrometry (GC-MS). The PFAS analytes included in and detectable by these methods depend on extraction methods and availability of reference standards.

RMP monitoring to date has focused on PFAAs, with consistent monitoring of nine PFCAs (C4-C12), three PFSA (C4, C6, and C8), and one PFOS precursor (PFOSA) since the first study in 2002. The list of targeted analytes has continued to grow, and the most recent studies include 40 analytes (EPA Method 1633), incorporating various PFAA precursors and PFOS replacement compounds (such as ADONA and GenX). Detection limits have also improved our ability to detect and quantify targeted analytes.

Recent developments in targeted methods mean a variety of novel analytical methods are becoming available that expand the list of PFAS analytes and subgroups that can be investigated (up to ~100 analytes available). The RMP is currently piloting a few newer targeted methods with extended PFAS analyte lists. However, considering the tens of thousands of PFAS potentially used in commerce and their transformation products, these expanded targeted methods may still underestimate the presence of PFAS in the Bay and its pathways. Various bulk organofluorine methods are being used and developed to quantify PFAS presence more broadly. The RMP is piloting the use of these bulk organofluorine methods to understand their applicability and limitations.

The **total oxidizable precursors (TOP) assay** transforms precursors to terminal PFCAs, which are then measured using a targeted PFAS method. The evaluation of pre- and post-oxidation data offers a clearer view of overall levels of PFAS present, though it does not identify which specific precursors are being converted. The RMP has applied the TOP assay to various matrices since 2014, with particular success in quantifying the significant presence of PFAS precursors in wastewater (Houtz et al., 2016; Lin et al., 2024). The RMP has also applied the TOP assay to stormwater, Bay surface water, and sediment samples, revealing the presence of PFAS precursors in these matrices (Méndez, Miller, et al., 2025; Sutton, Avellaneda, et al., 2024).

The TOP assay is not a standardized method, and RMP pilot studies have highlighted some challenges and limitations with this method. Importantly, the TOP assay does not differentiate between PFSA and PFCA precursors, as all PFAA precursors are oxidized to PFCAs under TOP assay conditions. The transformation pathways achieved during the TOP analysis may not fully reflect pathways that occur under environmental conditions. For example, FASA and FASE precursors predominantly transform to PFSAs under environmental conditions but can also form PFCAs as minor products. Under environmental conditions, precursors of a given chain length predominantly degrade to PFCAs of the same chain length, though transformation to shorter-chained products may also occur. However, all PFSA precursors degrade only to PFCAs in the TOP assay. Additionally, the TOP assay may underestimate the occurrence of precursors due to limitations in achieving complete oxidation in complex samples, exclusion of relevant transformation products in targeted analysis (e.g., ultra-short chain compounds), and sample extraction methods that may not capture all relevant PFAS in a matrix.

Variations of the TOP assay have emerged to better determine the concentrations of oxidizable precursors, including ultra-short chain TOP (which also analyzes the shortest PFAAs) and directTOP (directly oxidizing and analyzing a sample without any extractions). Both of these methods are currently in use or anticipated to be used in future RMP studies.

Total fluorine content methods measure the bulk fluorine content in samples, such as via combustion ion chromatography (CIC). The CIC analysis does not differentiate between inorganic and organic fluorine, and relies on the solution rinse and extraction method to differentiate between the inorganic or organic fluorine fraction. Extractable organic fluorine and adsorbable organic fluorine are two different approaches to separating the organic fluorine in the environmental sample prior to analysis with CIC. Detection limits tend to be significantly higher compared to LC-MS analysis. So far, the RMP has only piloted total fluorine content methods in wastewater and stormwater samples. The results obtained have been difficult to interpret due to concentrations that are orders of magnitude higher than those from targeted analyses and TOP assays, and the potential for interference from inorganic fluorine.

A newer bulk organofluorine method for PFAS analysis is **¹⁹F-nuclear magnetic resonance spectroscopy (¹⁹F-NMR)**, which directly observes fluorine atoms within molecules to provide detailed information about fluorinated functional groups without requiring compound-specific standards. ¹⁹F-NMR is a structural and quantitative technique used to characterize complex mixtures, confirm structures of newly discovered PFAS, and estimate bulk organofluorine content in extracts. This method provides direct structural insights, can detect fluorinated compounds that may not ionize well in mass spectrometry (such as fluoropolymers), and is non-destructive (compared to the total fluorine methods). However, there are limitations including high detection limits requiring concentrated samples, the high cost of the instrument, and the high level of expertise required; improvements to the method are in development (Gauthier & Mabury, 2022) and currently generally applied in research studies. The RMP is currently piloting this method in collaboration with University of Minnesota researchers using the most concentrated samples within biota, wastewater, and stormwater.

Unlike the targeted methods described above, **nontarget analysis (NTA)** detects known, unknown, and emerging fluorinated compounds without relying on a predefined target list. Instead of measuring only PFAS with available standards, NTA uses high-resolution mass spectrometry to capture full spectrum chemical information. This method collects full-scan data and fragmentation spectra, then applies computational workflows to detect features, assign molecular formulas, and compare results against suspect lists or PFAS databases. This exploratory method aims to discover new or unexpected PFAS, including transformation products. Analysis and interpretation requires a high level of expertise, and there are limitations to the method. Results are reported qualitatively or semi-quantitatively, with confidence levels specified for each identification to indicate the level of uncertainty and lack of confirmation. NTA serves as a useful method to complement other methods to build a more comprehensive characterization of PFAS present. The RMP is currently piloting this method in marine mammals and sport fish.

2 Synthesis of PFAS in the Bay

This section provides a high-level summary and synthesis of RMP monitoring data to date in the Bay. More detailed discussion of each matrix is provided in Sections 3–5.

2.1 Conceptual Model and Summary of RMP Monitoring

This section summarizes RMP monitoring data within a conceptual model framework (Figure 2.1). The purpose is to synthesize our current understanding of PFAS in the Bay relevant to Emerging Contaminants Workgroup management questions, and to identify priority data gaps that will guide our PFAS monitoring strategy. This conceptual model is most relevant for long-chain PFAAs (e.g., PFOS and PFOA), which have been the most consistently monitored and detected PFAS in the Bay. PFAS detections in the Bay may be from direct discharge or transformation from PFAS precursors to intermediate and terminal PFAS (i.e., PFAAs). Our conceptual understanding of PFAS transport from sources through pathways to the Bay, and their transport, transformation, and bioaccumulation within the Bay, is still incomplete and developing.

PFAS Sources and Pathways

PFAS can reach the Bay through municipal wastewater effluent discharge, urban stormwater runoff, wet and dry atmospheric deposition, riverine discharges, industrial wastewater, agricultural runoff, and contaminated sites and landfills that may contribute PFAS via contaminated groundwater, stormwater runoff, or wastewater (Figure 2.1).

In the highly urbanized San Francisco Bay region, which has relatively little history of PFAS-related manufacturing and limited PFAS contamination in drinking water, municipal wastewater and urban stormwater runoff are likely the primary pathways for PFAS to reach the Bay other than from AFFF-contaminated sites. PFAS have been and still are used in a diverse array of indoor and outdoor products, and PFAS from these products can ultimately be transported to the Bay through municipal wastewater and urban stormwater discharges. These products include additional PFAS subclasses not used in AFFF or captured by traditional targeted environmental monitoring methods. While typical analytes have been observed in many of these consumer products, much of the current use of PFAS in consumer products is compounds that can transform after their release, making source-tracing challenging.

Municipal wastewater effluent is considered a significant pathway for PFAS to reach aquatic environments, as conventional treatment technologies are generally ineffective in removing PFAS from wastewater (Liu et al., 2026). The sources of PFAS to municipal wastewater remain relatively poorly characterized. BACWA has taken leadership in coordinating wastewater monitoring in the region, and funded a study that identified domestic/residential discharges as major contributors of PFAS in municipal wastewater, suggesting consumer products and the resulting human exposures are likely an important source of PFAS entering municipal wastewater (Lin et al., 2024). Various indoor products known to contain PFAS include but are not limited to paints, sealants, varnishes, waxes, and adhesives; textiles and textile treatments; electronics and home appliances; cleaning products such as for laundry and dishwashing; food-contact

materials; and personal care and hygiene products. Commercial and industrial uses of PFAS may also be discharged to municipal wastewater; landfill leachate may also be a pathway to municipal wastewater.

RMP monitoring has shown that median urban stormwater runoff PFAS concentrations are comparable to those from municipal wastewater effluent (Peter et al., 2024). Because the region's annual stormwater runoff volumes exceed annual total wastewater discharges, this suggests that urban runoff contributes significantly to the quantity of PFAS in the Bay ecosystem. The sources of PFAS to stormwater runoff remain poorly characterized. Stormwater monitoring data, some of which involves sampling upstream of (excluding) areas potentially affected by historic contamination (e.g., use of AFFF) suggest PFAS-containing products as likely sources for detected PFAS. PFAS uses in outdoor products include but are not limited to paints and coatings, construction and building materials, outdoor textiles, vehicle products, and synthetic turf.

Historical use of PFAS in AFFF is widely understood to be a major source of legacy PFOA and PFOS site contamination. Many of the identified PFAS contaminated sites around the Bay are airports, military bases, bulk fuel terminals and storage facilities, and refineries where AFFF was frequently used for fire-suppression related activities. While PFOS AFFF was phased out in the early 2000s, facility stockpiles were not removed and could still be used. Later fluorotelomer manufacturing processes focused on short-chain PFAS and/or PFAS precursors, such as 6:2 FTS. The 2022 National Defense Authorization Act initially set a deadline to discontinue PFAS-containing AFFF by 2024; the deadline has been extended to 2026. Site contamination may be transported to the Bay through stormwater runoff, wastewater, interception of contaminated groundwater, and atmospheric deposition.

From 2019 to 2021, the SWRCB issued PFAS orders to airports, chrome plating facilities, landfills, refineries, and bulk fuel terminals throughout the Bay Area, requiring property owners to conduct PFAS sampling of soil, groundwater, stormwater runoff, and wastewater (if relevant) at their sites. PFAS have been used in chrome plating operations. Historically, PFOS and later 6:2 FTS were used as the active ingredient in the plating baths as a mist-suppressing agent to reduce exposure to toxic hexavalent chromium aerosols. Use of PFOS-based fume suppressants was banned in California in 2016. PFAS have also been widely documented in landfill leachate (Lang et al., 2017). The San Francisco Bay Regional Water Quality Control Board is managing the sites and requiring further investigation where appropriate, based on initial PFAS data. All of the PFAS site locations and data are available to the public through the SWRCB's interactive GeoTracker PFAS database and map (SWRCB, 2026). Some of these AFFF-contaminated sites have some of the highest levels of PFOS measured at industrial sites in the region, with PFOS levels in groundwater and stormwater ranging from below detection limits to over 500,000 nanograms/liter (ng/L).

An important challenge to inform effective source control and other management actions of ongoing sources of PFAS is to identify the major sources and pathways of PFAS to the Bay. The current USEPA-funded PFAS Sources to Solutions project will further develop detailed conceptual models describing consumer product sources and pathways to wastewater and stormwater (SFEI, 2026).

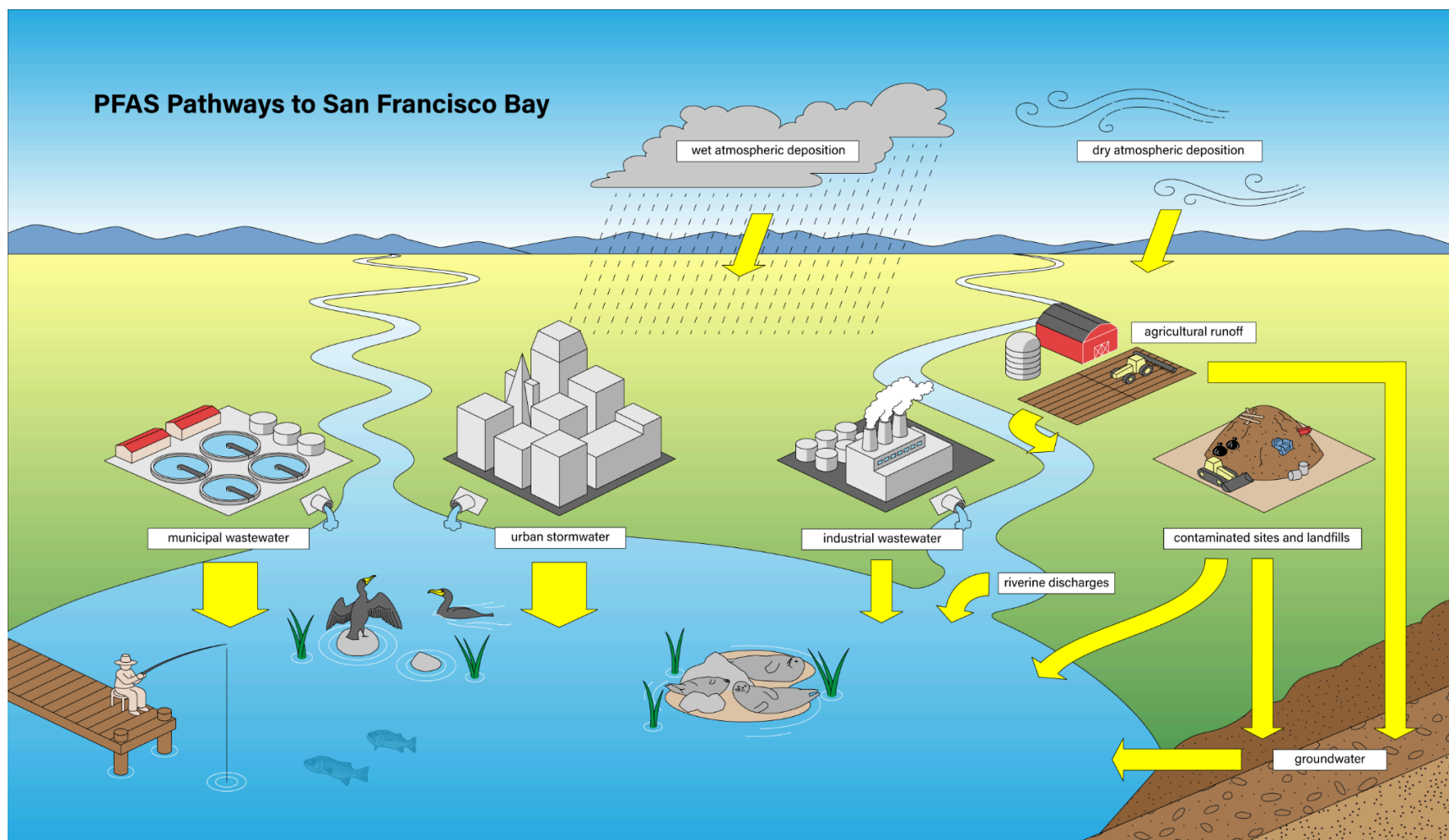


Figure 2.1. Conceptual Model of PFAS Pathways in San Francisco Bay.

Municipal wastewater effluent and urban stormwater runoff are major pathways that transport PFAS to the Bay. Additional transport pathways include wet and dry atmospheric deposition, riverine discharges, industrial wastewater, agricultural runoff, and contaminated sites and landfills that may contribute PFAS to the Bay via contaminated groundwater, stormwater runoff, or wastewater.

Urban Stormwater Runoff and Wastewater Transport Pathways

The RMP monitors wastewater and urban stormwater runoff as significant pathways for PFAS entering the Bay.

The most recent monitoring of PFAS from a representative, diverse range of publicly-owned treatment works (POTWs, i.e., municipal wastewater treatment plants) in the Bay region in 2020 and 2022 showed ng/L levels of PFOS, PFOA, and other PFAS in wastewater effluent. The PFAS detected consistently across all samples included two short-chain PFAS (PFHxA and PFPeA) and PFOA. The concentrations of PFHxA were highest, followed by PFPeA, PFOA, and PFOS, which was detected in over 95% of samples (median concentrations and range shown in Table 2.1). Bay Area effluent concentrations were in the lower range of those reported in California and other studies in the US (based on Geotracker summary data provided by SWRCB on April 3, 2024 via email).

Table 2.1. Comparative concentrations of PFAS in aqueous matrices. Median concentrations and range (minimum–maximum) of key PFAS including sum of short-chain PFCAs and sum of all PFAS (sumPFAS) detected in municipal wastewater (2020, 2022), stormwater runoff (2018–2025), and Bay Water (open water and near field during the wet and dry seasons from 2021–2024). Less than method detection limit (<MDL) is shown if the PFAS was not detected. The sum of short-chain PFCAs and sum of PFAS (quantified PFAS using EPA method 1633) are calculated with non-detects treated as zeros. PFPeA, PFHxA, PFOA, and PFOS had detection frequencies above 80% in effluent and stormwater runoff, and above 60% in Bay water.

PFAS Analytes	Effluent (ng/L)	Stormwater* (ng/L)	Bay Water (ng/L)
PFPeA	7.7 (2.7–30)	5.7 (<0.4–41)	2.0 (<0.8–28)
PFHxA	16 (<0.3–30)	7.4 (<0.4–180)	2.5 (<0.4–25)
PFOA	6.0 (3.5–12)	6.9 (<0.4–110)	1.8 (<0.4–20)
PFOS	4.5 (<0.3–13)	6.6 (<0.4–81)	2.6 (<0.4–20)
sum short-chain PFCAs	31 (14–69)	21 (<MDL–304)	6.8 (<MDL–91)
sumPFAS	57 (26–112)	46 (4.1–574)	15 (<MDL–179)

*Stormwater runoff data includes samples from urban and less-urban watersheds.

Levels of PFAS precursors that can transform to PFAAs were significant in influent, effluent, and biosolids, based on PFAS levels measured using the TOP assay. Target to TOP analysis revealed distinct increases in short-chain PFCAs depending on the matrix, indicating elevated levels of precursors for specific chain lengths entering wastewater streams. In influent samples, PFPeA (C5) exhibited the most substantial rise, averaging a 955% increase, while PFOA and other short-chain PFCAs (C8 or less) also recorded notable gains (averaging <400%). Effluent samples will have a significant portion of PFAS precursors converted to terminal PFAS through the treatment process. Effluent samples showed the largest increase in PFBA (C4) with a 289% average rise, alongside notable increases in PFPeA (C5) and PFHpA (C7) (both averaging ~150%).

Effluent from POTWs has shown declining levels of PFOA and PFNA by well over 50% between 2009 and 2022 (Lin et al., 2023, 2024). The decrease in PFOA and PFNA is statistically significant (Kruskall-Wallis: $p < 0.005$).

The RMP monitored PFAS levels in urban stormwater as part of a multi-year stormwater screening study from water year WY2019–WY2022, which included 21 highly urban watersheds and four reference watersheds that had minimal urban development. Urban stormwater runoff mobilizes contaminants contributed from wet deposition and wash-off of contaminants from the terrestrial landscape, predominantly from impervious surfaces. The RMP has continued investigation of PFAS in stormwater, including relevant stormwater studies in wet season 2024 and 2025 included in this report, although these recent stormwater monitoring data have not yet been thoroughly reviewed and analyzed (reports in progress). Similar to wastewater, measured PFAS concentrations in urban stormwater runoff were dominated by PFHxA, PFOS, PFOA, and PFPeA, with measured concentrations ranging by a factor of one hundred (Table 2.1). Overall, both pathways show comparable median concentrations (Table 2.1). The detection frequency of these analytes in wastewater and stormwater runoff are over 80%.

Other pathways, such as dry atmospheric deposition, industrial wastewater, agricultural runoff, and groundwater transport from contaminated sites have not been monitored by the RMP. The RMP has an ongoing special study on PFAS in atmospheric deposition through precipitation events.

Widespread Detections of PFAS in Bay Matrices

RMP monitoring of PFAS in the Bay, beginning in 2002, has documented widespread contamination. The RMP has measured PFAS in Bay surface water, sediment, sport and prey fish, bird eggs, and marine mammals. Bay wildlife are exposed to PFAS through water (via ingestion, aquatic respiration, and regulation of osmotic pressure), diet, and ingestion of sediment.

Within surface waters, PFHxA is the most dominant PFAS, detected in more than 60% of samples during the dry season and exhibiting the highest concentrations (Table 2.1). This likely reflects the shift in production from legacy C8 chemistries (e.g., PFOA and PFOS and their precursors) towards C6 (e.g., PFHxA and its precursors) and other shorter-chained chemistries. PFOA and PFOS followed with detections in approximately half of the samples at concentrations slightly lower or within range of PFHxA. These

results track with the detections of PFAS in the stormwater and wastewater pathways, though Bay concentrations are generally lower (Table 2.1). This can be partially attributed to the dilution of PFAS at sites sampled in the open Bay, characterized by deeper waters away from direct sources and pathways. Monitoring of near-field sites (i.e., proximal to stormwater and wastewater discharges) during the wet season found more PFAS and at levels generally higher than those in open Bay, dry season detections. The sum of PFAS at near-field sites (median 27 ng/L) was higher than that of open Bay water samples (median 9 ng/L), further indicating the dilution of pathways from near-field to open Bay sites. In contrast, these spatial variations were not observed in sediment in any subembayment.

The dominant PFAS detected in Bay sediment, bird eggs, sport and prey fish, and marine mammals is PFOS (Table 2.2 and Figure 2.2), which typically accounts for over half of the sum of PFAS quantified with targeted methods (EPA Method 1633) in biota. The dominance of PFOS in these matrices can be explained by its historical use in AFFF and consumer products, its persistence in the environment, its tendency to partition to sediment, and its high bioaccumulation potential relative to other PFAS. (Table 2.2 and Figure 2.3).

Table 2.2. Median concentrations and range (shown in parentheses) of key PFAS measured in Bay sediment (2023), prey fish (2023), sport fish (2024), bird eggs (2022 and 2024), and harbor seal serum (2023/2024). Less than method detection limit (<MDL) is shown if PFAS was not detected. Sum of PFAS (quantified PFAS with EPA method 1633) calculated with <MDL treated as zeros.

PFAS Analytes	Sediment (ng/g dw)	Prey Fish (ng/g ww)	Sport Fish (ng/g ww)	Bird Eggs (ng/g ww)	Harbor Seals (ng/g ww)
Bay-wide					Central Bay Only
PFOS	0.3 (0.05–0.67)	1.5 (0.31–10.1)	1.2 (<0.1–8.1)	74 (33–251)	52 (32–70)
PFOA	0.06 (<0.03–0.3)	<0.1 (<0.1–0.2)	<0.1 (<0.1–0.3)	1.1 (0.3–23)	0.65 (0.13–0.87)
sumPFAS	0.8 (<MDL–6)	3.4 (0.5–18.8)	3.3 (<MDL–70)	123 (76–399)	78 (47–97)
South and Lower South Bay					
PFOS	0.42 (0.05–0.7)	2.5 (0.4–10)	4.8 (2.8–8.1)	145 (96–251)	NA
PFOA	0.08 (<0.04–0.3)	0.1 (<0.1–0.2)	<0.1 (<0.1–0.3)	4.4 (3.1–6)	NA
sumPFAS	1.1 (0.1–15.9)	6.6 (0.54–19)	10 (5.3–16.6)	216 (152–399)	NA

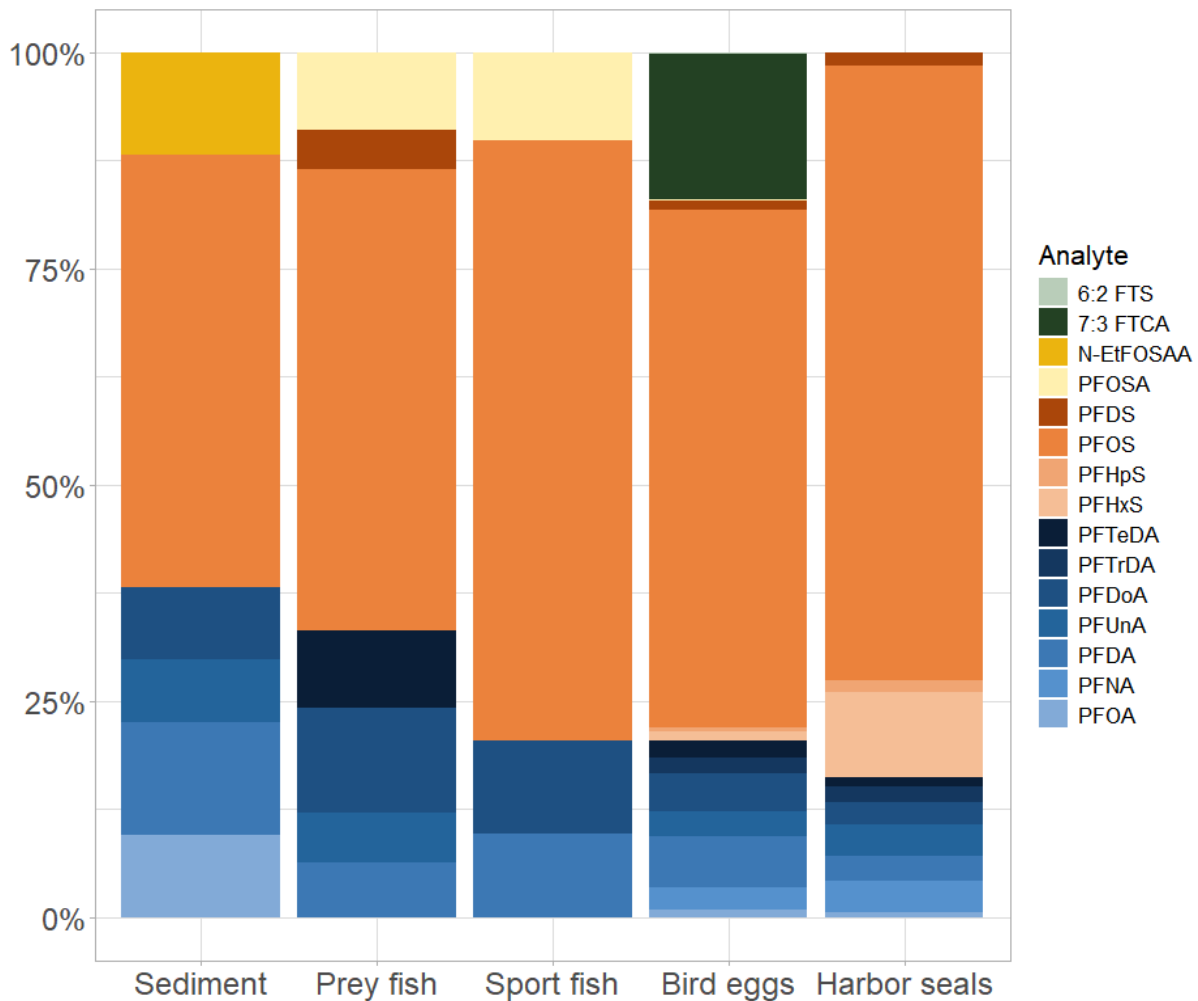


Figure 2.2. Composition of PFAS in Bay biota and sediment matrices is dominated by PFOS, which typically accounts for over half of the sum of PFAS quantified. Composition based on PFAS analytes with detection frequency >50% and median concentration. Composition based on sediment (2023), prey fish (2023), sport fish (2024), bird eggs (2022 and 2024), and harbor seal serum (2023/2024).

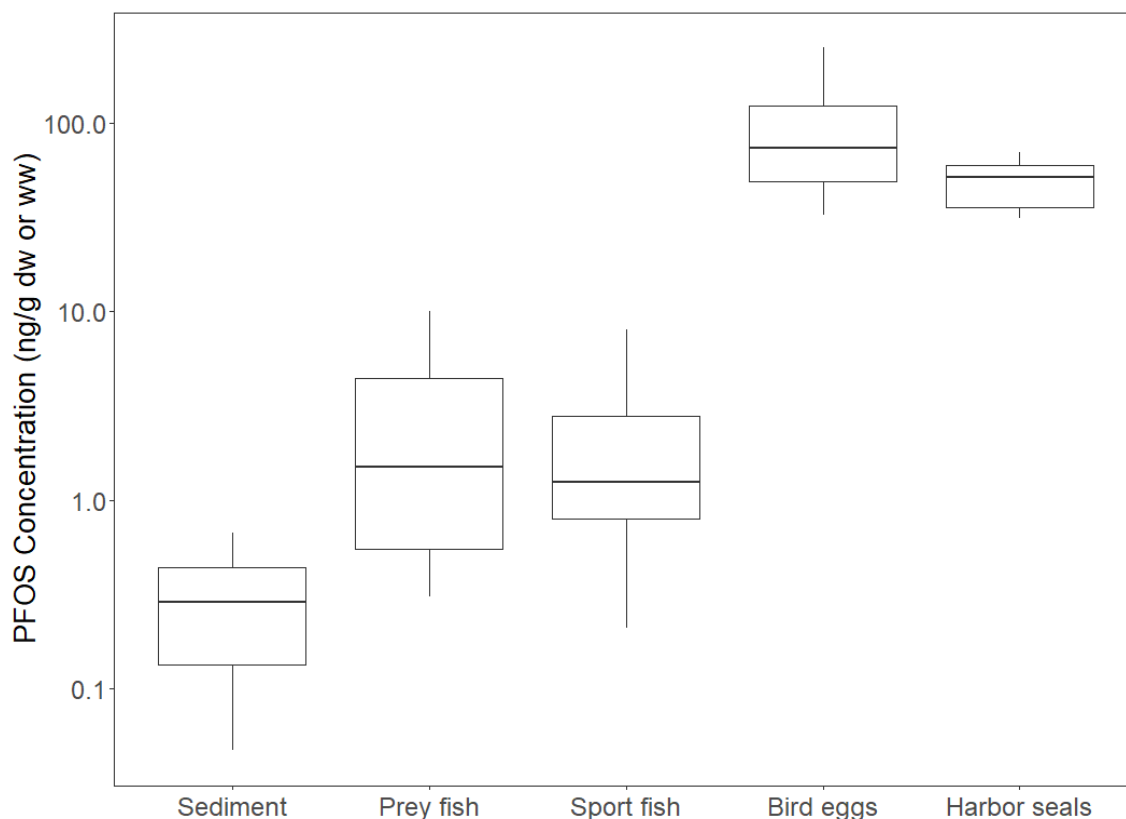


Figure 2.3. Boxplot showing the distribution of recent PFOS concentrations for sediment (2023), prey fish (2023), sport fish (2024), bird eggs (2022 and 2024), and harbor seal serum (2023/2024). The box represents the interquartile range (IQR), with the median line inside. Whiskers extend to the most extreme data points within 1.5*IQR.

Elevated Levels of PFOS Observed in South and Lower South Bay

Concentrations of commonly detected PFAS in Bay sediment, surface water, sport fish, prey fish, and bird eggs are higher in South and Lower South Bay compared to other subembayments (Table 2.2). Average concentrations of PFOS in shiner surfperch from South Bay (mean 3.2 ng/g ww) in 2024 were significantly higher (ANOVA with TukeyHSD, $p < 0.05$) than those from all other regions of the Bay (means=0.5–1.8 ng/g ww) except San Leandro Bay (mean 2.6 ng/g ww). Higher levels of PFAS in South and Lower South Bays are partially attributed to limited hydraulic flushing relative to other San Francisco Bay regions, due to less tidal flushing from the Pacific Ocean (Conomos, 1979). Potential sources of PFAS to receiving waters are prevalent across the southern Bay due to its urbanized landscape. Overall, both South and Lower South Bay are highly influenced by municipal wastewater effluents and urban stormwater runoff compared to other regions.

Decreasing Trends in Long-Chain PFAS Observed in South and Lower South Bay

While PFOS levels in South and Lower South Bay are highest in the Bay, this sub-region also shows the clearest decreases over time. Average concentrations of PFOS in sediment, bird eggs, and sport fish in South and Lower South Bay show statistically significant declines over the last ten years. Mean sediment concentrations of PFOS in South and Lower South Bay have declined from 1.5 ng/g dw in 2012 to 0.4 ng/g dw in 2023, representing a 73% decline (Figure 2.4). PFOS concentrations in bird eggs in Lower South Bay have decreased by 79% from peak median concentrations of 638 ng/g ww in 2016 to 135 ng/g in 2024. PFOS in shiner surfperch, an important sport fish species, declined by a more modest 15% in South Bay between 2014 (median 4.0 ng/g ww) and 2024 (3.4 ng/g ww). These declines may be attributed to management actions that phased out US production of PFOS.

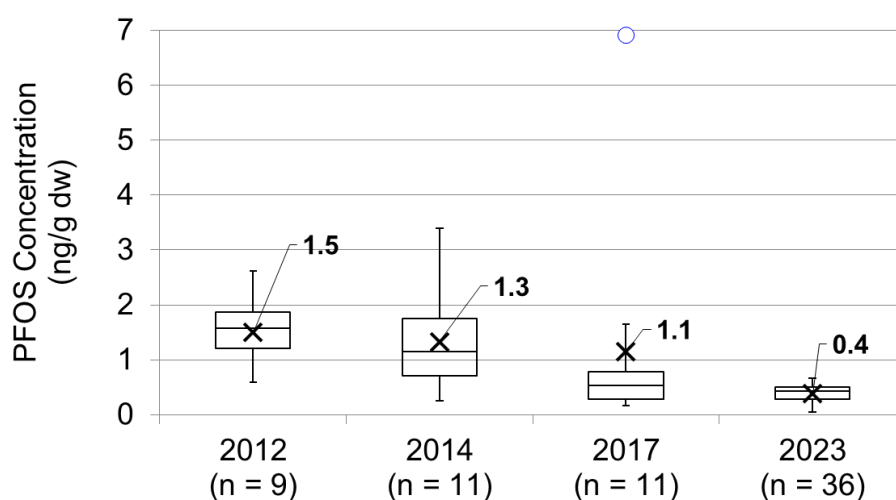


Figure 2.4. Boxplot of PFOS concentration in South and Lower South Bay sediment from samples collected in 2012, 2014, 2017, and 2023. The box represents the interquartile range (IQR), with the median line inside and X representing the average. Whiskers extend to the most extreme data points within 1.5*IQR. Outliers are plotted individually (Méndez, Miller, et al., 2025).

Trends for other matrices and other subembayments are less clear and require further monitoring to evaluate. PFOS concentrations in water and sediment were lower in the most recent datasets compared to previous monitoring efforts, but there are insufficient data to establish statistical significance. Trends for sediment, bird eggs, and sport fish are also not as pronounced in other subembayments, which tend to have lower starting concentrations than South Bay. Trends are less clear for PFOA and other PFAAs, which may be explained by lower concentrations and more diverse sources and contributions from precursors.

2.2 Risk Screening

For those contaminants of emerging concern known to occur in the Bay, the RMP establishes priorities and guides future monitoring using a tiered risk-based framework primarily based on comparison of observed concentrations to available toxicological thresholds (Sutton, Miller, et al., 2024). PFAS as a class is considered a High Concern in the Bay, which is defined as having occurrence data that suggest a high probability of an adverse impact on Bay wildlife or people, with sufficient data indicating occurrence concentrations between one and two orders of magnitude above ecological risk thresholds and above, but less than one order of magnitude greater than available human health thresholds for fish consumption. The PFAS evaluation is driven primarily by human health concerns for the occurrence of a few individual compounds (PFOS, PFOA, PFNA) in sport fish fillets. States, territories, and Tribes have the primary responsibility for issuing fish consumption advisories to protect people from the risks of eating contaminated fish caught in local waters. No human health or regulatory thresholds have yet been established for PFAS in California fish. However, 14 other states have issued fish consumption guidelines for PFOS and several other PFAS. These guidelines vary based on when they were established, the population they serve (e.g., general population vs. sensitive populations such as children), and the methods and underlying data used by each state to derive them. For example, “do not eat” state guidelines for PFOS range from 17 to 800 ng/g (Hughes & Henderson, 2025).

The lowest currently available state thresholds for PFAS were developed by Massachusetts (MA DPH, 2023) and are currently used for risk screening within the RMP’s tiered risk-based framework. For Bay-wide samples in 2024, 87% of Bay fish samples exceeded the one meal per day general population threshold for PFOS (0.50 ng/g), and 9% of samples exceeded the one meal per week general population threshold (3.52 ng/g).

Although placement in the RMP tiered risk-based framework for emerging contaminants is driven by concerns for human health and consumption of contaminated fish, PFAS observations in Bay fish also indicate a potential for ecotoxicity, especially for higher trophic level organisms like piscivorous birds and marine mammals (Méndez, Davis, et al., 2025). About eight percent (7.5%, all from Lower South Bay) of Bay sport fish samples from 2024 exceed the Canadian Federal Environmental Quality Guideline for PFOS in wildlife diet of 4.6 ng/g ww (ECCC, 2018).

PFAS concentrations in Bay water provide further evidence for the potential toxicity to people and sensitive aquatic organisms. While concentrations are below the USEPA chronic freshwater aquatic life water quality criteria for PFOA (100,000 ng/L) and for PFOS (250 ng/L) (USEPA, 2024d, 2024c), they far exceed the EU annual average environmental quality standard for the weighted sum of 25 PFAS in fresh and marine waters of 4.4 ng/L PFOA equivalents (EU Directive 2026/805). Many states have adopted EPA’s aquatic life water quality criteria or developed their own surface water thresholds; like state fish consumption thresholds, these vary by several orders of magnitude depending on their derivation methods and underlying data (Hughes & Henderson, 2025). The SF Bay Regional Water Quality Control Board has also

developed environmental screening levels (ESLs) for 13 PFAS in aquatic habitats for ecotoxicity (PFOS: 17 ng/L, PFOA: 3,200 ng/L) (SFBRWQCB, 2020, 2025). While these thresholds are not regulatory, they can be used for comparison with environmental concentrations. None of the individual PFAS concentrations observed in Bay water exceed the SF Bay Regional Water Quality Control Board ESLs for ecotoxicity. The sum of PFAS is also below all of these individual compound ecotoxicity thresholds except for that of PFOS.

Toxicity thresholds for PFAS in sediment are limited; neither the US EPA nor the EU have published benchmarks or standards. PFAS concentrations in Bay sediment do not exceed available toxicity thresholds for aquatic life (Ankley et al., 2021).

Secondary factors that support the High Concern classification with the RMP's tiered risk-based framework include persistence and the potential for cumulative (e.g., additive or synergistic) impacts within the PFAS class.

3 PFAS Pathways to the Bay

Wastewater effluent and urban stormwater runoff are important pathways for PFAS to be transported from urban sources to the Bay in the RMP's conceptual model.

Wastewater is the primary transport pathway for PFAS uses in the indoor environment that are directly or indirectly released to building drainage systems, including residential, industrial, and business uses discharged to municipal sewer systems and eventually publicly-owned treatment works (POTWs).

Urban stormwater runoff is the primary transport pathway for PFAS uses in the outdoor environment. Urban stormwater runoff mobilizes contaminants contributed from wet deposition and wash-off of contaminants from the terrestrial landscape, predominantly from impervious surfaces. The RMP's goals for monitoring PFAS transport pathways to the Bay are to better understand qualitatively the relative loadings from different pathways and to understand major sources and pathways to inform management actions. Transport pathways are important to link PFAS contamination in the Bay to upstream sources.

The RMP has conducted several studies to monitor wastewater effluent and urban runoff as important transport pathways to the Bay. This chapter focuses primarily on recent studies of PFAS in municipal wastewater and urban stormwater runoff. Additional transport pathways include riverine discharges, industrial wastewater, agricultural runoff, wet and dry atmospheric deposition, contaminated sites and landfills that may be transported via PFAS-contaminated groundwater (see Figure 2.1).

3.1 Wastewater

The RMP, in conjunction with the Bay Area Clean Water Agencies (BACWA), has monitored PFAS in wastewater several times since 2009. While the major sources of PFAS to municipal wastewater are poorly characterized, potential sources are extensive, chrome plating mist suppressant solvent, landfill leachate, manufacturing processes, consumer products and human excretion (i.e., human excretion due to ingestion of PFAS in drinking water, food, and household dust), and contaminated groundwater infiltration.

Wastewater Effluent

Wastewater effluent was studied across Bay Area POTWs in 2009, 2012, 2014, 2020, and 2022 (Houtz et al., 2016; Klosterhaus et al., 2013; Lin et al., 2024; Sedlak et al., 2017). In the most recent 2020/2022 study, 15 POTWs were selected to be representative of the diversity of POTWs in the region, and included a range in size, geographic, treatment type, service population and upstream industries serviced (Figure 3.1).



Figure 3.1: POTWs sampled during the most recent 2020 and 2022 Bay Area Clean Water Agencies PFAS study (orange dots).

PFCAs and PFSAAs are the PFAS subgroups that have been the most consistently monitored and detected in Bay wastewater effluent in studies since 2009. In the most recent 2020/2022 study (Figure 3.1), PFHxA (median: 16 ng/L, range: MDL–30 ng/L) was detected at the highest concentration, followed by PFPeA (8 ng/L, 3–30 ng/L), PFBA (6 ng/L, 2–36 ng/L), PFOA (6 ng/L, 3–12 ng/L) and PFOS (5 ng/L, MDL–13 ng/L). These PFAS were detected in all samples. The sum of PFAS quantified ranged from 26–112 ng/L (median: 57 ng/L; Figure 3.2).

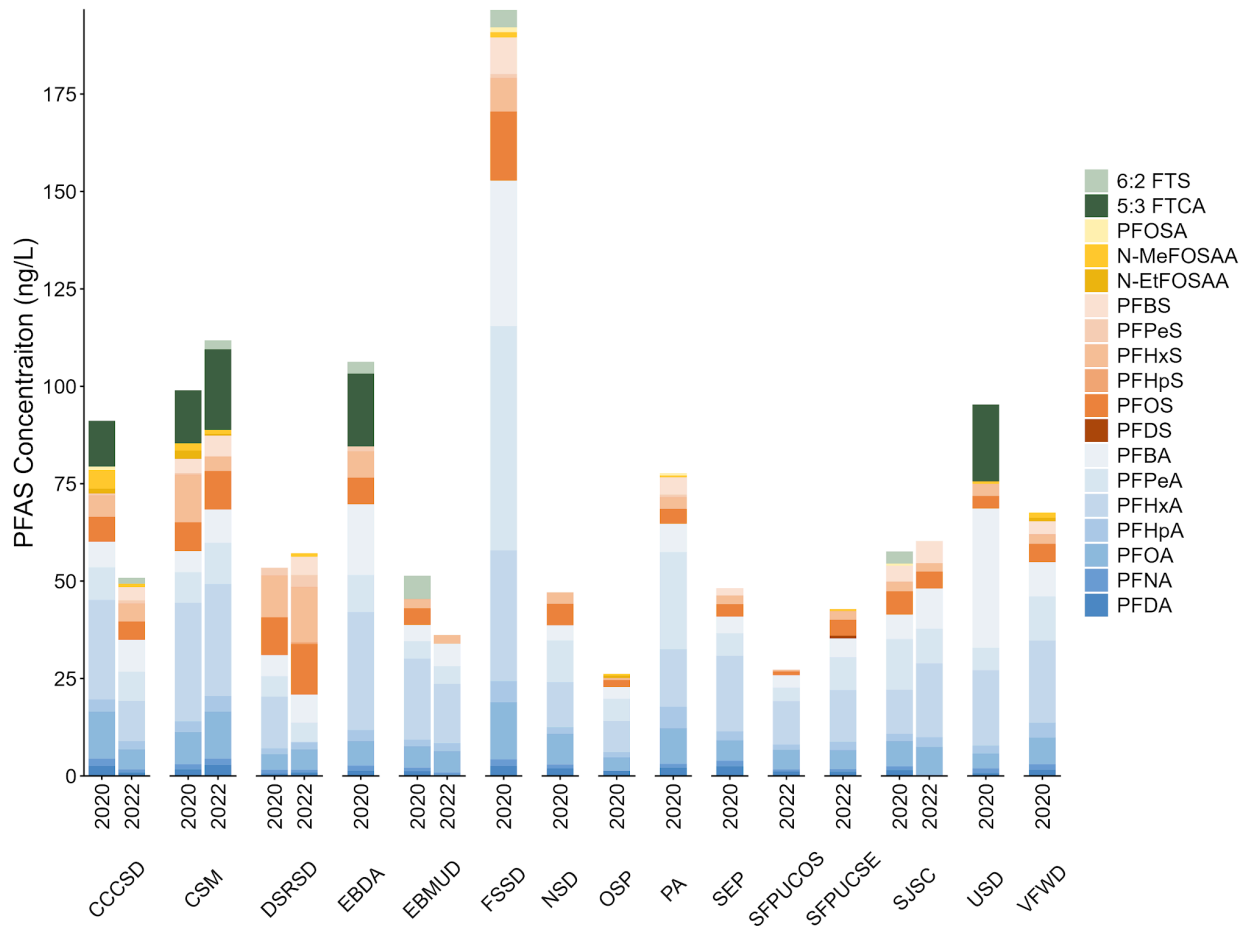


Figure 3.2: Concentrations of detected PFAS analytes in nanograms per liter in effluent from municipal POTWs from phase 1 (2020) and phase 2 (2022) sampling. Analytes below detection limits are treated as zero and not shown. Acronyms represent individual POTWs.

These concentration ranges show fairly consistent baselines, levels, and composition of PFAS in municipal wastewater. Wastewater effluent from San Francisco International Airport (SFO) showed significantly elevated levels of PFAS, including PFOS and 6:2 FTS, compared to other POTWs. Since wastewater from SFO appears to be an outlier (and may be explained by a significant portion of the wastewater influent coming from airport-related industrial activities), this section’s summaries of municipal wastewater results exclude SFO.

Comparison of effluent results from 2009 to 2022 indicate a continued decrease of long-chain PFAS (Figure 3.3). This trend is verified by analysis of the datasets indicating decreases by over 50% largely led by declines in long-chain compounds. Specifically, the average PFOS, PFOA, and PFNA concentrations declined by 78%, 81%, and 91%, respectively, from 2009 to 2022. A Kruskal-Wallis test was used to test the statistical significance of the PFOA, PFNA, and PFOS decreases between 2009 and 2022. Only the decreases in PFOA and PFNA concentrations were statistically significant

($p < 0.005$). Although there are less available data to compare PFAS quantified by the TOP assay, the comparison between 2014 and 2022 POTW effluent also indicates a decrease in PFAS quantified via TOP.

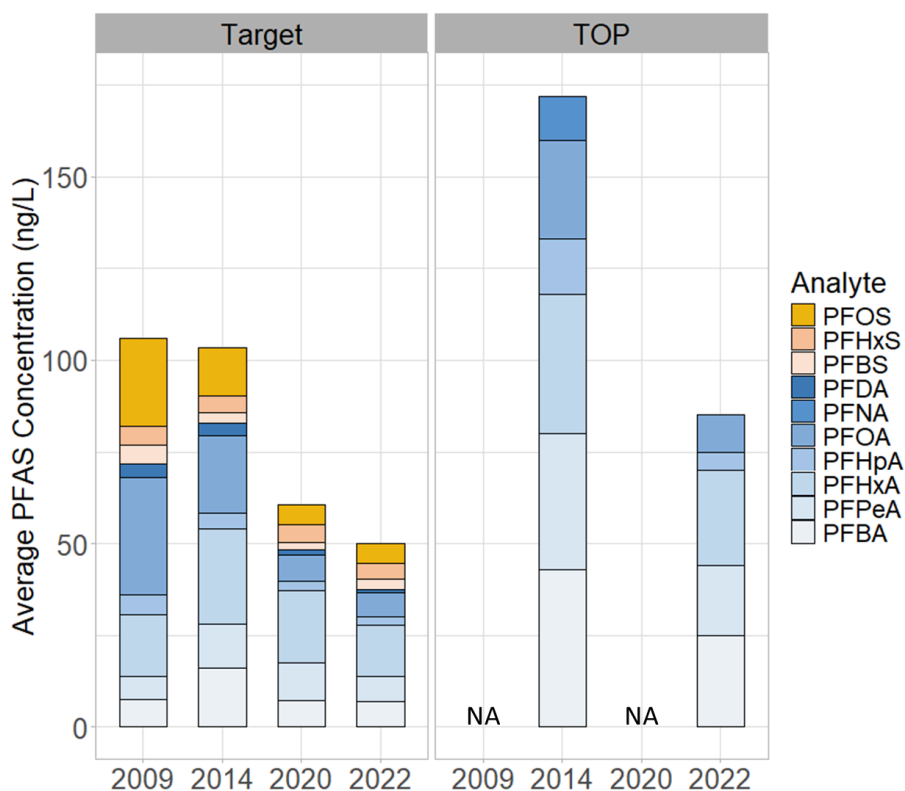


Figure 3.3: Average concentration of detected PFAS in San Francisco Bay POTW final effluent. The TOP assay was not applied for samples from 2009 and 2020.

Wastewater Influent

Wastewater influent was monitored during studies in 2020 and 2022 (Lin et al., 2024). Targeted analysis detected a variety of PFAS analytes in influent samples. Three analytes [PFHxA (median: 7.8 ng/L), PFPeA (6.3 ng/L), and PFOA (3.7 ng/L)] were quantified in all samples. PFHxS (median: 3.1 ng/L) and PFOS (median: MDL, maximum: 20.9 ng/L) were also frequently detected. The sum of PFAS quantified using targeted analysis ranged from 10–106 ng/L, with a median of 34 ng/L.

The TOP assay found significantly higher PFAS concentrations in influent than what was found using PFAS target analysis methods. The sum of PFAS quantified using TOP analysis ranged from 150–422 ng/L, with a median of 256 ng/L. These results indicate a significant presence of PFAS precursors in influent.

While sum of PFAS quantified in influent using EPA 1633 were not significantly different from levels in effluent, sum of PFAS quantified via TOP was higher in influent compared to effluent. This suggests significant levels of PFAS precursors or their terminal products partition to biosolids.

Sewershed Influent

The 2022 study to investigate upstream sewershed sources evaluated sewershed samples from residences, industrial laundries, hospitals, semiconductor-related manufacturing, chrome plating facilities, and others.

Residential customers represent a significant portion of sewage flows received by municipal POTWs, with the proportion of residential flows varying for each POTW. Upstream sewershed residential discharges from three POTWs were sampled to evaluate if residential flows were an important source of PFAS to the participating POTWs (Lin et al., 2024). Each of these POTWs receive over 80% of influent flows from residential customers. The sampled residential sewersheds represented a diversity of household types (i.e., multi-unit dwellings, single-family homes), infrastructure age, and geographic locations. 14 different residential locations were sampled for this study, with sample replicates taken at two locations.

Using the targeted method, the sum of PFAS levels in residential sewershed discharges varied significantly, ranging from under the method detection limit (<MDL) to 80 ng/L. PFHxA had the highest residential PFAS target analysis concentration range (<MDL–27 ng/L). The sum of PFAS quantified via TOP assay for the residential sewersheds ranged from 4–850 ng/L (median: 187 ng/L). First-order calculations estimated that at the three POTWs studied, residential customers accounted for a majority of estimated PFAS mass loadings in influent.

Five industrial laundry operations were sampled within the sewersheds of three POTWs. PFAS concentrations discharged from these operations were significantly higher than average levels in influent entering POTWs in a small screening study. The sum of PFAS for targeted analysis of industrial laundry ranged from 22–762 ng/L (median: 48 ng/L). PFAS measured via the TOP assay ranged between 58–100,369 ng/L (median: 892 ng/L), with one site having a PFAS concentration 27 times higher than the site with the next highest concentration. PFOA and 6:2 FTS were the most frequently detected analytes, with both detected in 86% of analyzed samples.

These industrial laundries serve a variety of business clients and industries. Restaurant linens, mops, and uniforms composed the majority of laundered items at most of the sampled facilities. Other commonly laundered items include business floor mats, and some facilities reported laundering refinery uniforms and rags, medical uniforms, patient gowns, vehicle shop towels, and laboratory coats.

Biosolids

Biosolids land application and subsequent stormwater runoff and groundwater transport represent potential pathways for PFAS entering the Bay. A small portion of Bay Area biosolids are land applied in the San Francisco Bay region, and most of what is applied in the region is in the North Bay along the Petaluma River and near Tubbs Island (BACWA, 2022). Biosolids may also be used as landfill Alternative Daily Cover (BACWA, 2024).

Biosolids from a representative subset of Bay Area POTWs were evaluated in 2020/2022 (Lin et al., 2024). The sum of PFAS quantified ranged from 44–323 ng/g dw (median: 202 ng/g dw). 5:3 FTCA was the highest detected PFAS analyte, having a detection rate of 95% and a median PFAS target concentration of 93 ng/g dw. Generally, 5:3 FTCA is considered an intermediate transformation product, particularly for 6:2 fluorotelomer structures, and is commonly observed in landfill leachates and food contact substances (Lang et al., 2017; Schaider et al., 2016), which may explain its widespread presence as the dominant detected PFAS in biosolids. Median concentrations of PFOS and PFOA in Bay biosolids were 13 and 1 ng/g respectively ([Appendix A](#)).

POTWs continue to plan for various changes to biosolids management practices in response to SB 1383 regulations intended to divert organics from landfills to reduce methane emissions, as well as increasing costs for transporting biosolids. BACWA will continue to track changes to biosolids management practices through surveys every two to three years to support ability to track changes in management due to water quality and air quality regulations and costs (BACWA, 2024).

While EPA has evaluated the potential human health and environmental risks associated with biosolids land application that contains PFOA and PFOS and determined there are potential risks, the agency has not established a federal criteria for land application of biosolids. States are taking leadership on this topic through various approaches. Michigan's Department of Environment, Great Lakes, and Energy (EGLE) is an agency that has developed a threshold of 20 ng/g of PFOA or PFOS in biosolids, above which there are restrictions to land application as part of its Land Application of Biosolids Containing PFAS Interim Strategy updated in 2024 (Michigan EGLE, 2022). Biosolids with PFOA or PFOS above 100 ng/g cannot be land applied. Some Bay Area POTW biosolids concentrations exceed the 20 ng/g thresholds, but are below the 100 ng/g threshold. PFOA concentrations do not exceed these thresholds.

PFAS TOP concentrations had a maximum of 1,565 ng/g dw (median 594 ng/g dw), showing high levels of PFAS precursors in biosolids.

The wide variation in biosolid PFAS concentrations can be partially explained by differences in treatment plant digestion and treatment processes. Biosolids that did not undergo digestion showed the lowest PFAS concentrations, while biosolids that underwent multiple digestion stages showed higher PFAS concentrations. This could be due to the reduction and concentration of organic solids as the biosolids go through the digestion stages.

Comparison with Other Locations

The PFAS effluent concentrations measured in 2020/2022 are generally consistent or in the lower range of PFAS concentrations reported by others, including municipal POTWs in the rest of California (Aflaki & Voskuhl, 2022), and studies of municipal wastewater from Chickasha, OK (Masoner et al., 2023) and Nevada (Thompson et al., 2022). The analyte composition observed in the current study was comparable to the Oklahoma study, which also observed frequent detections of PFOS.

The four individual PFAS analytes detected in 100% of BACWA influent samples (PFPeA, PFHxA, PFOA, PFHxS) were also detected in 100% of influent samples in a study from a regional city catchment in Queensland, Australia (Gallen et al., 2022). Interestingly, 6:2 FTS was frequently detected in the Gallen et al. (2022) study but was not observed in the BACWA influent samples. Another Australian study also found abundant levels of diPAPs (polyfluoroalkyl phosphoric acid diesters), which are generally used as grease-proofing agents in food contact materials (Szabo et al., 2023); these analytes were not included in the current BACWA study.

PFAS targeted analysis concentrations in Bay Area biosolids are also in the range of PFAS concentrations in biosolids reported by others. Thompson et al. (2023) quantified 92 PFAS analytes in municipal biosolids from eight POTWs in Florida, and the sum of PFAS ranged from 182–1,650 ng/g. Another study in Australia (Coggan et al., 2019; Moodie et al., 2021) reported the sum of PFAS concentrations ranging from 3–900 ng/g. In the Florida study, two facilities that used anaerobic digestion processes reported 5:3 FTCA concentrations of 21 and 65 ng/g dw. In the Australian study, 5:3 FTCA was detected in 88% of the biosolids samples ($n = 17$) with a mean concentration of 16 ± 21 ng/g (median: 4.6 ng/g) and a range of <MDL–61 ng/g (Moodie et al., 2021).

3.2 Urban Stormwater Runoff

As part of a stronger focus on emerging contaminants in stormwater, the RMP has been monitoring PFAS in urban stormwater runoff every wet season since water year (WY) 2019, except for WY2023 (Peter et al., 2024; Sutton, Avellaneda, et al., 2024). This synthesis summarizes data from WY2019–WY2025 targeted PFAS analysis (EPA method 1633). During this time, the RMP monitored PFAS in a total of 57 storm events at 43 spatially diverse urban watersheds, including four sites that drain more rural watersheds (Figure 3.4). Most sites were monitored during a single storm event, while 15 sites were monitored during two or three storms. Stormwater samples were collected as part of various stormwater sampling efforts, and utilized different sampling and compositing methods. These included traditional manual methods using a stainless steel bailer to collect stormwater aliquots, small automated remote samplers using the SFEI Mayfly and Omega sampling systems, and the Aquacell Portable Wastewater Sampler (Aquamatic) using a vacuum pump. We also summarize preliminary data from WY2025 using novel analytical methods to inform future stormwater monitoring studies. Further data analysis and interpretation to support understanding stormwater loadings to the Bay and linkage assessment to potential sources will be developed through the ongoing PFAS Sources to Solutions project and summarized in future reports.

The dominant PFAS quantified in urban stormwater runoff via the multi-year screening study included two PFCAs detected in all samples, PFHxA and PFOA, and PFOS, which was detected in >85% of samples. These three PFAS exhibited the highest maximum concentrations across all PFAS: PFHxA (MDL–180 ng/L, median 7.4 ng/L), PFOA (MDL–110 ng/L, median 6.9 ng/L), and PFOS (MDL–81 ng/L, median 6.6 ng/L). The median values are comparable to median concentrations in municipal wastewater effluent (see Table 2.1).

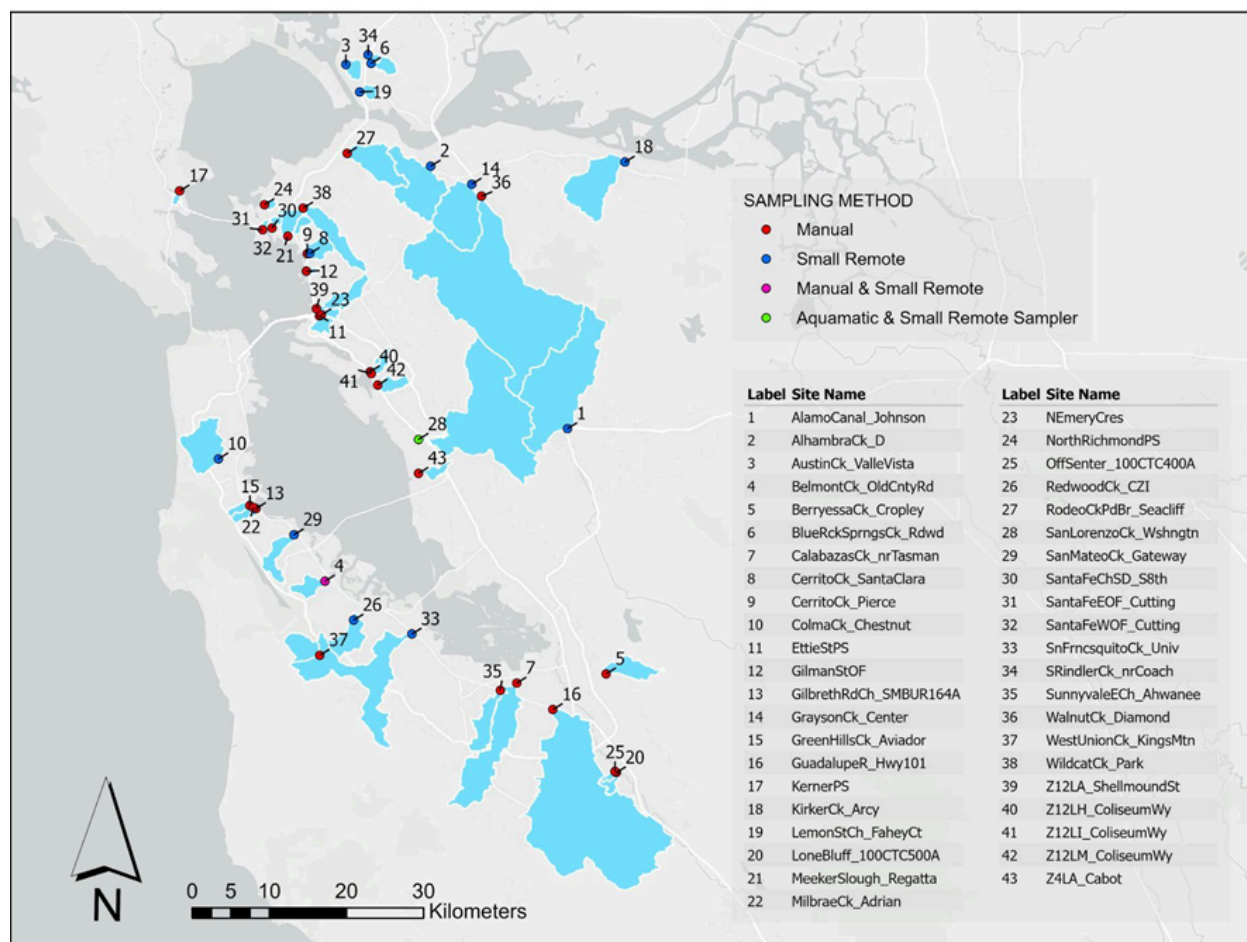


Figure. 3.4: Map of stormwater sites sampled between wet year 2019 to wet year 2025. Blue polygons represent watersheds, and ‘+’ symbol represents watershed drainage locations that were sampled during storm events. Site numbers adjacent to site markers on the map correspond to site names provided in the lower righthand table.

Additional PFAAs that were detected at lower levels in a majority of samples included the long-chain PFAAs PFDA (median 2.5 ng/L) and PFNA (1.4 ng/L), and short-chain PFAAs: PFBA (median 5.8 ng/L), PFPeA (5.7 ng/L), PFHpA (2.7 ng/L), PFBS (2.7 ng/L), and PFHxS (1.2 ng/L). Sum of PFAS quantified ranged from 4–574 ng/L. All other target analytes 6:2 FTS and 5:3 FTCA, FASAs, and FOSAAs, HFPO-DA, and ADONA included in EPA Method 1633 were detected infrequently or were not detected.

Levels of PFHxA, PFOA, and PFOS varied significantly across sites, with ranges spanning two orders of magnitude (Figure 3.3). The highest measured concentration for PFOS was 81 ng/L at the Santa Fe Channel East site, followed by 39 ng/L at Santa Fe Channel West and 19 ng/L at Line12H site. Emeryville Crescent North had the highest PFOA (4.2–110 ng/L) and PFHxA (MDL–180 ng/L) concentrations, with significant variability observed during four different storm events. The Emeryville Crescent North site was sampled three times during the period WY2020–WY2022 and twice at the nearby Ettie St Pump Station.

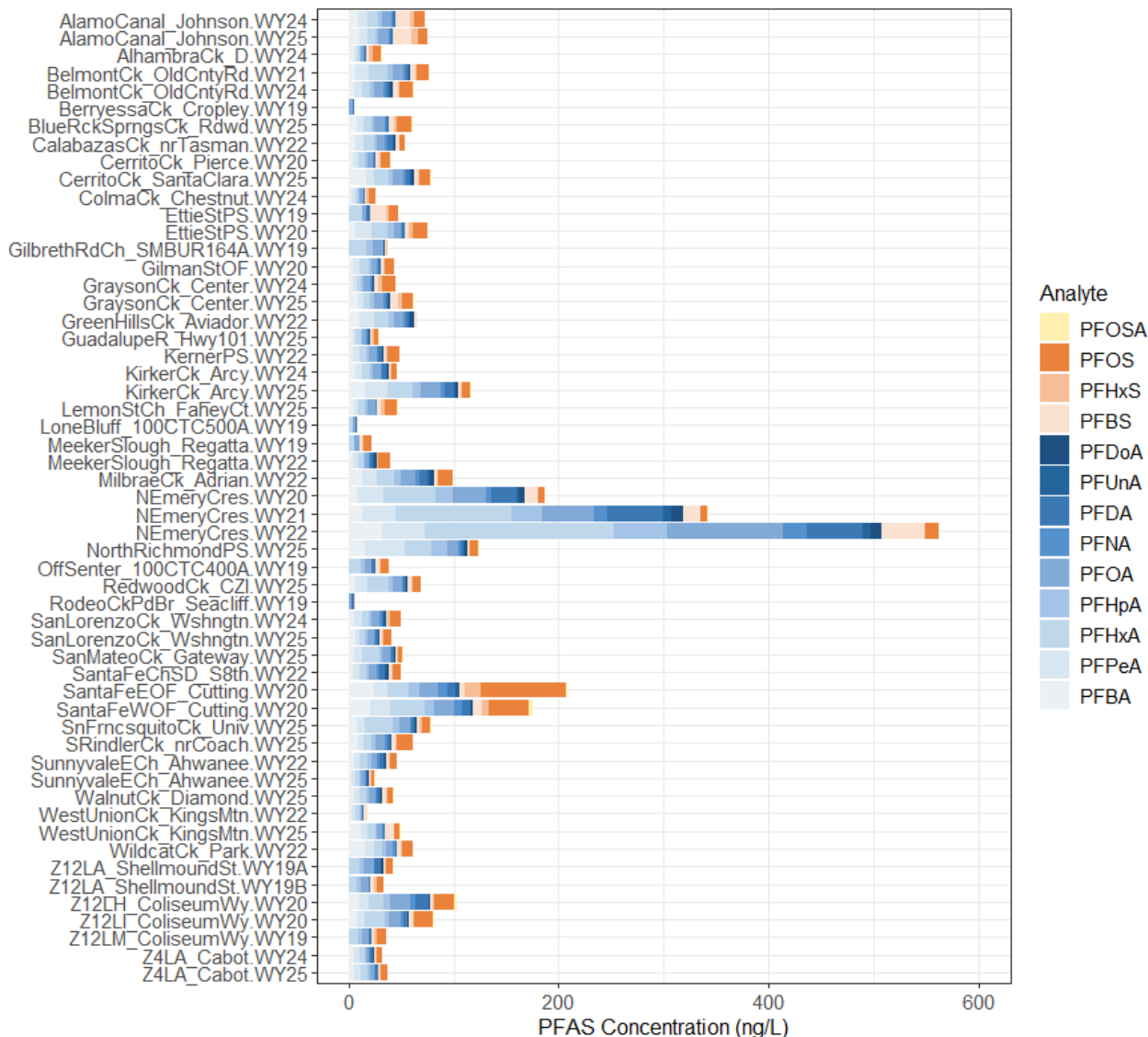


Figure 3.5. Bar charts summarizing concentrations of PFAS analytes from the EPA 1633 PFAS analyte list with detection frequencies in >15% of stormwater samples. Non-detects (i.e., concentrations below method detection limits) are not plotted. Sample names indicate SiteName.Water Year.

Evaluation of Ultra-Short Chain PFAS

Trifluoroacetic acid (TFA) is an ultra-short chain PFAS that has been found to be the most abundant PFAS in the environment globally with concentrations that pose potential irreversible adverse risk. High levels of TFA have been reported in precipitation, surface waters, and groundwater globally at levels of hundreds of ng/L (Arp et al., 2024; Ma et al., 2025; Zhi et al., 2024). The RMP has only recently started investigating TFA, and preliminary results of ultra-short chain PFAS in Bay stormwater samples from WY2025 (n = 6, plus two field duplicates) indicate notably high levels of TFA at hundreds of ng/L in all eight samples analyzed (maximum = 990 ng/L). Cahill (2022) found TFA

concentrations in surface water streams downwind (east) of the Bay Area increased tenfold from 1998 to 2021 (n = 4 pairs); 2021 median concentration was 145 ng/L, and the maximum was 2,790 ng/L. These high concentrations warrant further follow-up investigations of TFA in Bay wastewater and water.

Evaluation of PFAS Precursors

Also, targeted analysis of diPAPs in stormwater samples (n = 10 from the 2018–2022 CECs screening study and n = 6 for WY2025) found 6:2 diPAP in 94% of samples at low ng/L levels (Peter et al., 2024). DiPAPs are commercial surfactants used in cosmetic products, toilet paper, paper-based food packaging, textiles, and other consumer products, and can transform to PFCA terminal products in the environment, which are detected through the TOP assay. TFSI, a PFAS used in lithium ion batteries, was detected in 50% of six WY2025 samples at ~1 ng/L. Applying another newly developed method examining six common FTOHs, 6:2 FTOH and 4:2 FTOH were detected at elevated levels in one and two samples, respectively. FTOHs have also been found in consumer products similar to diPAPs as well as paints and coatings that are applied to large outdoor surfaces, and a potential path to stormwater runoff.

Applying the TOP assay to stormwater samples showed increases in PFCA levels at some sites after oxidation, which indicates the presence of unknown precursors (Sutton, Avellaneda, et al., 2024). However, we also observed some challenges and uncertainties in applying the TOP assay to stormwater samples. Some samples did not show significant increases in PFAS post oxidation, which may be partially explained by higher detection limits using the TOP assay. Additionally, the TOP assay had blank contamination that was not observed for targeted analysis. Samples analyzed in triplicate showed significant variability (relative standard deviation of 40–170%), although variability may be influenced by particulate matter that makes it challenging to create homogenous triplicates. Additional analysis is required to determine if the TOP assay should be a long-term component of stormwater monitoring.

Overall, these findings show the presence of a wide variety of PFAS precursors in stormwater runoff. We recommend continuing to pilot these analytical methods for stormwater monitoring and evaluation after the next round of WY2027 stormwater samples to make recommendations for long-term monitoring.

Comparison to Other Studies and Regions

While PFAS is less commonly monitored in the stormwater pathway, observations have been reported at some sites over the last decade. PFAS concentrations in urban runoff in the Bay region appear generally in line with other reported studies, although the dominant species may be quite different in different regions. For example, a study of urban stormwater runoff entering a roadside rain garden in New Jersey reported a sum of PFAS concentrations of approximately 1,500 ng/L, dominated by PFHxA and PFBS (Saleh et al., 2025). Another study evaluating major river basin systems in Alabama reported sum of PFAS concentrations ranging from non-detect to 237 ng/L, with a mean of 35.2 ng/L, with PFBS being the highest and PFPeA detected at the highest frequency (Viticoski et al., 2022).

4 PFAS in San Francisco Bay Abiotic Matrices

4.1 Sediment

The RMP started monitoring PFAS in sediment in 2009. Since then, monitoring has gradually increased the geographic coverage of the Bay through sampling events in 2012, 2014, 2017, 2020, and 2023 (Méndez, Miller, et al., 2025; Sedlak et al., 2017, 2018). This section focuses on synthesizing results and evaluating spatial trends from the 2017, 2020, and 2023 studies, while also incorporating results from the 2012 and 2014 to evaluate temporal trends. Sediment samples represent the top 5 centimeters of surface sediment in the Bay (Table 4.1; Figure 4.1).

Table 4.1. Summary of recent sediment sampling events.

Sampling Year	Sampling Region	N
2017	South and Lower South Bay margins ¹	11
2020	San Pablo and Suisun Bay margins	12
2023	Status & Trends dry season open water sites	16
2023	Central Bay, South Bay, and Lower South Bay “margins” and near-field ² sites	24 (M) 14 (NF)

¹Margins (M) are defined as areas one foot below mean lower low water up to the vegetated marsh edge, roughly the mean higher high water line, which was identified as a gap in previous RMP efforts that sampled deeper water locations.

²Near-field (NF) are Bay water locations near watershed and wastewater inputs into the Bay.

For all samples analyzed for target PFAS from the 2017, 2020, and 2023 sampling events (n = 77), PFOS was the predominant PFAS found in Bay sediment (Méndez, Miller, et al., 2025). PFOS concentrations ranged between 0.08–6.9 ng/g dw. PFOA and the PFOS precursor N-EtFOSAA were the only other analytes detected in over 60% of samples in each monitoring effort, although median concentrations were low, ranging from 0.04–0.10 ng/g dw. Several additional long-chain PFCAs (PFNA, PFDA, PFUnDA, and PFDoDA) and PFDS were detected in more than 60% of samples from South and Lower South Bay, but less frequently in the rest of the Bay.

Median PFOS concentrations in sediment were notably higher in South and Lower South Bay compared to the rest of the Bay (Méndez, Miller, et al., 2025). Additionally, a greater variety of PFAS were detected in South Bay and Lower South Bay sediment compared to the rest of the Bay, leading to sums of PFAS levels that were higher in South Bay and Lower South Bay samples. Median PFOS concentrations in South and Lower South Bay samples were 0.54 ng/g dw in 2017 margin samples, and 0.42 ng/g dw in 2023 combined open Bay, margins, and near-field samples. Sediment concentrations were fairly consistent within each southern subembayment, and no differences were observed between open Bay, margin, and near-field samples. This is compared to median PFOS concentrations of 0.1 ng/g dw in San Pablo and Suisun Bays 2020 margin sites and 0.14 ng/g dw in combined Central and San Pablo Bays in 2023 (Table 4.2).

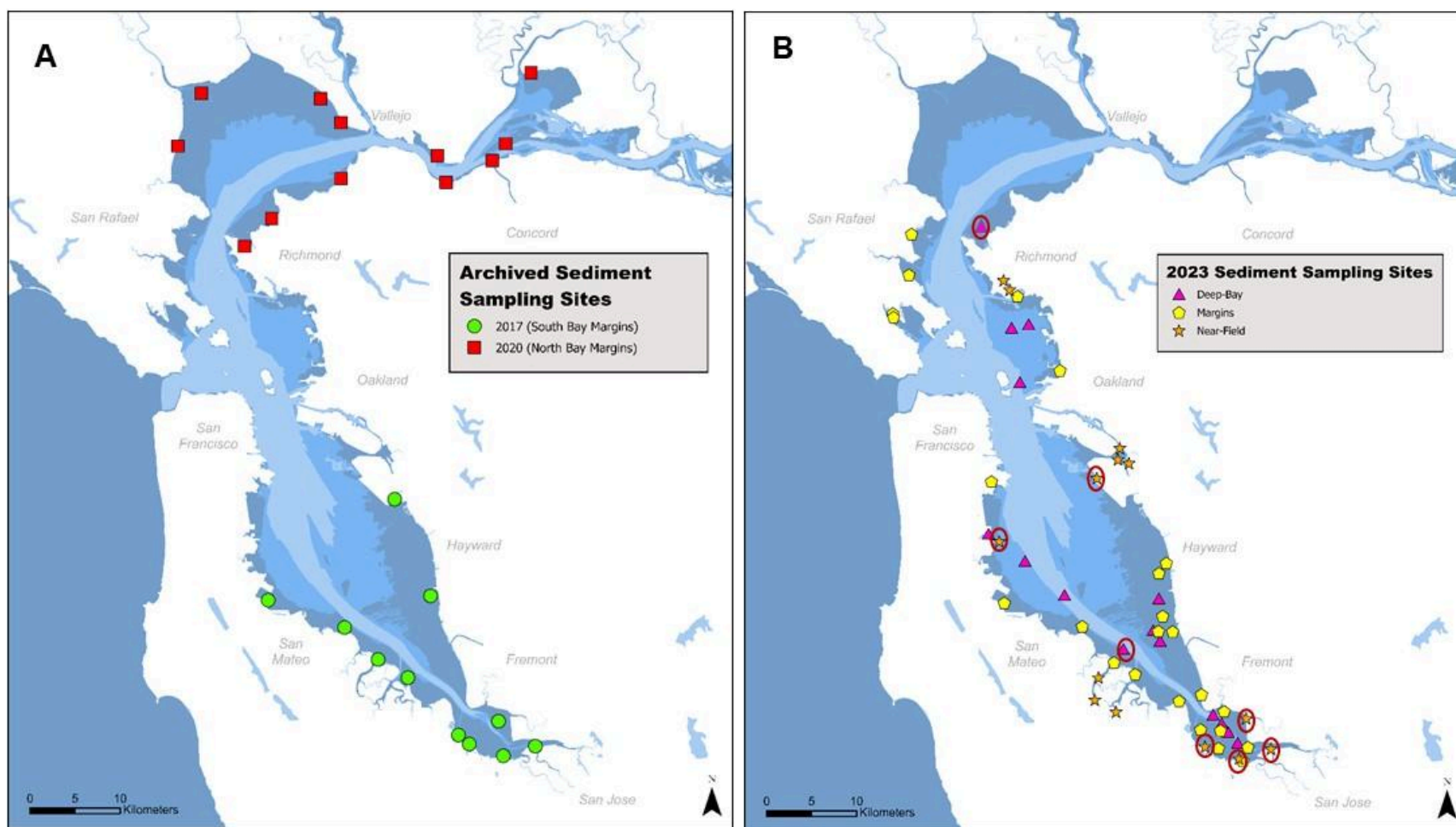


Figure 4.1. Maps of recent sediment sampling locations for analysis of PFAS. A: Sites where archived sediment samples were collected in 2017 and 2020. B: Locations sampled in 2023 for analysis of PFAS. Sites circled in red were also analyzed using the TOP assay.

Table 4.2. Median concentrations for PFAS detected above 60% in samples from 2017 (n = 11), 2020 (n = 12), and 2023 (n = 54). Samples from 2023 are combined as well as separated by subembayments (South and Lower South Bays, n = 36; Central and San Pablo Bays, n = 18) to compare with 2017 and 2020. Less than method detection limit (<MDL) is shown if PFAS was not detected. Sum of PFAS (quantified PFAS with EPA method 1633) calculated with <MDL treated as zeros.

PFAS	Median (ng/g dw)					
	MDL	2023 ¹ (All Sites)	2017 ² (South & Lower South Bays)	2023 ¹ (South & Lower South Bays)	2020 ² (San Pablo & Suisun Bays)	2023 ¹ (Central & San Pablo Bays)
PFOS	0.04	0.29	0.54	0.42	0.10	0.14
PFOA	0.04	0.06	0.10	0.08	<MDL	0.06
PFNA	0.04	<MDL	0.05	<MDL	<MDL	<MDL
PFDA	0.04	0.08	0.12	0.12	<MDL	<MDL
PFUnA	0.04	0.05	0.09	0.06	<MDL	<MDL
PFDoA	0.03	0.07	0.12	0.07	<MDL	<MDL
PFDS	0.04	<MDL	0.06	<MDL	<MDL	<MDL
N-EtFOSAA	0.04	0.06	0.10	0.10	<MDL	0.04
sumPFAS	NA	0.80	1.6	1.1	0.14	0.40

¹Data collected in 2023 were combined across open Bay, margins, and near-field sites with consistent levels across subembayments and no differences were observed between sites.

²Samples from 2017 and 2020 were collected only from margin sites in the Bay.

PFOS levels in sediment from South and Lower South Bay show declines between 2012 and 2023 (Figure 4.2; Méndez, Miller, et al., 2025). Concentrations are highest in these subembayments and the only area with consistent monitoring and detections since 2012. Mean concentrations have declined from 1.5 ng/g dw in 2012 to 0.4 ng/g dw in 2023. Conducting ANOVA on the normally-distributed datasets (which excludes the 2017 dataset), the decrease is statistically significant ($p < 0.05$) with a post-hoc Tukey-HSD test noting significance for declines from 2012 and 2014 to 2023.

TOP analysis of sediment of all 2017 and 2020 samples along with several from 2023 indicated the levels of PFAS precursors were quite variable throughout the Bay (Méndez, Miller, et al., 2025). Overall, 39% of all sediment samples analyzed (n= 31) showed measurable PFAS levels increased by over 35% after oxidation, with several showing substantial increases by an order of magnitude (range of increase: 35-1950%). PFOA showed the most consistently substantial increase post-oxidation (range of increase: 364-1,500%), indicating significant presence of long-chain PFAS precursors.

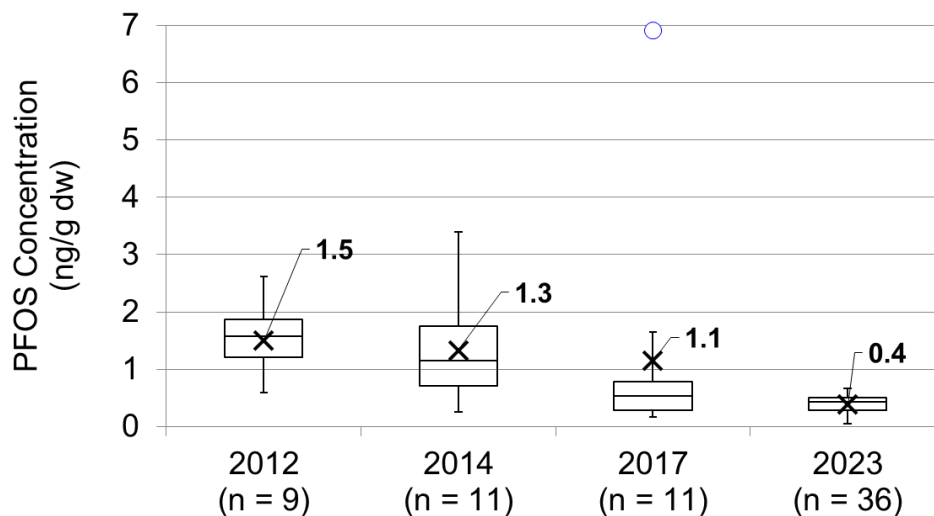


Figure 4.2. Boxplot showing distribution of PFOS concentrations in South and Lower South Bay sediment from samples collected in 2012, 2014, 2017, and 2023. The box represents the interquartile range (IQR), with the median line inside and X representing the average. Whiskers extend to the most extreme data points within 1.5*IQR. Outliers are plotted individually (Méndez, Miller, et al., 2025).

Comparison to Other Regions

Bay sediment PFAS concentrations align with global estuarine and marine levels ([Appendix Table B1](#)). In Florida, estuarine and marine sediments showed a similar trend of PFOS as the primary PFAS, with widespread long-chain PFAAs (Ahmadireskety et al., 2021; Griffin et al., 2023; Pulster et al., 2022). Most Florida sediments had lower average PFOS than South and Lower South Bays. Conversely, Bohai Sea (China) studies found PFOA predominant (Meng et al., 2021; Zhao et al., 2020); median PFOS was lower than the Bay's, though total PFAS ranges were comparable.

San Francisco Bay sediment PFAS levels were comparable to or lower than global freshwater sediments. PFOS also predominated in the Great Lakes and New Jersey, but average levels exceeded South and Lower South Bay levels (Codling et al., 2018; Goodrow et al., 2020). In France, PFOS was the primary freshwater PFAS with median concentrations similar to the Bay (Macorps et al., 2023). A national study of German freshwater sites found PFOS predominant at median levels above the Bay (Göckener et al., 2023), though a regional study (Guckert et al., 2022) found PFCAs more frequently than PFSAs, with higher total PFAS ranges. That study also frequently detected TFA (64%, 0.9–5.1 ng/g) and 6:2 diPAP (55%, up to 0.04 ng/g), compounds not previously monitored in Bay sediment.

Compared to the few published studies using the TOP assay on sediment, detections and overall concentrations of PFAA precursors are similar or lower in the Bay than other examined areas in Europe. The study in France (Macorps et al., 2023) found significant increases in 63% of samples while a regional study in Germany (Guckert et al., 2022) found only 29% of samples with a significant increase. Göckener et al., (2023)

developed and applied a modified version of the TOP assay by directly oxidizing the sample (dTOP, rather than applying oxidation on the sample extract as outlined by Houtz and Sedlak, 2012). This approach may provide a more comprehensive quantification of PFAS precursors in the sample with 73% of German freshwater sediments increasing from 44%–200% post-oxidation, a significantly greater frequency of increase compared to Bay samples. Because RMP TOP methods only oxidize sediment sample extractions, they may underestimate precursor concentrations.

4.2 Bay Water

In this synthesis, we focus on recent water sampling from 2021–2024 (Méndez et al., 2022; Méndez, Miller, et al., 2025). Open water samples were collected during the dry season of 2021 and 2023 as part of the biannual RMP Status and Trends monitoring cruise, which included 22 sites sampled across all five major Bay subembayments (Figure 4.3). A smaller set of open water and near-field water sites were sampled during WY2022, WY2023, and WY2024 (Figure 4.3). Within two hours after a storm event, near-field samples were collected at 3–4 sites, each near the mouth of a local tributary (Lion Creek/Arroyo Viejo in San Leandro Bay, Palo Alto Creek, Redwood Creek, and Stevens Creek), which were selected based on the high urban land use in the tributary watershed and accessibility for sampling ($n = 27$). Open water sites were selected from the closest existing sites to the near-field sites sampled independently by USGS, and were sampled within three weeks after the storm event ($n = 24$).

The predominant PFAS in Bay water samples were PFPeA, PFHxA, PFOS, and PFOA (Table 4.3; Méndez, Miller, et al., 2025). While only PFHxA was widely observed in 2023 dry season samples, four analytes (PFPeA, PFHxA, PFOA, and PFOS) were found in more than 60% of samples in 2021. 11 analytes were detected in dry season open Bay water samples. The median concentration of PFHxA and sum of PFAS was slightly lower in 2023 (PFHxA: 0.6 and sum of PFAS: 2 ng/L) compared to 2021 (PFHxA: 1.5 and sum of PFAS: 5.9 ng/L). Overall, the highest levels of PFAS were observed in South and Lower South Bay, leading to detections of the greatest diversity of PFAS in this subregion (Table 4.3).

Wet season near-field median levels of PFPeA, PFHxA, PFOA, PFOS, and sum of PFAS are comparable to median levels in stormwater (and wastewater effluent), indicating influence of stormwater as expected (Table 4.3). Wet season monitoring at near-field sites showed more frequent detection of PFAS (16 analytes) and at higher levels compared to dry season monitoring at open Bay sites, although the wet season near-field dataset is too small to conduct statistical analysis (Méndez, Miller, et al., 2025). PFHxA, PFPeA, PFOA, and PFOS were detected in all near-field samples, and median levels were generally above those found in open Bay water sites during both dry and wet seasons (Table 4.3). PFHpA, PFBS, and PFHxS were closely behind with detection frequencies above 90% and median levels in the 2 ng/L range. PFBA was detected in 78% of samples with median concentrations of 3 ng/L. TOP analysis of a subset ($n = 11$) of wet season samples indicated limited presence of PFAA precursors.

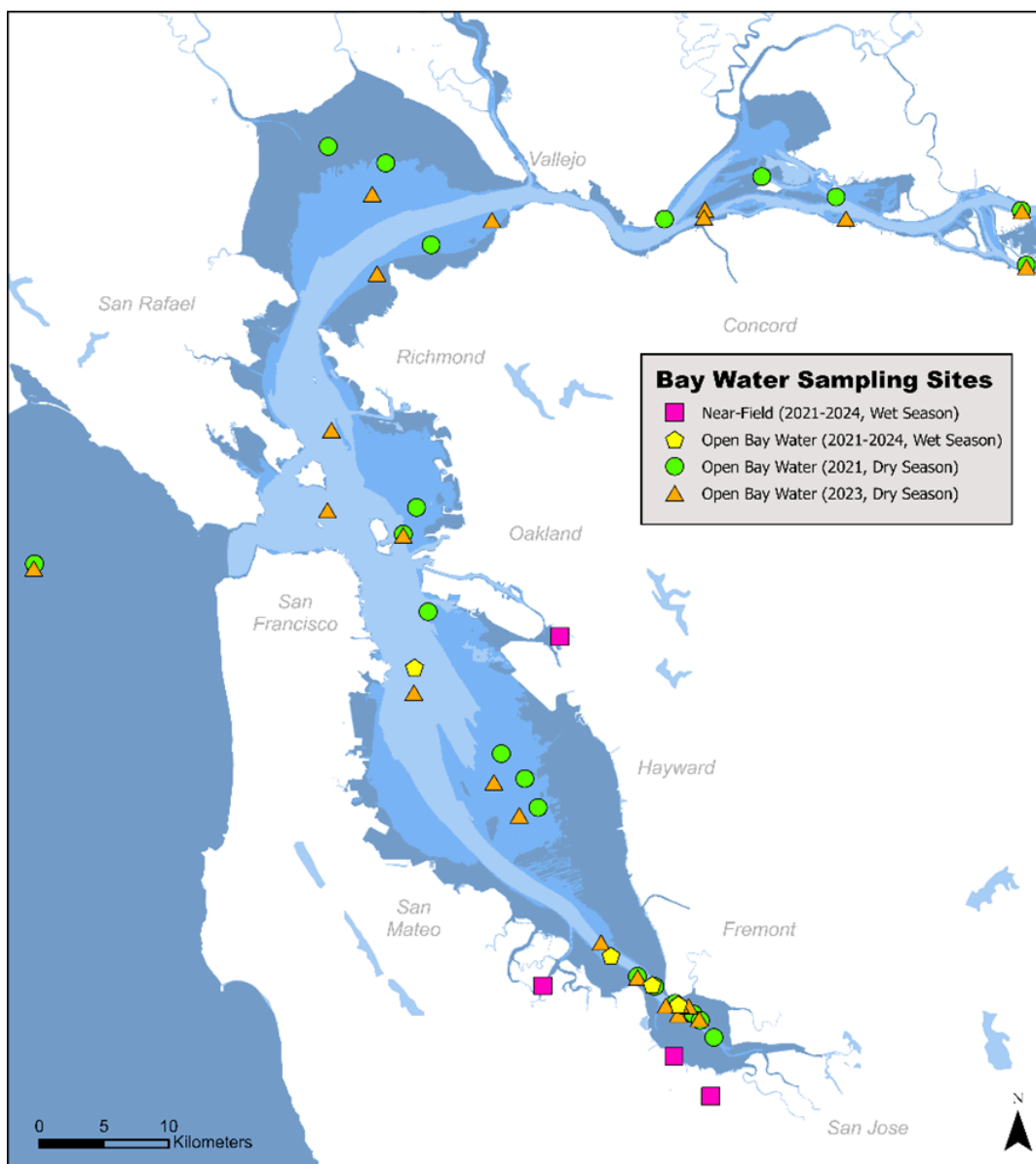


Figure 4.3. Bay water sampling locations from 2021–2024.

In the 2023 dry season samples, the TOP assay revealed sporadic, yet occasionally significant, increases in quantified PFAS (Méndez, Miller, et al., 2025). Seven out of 22 samples (32%) showed significant increases (>35%) ranging from 600% to 24,000%. Initial findings suggest that while South and Lower South Bays tend to exhibit higher concentrations of many PFAS, other subembayments may also contain smaller, localized areas with elevated levels of PFAS precursors. After oxidation, PFBA became the dominant form of PFAS detected in the samples, which suggests that the precursor compounds in the water may be different from those in the sediment, where PFOA exhibited the most significant change following oxidation. TOP analysis of a subset ($n = 11$) of wet season samples indicated limited presence of PFAS precursors.

Table 4.3. Median concentrations of PFPeA, PFHxA, PFOA, and PFOS in samples from 2021 and 2023 during the dry season (open Bay, n = 44) and from fall 2021 to spring 2024 for the wet season (near-field, n = 25; deep Bay, n = 24). South and Lower South Bay samples from 2023 (n = 18) were separated to compare with the wet season where most sites were in these two subembayments. Sum of PFAS (quantified PFAS with EPA method 1633) calculated with nondetects treated as zeros. Median concentrations measured in municipal wastewater effluent in 2020 and 2022 and stormwater runoff samples from 2018–2025 are also included for comparison.

PFAS	Median (ng/L)						
	MDL	2021 & 2023 Dry Season Open Bay Water (All Sites)	2021 & 2023 Dry Season Open Bay Water (South and Lower South Bays)	2021– 2024 Wet Season Open Bay	2021– 2024 Wet Season Near-field	2020 & 2022 Effluent	2018– 2025 Storm- water
PFPeA	0.96	1.1	2.5	2.0	4.5	7.7	5.7
PFHxA	0.5	1.1	2.8	2.7	5.3	16	7.4
PFOA	0.5	0.7	2.0	2.1	3.8	6.0	6.9
PFOS	0.5	0.6	3.1	2.7	4.6	4.5	6.6
sumPFAS	NA	4.3	18	15	32	57	46

Comparison to Other Regions

Observations in estuarine and coastal waters indicate that short-chain PFCAs, PFOS, and PFOA predominate globally, with concentrations similar to or higher than in the Bay ([Appendix Table B2](#)). In US surface waters, several Florida studies found total PFAS levels aligned with the Bay including Pensacola Bay in 2020 (da Silva et al., 2022), Biscayne Bay in 2022 (Ogunbiyi et al., 2024), and a statewide study in 2022 (Griffin et al., 2023). Pensacola's higher levels are attributed to major Naval contamination sources. Alabama estuarine and coastal studies, including Perdido Bay in 2016 (Mulabagal et al., 2018) and Mobile Bay in 2020 (Viticoski et al., 2022), showed concentrations generally on the lower end of the ranges detected in the Bay. A review that compiled and analyzed all peer reviewed literature to 2020 on PFAS in coastal marine water and open ocean waters found seven PFASs and nine PFCAs comprising ~83% of total measurements (Muir & Miaz, 2021). Highest median concentrations occurred near China and Europe, though data scarcity in the Americas limited analysis. Western US waters (Puget Sound and SF Bay) showed relatively high mean total PFASs from 2010 to 2019 (3–6 ng/L), though RMP 2023 levels were generally on the lower end.

Some studies identified novel PFAS not yet measured in Bay water. In Pensacola Bay, PFPrS and 6:2 diPAP appeared in over a third of samples (<6 ng/L; da Silva et al., 2022), alongside FASA subclasses PFBSA and PFHxSA (up to 16 ng/L). Global surface

waters (mostly freshwaters) exhibit wide ranges (tens to millions of ng/L) of ultra-short chain PFAS, particularly TFA (Arp et al., 2024; Ma et al., 2025; Zhi et al., 2024).

Surface water TOP assay studies showed unknown PFAA precursor concentrations comparable to or exceeding the Bay. German drinking water sources showed minimal post-TOP increases (Neuwald et al., 2022). Conversely, studies of European and Chinese freshwaters have reported consistent oxidation increases up to 5,259% (Foddy et al., 2024; Joeress et al., 2022), likely due to active PFAS sources near sampling sites.

PFAS in Water May Pose Risk

The SF Bay Regional Water Quality Control Board has developed environmental screening levels (ESLs) for the protection of human health via seafood ingestion for PFOA, PFBS, and PFOS of 0.0036 ng/L, 500 ng/L, and 0.07 ng/L, respectively (SFBRWQCB, 2020, 2025). These thresholds do not constitute policy or regulation, and their use is optional. Bay waters far exceed these PFOA and PFOS screening levels, consistent with fish tissue exceedances of human health thresholds (see section 5.1). Of note, these PFOA and PFOS screening levels are below method detection limits of 0.5 ng/L in Bay water (Méndez, Miller, et al., 2025). Other states have also developed similar guidelines that are exceeded by San Francisco Bay water concentrations; for example, Minnesota has a water quality criterion for the protection of fish consumption and recreational exposure surface water uses for PFOS of 0.05 ng/L (Preimesberger, 2020). Similarly, the USEPA has developed draft Human Health Criteria (HHC) recommendations for the same three PFAS, designed to protect human health from adverse effects associated with the ingestion of water, fish, and shellfish from inland and coastal waterbodies (USEPA, 2024a, 2024b). For each PFAS, EPA derived both a “water + organism” HHC, based on the consumption of both water and fish and/or shellfish and an organism-only HHC, which is based on the consumption of fish and/or shellfish alone. The organism-only HHCs are more relevant to the RMP, as the Bay is not a drinking water source, and are the same as the SF Bay Regional Water Quality Control Board ESLs (PFOA: 0.0036 ng/L, PFBS: 500 ng/L, and PFOS: 0.07 ng/L).

The SF Bay Regional Water Quality Control Board has also developed ecotoxicity ESLs for 13 PFAS in freshwater and 9 PFAS in saltwater (SFBRWQCB, 2020, 2025). These thresholds are generally much higher than those for the protection of human health, from 17 ng/L for PFOS to 540,000 ng/L for PFHxA. The 90th percentile near-field Bay water concentrations (13 ng/L) are approaching the PFOS fresh and saltwater ESLs.

There is evidence that many individual PFAS act through the same mechanism(s) of action to cause toxicity in wildlife and humans. Thresholds for individual PFAS may therefore underestimate cumulative PFAS risks. The European Union has developed environmental quality standards for the protection of both human health and aquatic organisms for the weighted sum of 25 PFAS, expressed as PFOA-equivalents based on the potencies of each PFAS relative to that of PFOA. The environmental quality standard for the weighted sum of PFAS is 4.4 ng/L for both fresh and marine waters. Bay waters exceed this threshold, with a 90th percentile weighted sum for all sites of 43 ng/L, and even the median weighted sum for all sites at 8.7 ng/L, indicating that together, these 25 PFAS may pose adverse risk.

5 PFAS in San Francisco Bay Biota

The widespread presence of PFAS throughout Bay surface water and sediment warrants robust monitoring of Bay biota to understand potential impacts on the Bay ecosystem. PFAS have been shown to biomagnify and may adversely affect higher-order predators within an ecosystem. The RMP regularly monitors PFAS levels and trends in sport fish and cormorant eggs through the Status and Trends (S&T) program. The RMP has also monitored bivalves, prey fish, and harbor seals through more opportunistic studies.

An important consideration in interpreting results from studies across years is that PFAS analytical methods continue to improve, leading to greater ability to measure and quantify PFAS in tissue. The 2014 RMP analysis of PFAS in biota included 13 PFAS. Methods continued to advance, with the target analyte list expanding to 33 in 2019. In addition, reporting limits have also become more sensitive over time across all analytes. For example, the reporting limit for PFOS decreased from 1 ng/g (2014) to 0.2 ng/g (2019) to 0.1 ng/g (2022).

5.1 Fish

As sensitive indicators of exposure to aquatic or sediment-bound contaminants that relate to both wildlife and human health, fish are key biota to monitor throughout the Bay. Fish exposure to contaminants can occur via ingestion, aquatic respiration, and regulation of osmotic pressure. Contaminants can have negative effects on fish and can biomagnify to impact higher trophic level species that consume them such as striped bass, seals, birds, and humans.

Concern for impacts to wildlife and human health has led the RMP to monitor sport fish and prey fish for a variety of contaminants, including PFAS. Sport fish are defined as those with lengths greater than 100 millimeters that are typically caught and consumed by fishers in the Bay. Sport fish are primarily monitored to evaluate human exposure from ingestion of these fish. Fish with lengths below 100 millimeters are considered prey fish, which are monitored to evaluate spatial and temporal trends in contaminant exposure and potential impacts across the Bay food web. San Francisco Bay prey fish are also believed to have high site fidelity, with home ranges of 10 kilometers or less (Melwani et al., 2012), making them suitable for assessing spatial trends.

5.1.2 Sport Fish

Beginning in 2009, sport fish have been monitored for PFAS every five years at various stations throughout the Bay (Figure 5.1; (Buzby et al., 2021; Davis et al., 2011; Méndez, Davis, et al., 2025; J. Sun et al., 2017). Five stations are popular fishing locations (San Pablo Bay, Berkeley, Oakland, San Francisco, Redwood Creek), although San Pablo Bay was not measured in 2019. The Central Bay station was sampled in 2009 and 2024 for target species found primarily in deeper waters. The remaining eight stations (Artesian Slough, Carquinez Strait, Coyote Creek, Richmond Harbor, Hunter's Point A & B, San Leandro Bay, and Suisun Bay) represent areas of interest to more fully understand contaminant concentrations across the Bay. Suisun Bay and Carquinez

Strait capture additional regions in the North Bay, while the remaining sites were selected for fish monitoring due to proximity to contaminated sites with elevated mercury or PCB concentrations. Some of these sites are also in proximity to known PFAS pathways, including wastewater effluent outflows and stormwater runoff.



Figure 5.1. Sport fish sampling locations in San Francisco Bay since 2009.

This synthesis focuses on sport fish data collected between 2014 and 2024, including samples analyzed using an updated analytical method consistent with the EPA standard method (Table 5.1). Samples analyzed in 2009 and 2014 were analyzed using older analytical methods, with higher detection limits in 2009 and many fewer analytes (Mendez et al. 2025). Samples ($n = 12$) from five fish species were collected and analyzed for 33 PFAS in 2019. Archived samples from 2014 ($n = 27$) of four species and 2019 ($n = 21$) of two species were also analyzed in 2022 using a similar method to the method used in 2019, but with an expanded list of 40 analytes. In 2024, sport fish monitoring was expanded to include 12 species ($n = 67$), measured using the same analytical method used in 2022. To ensure the comparability of reported data, temporal trends include data analyzed in 2019 or later.

Table 5.1. Summary of recent sportfish studies.

PFAS Study	N	Species	# of PFAS Analytes
2019 Study	12	Largemouth bass (<i>Micropterus salmoides</i>) Shiner surfperch (<i>Cymatogaster aggregata</i>) Striped bass (<i>Morone saxatilis</i>) White croaker (<i>Genyonemus lineatus</i>) White sturgeon (<i>Acipenser transmontanus</i>)	33
2022 Study of Archived Samples	27 (2014) 21 (2019)	4 (2014), 2 (2019) Pacific staghorn sculpin (<i>Leptocottus armatus</i>) Shiner surfperch (both years) Striped bass White croaker (both years)	40
2024 Study	67	Bat ray (<i>Myliobatis californica</i>) California halibut (<i>Paralichthys californicus</i>) Diamond turbot (<i>Hypsopsetta guttulata</i>) Jacksmelt (<i>Atherinopsis californiensis</i>) Largemouth bass Monkeyface prickleback (<i>Cebidichthys violaceus</i>) Northern anchovy (<i>Engraulis mordax</i>) Pacific staghorn sculpin Shiner surfperch Striped bass Walleye surfperch (<i>Hyperprosopon argenteum</i>) White croaker	40

In general, fish were dissected skin-off, and only the fillet muscle tissue was used for analysis, representing the section of the fish commonly consumed by many people. Like most other sport fish sampled, white croakers were prepared as skinless fillets, except in 2014 when they were processed whole with head, tail, and viscera removed (which becomes important for data interpretation). Pacific staghorn sculpin, northern anchovy, and shiner surfperch were too small to be filleted and were processed whole, but with head, tail, and viscera removed. Fish samples were analyzed as individuals for common carp. For all other species, composites were prepared by homogenization of equally weighted aliquots from each fish by station, species, and size class.

PFOS Predominant PFAS Measured in Sport Fish

PFOS was the predominant PFAS found in Bay sport fish, and was detected in ≥90% of samples between 2014 and 2024. In 2024, detected concentrations of PFOS ranged from 0.21 to 8.1 ng/g ww, with an average concentration of 1.9 ng/g ww. Variations in PFAS concentration among species are likely attributable to a combination of factors, including fish species (e.g., life history, feeding habits, trophic level), sample preparation method, and sampling location. Although not all PFAS bioaccumulate or biomagnify, many long-chain PFAS have been observed to biomagnify in fish (Miranda et al. 2022), contributing to higher concentrations in higher trophic level species.

In 2024, the highest average PFOS concentrations were observed in largemouth bass and striped bass, which are both relatively high trophic level species and the only species sampled in the Artesian Slough (Figure 5.2). Elsewhere in South Bay (Redwood Creek), PFOS in a sample of another higher trophic level species, bat ray, was also notably higher than concentrations observed in South Bay shiner surfperch (2.8–3.5 ng/g ww).

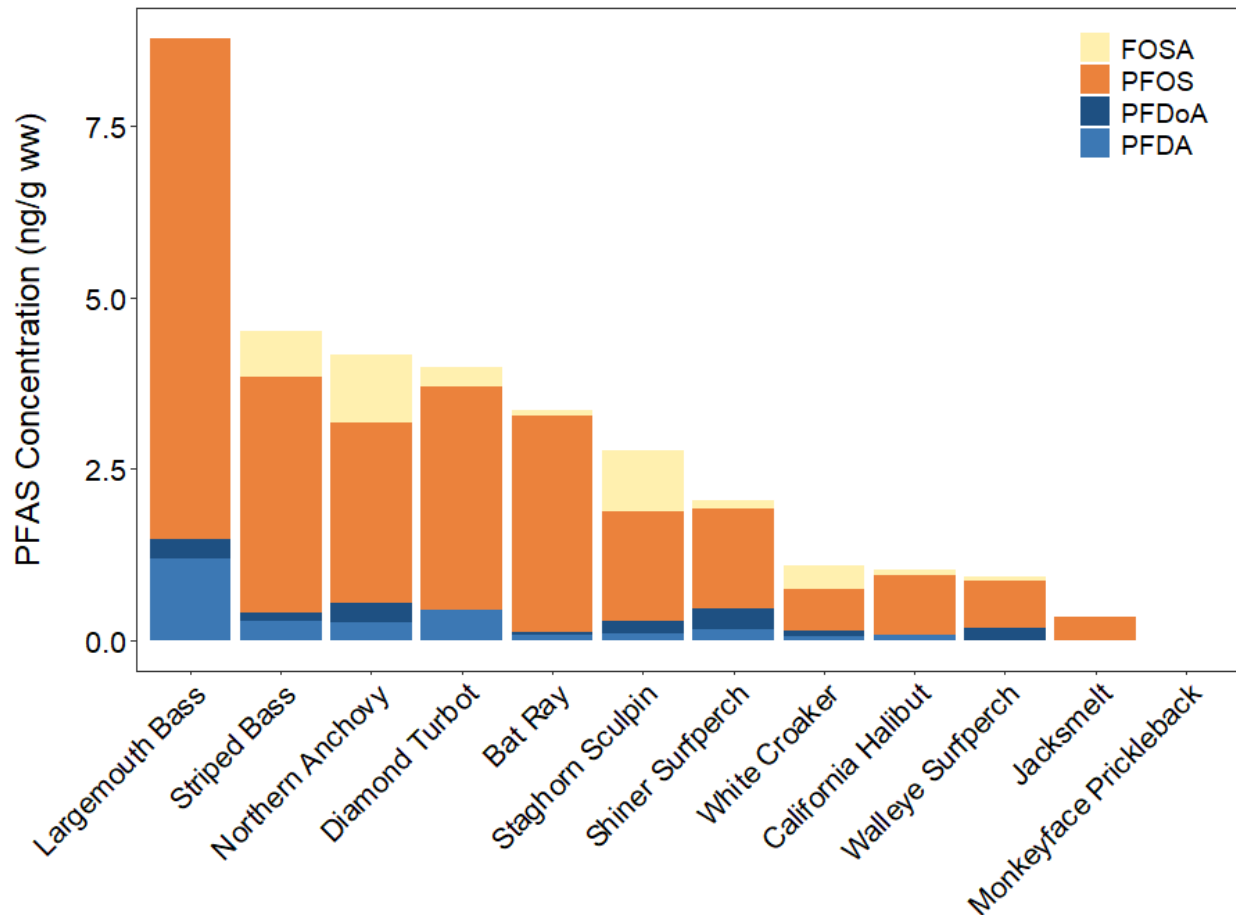


Figure 5.2. Average PFAS concentration and composition for sport fish monitored in 2024. Only PFAS detected in > 50% of fish samples in 2024 are shown. Concentrations for each PFAS were averaged within species, with non-detect values (below method detection limits) treated as zeros. Only one monkeyface prickleback sample was measured, and no PFAS were detected.

The next highest average PFOS concentrations (aside from a single sample of diamond turbot) were observed in northern anchovy and staghorn sculpin, which are small fish that are processed whole, followed by shiner surfperch, which are processed whole with no head, tail, or guts (Figure 5.2). The largest number of unique PFAS were also detected in these smaller species (13 in northern anchovy and staghorn sculpin, 17 in shiner surfperch). In contrast to mercury and other persistent organic pollutants, PFAS tend to bind to binding proteins, and preferentially accumulate in tissues like blood, liver, and kidneys rather than muscle or lipids. This can result in much higher

concentrations in whole body fish tissue samples compared to fillets. Fish species that are commonly consumed whole, and processed whole by the RMP, tend to be small low-trophic level species.

Overall, 21 total PFAS were observed in Bay sport fish in 2024. Beyond PFOS, three other PFAS were detected in >50% of samples, including the PFOS precursor PFOSA (average: 0.35 ng/g ww, max: 1.35 ng/g ww) and two long-chain PFCAs, PFDA (average: 0.18 ng/g ww, max: 1.24 ng/g ww) and PFDoA (average: 0.20 ng/g ww, max: 0.83 ng/g ww). Other long-chain PFCAs (e.g., PFOA, PFNA, PFUnA, PFTrDA, PFTeDA, PFDS), as well as PFHxS, were regularly detected (20-45%) across species and locations. In comparison, short-chain PFCAs (e.g., PFPeS, PFHxA, PFHpA) and precursors (e.g., 6:3 FTCA, 7:3 FTCA, 6:2 FTS, N-EtFOSAA, N-EtFOSE, N-MeFOSA) were detected sporadically (<10%). Ether PFAS, some of which were produced as replacements for legacy PFAS and have been detected in other tissue matrices, were not detected in any samples.

Two PFCA precursors that were added to the analytical method in 2022 were detected at notable levels in several samples. In 2024, 6:2 FTS was detected for the first time at unusually high concentrations in five fish samples (0.52–66.6 ng/g ww), with concentrations above 30 ng/g ww in three samples from near-shore priority margin unit sites (Richmond Harbor and Hunter's Point B). 6:2 FTS has previously been detected in sediment, and fish detections were found only in shiner surfperch and staghorn sculpin, which feed at least partially in benthic environments. However, it is unusual to find such high concentrations of 6:2 FTS in fish tissues, as this compound is known to be readily metabolized. Continued monitoring is needed to evaluate whether these detections are anomalous events or indicative of ongoing exposure risks.

Additionally, 7:3 fluorotelomer carboxylic acid (7:3 FTCA) has also been regularly detected in samples collected since 2014, though the detection frequency in 2024 was fairly low (6% of samples). Although detected concentrations were relatively high (3.18–5.07 ng/g ww) compared to other PFAS, including PFOS, QA/QC results suggest these concentrations may be overestimated (matrix spike recovery 191%). Additionally, 7:3 FTCA has only been detected in white croaker and shiner surfperch processed as whole fish (with head, tail and viscera removed) rather than as fillets (e.g., white croaker samples processed as fillets in 2019 and 2024 showed no detections). Detections were observed in five shiner surfperch samples in 2024, and only at the San Francisco Waterfront and South Bay, in contrast with additional detections in North Bay and Oakland observed in 2014. Detections of 7:3 FTCA in freshwater fish have been linked to its role as a transformation product of 8:2 fluorotelomer alcohol and fluorotelomer acrylates (Butt et al., 2013; Eriksson et al., 2017), which are long-chain legacy PFAS precursors. Continued monitoring will help indicate whether detections will decrease, or whether current levels reflect ongoing residual environmental releases of these legacy compounds.

Several other precursors have also been sporadically detected. In 2024, 5:3 FTCA was detected for the first time along with 7:3 FTCA in three shiner surfperch samples in South Bay (2.7–3.4 ng/g ww), both at concentrations higher than PFOS. Several PFOS precursors have also been variably detected in species and locations throughout the

Bay, not concurrently, at relatively low concentrations (in 2024: N-EtFOSE detected in six samples, 1.01–2.17 ng/g ww; N-EtFOSAA detected in three samples, 0.1–0.16 ng/g ww; N-MeFOSA detected in one sample, 0.27 ng/g ww).

Higher PFAS Concentrations Observed in South Bay

Fish caught in South Bay generally had the highest PFAS concentrations compared to fish caught in other regions of the Bay (Figure 5.3). This pattern remained consistent in all monitoring years and among most species examined, particularly for PFOS and its precursor FOSA. In shiner surfperch collected in 2024, PFOS concentrations in South Bay (Redwood Creek) samples were significantly higher than all other regions except San Leandro Bay (ANOVA with Tukey HSD, $p=0.05$). Similar patterns were also observed in other species measured in South Bay, Lower South Bay or Artesian Slough in 2024, with the 5 highest concentration samples occurring in high trophic level species in these regions. The highest concentrations of striped bass and bat rays were observed in the Lower South Bay or South Bay (3.78–8.02 ng/g ww) compared to other regions of the Bay (0.5–3.6 ng/g ww). Largemouth bass, which are resident in the freshwater Artesian Slough that discharges into Lower South Bay, had the highest average PFOS concentration among species (average=7.30 ng/g ww; range 7.07–7.54 ng/g ww). These findings are in line with higher concentrations of PFAS in other biota as well as in surface water and sediment in South and Lower South Bays compared to the rest of the Bay.

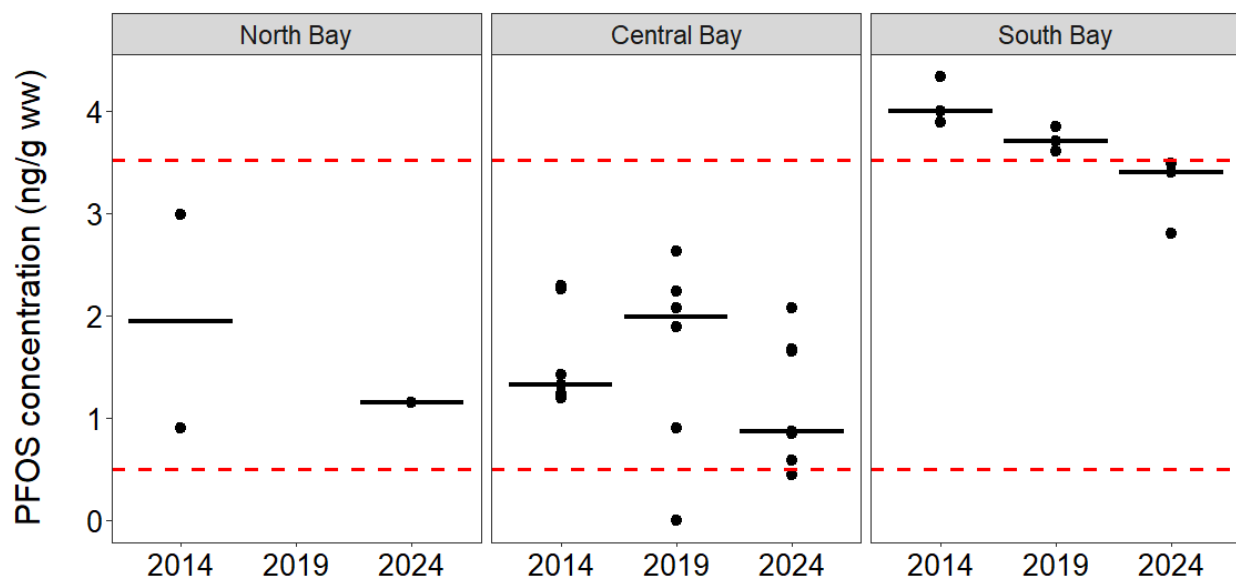


Figure 5.3. Comparison of concentrations of PFOS in shiner surfperch caught in North Bay (San Pablo Bay station), Central Bay (Berkeley, Oakland, and San Francisco Waterfront stations), and South Bay (Redwood Creek) between 2014 and 2024. The bars shown represent the median concentration (non-detect = 0). Samples only include those analyzed in 2019 or later, using the modern analytical method. The red dashed lines represent the Massachusetts one meal per day (0.5 nanograms/gram wet weight) and one meal per week (3.52 ng/g ww) consumption advisories for the general population.

Beyond PFOS, the largest number of PFAS were detected in at least one sample in South Bay (n=17), followed by Central Bay (n=16). The 5:3 and 7:3 fluorotelomer carboxylic acids (FTCA) were almost exclusively detected in South Bay shiner surfperch, with a single additional detection of 7:3 FTCA in a shiner surfperch sample from the San Francisco Waterfront site. 6:2 FTS and N-EtFOSE were also sporadically detected in South Bay.

Temporal Trends Vary across Subembayment

Temporal trends in fish PFAS concentrations are largely driven by trends in PFOS, and can be most clearly observed in shiner surfperch, which have been most consistently measured across multiple locations over time and are known to have high site fidelity. In South Bay, PFOS concentrations in shiner surfperch have decreased from a median value of 4.0 ng/g ww (3.9–4.3 ng/g ww) in 2014 to 3.4 ng/g ww (2.8–3.5 ng/g ww) in 2024 (ANOVA with Tukey HSD, $p < 0.05$). Evidence of similar decreases is not clear in Central Bay, and current data are not sufficient to evaluate trends in North Bay (Figure 5.3). This variation in trends among subembayments is similar to the pattern observed in Bay bird eggs, although the rate of decrease in PFOS concentrations in South Bay is much smaller in fish.

Similar decreases in shiner surfperch in South Bay are also observed in the PFOS precursor, FOSA ([Appendix Figure C1](#)). In contrast, no evidence of decreasing concentrations is observed among long-chain PFCAs in Bay shiner surfperch ([Appendix Figures C2-C3](#)). The difference in trends among PFOS and other long-chain PFCAs observed in South Bay could potentially reflect differences in the production phaseout and use patterns of PFSA and PFCA compounds and their precursors. However, concentrations of long-chain PFCAs are generally low and longer-term datasets are needed for more robust evaluation of trends.

Comparison to Other Studies and Regions

Global studies of PFAS in marine and freshwater fish tissue typically identify PFOS as the dominant contaminant, with concentrations typically ranging from a few ng/g ww up to over 200 ng/g ww ([Appendix Table B3](#)). In San Francisco Bay fish, PFOS levels have generally been comparable to or lower than those reported in other US studies. In marine fish from Charleston Harbor, South Carolina (Fair et al., 2019), PFOS concentrations were similar to San Francisco Bay fish during the same sample time period. Marine fish from Rhode Island from 2006 to 2014 were noticeably different from most studies, as PFOS was not the dominant PFAS detected and was generally found at low concentrations (Hedgspeth et al., 2023). The concentrations of PFOS in Tampa Bay, Florida (2021–2022) were generally in a higher range from those in 2024 San Francisco Bay fish. Similarly, freshwater fish in the U.S. generally had PFOS concentrations that ranged higher relative to the Bay, including the US Great Lakes in 2015 (Barbo et al., 2023), US rivers and streams (encompassing 48 states) from 2013 to 2015 (Barbo et al., 2023), New Jersey in 2015–2016 (Goodrow et al., 2020), North Carolina in 2020 (Boatman et al., 2024), and the New Hampshire Merrimack rivershed in 2019 (Pickard et al., 2022).

Likewise, many of the long-chain PFCAs and other detected PFAS have been reported in studies of freshwater and marine fish at low concentrations in the US and internationally. Currently reported concentrations of 7:3 FTCA are also similar to those found in the few available studies of freshwater fish around Michigan, eastern Canada, and Hong Kong (George et al., 2023; Kaboré et al., 2022; Loi et al., 2011).

PFAS in Bay Fish Above Some Fish Consumption Advisory Thresholds, Indicating Human Health Risk

Although California does not yet have fish consumption thresholds for PFAS, 14 other states have issued fish consumption guidelines for PFOS and several other PFAS (Hughes & Henderson, 2025). State fish consumption guidelines vary dramatically due to differences in reference dose values used to determine the guidelines, which have generally declined over time as more toxicity data have indicated adverse effects at lower exposure levels. Without California-specific values, the RMP uses the lowest (most protective) human health thresholds from other states. Massachusetts has developed the most stringent fish consumption advisories in the US, with PFOS thresholds of ≤ 0.22 ng/g for sensitive populations (children 12 or younger and people who are nursing, pregnant, or may become pregnant) and ≤ 0.50 ng/g for the general population (children older than 12 and most adults) for one eight-ounce serving per day, and ≤ 1.56 ng/g and ≤ 3.52 ng/g for one serving per week for sensitive and general populations respectively (MA DPH, 2023). In 2024, 9% of Bay fish samples exceeded the general one meal per week threshold, including samples from 6 out of the 12 species sampled and all 5 samples collected from the Artesian Slough (largemouth and striped bass). 87% of samples exceeded the general one meal per day threshold, including samples from 10 out of 12 species, as well as all samples from San Pablo Bay, Oakland, South Bay (including the Artesian Slough), and all priority margin unit sites. All but one sample (California Halibut from Central Bay) exceeded the sensitive population one meal per day threshold ([Appendix Figure C4](#)).

One meal per day advisory thresholds from Massachusetts for the sensitive or general populations (shown respectively) for PFBA (≤ 111 ng/g; ≤ 250 ng/g), PFOA (≤ 0.33 ng/g; ≤ 0.75 ng/g), PFNA (≤ 0.33 ng/g; ≤ 0.75 ng/g), PFBS (≤ 33 ng/g; ≤ 75 ng/g), and PFHxS (≤ 2.22 ng/g; ≤ 5.01 ng/g) were generally not exceeded in 2024 samples. Exceptions included three exceedances of the sensitive population threshold for PFOA (0.34 ng/g ww) and two exceedances of the sensitive population threshold and one exceedance of the general population threshold for PFNA (0.36–0.97 ng/g). No one meal per week thresholds were exceeded.

PFAS in Bay Fish Above Some Ecotoxicity Thresholds, Indicating Ecological Risk

In addition to potential human health impacts, Bay sport fish PFAS concentrations may also pose risk to marine mammals such as harbor seals and California sea lions that prey on them. Five Bay sport fish samples (7.5%), all sampled from the Artesian Slough and South Bay, exceeded the Canadian Federal Environmental Quality Guideline for PFOS in wildlife diet of 4.6 ng/g ww (ECCC, 2018).

5.1.3 Prey Fish

In contrast to sport fish, the RMP has not conducted consistent monitoring of prey fish for contaminants. The first study was conducted in 2009; with an additional study that ran from 2012 to 2013 (Sedlak et al., 2017; Sedlak & Greig, 2012). The latter effort targeted five sites near cormorant nesting sites and seal haul-out sites in Central, South, and Lower South Bays. Fish were opportunistically collected by the California Department of Fish and Wildlife from July 2012 through April 2013. Twenty-four composites consisting of yellowfin gobies (*Acanthogobius flavimanus*), chameleon/cheekspot gobies (*Tridentiger trigonocephalus/Illypnus gilbert*), northern anchovy, shiner surfperch, and Pacific staghorn sculpin were analyzed for 13 PFAS.

In 2023, prey fish were included in the RMP S&T biota monitoring program to pilot the feasibility and utility of routine monitoring. Prey fish were sampled at 12 stations that overlap with or span near-field sediment stations sampled in 2023 (Figure 5.4). Thirty-five composites of topsmelt (*Atherinops affinis*) and four composites of staghorn sculpin were analyzed for 40 PFAS.

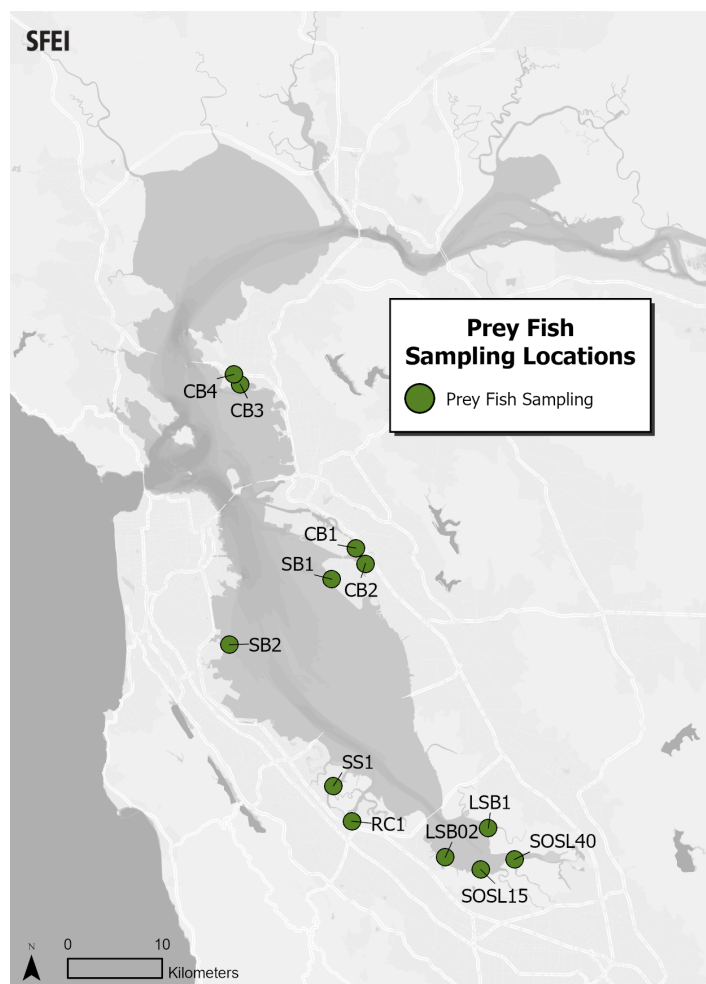


Figure 5.4. Sites sampled for prey fish in 2023.

Prey fish under 100 mm were processed whole as they are too small to be filleted, while any that are larger had their head, tail, and viscera removed. Composites were prepared by homogenization by station, species, and size class, varying between one individual to as many as 30 individuals.

PFOS Detected Predominantly, Decreases in Several PFAS Observed Between 2012 and 2023

PFOS was the major PFAS detected in all prey fish monitored since 2012, with a detectable level in nearly all samples (except one in 2012). PFOS concentrations in composites ranged from non-detect ($MDL \leq 2$ ng/g ww) to 241 ng/g ww in 2012/2013 and from 0.3 ng/g ww to 10 ng/g ww in 2023. The median concentrations across all fish show notable declines from 2012/2013 (11 ng/g ww) to 2023 (1.5 ng/g ww). Further, the average concentrations of PFOS in Pacific staghorn sculpin, sampled during both periods and found to have the highest concentrations in the first study, decreased markedly across this period from 77 ng/g ww (2012/2013) to 5 ng/g ww (2023).

Across both sampling periods, seven PFAS were also detected beyond PFOS. PFOSA was the second most detected compound (detection frequency >90%) across both sampling years, with median concentrations an order of magnitude higher in 2012/2013 (2.4 ng/g ww) than in 2023 (0.25 ng/g ww). Two long-chain PFCAs (PFDA and PFDoA) were detected in >60% of samples during both monitoring years, with median concentrations also generally lower in 2023 compared to 2012/2013. PFNA, another long-chain PFCAs, decreased in detection frequency from 55% in 2012/2013 to 26% in 2023, with concentrations also lower. PFOA and PFHxS were more sporadically detected in both sample years, though the range of detections was notably higher in 2012/2013 (detected range 0.5–15 ng/g ww) than in 2023 (<0.43 ng/g ww). PFUnDA was the only analyte with increased detection frequency in 2023 (82%) compared to 2012/2013 (29%), though concentrations were generally low (<1.2 ng/g ww).

The expanded method in 2023 allowed detection of eight additional PFAS. PFTeDA and PFDS were commonly detected (>66% of samples) at relatively low concentrations (<0.60 ng/g ww). Other PFAS detected sporadically at low levels in 2023 include PFTTrDA, a few short-chain PFCAs (PFBA, PFHxA) and multiple PFAS precursors (6:2 FTS, N-MeFOSAA, N-EtFOSAA). Newer PFAS such as HFPO-DA (GenX) and ADONA were not detected in any prey fish.

Higher PFAS Concentrations Observed in Lower South Bay

In the 2023 sampling, the highest concentrations of both PFOS and the sum of all PFAS detected were observed in fish caught in Lower South Bay compared to South and Central Bay samples (Figure 5.5). The median concentration (3.4 ng/g ww) in Lower South Bay samples was over three times greater than in Central Bay (0.8 ng/g ww) and South Bay samples (0.7 ng/g ww).

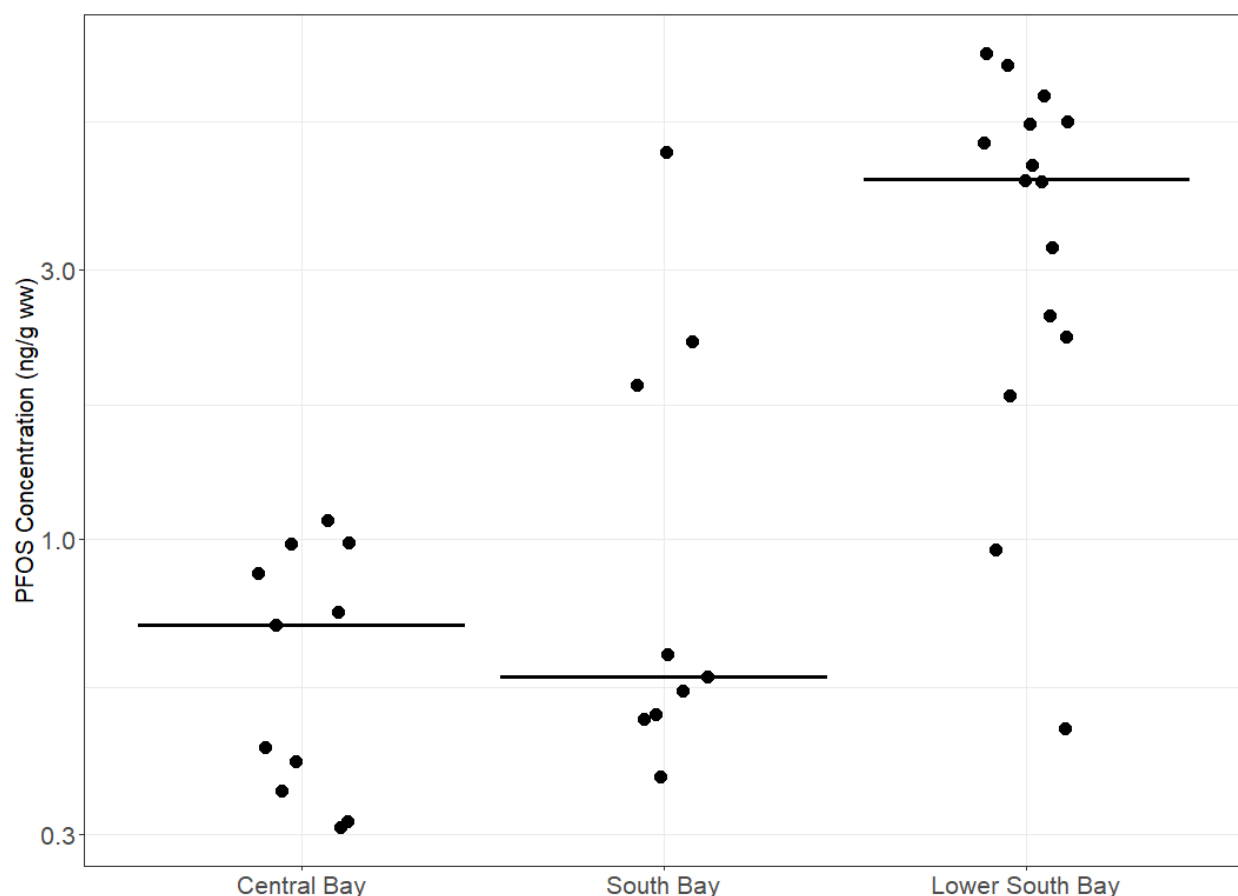


Figure 5.5. Comparison of concentrations of PFOS in top smelt prey fish from Central Bay, South Bay, and Lower South Bay in 2023. The bars represent the median concentration (where non-detect = 0).

Applying similar risk thresholds as used for sport fish risk screening, 24% of prey fish composites, all of which were from South and Lower South Bay, exceeded the Canadian Federal Environmental Quality Guideline for PFOS in wildlife diet of 4.6 ng/g ww (ECCC, 2018). Although prey fish (< 100 mm) are categorized differently from sport fish (> 100 mm) and are primarily tracked to assess ecological health and temporal trends rather than human exposure, local communities in the Bay area consume prey fish as well. Therefore, comparing prey fish PFAS levels against fish consumption advisory thresholds remain relevant. The vast majority (74%) of prey fish sample composites exceeded the Massachusetts general population threshold for one meal per day (≤ 0.5 ng/g; MA DPH, 2023); and 29% of samples exceeded the general one meal per week threshold (Figure 5.5). These threshold exceedances indicate a risk for both piscivorous birds and mammals and humans through consumption of PFAS-contaminated fish.

5.2 Bird Eggs

The RMP monitors double-crested cormorant (*Phalacrocorax auritus*) eggs to assess contaminant exposure from the open water habitats of the Bay. Cormorants are year-round avian residents that forage for food in shallow, open waters close to shore. Their feeding range can be quite large; they are known to forage in an approximately 20-mile radius from their nesting sites (Melwani et al. 2012). As such, they are excellent regional integrators of contaminants. Beginning in 2006, bird eggs have been monitored every two to four years for PFAS from three general Bay sites (Figure 5.6): the North Bay site in the Delta-influenced Suisun Bay at or near Wheeler Island, the Central Bay site on the Richmond Bridge, and the South Bay sites at Don Edwards Pond or Newark.



Figure 5.6. Sites sampled for bird eggs since 2006.

This synthesis focuses on cormorant egg data from S&T monitoring efforts in 2012, 2016, 2018, 2022, and 2024 (Sedlak et al., 2017, 2018). In most instances, three composites of seven eggs each were collected at each site. During 2018, no eggs were collected from the North Bay site at Wheeler Island. The 2022 samples were collected from neighboring Rich Island, approximately 1.5 miles west of the Wheeler Island colony.

Eggs from each site are weighed and randomly sorted into three composite samples with seven eggs each. During each monitoring effort, except for 2018, a total of nine composite samples were collected across the three sites. Analyses from 2012 to 2018 included 13 PFAS, with advancements in analytical methods expanding the list to 40 in 2022.

PFOS Predominant Across All Bay Regions

PFOS consistently dominates total PFAS found in Bay bird eggs, accounting for well over half of the sum of PFAS concentrations in 2024 (33–165 ng/g ww). Aside from PFOS, 7 additional PFCAs (PFOA, PFNA, PFDA, PFUnA, PFDoA, PFTrDA, PFTeDA) have been consistently detected between 2012–2024, although PFTrDA and PFTeDA have only been measured since 2022. Of these, PFDA (maximum = 13.8 ng/g ww) and PFDoA (maximum = 8.8 ng/g ww) are the next most abundant compounds after PFOS. Additionally, PFHxS, PFHpS, and PFDS were also detected in 100% of samples in 2022 and 2024, although concentrations were lower than other long-chain PFCAs. This PFAS composition is fairly consistent with those in sport fish, with PFOS and sum of PFCAs making up a majority of the PFAS quantified with EPA Method 1633 (sum of PFAS).

Two precursors were detected in most samples in 2024. 7:3 FTCA (PFCA precursor, including PFOA) was detected at relatively high levels across samples (10–28 ng/g ww), with similar concentration ranges across subembayments, although replicate precision in these samples was low. PFOSA (PFOS precursor) was detected in 7 of 9 samples at concentrations up to 0.25 ng/g ww, while 8:2 FTS, N-MeFOSAA and 9CI-9F3ONS were detected sporadically at low levels in either Central or Suisun Bay. In general, precursors are detected less frequently in fish: while PFOSA has been regularly detected across multiple species, 7:3 FTCA has been sporadically detected in shiner surfperch and white croaker, and 8:2 FTS, N-MeFOSAA, and 9CI-9F3ONS have not been detected.

Temporal PFAS Trends Vary Across Subembayments

PFAS temporal trends are strongly driven by PFOS, which exhibits contrasting patterns across subembayments. In South Bay, PFOS concentrations decreased from a peak concentration range of 570–654 ng/g ww in 2016 to 96–165 ng/g ww in 2024, with median concentrations decreasing by 79% (Kruskall-Wallis with Dunn's post-hoc test, $p < 0.05$). In Central Bay, PFOS concentrations in 2024 (33–63 ng/g ww) were also lower than concentrations in 2016 (78–152 ng/g ww). In comparison, concentrations in Suisun Bay have not changed substantially over time. Although a high outlier sample was observed in 2012 (185 ng/g ww), concentrations among other samples have not changed substantially between 2012 (36–88 ng/g ww) and 2024 (53–84 ng/g ww).

Temporal trends for other PFAAs are less clear, partially due to lower concentrations compared to PFOS and insufficient data for robust statistical analysis.

Higher PFAS Concentrations Observed in South Bay

Cormorant eggs in South Bay generally had the highest individual and sum of PFAS concentrations and the greatest variety of PFAAs detected compared to other regions in the Bay. PFNS and PFHpA were detected only in South Bay. PFOS has been particularly elevated in South Bay, although the relative difference in concentrations is declining as concentrations continue to decrease (Figure 5.7). In 2024, the average and median concentrations in samples from South Bay was nearly double the concentrations observed in samples from the Central and North Bay sites.

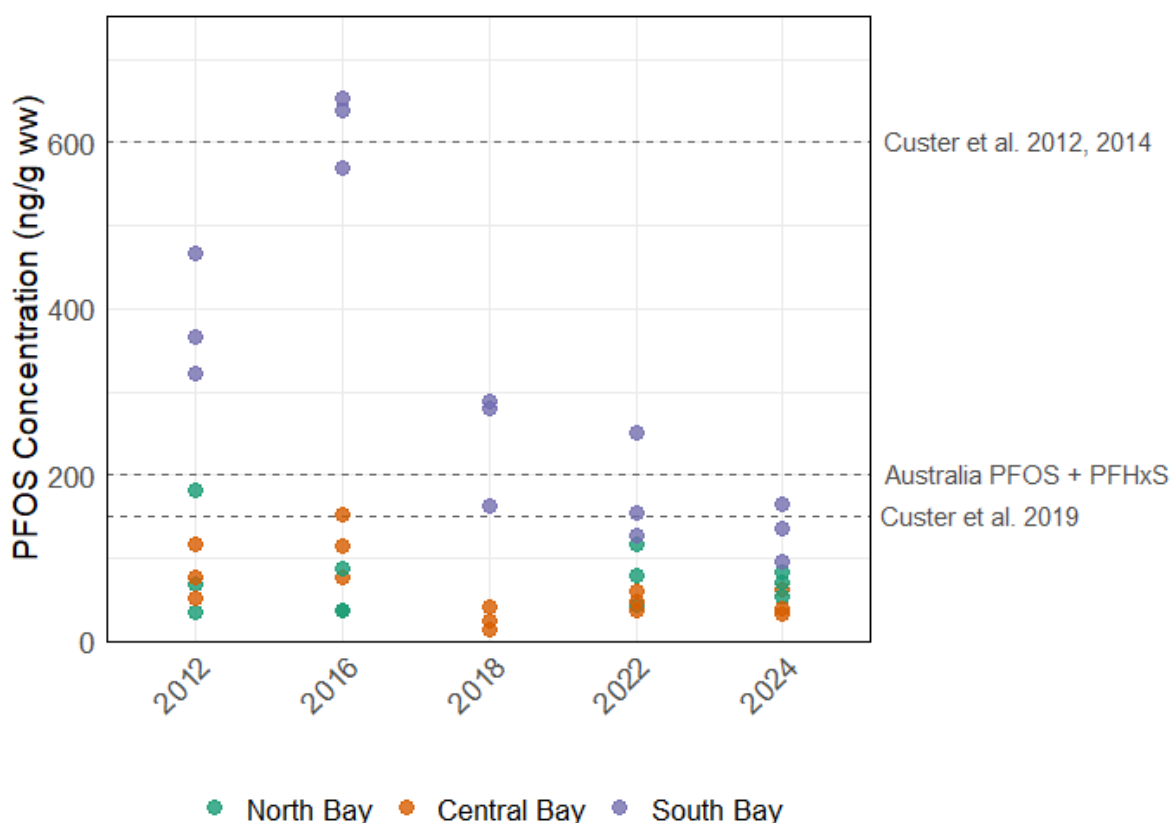


Figure 5.7. Concentrations of PFOS in composite cormorant eggs collected between 2012–2024. Horizontal lines represent health-based thresholds for bird eggs that have been linked to negative impacts in birds. The Custer et al. guidelines are non-regulatory thresholds, while the Australia value represents a guideline for ecological exposure of the combination of PFOS and PFHxS protective of birds. The Canadian Federal Environmental Quality Guideline for PFOS in bird eggs (1,900 ng/g ww) is much higher than concentrations observed in the Bay and is not shown.

Some precursors and alternative PFAS show contrasting spatial patterns, although overall detections and concentrations are low. For example, 6:2 FTS which was first analyzed in 2022, was high in all North Bay samples and at similar levels in South Bay samples, although 6:2 FTS was not detected at any sites in 2024. Other sporadically detected PFAS, including N-MeFOSAA (PFOS precursor), 8:2 FTS (PFCA precursor), and 9CI-PF3ONS (ether PFAS) have only been detected in Central or North Bay samples.

Comparison to Other Studies and Regions

PFOS is the primary PFAS measured in Bay bird eggs, with concentrations in Bay cormorant eggs greater than more rural areas but generally comparable to global detections in other birds ([Appendix Table B4](#)). Average PFOS levels were higher in other US areas including Great Lakes herring gulls (Letcher et al., 2015), Michigan Caspian terns (Su et al., 2017), and Baltic Sea common guillemots (de Wit et al., 2020; Gebbink et al., 2016) than in the Bay. New Hampshire common loons and South Carolina brown pelicans had PFOS levels similar to the Bay. Additionally, 7:3 FTCA levels were lower in Svalbard kittiwake eggs (Jouanneau et al., 2022) than in the Bay, but higher in New Hampshire loons (Grade & Vogel, 2023).

While PFOS remains dominant across regions, long-term studies in Canadian seabirds (Kesic et al., 2023) and Spain gull eggs (Colomer-Vidal et al., 2022) found odd-chain PFCAs (PFUdA, PFTriDA) predominant due to fluorotelomer oxidation and atmospheric transport. Conversely, even-chain PFCAs (PFDA, PFDoA) predominate in San Francisco Bay, suggesting local sources. These studies also found PFOS declines that likely stem from long-chain PFAS management in the EU, US, and Canada.

PFAS precursor 6:2 FTS was detected in all 2022 Bay samples (mean: 3.2 ng/g ww), matching New Hampshire levels (Grade & Vogel, 2023) but well below Great Lakes bald eagle eggs (Wu et al., 2020). However, 6:2 FTS was absent in 2024 Bay samples, requiring continued monitoring to confirm sustained reductions.

PFAS in Bird Eggs May Pose Risk

Laboratory and field data have associated exposure to PFAS with a variety of adverse effects in birds, from oxidative stress to reproductive impacts (Ankley et al., 2021). However, because mechanism-based quantitative linkages between exposure and effects have yet to be established, there is considerable uncertainty in the literature regarding developing a protective threshold for the concentrations of PFAS in bird eggs. The Canadian Federal Environmental Quality Guideline for PFOS in bird eggs is 1,900 ng/g ww (ECCC, 2018), based on the lowest observable adverse effect level causing decreased hatching success in quail. This laboratory study was also used with additional assessment factors to develop Australia's guideline of 200 ng/g ww for the sum of PFOS and PFHxS (HEPA, 2020). Field studies of Great Lakes tree swallows have shown impacts on hatching success as low as 150 ng PFOS/g ww in some areas (Custer et al., 2012, 2014) but no impact on hatching success at concentrations exceeding 600 ng/g ww in others (Custer et al., 2019). These differences could be due

to effects from co-exposure to other contaminants and/or variations in sensitivity between populations.

Given this uncertainty, we recommend RMP use of the Australian guideline to be more protective and better account for potential mixture effects. Although median PFOS concentrations in Bay bird eggs as a whole have been below this threshold since 2012, 90th percentile concentrations exceeded this concentration through 2018, and one South Bay 2022 sample exceeded this threshold. All samples were below this threshold for the first time in 2024. In addition, the Canadian Federal Environmental Water Quality Guideline for PFOS for an avian wildlife diet is 8.2 ng/g (ECCC, 2018). Some of the prey fish analyzed in the Bay exceed this PFOS threshold. Ongoing monitoring is needed to assess whether PFAS levels in Bay bird eggs remain consistently below levels that may cause adverse impacts.

5.3 Harbor Seals

The Pacific harbor seal (*Phoca vitulina richardii*), the San Francisco Bay Area's only resident pinniped, lives year-round in the Bay and surrounding coastal waters. Seals primarily forage and rest (haul out) in the central and southern Bay, using rocky shoals or mudflats to rest, birth, and nurture pups. Major haul-out sites include Castro Rocks, Yerba Buena Island, and Mowry Slough. Seals exhibit high site fidelity, typically foraging within 20 kilometers of haul-out sites (Melwani et al., 2012; Nickel & Grigg, 2002). They are carnivorous opportunists, eating a variety of fish like goby, topsmelt, and anchovy; their diet reflects available fish distribution (Gibble & Harvey, 2015). As apex predators with narrow spatial ranges, harbor seals are ideal biosentinels for contaminant monitoring.

The RMP has opportunistically sampled Pacific harbor seals through special studies primarily by leveraging existing recovery and sample collection efforts or by using archived biological samples. From 2004 to 2011, the RMP analyzed PFAS in the serum of Pacific harbor seals at Castro Rocks (Central Bay) and Mowry/Corkscrew Slough (Lower South Bay), plus Tomales Bay as a reference site (Sedlak & Greig, 2012; Sedlak et al., 2017). In 2014, samples were collected only from Lower South Bay. Thirteen PFAS analytes were examined during this decade of studies.

In 2023 and 2024, the RMP piloted a study of marine mammal tissues to inform their potential addition to the S&T monitoring program. Serum samples were collected similar to previous efforts, as well as liver to examine the tissues with highest reported body burden for PFAS (38% in blood; 36% in liver) in harbor seals (Ahrens et al., 2009b). Harbor seals were opportunistically collected from sites in Central Bay and analyzed for 40 PFAS thanks to advancements in analytical methods. Serum samples were from seals who needed rehabilitation (e.g., sick adults or orphaned pups), and liver samples were from recently deceased seals.

This synthesis focuses on harbor seal data from three monitoring efforts: 2009–2011, 2014, and 2023/2024. Serum samples across all years and liver samples from 2023/2024 were collected and analyzed for individual harbor seals of varying lifestages (juveniles and adults) and sex (male and female).

PFOS Predominant in Serum Across All Bay Regions

In harbor seal serum collected from the Bay, PFOS was the most prevalent PFAS detected. Concentrations of PFOS in 2023/2024 serum samples (n = 6) ranged from 32 to 70 ng/g and consistently accounted for over 50% of the sum of PFAS. Six other PFAS (PFHxS, PFOA, PFNA, PFDA, PFUnDA, and PFDoDA) have been frequently detected between 2009–2024 with PFHxS as the next most abundant compound across all monitoring years. This pattern, dominated by PFOS and the sum of PFCAs, aligns with the chemical fingerprint observed in other taxa of biota.

In 2023/2024, an additional 12 PFAS were detected in the serum of Bay harbor seals and generally at levels an order of magnitude lower than PFOS. Nearly all long-chain PFCAs (C7 and above) were found in 100% of samples, with the exception of PFTrDA (5/6 detections). Individual PFCA concentrations were typically below 3 ng/g, ranging from 0.13 to 6.2 ng/g. In addition, three long-chain PFSAAs (PFHxS, PFHpS, and PFDS) were also found in all samples, with the latter two at low concentrations (<1.8 ng/g). PFHxS was found at the second highest concentration of any PFAS, with a median of 9.3 ng/g ww (range 3.9–14 ng/g ww). Two short-chain compounds (PFHpA and PFPeS) were detected sporadically at low levels (<0.23 ng/g ww). No precursors were found in any serum samples.

Higher PFAS Concentrations in Serum Observed in Lower South Bay

Similar to other biota, the greatest individual and sum detections of PFAS were in serum samples from Lower South Bay from 2009 to 2014. PFOS was particularly elevated with median levels from 2009–2011 (n = 20) of 1,050 ng/g ww, although this decreased to 331 ng/g ww in 2014 (n = 7). Levels in South Bay were generally over an order of magnitude greater than those in Central Bay samples, although limited collection opportunities of harbor seals in Lower South Bay have restricted further comparison between subembayments in more recent sampling efforts.

Levels of PFAS in Serum Are Stable Over Time in Central Bay

Recent data (2023/2024) indicate that PFAS levels in serum from Central Bay, including PFOS, are comparable to those detected over a decade ago. The median PFOS concentration in 2023/2024 was 52 ng/g ww, which is similar to the 70 ng/g ww recorded in 2009–2011. While a few samples in the earlier period (4 out of 13) exceeded 125 ng/g ww, most samples generally fall within the same range: 32–70 ng/g ww in the recent projects compared to 12–89 ng/g ww previously. Other PFAAs also follow this trend, showing no significant shift in concentrations in seals collected in Central Bay.

PFAS Fingerprint in Liver Samples

Liver samples from harbor seals were examined for the first time in 2023/2024. The detection of several additional PFAS, particularly precursors, provides a more complex picture of PFAS in Bay marine mammals. PFOS was still the dominant PFAS detected in liver samples (n = 7), ranging from 31 to 74 ng/g ww, and levels generally accounted for about half of the sum of PFAS. However, in two of the adult females, PFOS

accounted for less than a third of the sum of PFAS. This may be due to variable differences in loss of individual PFAS compounds via milk; Grønnestad et al.(2017) observed lower ratios of PFOS to sum of PFAS in mothers compared to pups.

Beyond PFOS, a total of 16 other PFAS were detected in liver samples, with most well below concentrations of PFOS. All long-chain PFCAs (C7 and above) were found in 100% of samples at levels between 0.12 and 5.5 ng/g ww. In addition, three long-chain PFSA's (PFHxS, PFHpS, and PFDS) were also found in all samples, with the latter two at low concentrations (<1 ng/g ww). PFHxS was the second highest PFAA, with a median of 3.6 ng/g ww (range 0.52–7.3 ng/g ww). Three short-chain compounds (PFBA, PFPeA and PFPeS) were detected sporadically at low levels (<1 ng/g ww).

Three PFAS precursors or alternative PFAS (7:3 FTCA, PFOSA, and 9CI-PF3ONS) were also found, with the latter two detected occasionally at low levels (<0.5 ng/g ww). In contrast, 7:3 FTCA was found at levels similar to those of PFOS, with a median of 26 ng/g ww (range 9–116 ng/g ww), and in some samples could account for up to half of the sum of PFAS. 7:3 FTCA has been sporadically detected in sport fish, and more consistently in bird eggs, noting its presence across the Bay food web. The detection of precursors in liver samples is expected, as the liver is where many exogenous compounds are metabolized and can be broken down into detectable precursor compounds.

Comparison to Other Studies and Regions

Comparison to other studies of PFAS in pinnipeds is difficult because concentrations of PFAS found in seals vary not only by location, but also by time, species, animal age, and sometimes animal sex. Additionally, the sum of PFAS is highly dependent on the analytical methods and number of analytes, which have generally improved over time. PFAS in blood is less frequently measured than PFAS in liver, and is less directly comparable across studies because blood may be analyzed whole (e.g., Ahrens et al., 2009b) or separated into plasma (contains clotting factors; e.g., Routti et al., 2014) or serum (does not contain clotting factors; e.g., Sedlak et al., 2017).

Despite study differences, some useful comparisons can be made. PFOS is generally the predominant PFAS observed in seals around the world. PFOS concentrations in Bay of Fundy harbor seals are generally similar to or lower than concentrations observed in harbor seals in other parts of the world over the last few decades (Table B1). Compared to other pinniped species, harbor seals appear to have generally elevated PFAS concentrations, perhaps due to their propensity to live in more urbanized, temperate environments and to spend more time closer to shore.

PFAS in Harbor Seals May Pose Risk

Few guidelines and standards exist for PFAS that protect seals and other marine mammals. The Canadian Federal Environmental Water Quality Guideline for PFOS for a mammalian wildlife diet is 4.6 ng/g (ECCC, 2018). The fact that many of the fish consumed by Bay seals have concentrations exceeding this level raises a potential concern for Bay harbor seal health.

Very few studies have evaluated the toxicological effects of PFAS on seals. Inference from data for other organisms to effects in seals is challenging given the current state of knowledge. In general, there appears to be strong agreement between adverse health effects observed in epidemiological research in humans and adverse effects observed in studies of other taxa exposed to PFAS (Andrews et al., 2023). PFOA, PFOS, and a variety of less well studied PFAS have been linked with a broad array of toxic effects in humans and laboratory animals, including impacts to immune, nervous, and endocrine systems, reproduction and development, metabolism, and carcinogenicity.

In a study of Baikal seals (*Pusa sibirica*), (Ishibashi, Iwata, Kim, Tao, Kannan, Amano, et al., 2008) found PFAAs activate peroxisome proliferator-activated receptors that are important for the regulation of gene functions. The researchers observed the activation of the peroxisome proliferator receptors in Baikal seals in the wild that had much lower liver concentrations of PFNA (3.3–72 ng/g) and PFDA (0.56–35 ng/g), suggesting that lower concentrations may have impacts as well (Ishibashi, Iwata, Kim, Tao, Kannan, Tanabe, et al., 2008). However, these seals may have also been exposed to other co-occurring contaminants (Polasko et al., 2024), which could lead to overestimation of toxic effects. San Francisco Bay harbor seals have liver PFNA and PFDA concentrations in the low end of the observed concentrations associated with peroxisome proliferator-activated receptors activation in wild Baikal seals.

Based on a review of human epidemiological studies of PFOS and PFOA and laboratory studies of mice, Grandjean & Clapp (2015) proposed a benchmark dose level serum concentration of 1.3 ng/g for PFOS and 0.8 ng/g for PFOA. Benchmark dose is defined as the dose that elicits a response as compared to a background concentration. These benchmark dose levels are significantly below the mean serum PFOS concentration observed in 2023 (52 ng/g), the mean serum PFOS and PFOA concentrations observed in 2014 in South Bay seals (184 ng/g and 4.5 ng/g, respectively), and the mean serum concentrations observed in 2009–2011 in Central Bay seals (56.8 ng/g PFOS and 1.8 ng/g PFOA). Assuming similar toxicological responses between humans and pinnipeds, these findings add to the evidence suggesting potential risk of adverse effects in Bay harbor seals at PFAS concentrations currently observed.

6 Monitoring and Science Strategy

This monitoring strategy for the next five years outlines the RMP's approach to increased regional and statewide management information needs on PFAS in the Bay region. While PFOS levels in the Bay are the major risk driver, water quality managers and policy makers are keeping a broad view of PFAS as a class to evaluate risk and inform source control and other management actions.

This monitoring strategy is meant to be a living document and continuously adapt to updated understanding of PFAS in the Bay, development of new analytical tools for PFAS monitoring, and management information needs. Therefore, we anticipate an update to this strategy within the next five years. In particular, a major focus on urban stormwater runoff and wastewater monitoring and modeling efforts, currently underway through the PFAS Sources to Solutions project, is anticipated to further inform future monitoring and modeling needs and efforts. The current monitoring and science strategy incorporates the four major elements described below.

1) Utilize both standardized and complementary PFAS analytical methods for monitoring in the Bay to support management of PFAS as a class.

Baywide monitoring of all matrices using standardized targeted methods, particularly EPA Method 1633, is an important part of the Status & Trends monitoring program to continue observations of levels of PFOS and other PFAS in Bay water, sediment, fish, and bird eggs. However, EPA Method 1633 only quantifies a small fraction of PFAS used in commerce or observed in the environment, and applying complementary methods to improve evaluation of PFAS as a class is important for informing management questions.

Identifying and quantifying additional novel PFAS subclasses can provide more context for evaluating risks, as well as more information to identify potential sources and transport/transformation pathways. However, novel methods to quantify additional PFAS subclasses are in different stages of scientific development. The RMP is piloting additional targeted analytical methods available through commercial and academic laboratories to quantify PFAS precursors and subgroups that have been reported in other locations but not previously measured in the Bay. These include ultra-short chain PFAS, PAPs (mono- and diPAPs), FASAs, FTCAs, and FTOHs (see acronym list for analytes). For example, TFA, an ultra-short chain PFAS, has been reported as the most abundant PFAS in the environment due to a wide range of sources and precursors. The RMP is piloting methods to measure levels in the Bay water, wastewater, urban runoff, and precipitation to inform future monitoring recommendations. Pending results, these methods could be applied to other Bay matrices, although method detection limits and observations in studies elsewhere should be considered.

Periodic application of PFAS-specific nontarget analysis (NTA) methods is also recommended to screen for novel PFAS not included in available targeted methods, as these methods can help identify PFAS analytes and subclasses that may warrant further investigation. NTA is an exploratory approach that can identify new or unexpected PFAS, including transformation products and other compounds currently lacking

analytical standards. The RMP is applying these methods to marine mammal and sport fish samples.

In addition to targeted and nontarget methods, bulk PFAS quantification methods have the potential to provide complementary information on PFAS as a class. The RMP has applied the TOP assay to various Bay matrices, which has demonstrated significant levels of PFAS precursors in wastewater, stormwater runoff, sediment, and surface water, with wastewater showing the most consistent observations. The TOP assay is currently being applied to wastewater and urban stormwater runoff monitoring, and may also be useful for future study of other matrices such as sediment when methods are further developed. Additionally, the RMP is piloting a bulk organofluorine method using ¹⁹F-NMR on a variety of samples and extracts. SFEI staff will continue to evaluate the applicability of this method and others (e.g., adsorbable and extractable organofluorine; AOF and EOF) for RMP monitoring.

2) Build upon the Status & Trends program to continue evaluating PFAS temporal and geographic trends to inform risks and management decisions.

Due to the RMP's visionary CECs monitoring program, the RMP has the most comprehensive long-term multi-matrix PFAS dataset in California. We recommend building upon the RMP's Status & Trends Program to expand monitoring that facilitates evaluating temporal and geographic trends in various Bay matrices. Developing these data is essential for evaluation of the current risks posed by PFOS and other PFAS in the Bay. They also allow the RMP to evaluate long-term trends in response to past and potential future management actions. To date, Status and Trends Program monitoring has documented PFOS declines in South Bay, although trends in other PFAS are less consistent or clear.

As PFAS levels in fish are a major driver of risk, we recommend increasing the frequency of sport fish monitoring to every three years and prey fish monitoring to every two to three years. This will improve datasets available to evaluate temporal trends and more quickly inform management relevant to human health concerns. Currently, PFAS monitoring in sport fish occurs every five years and is next planned for 2029 and 2034 (existing dataset includes 2009, 2014, 2019, and 2024). The recently funded and currently underway "Protecting Bay Fishers" project, funded by the USEPA Program Office for San Francisco Bay, will implement a significant sport fish monitoring effort throughout the Bay in 2027 and jump-start the process for more frequent monitoring.

We recommend a special study on prey fish in 2030 with an evaluation to update the Status and Trends program's monitoring design for sport fish and prey fish. To maximize interpretive value, the study designs for sport and prey fish monitoring may be coordinated with water and sediment monitoring to inform our understanding of the exposure pathways that lead to bioaccumulation of PFOS and other PFAS in biota. This co-matrix study design will be informed by fish bioaccumulation modeling efforts described under element 4.

3) Continue pathways monitoring to understand major PFAS sources, pathways, and loadings to the Bay, with recommended re-evaluation of pathways monitoring needs in 2030.

The RMP has several ongoing and planned efforts to investigate and evaluate PFAS sources, pathways, and loadings to the Bay, which are important for informing management actions. This includes evaluation of the major pathways, municipal wastewater and urban stormwater runoff; as well as an ongoing precipitation study, and planned special study to evaluate available data on local contaminated sites. The current PFAS Sources to Solutions project includes a major focus on understanding relative PFAS loadings from wastewater and urban stormwater runoff, and an evaluation of potential sources for each pathway. Future studies will continue to build on these findings.

The RMP should also continue to coordinate with Bay Area Clean Water Agencies (BACWA) members and collaborate on future prioritized concerns. BACWA has been proactive in independently funding and implementing wastewater monitoring studies to evaluate PFAS levels and trends and identify major sources. BACWA members are implementing individual and regional studies of PFAS in wastewater to inform management strategies to protect drinking water sources and aquatic and terrestrial environments receiving wastewater discharges and biosolids. The current phase 3 BACWA PFAS study will continue evaluating temporal trends of PFAS in wastewater and investigating major PFAS sewershed sources, as well as piloting additional targeted PFAS analytical methods to improve quantification of PFAS as a class.

Future collaboration could include evaluating temporal trends in PFAS levels and composition in wastewater; refining regional loading estimates from wastewater to the Bay relative to other pathways; or monitoring and data analysis needed to identify major sewershed sources. In particular, we recommend increased coordination between the RMP and BACWA to evaluate PFAS trends in wastewater due to anticipated reductions following regulations prohibiting PFAS uses in certain consumer products, such as carpets and rugs, textiles, and personal care products.

In parallel, the RMP's multi-year stormwater CECs monitoring activities will continue to prioritize PFAS monitoring, with particular emphasis on gathering data through Water Year 2027 to inform data analysis and loads modeling supported by the PFAS Sources to Solutions project. Over the next five years, we anticipate a transition from broader reconnaissance-based occurrence screening and evaluation to more nuanced studies that explore specific questions relating to watershed transport and sources or land features to inform modeling and load estimates.

Air transport of PFAS through wet and dry deposition is another direct pathway to the Bay and an indirect pathway through stormwater runoff. Studies from other regions and preliminary Bay data suggest wet deposition contains ultra-short and short-chain PFAS and precursors. Evaluation of air transport is important to better understand the relative contribution of precipitation versus urban washoff from the landscape into urban stormwater runoff.

PFAS transport and loadings to the Bay from contaminated sites (e.g., airports, military bases, bulk fuel terminals, refineries, chrome platers, and landfills) have not yet been sufficiently evaluated at a regional level. While to date Water Boards have required on-site PFAS investigations, further data analysis and development of conceptual models is needed to better understand how these contaminated sites may contribute to stormwater, wastewater, and groundwater transport to the Bay. We recommend an RMP study to support SFEI staff collaboration with Regional Water Board teams to synthesize available monitoring data and develop a regional conceptual model of potential PFAS loadings from contaminated sites. This coordinated effort would also inform RMP's stormwater monitoring approach and interpretation by clearly identifying how known contaminated sites may influence watersheds around the Bay.

By 2030, this focused work on pathways monitoring and modeling will provide foundational data and information to inform development of a robust RMP pathways monitoring strategy for PFAS and other emerging contaminants. Priorities for PFAS will reflect updated management information needs.

4) Develop qualitative and quantitative models linking PFAS loadings and pathways to the Bay with fish bioaccumulation to evaluate in-Bay recovery scenarios to inform management decisions.

Conceptual and quantitative models are important for synthesizing available information about the Bay and identifying data gaps. In anticipation of a framework needed to manage PFAS entering the Bay, we recommend developing quantitative models for PFOS and other high priority PFAS to inform management decisions. We recommend starting with a simple modeling framework that can be expanded to match the availability of monitoring data and management information needs and priorities. This starts with developing an in-Bay mass balance model to understand relative loadings from major PFAS pathways in relation to current levels in Bay sediment, water, and biota. The model will aim to address the following questions:

- Have we accounted for the major transport pathways for PFOS and other high priority PFAS to the Bay? Are there any other important pathways and transformation processes?
- How will Bay sediment and water concentrations change relative to changes in loadings to the Bay, including projected in-Bay recovery scenarios? What pathways and loadings can be reduced to manage levels of PFOS and other high priority PFAS in fish?

Current and planned studies address important data gaps to support a Bay PFAS/PFOS modeling framework. Through the PFAS Sources to Solutions project, SFEI will develop first order estimates of the relative loadings of specific PFAS, including PFOS, from wastewater effluent and stormwater runoff to the Bay. In particular, the approach for modeling stormwater loads is still in development, with a preference for options that are data-driven, spatially distributed, and leverage existing RMP's watershed models.

Meanwhile, the RMP and the Nutrient Management Strategy are investing in in-Bay hydrodynamic and sediment transport models to understand sediment-bound

contaminants (e.g., mercury and PCBs) and dissolved contaminants (e.g., nutrients, specific tire-derived contaminants). PFAS may require a combination of sediment transport and hydrodynamic models. The RMP is has developed an integrated watershed-Bay modeling framework to allow evaluation of management scenarios and their implications for trends in Bay contamination. The RMP's food-web transfer bioaccumulation model for PCBs in the Bay, which forecasts PCB concentrations in sport fish based on reductions in the Bay food web from reduced loadings, could also be applied. This framework is currently being developed for PCBs, and could be adapted to address management questions related to PFOS and other bioaccumulating PFAS.

In parallel with the planned 2028 prey fish Status and Trends monitoring effort, we recommend developing a conceptual model of fate and transport of inputs and within the Bay and bioaccumulation of PFOS and other PFAS driving risk in key Bay fish species. The conceptual model development would inform future quantitative fate and transport and bioaccumulation models, which would link PFAS presence in sediment and water to levels in fish as well as associated levels in pathways (municipal wastewater and urban stormwater), ultimately informing actions that may be needed to manage human health risks from consuming Bay fish.

The RMP may prioritize shiner surfperch in developing a bioaccumulation model as a popular sport fish species that feeds primarily on benthos and exhibits high site fidelity (enabling assessment of spatial differences due to different exposure levels from sediment and water in different subembayments). The species is already used in Status and Trends monitoring to evaluate PCBs and other synthetic organic contaminants. Other important considerations for building this model include determining whether sufficient co-located sediment and water monitoring data exist to evaluate exposure for each modeled fish species at different Bay locations, and evaluating PFOS precursors. The Status & Trends monitoring program design could be adjusted to fill data gaps.

PFAS Multi-Year Plan

Table 6.1. Special studies and monitoring in the RMP ECWG and SPLWG focused on or relevant to PFAS from 2024 to 2031. Numbers indicate proposed budget allocations in \$1000s. Budgets in parentheses represent funding or in-kind services from external sources (e.g., BACWA, USEPA funds). Budgets that are starred represent funding that has been allocated for the given study within SPLWG, while all other RMP funded studies are allocated from ECWG. Bold boxes indicate multi-year studies. Items shaded in yellow are considered high priority for 2028 funding and beyond. The RMP ECWG and SPLWG have not yet fully planned out 2031 studies.

Workgroup	Study	Funder	2024	2025	2026	2027	2028	2029	2030	2031*
ECWG	RMP Status and Trends	S&T	Water (wet) Eggs Sport fish Seals	Water (dry)	Water (wet)	Water (dry) Eggs	Water (wet) Sediment Prey Fish	Water (dry) Sport fish	Water (wet) Eggs	Water (dry)
	PFAS: Synthesis and Strategy	RMP	125							120
	Bay Water TOP Assay	RMP	67							
	PFAS NMR Analysis in Wastewater, Stormwater, Other Matrices	USEPA†		(125)	(256)					
	PFAS Stormwater Depth Pilot	RMP		55						
	PFAS in Precipitation Pilot	RMP			89	62	121			
	PFAS Sources to Solutions (includes conceptual model)	USEPA§	(547)	(751)	(799)	(502)				
	PFAS in Influent, Effluent	BACWA			(100)					
	Ultra-short and Novel PFAS in Bay Water	RMP				140			50	80
	Fish Bioaccumulation Conceptual Model	RMP					100			
	Sediment Novel PFAS Analytes and Direct TOP Assay	RMP					100			
	Wastewater follow-up and New Concerns	RMP						100		60
	Novel PFAS in Sport Fish (NTA Follow-up)	RMP						60		
	PFAS in Prey Fish Temporal Trends	RMP							60	
	PFAS in-Bay Model, Mass Balance	RMP							100	
	Protecting Bay Fishers*	USEPA				(102)	(382)	(381)	(381)	(344)
ECWG	NTA of Bay Fish	RMP USEPA†	23	-75						
	NTA of Microfibers and Stormwater	RMP		124						
	NTA of Stormwater	RMP USEPA†		36	21 (45)	156				
	Stormwater CECs Monitoring and Modeling (multiple classes)	RMP USEPA‡	300 (100)	300 (100)	325 (53)	338	325	325	325	325
	Expanded Efforts with Additional Resources	RMP		150	50	113	250	250	250	
	PFAS Contaminated Sites Conceptual Model, Data Analysis	RMP					80			
SPLWG	Stormwater CECs Modeling and Data Analysis	USEPA† RMP		(37)*			100*	150*	100*	
	Remote Sampler Purchase	RMP	180*							
	Remote Sampler Improvements	RMP	51*			60*	30*	30*	40*	
	Dry Weather Monitoring for CECs	RMP					50*	75*	75*	
PCBWG		SEP USEPA‡ USEPA†	(136)* (350)*	(136)* (340)*	(235)* (200)*	(200)*	(200)*			
	In-Bay Fate Model									
RMP-funded Special Studies Subtotal			746	665	485	869	1,176	990	1,000	180
Externally Funded Studies/Proposals Subtotal			1,133	1,564	1,688	804	582	381	381	344
TOTAL			1,879	2,229	2,173	1,673	1,758	1,371	1,381	524

† "Tracking Bay Health: Water Quality Science to Inform Management" grant

‡ "Destination Clean Bay" grant

§ "PFAS Sources to Solutions" grant

|| "Protection Bay Fishers" grant

References

- Aflaki, R., & Voskuhl, J. (2022, April 5). *What can the Geotracker PFAS data tell us?* 68th California Association of Sanitation Agencies, San Diego, CA.
- Ahmadireskety, A., Da Silva, B. F., Awkerman, J. A., Aufmuth, J., Yost, R. A., & Bowden, J. A. (2021). Per- and polyfluoroalkyl substances (PFAS) in sediments collected from the Pensacola Bay System watershed. *Environmental Advances*, 5, 100088. <https://doi.org/10.1016/j.envadv.2021.100088>
- Ahrens, L., Siebert, U., & Ebinghaus, R. (2009a). Temporal trends of polyfluoroalkyl compounds in harbor seals (*Phoca vitulina*) from the German Bight, 1999–2008. *Chemosphere*, 76(2), 151–158. <https://doi.org/10.1016/j.chemosphere.2009.03.053>
- Ahrens, L., Siebert, U., & Ebinghaus, R. (2009b). Total body burden and tissue distribution of polyfluorinated compounds in harbor seals (*Phoca vitulina*) from the German Bight. *Marine Pollution Bulletin*, 58(4), 520–525. <https://doi.org/10.1016/j.marpolbul.2008.11.030>
- Andrews, D. Q., Stoiber, T., Temkin, A. M., & Naidenko, O. V. (2023). Discussion. Has the human population become a sentinel for the adverse effects of PFAS contamination on wildlife health and endangered species? *Science of The Total Environment*, 901, 165939. <https://doi.org/10.1016/j.scitotenv.2023.165939>
- Ankley, G. T., Cureton, P., Hoke, R. A., Houde, M., Kumar, A., Kurias, J., Lanno, R., McCarthy, C., Newsted, J., Salice, C. J., Sample, B. E., Sepúlveda, M. S., Steevens, J., & Valsecchi, S. (2021). Assessing the Ecological Risks of Per- and Polyfluoroalkyl Substances: Current State-of-the Science and a Proposed Path Forward. *Environmental Toxicology and Chemistry*, 40(3), 564–605. <https://doi.org/10.1002/etc.4869>
- Arp, H. P. H., Gredelj, A., Glüge, J., Scheringer, M., & Cousins, I. T. (2024). The Global Threat from the Irreversible Accumulation of Trifluoroacetic Acid (TFA). *Environmental Science & Technology*, 58(45), 19925–19935. <https://doi.org/10.1021/acs.est.4c06189>
- ATSDR. (2021). *Toxicological Profile for Perfluoroalkyls*. Agency for Toxic Substances and Disease Registry. <https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf>
- BACWA. (2022). *Biosolids in the Baylands: White Paper*. Bay Area Clean Water Agencies. <https://bacwa.org/wp-content/uploads/2022/07/Biosolids-in-the-Baylands-White-Paper-March-2022.pdf>
- BACWA. (2024). *2024 Biosolids Trends Survey Report*. Bay Area Clean Water Agencies. <https://bacwa.org/wp-content/uploads/2024/09/BACWA-2024-Biosolids-Trends-Survey-Report-2024-09-10.pdf>

- Bălan, S. A., Mathrani, V. C., Guo, D. F., & Algazi, A. M. (2021). Regulating PFAS as a chemical class under the California Safer Consumer Products Program. *Environmental Health Perspectives*, 129(2), 025001. <https://doi.org/10.1289/EHP7431>
- Barbo, N., Stoiber, T., Naidenko, O. V., & Andrews, D. Q. (2023). Locally caught freshwater fish across the United States are likely a significant source of exposure to PFOS and other perfluorinated compounds. *Environmental Research*, 220, 115165. <https://doi.org/10.1016/j.envres.2022.115165>
- Boatman, A. K., Chappel, J. R., Polera, M. E., Dodds, J. N., Belcher, S. M., & Baker, E. S. (2024). Assessing Per- and Polyfluoroalkyl Substances in Fish Fillet Using Non-Targeted Analyses. *Environmental Science & Technology*, 58(32), 14486–14495. <https://doi.org/10.1021/acs.est.4c04299>
- Brendel, S., Fetter, É., Staude, C., Vierke, L., & Biegel-Engler, A. (2018). Short-chain perfluoroalkyl acids: Environmental concerns and a regulatory strategy under REACH. *Environmental Sciences Europe*, 30(1). <https://doi.org/10.1186/s12302-018-0134-4>
- Britton, K. N., Judson, R. S., Hill, B. N., Jarema, K. A., Olin, J. K., Knapp, B. R., Lowery, M., Feshuk, M., Brown, J., & Padilla, S. (2024). Using Zebrafish to Screen Developmental Toxicity of Per- and Polyfluoroalkyl Substances (PFAS). *Toxics*, 12(7), 501. <https://doi.org/10.3390/toxics12070501>
- Buck, R. C., Franklin, J., Berger, U., Conder, J. M., Cousins, I. T., de Voogt, P., Jensen, A. A., Kannan, K., Mabury, S. A., & van Leeuwen, S. P. (2011). Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. *Integrated Environmental Assessment and Management*, 7(4), 513–541. <https://doi.org/10.1002/ieam.258>
- Butt, C. M., Mabury, S. A., Kwan, M., Wang, X., & Muir, D. C. G. (2008). Spatial trends of perfluoroalkyl compounds in ringed seals (*phoca hispida*) from the canadian arctic. *Environmental Toxicology and Chemistry*, 27(3), 542–553.
- Buzby, N., Davis, J., Sutton, R., Miller, E., Yee, D., Wong, A., Sigala, M., Bonnema, A., Heim, W., & Grace, R. (2021). *Contaminant Concentrations in Sport Fish from San Francisco Bay: 2019* (SFEI Contribution No. 1036). San Francisco Estuary Institute. <https://www.sfei.org/documents/contaminant-concentrations-sport-fish-san-francisco-bay-2019>
- Byrne, S., Seguinot-Medina, S., Waghiyi, V., Apatiki, E., Immingan, T., Miller, P., Von Hippel, F. A., Buck, C. L., & Carpenter, D. O. (2022). PFAS and PBDEs in traditional subsistence foods from Sivuqaq, Alaska. *Environmental Science and Pollution Research International*, 29(51), 77145–77156. <https://doi.org/10.1007/s11356-022-20757-2>

- Cahill, T. M. (2022). Increases in Trifluoroacetate Concentrations in Surface Waters over Two Decades. *Environmental Science & Technology*, 56(13), 9428–9434. <https://doi.org/10.1021/acs.est.2c01826>
- Codling, G., Hosseini, S., Corcoran, M. B., Bonina, S., Lin, T., Li, A., Sturchio, N. C., Rockne, K. J., Ji, K., Peng, H., & Giesy, J. P. (2018). Current and historical concentrations of poly and perfluorinated compounds in sediments of the northern Great Lakes – Superior, Huron, and Michigan. *Environmental Pollution*, 236, 373–381. <https://doi.org/10.1016/j.envpol.2018.01.065>
- Coggan, T. L., Moodie, D., Kolobaric, A., Szabo, D., Shimeta, J., Crosbie, N. D., Lee, E., Fernandes, M., & Clarke, B. O. (2019). An investigation into per- and polyfluoroalkyl substances (PFAS) in nineteen Australian wastewater treatment plants (WWTPs). *Heliyon*, 5(8), e02316. <https://doi.org/10.1016/j.heliyon.2019.e02316>
- Colomer-Vidal, P., Bertolero, A., Alcaraz, C., Garreta-Lara, E., Santos, F. J., & Lacorte, S. (2022). Distribution and ten-year temporal trends (2009–2018) of perfluoroalkyl substances in gull eggs from Spanish breeding colonies. *Environmental Pollution*, 293, 118555. <https://doi.org/10.1016/j.envpol.2021.118555>
- Conder, J. M., Hoke, R. A., Wolf, W. de, Russell, M. H., & Buck, R. C. (2008). Are PFCAs Bioaccumulative? A Critical Review and Comparison with Regulatory Criteria and Persistent Lipophilic Compounds. *Environmental Science & Technology*, 42(4), 995–1003. <https://doi.org/10.1021/es070895g>
- Conomos, T. J. (1979). Properties and circulation of San Francisco Bay waters. In American Association for the Advancement of Science- Pacific Division (Ed.), *San Francisco Bay—The urbanized estuary*. American Association for the Advancement of Science- Pacific Division. USGS Publications Warehouse. <https://pubs.usgs.gov/publication/70156929>
- Custer, C. M., Custer, T. W., Delaney, R., Dummer, P. M., Schultz, S., & Karouna-Renier, N. (2019). Perfluoroalkyl Contaminant Exposure and Effects in Tree Swallows Nesting at Clarks Marsh, Oscoda, Michigan, USA. *Archives of Environmental Contamination and Toxicology*, 77(1), 1–13. <https://doi.org/10.1007/s00244-019-00620-1>
- Custer, C. M., Custer, T. W., Dummer, P. M., Etterson, M. A., Thogmartin, W. E., Wu, Q., Kannan, K., Trowbridge, A., & McKann, P. C. (2014). Exposure and Effects of Perfluoroalkyl Substances in Tree Swallows Nesting in Minnesota and Wisconsin, USA. *Archives of Environmental Contamination and Toxicology*, 66(1), 120–138. <https://doi.org/10.1007/s00244-013-9934-0>
- Custer, C. M., Custer, T. W., Schoenfuss, H. L., Poganski, B. H., & Solem, L. (2012). Exposure and effects of perfluoroalkyl compounds on tree swallows nesting at Lake Johanna in east central Minnesota, USA. *Reproductive Toxicology*, 33(4), 556–562. <https://doi.org/10.1016/j.reprotox.2011.01.005>

- da Silva, B. F., Aristizabal-Henao, J. J., Aufmuth, J., Awkerman, J., & Bowden, J. A. (2022). Survey of per- and polyfluoroalkyl substances (PFAS) in surface water collected in Pensacola, FL. *Heliyon*, 8(8), e10239. <https://doi.org/10.1016/j.heliyon.2022.e10239>
- Davis, J., Schiff, K., Melwani, A., Bezalel, S. N., Hunt, J., Allen, R. M., Ichikawa, G., Bonnema, A., Heim, W., Crane, D., Swenson, S., Lamerdin, C., & Stephenson, M. (2011). *Contaminants in Sport Fish from the California Coast, 2009: Summary Report on Year One of a Two-Year Screening Survey*. SWRCB SWAMP. <https://www.sfei.org/documents/contaminants-sport-fish-california-coast-2009-summary-report-year-one-two-year-screening>
- de Wit, C. A., Bossi, R., Dietz, R., Dreyer, A., Faxneld, S., Garbus, S. E., Hellström, P., Koschorreck, J., Lohmann, N., Roos, A., Sellström, U., Sonne, C., Treu, G., Vorkamp, K., Yuan, B., & Eulaers, I. (2020). Organohalogen compounds of emerging concern in Baltic Sea biota: Levels, biomagnification potential and comparisons with legacy contaminants. *Environment International*, 144, 106037. <https://doi.org/10.1016/j.envint.2020.106037>
- ECCC. (2018). *Canadian Environmental Protection Act, 1999 Federal Environmental Quality Guidelines Perfluorooctane Sulfonate (PFOS)*. Environment and Climate Change Canada. <https://www.canada.ca/en/environment-climate-change/services/evaluating-existing-substances/federal-environmental-quality-guidelines-perfluorooctane-sulfonate.html>
- Ehsan, M. N. (2026). Per- and polyfluoroalkyl substances in aquatic sediments: Global distribution, partitioning behavior, and ecological consequences. *Journal of Environmental Quality*, 55(1), e70148. <https://doi.org/10.1002/jeq2.70148>
- Evich, M. G., Davis, M. J. B., McCord, J. P., Acrey, B., Awkerman, J. A., Knappe, D. R. U., Lindstrom, A. B., Speth, T. F., Tebes-Stevens, C., Strynar, M. J., Wang, Z., Weber, E. J., Henderson, W. M., & Washington, J. W. (2022). Per- and polyfluoroalkyl substances in the environment. *Science*, 375(6580), eabg9065. <https://doi.org/10.1126/science.abg9065>
- Fair, P. A., & Houde, M. (2018). Poly- and Perfluoroalkyl Substances in Marine Mammals. In *Marine Mammal Ecotoxicology* (pp. 117–145). Elsevier. <https://doi.org/10.1016/B978-0-12-812144-3.00005-X>
- Fair, P. A., Wolf, B., White, N. D., Arnott, S. A., Kannan, K., Karthikraj, R., & Vena, J. E. (2019). Perfluoroalkyl substances (PFASs) in edible fish species from Charleston Harbor and tributaries, South Carolina, United States: Exposure and risk assessment. *Environmental Research*, 171, 266–277. <https://doi.org/10.1016/j.envres.2019.01.021>

- Fischer, F. C., Ludtke, S., Thackray, C., Pickard, H. M., Haque, F., Dassuncao, C., Endo, S., Schaider, L., & Sunderland, E. M. (2024). Binding of Per- and Polyfluoroalkyl Substances (PFAS) to Serum Proteins: Implications for Toxicokinetics in Humans. *Environmental Science & Technology*, 58(2), 1055–1063. <https://doi.org/10.1021/acs.est.3c07415>
- Foddy, H., Cojocariu, C., & Nikiforov, V. (2024). *Routine PFAS Testing in Surface Water Using TOP Assay and ACQUITY™ QDa™ II Mass Detector*. Waters Corporation.
- Gallen, C., Bignert, A., Taucare, G., O'Brien, J., Braeunig, J., Reeks, T., Thompson, J., & Mueller, J. F. (2022). Temporal trends of perfluoroalkyl substances in an Australian wastewater treatment plant: A ten-year retrospective investigation. *Science of The Total Environment*, 804, 150211. <https://doi.org/10.1016/j.scitotenv.2021.150211>
- Gauthier, J. R., & Mabury, S. A. (2022). Noise-Reduced Quantitative Fluorine NMR Spectroscopy Reveals the Presence of Additional Per- and Polyfluorinated Alkyl Substances in Environmental and Biological Samples When Compared with Routine Mass Spectrometry Methods. *Analytical Chemistry*, 94(7), 3278–3286. <https://doi.org/10.1021/acs.analchem.1c05107>
- Gebbink, W. A., Bignert, A., & Berger, U. (2016). Perfluoroalkyl Acids (PFAAs) and Selected Precursors in the Baltic Sea Environment: Do Precursors Play a Role in Food Web Accumulation of PFAAs? *Environmental Science & Technology*, 50(12), 6354–6362. <https://doi.org/10.1021/acs.est.6b01197>
- George, S. E., Baker, T. R., & Baker, B. B. (2023). Nonlethal detection of PFAS bioaccumulation and biomagnification within fishes in an urban- and wastewater-dominant Great Lakes watershed. *Environmental Pollution*, 321, 121123. <https://doi.org/10.1016/j.envpol.2023.121123>
- Gibble, C. M., & Harvey, J. T. (2015). Food habits of harbor seals (*Phoca vitulina richardii*) as an indicator of invasive species in San Francisco Bay, California. *Marine Mammal Science*, 31(3), 1014–1034. <https://doi.org/10.1111/mms.12214>
- Göckener, B., Fliedner, A., Weinfurter, K., Rüdell, H., Badry, A., & Koschorreck, J. (2023). Tracking down unknown PFAS pollution – The direct TOP assay in spatial monitoring of surface waters in Germany. *Science of The Total Environment*, 898, 165425. <https://doi.org/10.1016/j.scitotenv.2023.165425>
- Goodrow, S. M., Ruppel, B., Lippincott, R. L., Post, G. B., & Procopio, N. A. (2020). Investigation of levels of perfluoroalkyl substances in surface water, sediment and fish tissue in New Jersey, USA. *Science of The Total Environment*, 729, 138839. <https://doi.org/10.1016/j.scitotenv.2020.138839>
- Grade, T., & Vogel, H. (2023). *PFAS in New Hampshire Loon Eggs* (p. 27).

- Grandjean, P., & Clapp, R. (2015). Perfluorinated Alkyl Substances: Emerging Insights Into Health Risks. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, 25(2), 147–163.
<https://doi.org/10.1177/1048291115590506>
- Griffin, E. K., Hall, L. M., Brown, M. A., Taylor-Manges, A., Green, T., Suchanec, K., Furman, B. T., Congdon, V. M., Wilson, S. S., Osborne, T. Z., Martin, S., Schultz, E. A., Lukacs, D. T., Greenberg, J. A., & Bowden, J. A. (2023). PFAS surveillance in abiotic matrices within vital aquatic habitats throughout Florida. *Marine Pollution Bulletin*, 192, 115011.
<https://doi.org/10.1016/j.marpolbul.2023.115011>
- Grønnestad, R., Villanger, G. D., Polder, A., Kovacs, K. M., Lydersen, C., Jenssen, B. M., & Borgå, K. (2017). Maternal transfer of perfluoroalkyl substances in hooded seals. *Environmental Toxicology and Chemistry*, 36(3), 763–770.
<https://doi.org/10.1002/etc.3623>
- Guckert, M., Scheurer, M., Schaffer, M., Reemtsma, T., & Nödler, K. (2022). Combining target analysis with sum parameters—A comprehensive approach to determine sediment contamination with PFAS and further fluorinated substances. *Environmental Science and Pollution Research*, 29(57), 85802–85814.
<https://doi.org/10.1007/s11356-022-21588-x>
- Hall, J., Bengtson Nash, S., Gautam, A., Bender, H., Pitcher, B. J., McCallum, H., & Doyle, C. (2023). Persistent organic pollutants and trace elements detected in New Zealand fur seals (long-nosed fur seal; *Arctocephalus forsteri*) from New South Wales, Australia, between 1998 and 2019. *Science of The Total Environment*, 902, 166087. <https://doi.org/10.1016/j.scitotenv.2023.166087>
- Hedgespeth, M. L., Taylor, D. L., Balint, S., Schwartz, M., & Cantwell, M. G. (2023). Ecological characteristics impact PFAS concentrations in a U.S. North Atlantic food web. *The Science of the Total Environment*, 880, 163302.
<https://doi.org/10.1016/j.scitotenv.2023.163302>
- HEPA. (2020). *PFAS National Environmental Management Plan Version 2.0* (p. 134). Heads of EPAs Australia and New Zealand.
- Heuvel-Greve, M., Schotanus, J., Kotterman, M. J. J., Kwadijk, C., Brasseur, S., Leopold, M., van Zwol, J., Ende, D., Cornelisse, S., de Froe, E., & Foekema, E. (2026). Exposure and magnification of PFAS in a temperate estuarine food web, including top predators. *Marine Pollution Bulletin*, 224, 119053.
<https://doi.org/10.1016/j.marpolbul.2025.119053>
- Houde, M., De Silva, A. O., Muir, D. C. G., & Letcher, R. J. (2011). Monitoring of perfluorinated compounds in aquatic biota: An updated review: PFCs in aquatic biota. *Environmental Science & Technology*, 45(19), 7962–7973.
<https://doi.org/10.1021/es104326w>

- Houtz, E. F., Sutton, R., Park, J.-S., & Sedlak, M. (2016). Poly- and perfluoroalkyl substances in wastewater: Significance of unknown precursors, manufacturing shifts, and likely AFFF impacts. *Water Research*, 95, 142–149. <https://doi.org/10.1016/j.watres.2016.02.055>
- Hughes, S. G., & Henderson, J. (2025). *ECOS Compendium of State PFAS Actions*. <https://www.ecos.org/wp-content/uploads/2025/04/ECOS-PFAS-Compendium-FINAL.pdf>
- Ishibashi, H., Iwata, H., Kim, E.-Y., Tao, L., Kannan, K., Amano, M., Miyazaki, N., Tanabe, S., Batoev, V. B., & Petrov, E. A. (2008). Contamination and Effects of Perfluorochemicals in Baikal Seal (*Pusa sibirica*). 1. Residue Level, Tissue Distribution, and Temporal Trend. *Environmental Science & Technology*, 42(7), 2295–2301. <https://doi.org/10.1021/es072054f>
- Ishibashi, H., Iwata, H., Kim, E.-Y., Tao, L., Kannan, K., Tanabe, S., Batoev, V. B., & Petrov, E. A. (2008). Contamination and Effects of Perfluorochemicals in Baikal Seal (*Pusa sibirica*). 2. Molecular Characterization, Expression Level, and Transcriptional Activation of Peroxisome Proliferator-Activated Receptor α . *Environmental Science & Technology*, 42(7), 2302–2308. <https://doi.org/10.1021/es0720558>
- Joerss, H., Menger, F., Tang, J., Ebinghaus, R., & Ahrens, L. (2022). Beyond the Tip of the Iceberg: Suspect Screening Reveals Point Source-Specific Patterns of Emerging and Novel Per- and Polyfluoroalkyl Substances in German and Chinese Rivers. *Environmental Science & Technology*, 56(9), 5456–5465. <https://doi.org/10.1021/acs.est.1c07987>
- Jouanneau, W., Léandri-Breton, D.-J., Corbeau, A., Herzke, D., Moe, B., Nikiforov, V. A., Gabrielsen, G. W., & Chastel, O. (2022). A Bad Start in Life? Maternal Transfer of Legacy and Emerging Poly- and Perfluoroalkyl Substances to Eggs in an Arctic Seabird. *Environmental Science & Technology*, 56(10), 6091–6102. <https://doi.org/10.1021/acs.est.1c03773>
- Kaboré, H. A., Goeury, K., Desrosiers, M., Vo Duy, S., Liu, J., Cabana, G., Munoz, G., & Sauvé, S. (2022). Novel and legacy per- and polyfluoroalkyl substances (PFAS) in freshwater sporting fish from background and firefighting foam impacted ecosystems in Eastern Canada. *Science of The Total Environment*, 816, 151563. <https://doi.org/10.1016/j.scitotenv.2021.151563>
- Kesic, R., Elliott, J. E., Elliott, K. H., Lee, S. L., & Maisonneuve, F. (2023). Perfluoroalkyl Substances in Seabird Eggs from Canada's Pacific Coast: Temporal Trends (1973–2019) and Interspecific Patterns. *Environmental Science & Technology*, 57(29), 10792–10803. <https://doi.org/10.1021/acs.est.3c02965>

- Khalid, N. K., Le Calvez, M., Lemire, M., Dinh, Q. T., Fontaine, J., Lair, S., & Sauvé, S. (2024). Occurrence of 80 per and polyfluorinated alkyl substances (PFAS) in muscle and liver tissues of marine mammals of the St. Lawrence Estuary and Gulf, Quebec, Canada. *Frontiers in Environmental Chemistry*, 5. <https://doi.org/10.3389/fenvc.2024.1403728>
- Khan, B., Burgess, R. M., & Cantwell, M. G. (2023). Occurrence and Bioaccumulation Patterns of Per- and Polyfluoroalkyl Substances (PFAS) in the Marine Environment. *ACS ES&T Water*, 3(5), 1243–1259. <https://doi.org/10.1021/acsestwater.2c00296>
- Klosterhaus, S., Yee, D., Sedlak, M., Wong, A., & Sutton, R. (2013). *Contaminants of Emerging Concern in San Francisco Bay: A Summary of Occurrence Data and Identification of Data Gaps* (Contribution No. 698). San Francisco Estuary Institute. https://www.sfei.org/sites/default/files/biblio_files/698_CEC_synthesis.pdf
- Lang, J. R., Allred, B. M., Field, J. A., Levis, J. W., & Barlaz, M. A. (2017). National Estimate of Per- and Polyfluoroalkyl Substance (PFAS) Release to U.S. Municipal Landfill Leachate. *Environmental Science & Technology*, 51(4), 2197–2205. <https://doi.org/10.1021/acs.est.6b05005>
- Letcher, R. J., Su, G., Moore, J. N., Williams, L. L., Martin, P. A., de Solla, S. R., & Bowerman, W. W. (2015). Perfluorinated sulfonate and carboxylate compounds and precursors in herring gull eggs from across the Laurentian Great Lakes of North America: Temporal and recent spatial comparisons and exposure implications. *Science of The Total Environment*, 538, 468–477. <https://doi.org/10.1016/j.scitotenv.2015.08.083>
- Lin, D., Méndez, M., & Paterson, K. (2023). *Study of Per- and Polyfluoroalkyl Substances in Bay Area POTWs Final Report* (SFEI Contribution No. 1145). San Francisco Estuary Institute. <https://www.sfei.org/documents/study-and-polyfluoroalkyl-substances-bay-area-potws-final-report>
- Lin, D., Méndez, M., Paterson, K., Wong, A., Yee, D., Sutton, R., Houtz, E., Cousins, M., & Fono, L. (2024). Residential Wastewater as a Major Source of Per- and Polyfluoroalkyl Substances to Municipal Wastewater. *ACS ES&T Water*. <https://doi.org/10.1021/acsestwater.4c00507>
- Liu, Y., Wang, B., Guo, J., Sun, H., Zhang, Y., Wang, G., Yang, X., & Duan, J. (2026). Per- and polyfluoroalkyl substances (PFAS) in wastewater treatment plants: An overview of their occurrence, fate, effects, and ecological risks. *Journal of Environmental Management*, 399, 128630. <https://doi.org/10.1016/j.jenvman.2026.128630>

- Lohmann, R., & Letcher, R. J. (2023). The universe of fluorinated polymers and polymeric substances and potential environmental impacts and concerns. *Current Opinion in Green and Sustainable Chemistry*, 41, 100795. <https://doi.org/10.1016/j.cogsc.2023.100795>
- Loi, E. I. H., Yeung, L. W. Y., Taniyasu, S., Lam, P. K. S., Kannan, K., & Yamashita, N. (2011). Trophic Magnification of Poly- and Perfluorinated Compounds in a Subtropical Food Web. *Environmental Science & Technology*, 45(13), 5506–5513. <https://doi.org/10.1021/es200432n>
- MA DPH. (2023). *Evaluation of PFAS in Recreational Waterbodies in Massachusetts Technical Support Document DRAFT v. 2.0*. <https://www.mass.gov/doc/technical-basis-for-issuing-fish-advisories-0/download>
- Ma, W., Li, F., Tian, Y., Xu, H., Luo, J., Shi, D., Dai, J., Tang, J., & Pan, Y. (2025). Simultaneous Analysis of Ultrashort- to Long-Chain PFAS in Multisalinity Aquatic Systems: Methodology, Spatial Profiling, and Risk Prioritization. *Environmental Science & Technology*, 59(49), 26770–26780. <https://doi.org/10.1021/acs.est.5c11234>
- Macorps, N., Labadie, P., Lestremau, F., Assoumani, A., & Budzinski, H. (2023). Per- and polyfluoroalkyl substances (PFAS) in surface sediments: Occurrence, patterns, spatial distribution and contribution of unattributed precursors in French aquatic environments. *Science of The Total Environment*, 874, 162493. <https://doi.org/10.1016/j.scitotenv.2023.162493>
- Masoner, J. R., Kolpin, D. W., Cozzarelli, I. M., Bradley, P. M., Arnall, B. B., Forshay, K. J., Gray, J. L., Groves, J. F., Hladik, M. L., Hubbard, L. E., Iwanowicz, L. R., Jaeschke, J. B., Lane, R. F., McCleskey, R. B., Polite, B. F., Roth, D. A., Pettijohn, M. B., & Wilson, M. C. (2023). Contaminant Exposure and Transport from Three Potential Reuse Waters within a Single Watershed. *Environmental Science & Technology*, acs.est.2c07372. <https://doi.org/10.1021/acs.est.2c07372>
- Melwani, A., Greenfield, B. K., Yee, D., & Davis, J. A. (2012). *Conceptual Foundations for Modeling Bioaccumulation in San Francisco Bay* (SFEI Contribution No. 676). San Francisco Estuary Institute.
- Méndez, M., Davis, J., Miller, E. L., Grace, R., & Sutton, R. (2025). Spatial Trends and Health Risks of Per- and Polyfluoroalkyl Substances in San Francisco Bay Fish from 2009 to 2019. *ACS ES&T Water*. <https://doi.org/10.1021/acsestwater.4c00999>
- Méndez, M., Miller, E., Lin, D., & Sutton, R. (2025). *Emerging PFAS Trends in Water and Sediment from San Francisco Bay* (SFEI Contribution #1245). San Francisco Estuary Institute.
- Méndez, M., Trinh, M., Miller, E., Lin, D., & Sutton, R. (2022). *PFAS in San Francisco Bay Water* (SFEI Contribution No. 1094). San Francisco Estuary Institute. <https://www.sfei.org/documents/pfas-san-francisco-bay-water>

- Meng, L., Song, B., Zhong, H., Ma, X., Wang, Y., Ma, D., Lu, Y., Gao, W., Wang, Y., & Jiang, G. (2021). Legacy and emerging per- and polyfluoroalkyl substances (PFAS) in the Bohai Sea and its inflow rivers. *Environment International*, 156, 106735. <https://doi.org/10.1016/j.envint.2021.106735>
- Michigan EGLE. (2022). *Land Application of Biosolids Containing PFAS Interim Strategy*. Michigan Department of Environment, Great Lakes, and Energy. <https://www.michigan.gov/egle/-/media/Project/Websites/egle/Documents/Programs/WRD/Biosolids/PFAS-Biosolids-Interim-Strategy-2022.pdf?rev=55c495dd264c4e44878715087e63e159&hash=DA6EE9143595604F8DAA5582C9EE4356>
- Miller, E., Méndez, M., Shimabuku, I., Buzby, N., & Sutton, R. (2020). *Contaminants of Emerging Concern in San Francisco Bay: A Strategy for Future Investigations 2020 Update* (No. 1007). San Francisco Estuary Institute. <https://www.sfei.org/documents/contaminants-emerging-concern-san-francisco-bay-strategy-future-investigations-2020-update>
- Moodie, D., Coggan, T., Berry, K., Kolobaric, A., Fernandes, M., Lee, E., Reichman, S., Nuggeoda, D., & Clarke, B. O. (2021). Legacy and emerging per- and polyfluoroalkyl substances (PFASs) in Australian biosolids. *Chemosphere*, 270, 129143. <https://doi.org/10.1016/j.chemosphere.2020.129143>
- Muir, D., & Miaz, L. T. (2021). Spatial and Temporal Trends of Perfluoroalkyl Substances in Global Ocean and Coastal Waters. *Environmental Science & Technology*, 55(14), 9527–9537. <https://doi.org/10.1021/acs.est.0c08035>
- Mulabagal, V., Liu, L., Qi, J., Wilson, C., & Hayworth, J. S. (2018). A rapid UHPLC-MS/MS method for simultaneous quantitation of 23 perfluoroalkyl substances (PFAS) in estuarine water. *Talanta*, 190, 95–102. <https://doi.org/10.1016/j.talanta.2018.07.053>
- Neuwald, I. J., Hübner, D., Wiegand, L., Valkov, V., Borchers, U., Nödler, K., Scheurer, M., Hale, S. E., Arp, H. P. H., & Zahn, D. (2022). Ultra-short chain PFAS in the sources of German drinking water: Prevalent, overlooked, difficult to remove, and unregulated. *Vom Wasser*, 120(4), 97–100. <https://doi.org/10.1002/vomw.202200018>
- Nickel, B., & Grigg, B. (2002). *Telemetry technology in biomarker studies of Pacific harbor seals (Phoca vitulina richardii) in the San Francisco Estuary, California*. San Francisco Estuary Institute.
- OECD. (2021). *Reconciling Terminology of the Universe of Per- and Polyfluoroalkyl Substances: Recommendations and Practical Guidance* (Series on Risk Management No. 61., p. 45). Organisation for Economic Co-operation and Development (OECD). <https://www.oecd.org/chemicalsafety/portal-perfluorinated-chemicals/terminology-per-and-polyfluoroalkyl-substances.pdf>

- Ogunbiyi, O. D., Massenat, N., & Quinete, N. (2024). Dispersion and stratification of Per- and polyfluoroalkyl substances (PFAS) in surface and deep-water profiles: A case study of the Biscayne Bay area. *Science of The Total Environment*, 909, 168413. <https://doi.org/10.1016/j.scitotenv.2023.168413>
- Peter, K. T., Gilbreath, A., Gonzalez, M., Tian, Z., Wong, A., Yee, D., Miller, E. L., Pedro Avellaneda, Chen, D., Patterson, A., Fitzgerald, N., Higgins, C. P., Kolodziej, E. P., & Sutton, R. (2024). Storms Mobilize Organophosphate Esters, Bisphenols, PFASs, and Vehicle-derived Contaminants to San Francisco Bay Watersheds. *Environmental Science: Processes & Impacts*, 26, 1760–1779. <https://doi.org/10.1039/d4em00117f>
- Pickard, H. M., Haque, F., & Sunderland, E. M. (2024). Bioaccumulation of Perfluoroalkyl Sulfonamides (FASA). *Environmental Science & Technology Letters*, 11(4), 350–356. <https://doi.org/10.1021/acs.estlett.4c00143>
- Pickard, H. M., Ruyle, B. J., Thackray, C. P., Chovancova, A., Dassuncao, C., Becanova, J., Vojta, S., Lohmann, R., & Sunderland, E. M. (2022). PFAS and Precursor Bioaccumulation in Freshwater Recreational Fish: Implications for Fish Advisories. *Environmental Science & Technology*, acs.est.2c03734. <https://doi.org/10.1021/acs.est.2c03734>
- Polasko, A. L., Koutnik, V. S., Tsai, K., Alkidim, S., Borthakur, A., Mohanty, S., & Mahendra, S. (2024). Evaluation of historical data on persistent organic pollutants and heavy metals in Lake Baikal: Implications for accumulation in marine environments. *Environmental Research*, 252, 119035. <https://doi.org/10.1016/j.envres.2024.119035>
- Preimesberger, A. L. H. (2020). *Water Quality Standards Technical Support Document: Human Health Protective Water Quality Criteria for Perfluorooctane Sulfonate (PFOS)* (Wq-S6-61a; p. 38). Minnesota Pollution Control Agency.
- Prevedouros, K., Cousins, I. T., Buck, R. C., & Korzeniowski, S. H. (2006). Sources, Fate and Transport of Perfluorocarboxylates. *Environmental Science & Technology*, 40(1), 32–44. <https://doi.org/10.1021/es0512475>
- Pulster, E. L., Rullo, K., Gilbert, S., Ash, T. M., Goetting, B., Campbell, K., Markham, S., & Murawski, S. A. (2022). Assessing per- and polyfluoroalkyl substances (PFAS) in sediments and fishes in a large, urbanized estuary and the potential human health implications. *Frontiers in Marine Science*, 9. <https://doi.org/10.3389/fmars.2022.1046667>
- Rigét, F., Bossi, R., Sonne, C., Vorkamp, K., & Dietz, R. (2013). Trends of perfluorochemicals in Greenland ringed seals and polar bears: Indications of shifts to decreasing trends. *Chemosphere*, 93(8), 1607–1614. <https://doi.org/10.1016/j.chemosphere.2013.08.015>

- Rotander, A., Kärrman, A., Bavel, B. van, Polder, A., Rigét, F., Auðunsson, G. A., Víkingsson, G., Gabrielsen, G. W., Bloch, D., & Dam, M. (2012). Increasing levels of long-chain perfluorocarboxylic acids (PFCAs) in Arctic and North Atlantic marine mammals, 1984–2009. *Chemosphere*, 86(3), 278–285. <https://doi.org/10.1016/j.chemosphere.2011.09.054>
- Routti, H., Gabrielsen, G. W., Herzke, D., Kovacs, K. M., & Lydersen, C. (2016). Spatial and temporal trends in perfluoroalkyl substances (PFASs) in ringed seals (*Pusa hispida*) from Svalbard. *Environmental Pollution*, 214, 230–238. <https://doi.org/10.1016/j.envpol.2016.04.016>
- Routti, H., Krafft, B. A., Herzke, D., Eisert, R., & Oftedal, O. (2015). Perfluoroalkyl substances detected in the world's southernmost marine mammal, the Weddell seal (*Leptonychotes weddellii*). *Environmental Pollution*, 197, 62–67. <https://doi.org/10.1016/j.envpol.2014.11.026>
- Routti, H., Lydersen, C., Hanssen, L., & Kovacs, K. M. (2014). Contaminant levels in the world's northernmost harbor seals (*Phoca vitulina*). *Marine Pollution Bulletin*, 87(1–2), 140–146. <https://doi.org/10.1016/j.marpolbul.2014.08.001>
- Safer States: Bill Tracker for PFAS*. (2026). Safer States. https://www.saferstates.org/bill-tracker/?toxic_chemicals=PFAS
- Saleh, H., Sarkar, D., Zhang, Z., Boufadel, M., & Datta, R. (2025). Per- and Polyfluoroalkyl Substances (PFAS) in Urban Stormwater Runoff: Insights from a Roadside Rain Garden. *Water*, 17(20), 2982. <https://doi.org/10.3390/w17202982>
- Schaider, L. A., Ackerman, J. M., & Rudel, R. A. (2016). Septic systems as sources of organic wastewater compounds in domestic drinking water wells in a shallow sand and gravel aquifer. *Science of The Total Environment*, 547, 470–481. <https://doi.org/10.1016/j.scitotenv.2015.12.081>
- Sedlak, M. D., Benskin, J. P., Wong, A., Grace, R., & Greig, D. J. (2017). Per- and polyfluoroalkyl substances (PFASs) in San Francisco Bay wildlife: Temporal trends, exposure pathways, and notable presence of precursor compounds. *Chemosphere*, 185, 1217–1226. <https://doi.org/10.1016/j.chemosphere.2017.04.096>
- Sedlak, M. D., & Greig, D. J. (2012). Perfluoroalkyl compounds (PFCs) in wildlife from an urban estuary. *J. Environ. Monit.*, 14(1), 146–154. <https://doi.org/10.1039/C1EM10609K>
- Sedlak, M. D., Sutton, R., Wong, A., & Lin, D. (2018). *Per and Polyfluoroalkyl Substances (PFASs) in San Francisco Bay: Synthesis and Strategy* (No. 867). San Francisco Estuary Institute. <https://www.sfei.org/documents/and-polyfluoroalkyl-substances-pfass-san-francisco-bay-synthesis-and-strategy>

- SFBRWQCB. (2020). *MEMO: Interim Final ESLs for PFOS and PFOA*. San Francisco Bay Regional Water Quality Control Board.
https://sj-admin.s3-us-west-2.amazonaws.com/2020_0527_SFRWQCB_Interim%20Final%20Memo.pdf
- SFBRWQCB. (2025). *MEMO: Update to Environmental Screening Levels (ESLs)*. San Francisco Bay Regional Water Quality Control Board.
https://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/ESL/ESL_Summary_Tables_Rev3.pdf
- SFEI. (2026). *Sources to Solutions: Identifying and Preventing PFAS Pollution in San Francisco Bay*. San Francisco Estuary Institute.
<https://www.sfei.org/projects/sources-solutions-identifying-and-preventing-pfas-pollution-san-francisco-bay>
- Simola, J., Kunnasranta, M., Auriola, S., & Akkanen, J. (2026). Perfluorooctanesulfonate (PFOS) exposure in ringed seals (*Pusa saimensis*) in Lake Saimaa, Finland. *Marine Pollution Bulletin*, 224, 119150.
<https://doi.org/10.1016/j.marpolbul.2025.119150>
- Simon, P., Ramacher, M. O. P., Hagemann, S., Matthias, V., Joerss, H., & Bieser, J. (2026). POPE: An annual global half degree emission inventory for PFAS 1950–2020. *Earth System Science Data*, 18(8), 5713–5737.
<https://doi.org/10.5194/essd-18-5713-2026>
- Spaan, K. M., van Noordenburg, C., Plassmann, M. M., Schultes, L., Shaw, S., Berger, M., Heide-Jørgensen, M. P., Rosing-Asvid, A., Granquist, S. M., Dietz, R., Sonne, C., Rigét, F., Roos, A., & Benskin, J. P. (2020). Fluorine Mass Balance and Suspect Screening in Marine Mammals from the Northern Hemisphere. *Environmental Science & Technology*, 54(7), 4046–4058.
<https://doi.org/10.1021/acs.est.9b06773>
- Spaan, K. M., van Noordenburg, C., Plassmann, M. M., Schultes, L., Shaw, S., Berger, M., Heide-Jørgensen, M. P., Rosing-Asvid, A., Granquist, S. M., Dietz, R., Sonne, C., Rigét, F., Roos, A., & Benskin, J. P. (2021). Correction to Fluorine Mass Balance and Suspect Screening in Marine Mammals from the Northern Hemisphere. *Environmental Science & Technology*, 55(9), 6518–6520.
<https://doi.org/10.1021/acs.est.0c06864>
- Su, G., Letcher, R. J., Moore, J. N., Williams, L. L., & Grasman, K. A. (2017). Contaminants of emerging concern in Caspian tern compared to herring gull eggs from Michigan colonies in the Great Lakes of North America. *Environmental Pollution*, 222, 154–164. <https://doi.org/10.1016/j.envpol.2016.12.061>
- Sun, J., Davis, J., Shira, B., Ross, J., Wong, A., Yee, D., Fairey, R., Bonnema, A., Crane, D., Grace, R., Mayfield, R., & Hobbs, J. (2017). *Contaminant Concentrations in Sport Fish from San Francisco Bay: 2014* (No. 806). San Francisco Estuary Institute.

- Sun, J. M., Kelly, B. C., Gobas, F. A. P. C., & Sunderland, E. M. (2022). A food web bioaccumulation model for the accumulation of per- and polyfluoroalkyl substances (PFAS) in fish: How important is renal elimination? *Environmental Science: Processes & Impacts*, 24(8), 1152–1164. <https://doi.org/10.1039/D2EM00047D>
- Sutton, R., Avellaneda, P. M., Mendez, M., Dougherty, J., & Gilbreath, A. N. (2024). *Screening Bay Area Stormwater for Emerging Contaminants: Summary for Managers* (SFEI Contribution No. 1214). San Francisco Estuary Institute. <https://www.sfei.org/documents/screening-bay-area-stormwater-emerging-contaminants-summary-managers>
- Sutton, R., Miller, E., Méndez, M., Wong, A., Paterson, K., Dougherty, J., & Lin, D. (2024). *Contaminants of Emerging Concern in San Francisco Bay: A Strategy for Future Investigations 2024 Revision* (SFEI Contribution No. 1203). San Francisco Estuary Institute. <https://www.sfei.org/documents/contaminants-emerging-concern-san-francisco-bay-strategy-future-investigations-2024>
- SWRCB. (2026). *GeoTracker: PFAS Map*. State Water Resources Control Board. https://geotracker.waterboards.ca.gov/map/pfas_map#
- Szabo, D., Marchiandi, J., Samandra, S., Johnston, J. M., Mulder, R. A., Green, M. P., & Clarke, B. O. (2023). High-resolution temporal wastewater treatment plant investigation to understand influent mass flux of per- and polyfluoroalkyl substances (PFAS). *Journal of Hazardous Materials*, 447, 130854. <https://doi.org/10.1016/j.jhazmat.2023.130854>
- Thompson, K. A., Mortazavian, S., Gonzalez, D. J., Bott, C., Hooper, J., Schaefer, C. E., & Dickenson, E. R. V. (2022). Poly- and Perfluoroalkyl Substances in Municipal Wastewater Treatment Plants in the United States: Seasonal Patterns and Meta-Analysis of Long-Term Trends and Average Concentrations. *ACS ES&T Water*, 2(5), 690–700. <https://doi.org/10.1021/acsestwater.1c00377>
- USEPA. (2024a). *DRAFT Human Health Ambient Water Quality Criteria: PFOA and Related Salts* (EPA 822P24001). Health and Ecological Criteria Division, Office of Science and Technology, Office of Water of the U.S. Environmental Protection Agency. <https://www.epa.gov/system/files/documents/2024-12/pfoa-hhc-draft.pdf>
- USEPA. (2024b). *Draft Human Health Ambient Water Quality Criteria: PFOS and Related Salts* (EPA 822P24002). Health and Ecological Criteria Division, Office of Science and Technology, Office of Water of the U.S. Environmental Protection Agency. <https://www.epa.gov/system/files/documents/2024-12/pfos-hhc-draft.pdf>
- USEPA. (2024c). *Final Freshwater Aquatic Life Ambient Water Quality Criteria and Acute Saltwater Aquatic Life Benchmark for Perfluorooctanoic Acid* (EPA-842-R-24-002). U.S. Environmental Protection Agency Office of Water, Office of Science and Technology, Health and Ecological Criteria Division.

- USEPA. (2024d). *Final Freshwater Aquatic Life Ambient Water Quality Criteria and Acute Saltwater Benchmark for PFOS* (EPA-842-R-24-003). U.S. Environmental Protection Agency Office of Water, Office of Science and Technology, Health and Ecological Criteria Division.
<https://www.epa.gov/system/files/documents/2024-09/pfos-report-2024.pdf>
- Viticoski, R. L., Wang, D., Feltman, M. A., Mulabagal, V., Rogers, S. R., Bliersch, D. M., & Hayworth, J. S. (2022). Spatial distribution and mass transport of Perfluoroalkyl Substances (PFAS) in surface water: A statewide evaluation of PFAS occurrence and fate in Alabama. *Science of The Total Environment*, 836, 155524.
<https://doi.org/10.1016/j.scitotenv.2022.155524>
- Wang, Y., Chang, W., Wang, L., Zhang, Y., Zhang, Y., Wang, M., Wang, Y., & Li, P. (2019). A review of sources, multimedia distribution and health risks of novel fluorinated alternatives. *Ecotoxicology and Environmental Safety*, 182, 109402.
<https://doi.org/10.1016/j.ecoenv.2019.109402>
- Wang, Z., Boucher, J. M., Scheringer, M., Cousins, I. T., & Hungerbühler, K. (2017). Toward a Comprehensive Global Emission Inventory of C₄–C₁₀ Perfluoroalkanesulfonic Acids (PFSA) and Related Precursors: Focus on the Life Cycle of C₈-Based Products and Ongoing Industrial Transition. *Environmental Science & Technology*, 51(8), 4482–4493.
<https://doi.org/10.1021/acs.est.6b06191>
- Wang, Z., Cousins, I. T., Scheringer, M., Buck, R. C., & Hungerbühler, K. (2014). Global emission inventories for C₄–C₁₄ perfluoroalkyl carboxylic acid (PFCA) homologues from 1951 to 2030, Part I: Production and emissions from quantifiable sources. *Environment International*, 70, 62–75.
<https://doi.org/10.1016/j.envint.2014.04.013>
- Wang, Z., Cousins, I. T., Scheringer, M., & Hungerbuehler, K. (2015). Hazard assessment of fluorinated alternatives to long-chain perfluoroalkyl acids (PFAAs) and their precursors: Status quo, ongoing challenges and possible solutions. *Environment International*, 75, 172–179.
<https://doi.org/10.1016/j.envint.2014.11.013>
- Water Code Sections 13267 and 13383 Order for the Determination of the Presence of Per- and Polyfluoroalkyl Substances at Publicly Owned Treatment Works, Pub. L. No. ORDER WQ 2020-0015-DWQ, Water Code sections 13267 and 13383 (2020).
- Wu, Y., Simon, K. L., Best, D. A., Bowerman, W., & Venier, M. (2020). Novel and legacy per- and polyfluoroalkyl substances in bald eagle eggs from the Great Lakes region. *Environmental Pollution*, 260, 113811.
<https://doi.org/10.1016/j.envpol.2019.113811>

- Zhao, Z., Cheng, X., Hua, X., Jiang, B., Tian, C., Tang, J., Li, Q., Sun, H., Lin, T., Liao, Y., & Zhang, G. (2020). Emerging and legacy per- and polyfluoroalkyl substances in water, sediment, and air of the Bohai Sea and its surrounding rivers. *Environmental Pollution*, 263, 114391. <https://doi.org/10.1016/j.envpol.2020.114391>
- Zhi, Y., Lu, X., Munoz, G., Yeung, L. W. Y., De Silva, A. O., Hao, S., He, H., Jia, Y., Higgins, C. P., & Zhang, C. (2024). Environmental Occurrence and Biotic Concentrations of Ultrashort-Chain Perfluoroalkyl Acids: Overlooked Global Organofluorine Contaminants. *Environmental Science & Technology*, 58(49), 21393–21410. <https://doi.org/10.1021/acs.est.4c04453>

Appendix A: PFAS Summary Tables by Matrix

Table A1: 2020 and 2022 Effluent Concentrations (ng/L, municipal POTWs only)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/L	22	1.1	25.9	100%	2.3	35.8	18.6	6.1	8.8
PFPeA	ng/L	22	0.5	11.5	100%	2.7	29.6	23.7	7.7	9.9
PFHxA	ng/L	22	0.3	71.6	95%	MDL	30.4	27.9	15.7	16.2
PFHpA	ng/L	22	0.3	5.8	100%	1.3	5.5	3.8	2.0	2.4
PFOA	ng/L	22	0.3	5.9	100%	3.5	12.1	8.3	6.1	6.3
PFNA	ng/L	22	0.3	7.5	91%	MDL	1.6	1.4	0.9	0.9
PFDA	ng/L	22	0.3	9.5	91%	MDL	2.8	2.1	1.3	1.3
PFUnA	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
PFDoA	ng/L	22	0.2	4.6	0%	MDL	MDL	MDL	MDL	MDL
PFTTrDA	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
PFTeDA	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
PFBS	ng/L	22	0.3	19.9	50%	MDL	5.7	4.9	0.9	2.1
PFPeS	ng/L	22	0.3	5.8	41%	MDL	3.0	1.2	MDL	0.4
PFHxS	ng/L	22	0.3	42.9	100%	0.4	14.3	10.3	2.9	4.2
PFHpS	ng/L	22	0.3	5.8	5%	MDL	0.4	MDL	MDL	0.0
PFOS	ng/L	22	0.3	17.6	95%	MDL	13.0	9.7	4.5	5.4
PFNS	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/L	22	0.3	5.8	5%	MDL	0.7	MDL	MDL	0.0
PFDoS	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/L	22	1.1	23	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/L	22	1	20.7	32%	MDL	6.0	3.1	MDL	1.0
8:2 FTS	ng/L	22	0.9	19.5	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/L	22	1.1	23	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/L	22	6.7	237	23%	MDL	20.6	18.2	0.0	3.8
7:3 FTCA	ng/L	22	6.7	237	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/L	22	0.3	5.8	27%	MDL	0.7	0.6	MDL	0.2
N-MeFOSA	ng/L	22	0.3	5.8	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/L	22	0.7	16.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/L	22	0.3	18.6	59%	MDL	2.5	1.8	0.5	0.6
N-EtFOSAA	ng/L	22	0.3	5.8	27%	MDL	2.1	0.7	MDL	0.2
N-MeFOSE	ng/L	22	2.7	57.8	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/L	22	2.5	57.8	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/L	22	1.1	18.1	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/L	22	1.1	18.1	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/L	22	0.5	11.6	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/L	22	0.3	4.5	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/L	22	0.5	9.1	5%	MDL	1.1	MDL	MDL	0.1
9CI-PF3ONS	ng/L	22	1.1	18.2	0%	MDL	MDL	MDL	MDL	MDL
11CI-PF3OUdS	ng/L	22	1.1	18.2	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/L	22	0.3	4.5	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/L	22				26.3	111.8	102.4	57.4	63.9

Table A2: 2020 and 2022 Influent Concentrations (ng/L municipal POTWs only)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/L	21	5.7	26.6	48%	MDL	23.4	15.4	MDL	5.4
PFPeA	ng/L	21	2.8	13.3	100%	3	14.2	9.2	6.3	6.7
PFHxA	ng/L	21	1.4	6.6	100%	3.8	17.4	10.3	7.8	7.9
PFHpA	ng/L	21	1.4	6.6	62%	MDL	3.2	2.9	1.7	1.4
PFOA	ng/L	21	1.4	6.6	100%	1.73	6.6	6.1	3.7	4.0
PFNA	ng/L	21	1.4	12.2	5%	MDL	1.8	MDL	MDL	0.1
PFDA	ng/L	21	1.4	9.7	14%	MDL	2.2	1.6	MDL	0.3
PFUnA	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFDaA	ng/L	21	1.2	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFTTrDA	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFTeDA	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFBS	ng/L	21	1.4	13	24%	MDL	10.5	5.3	MDL	1.4
PFPeS	ng/L	21	1.4	6.7	5%	MDL	2.3	MDL	MDL	0.1
PFHxS	ng/L	21	1.4	13.9	57%	MDL	12.6	5.3	3.1	3.0
PFHpS	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/L	21	1.4	24.9	38%	MDL	20.9	15.7	MDL	5.5
PFNS	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/L	21	1.4	9.8	0%	MDL	MDL	MDL	MDL	MDL
PFDoS	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/L	21	5.7	26.6	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/L	21	5.1	23.9	0%	MDL	MDL	MDL	MDL	MDL
8:2 FTS	ng/L	21	5.1	26.6	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/L	21	5.7	26.6	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/L	21	35.5	166	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/L	21	35.5	166	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/L	21	1.4	6.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSA	ng/L	21	1.5	7.6	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/L	21	3.6	16.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/L	21	1.4	6.6	5%	MDL	1.6	MDL	MDL	0.1
N-EtFOSAA	ng/L	21	1.4	6.6	5%	MDL	2.9	MDL	MDL	0.1
N-MeFOSE	ng/L	21	14.2	66.4	5%	MDL	40.5	MDL	MDL	1.9
N-EtFOSE	ng/L	21	10.6	49.7	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/L	21	5.4	7.7	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/L	21	5.7	8.1	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/L	21	2.8	4.1	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/L	21	1.4	2	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/L	21	2.8	4	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/L	21	5.7	8.1	0%	MDL	MDL	MDL	MDL	MDL
11CI-PF3OUdS	ng/L	21	5.7	8	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/L	21	1.4	2	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/L	21				9.5	106.4	58.6	34.4	37.9

Table A3: 2020 and 2022 Biosolids Concentrations (ng/g, municipal POTWs only)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	19	1.1	18.4	16%	MDL	3.0	1.6	MDL	0.4
PFPeA	ng/g	19	0.5	9.2	26%	MDL	4.6	1.6	MDL	0.6
PFHxA	ng/g	19	0.3	29.2	95%	MDL	7.3	5.5	1.3	2.2
PFHpA	ng/g	19	0.3	4.6	16%	MDL	2.2	0.9	MDL	0.2
PFOA	ng/g	19	0.3	4.6	100%	0.5	14.7	9.5	0.9	2.7
PFNA	ng/g	19	0.3	4.6	89%	MDL	6.6	5.8	0.9	1.6
PFDA	ng/g	19	0.3	4.6	100%	1.2	14.7	9.7	3.9	4.7
PFUnA	ng/g	19	0.3	4.6	95%	MDL	4.4	3.1	2.0	1.9
PFDoA	ng/g	19	0.2	3.9	100%	0.8	6.0	5.5	3.3	3.4
PFTTrDA	ng/g	19	0.3	4.6	89%	MDL	3.3	1.8	1.3	1.2
PFTeDA	ng/g	19	0.3	4.6	79%	MDL	2.2	2.0	1.5	1.3
PFBS	ng/g	19	0.3	66.8	5%	MDL	4.8	MDL	MDL	0.3
PFPeS	ng/g	19	0.3	10.9	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	19	0.3	4.6	11%	MDL	1.9	MDL	MDL	0.2
PFHpS	ng/g	19	0.3	4.6	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/g	19	0.3	4.6	89%	MDL	49.0	30.9	12.6	13.8
PFNS	ng/g	19	0.3	4.6	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	19	0.3	4.6	11%	MDL	3.9	MDL	MDL	0.2
PFDoS	ng/g	19	0.3	4.6	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	19	1.1	18.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	19	1	16.6	5%	MDL	7.2	MDL	MDL	0.4
8:2 FTS	ng/g	19	0.9	15.6	16%	MDL	2.2	1.1	MDL	0.3
3:3 FTCA	ng/g	19	1.1	18.4	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	19	6.8	115	95%	MDL	154.0	149.4	92.6	92.1
7:3 FTCA	ng/g	19	6.8	115	79%	MDL	30.6	28.0	18.7	15.6
PFOSA	ng/g	19	0.3	4.6	95%	MDL	3.6	1.7	1.1	1.1
N-MeFOSA	ng/g	19	0.3	4.6	5%	MDL	0.5	MDL	MDL	0.0
N-EtFOSA	ng/g	19	0.7	12.9	5%	MDL	1.0	MDL	MDL	0.1
N-MeFOSAA	ng/g	19	0.3	5.5	84%	MDL	40.8	29.5	7.6	11.0
N-EtFOSAA	ng/g	19	0.3	4.6	95%	MDL	122.0	20.5	6.9	13.4
N-MeFOSE	ng/g	19	2.7	46	89%	MDL	25.0	20.9	13.2	13.0
N-EtFOSE	ng/g	19	2.2	46	84%	MDL	41.7	8.7	5.6	7.1
HFPO-DA	ng/g	19	1.1	1.7	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	19	1.1	1.8	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	19	0.5	0.9	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	19	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	19	0.5	0.9	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/g	19	1.1	1.8	0%	MDL	MDL	MDL	MDL	MDL
11CI-PF3OUdS	ng/g	19	1.1	1.8	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	19	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	19				43.9	323.3	243.3	201.6	188.4

Table A4: 2018 - 2025 Stormwater Concentrations (ng/L)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/L	72	1.5	5.4	82%	MDL	32	15.3	5.8	7.3
PFPeA	ng/L	72	0.4	22.4	89%	MDL	41	14	5.7	7.3
PFHxA	ng/L	72	0.4	2.2	86%	MDL	180	21.1	7.4	13.3
PFHpA	ng/L	72	0.2	2.2	93%	MDL	51	6.6	2.7	4.3
PFOA	ng/L	72	0.4	1.8	90%	MDL	110	13.7	6.9	9.2
PFNA	ng/L	72	0.2	11.2	75%	MDL	22	3.2	1.4	2
PFDA	ng/L	72	0.2	1.3	93%	MDL	54	7.6	2.5	4.6
PFUnA	ng/L	72	0.4	2.4	42%	MDL	7.5	1.5	MDL	0.6
PFDaA	ng/L	72	0.3	1.2	63%	MDL	11	2.5	0.7	1.1
PFTTrDA	ng/L	72	0.4	2.8	3%	MDL	4.1	MDL	MDL	0.1
PFTeDA	ng/L	72	0.4	3.1	10%	MDL	1.74	MDL	MDL	0.1
PFBS	ng/L	72	0.1	1.3	100%	0.62	41	7.9	2.7	4.3
PFPeS	ng/L	72	0.2	1.4	8%	MDL	1.8	MDL	MDL	0.1
PFHxS	ng/L	72	0.4	1.5	79%	MDL	15	3.4	1.2	1.6
PFHpS	ng/L	72	0.1	1.3	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/L	72	0.4	5.6	90%	MDL	81	14	6.6	8.7
PFNS	ng/L	72	0.3	1.3	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/L	72	0.2	1.3	1%	MDL	0.62	MDL	MDL	0.01
PFDoS	ng/L	72	0.4	2.2	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/L	72	0.2	5.6	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/L	72	1.3	5.4	10%	MDL	5.2	MDL	MDL	0.34
8:2 FTS	ng/L	72	0.3	4.6	3%	MDL	0.83	MDL	MDL	0.02
3:3 FTCA	ng/L	72	0.3	11.2	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/L	72	0.2	33.5	4%	MDL	5.2	MDL	MDL	0.1
7:3 FTCA	ng/L	72	0.4	33.5	1%	MDL	1.4	MDL	MDL	0.02
PFOSA	ng/L	72	0.4	2.2	40%	MDL	3.6	1.1	MDL	0.4
N-MeFOSA	ng/L	72	0.3	5.6	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/L	72	0.6	5.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/L	72	0.4	224.4	3%	MDL	1.52	MDL	MDL	0.03
N-EtFOSAA	ng/L	43	0.4	1.3	2%	MDL	0.524	MDL	MDL	0.01
N-MeFOSE	ng/L	52	1	13.4	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/L	52	0.6	13.4	4%	MDL	3.6	MDL	MDL	0.1
HFPO-DA	ng/L	43	1.1	5.6	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/L	72	0.3	5.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/L	52	0.4	2.7	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/L	32	0.2	0.8	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/L	32	0.2	1.7	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/L	72	0.2	5.4	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/L	72	0.2	5.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/L	52	0.2	1.3	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/L	72				4.1	573.6	103.9	46.2	64.3

Table A5: 2023 Sediment Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	54	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOA	ng/g	54	0.03	0.1	76%	MDL	0.33	0.15	0.06	0.07
PFNA	ng/g	54	0.03	0.1	44%	MDL	0.14	0.07	MDL	0.04
PFDA	ng/g	54	0.03	0.1	69%	MDL	0.24	0.17	0.11	0.11
PFUnA	ng/g	54	0.03	0.1	67%	MDL	0.09	0.08	0.06	0.05
PFDoA	ng/g	54	0.03	0.1	76%	MDL	0.23	0.11	0.07	0.07
PFTTrDA	ng/g	54	0.03	0.1	19%	MDL	0.15	0.02	MDL	0.01
PFTeDA	ng/g	54	0.03	0.1	28%	MDL	0.22	0.05	MDL	0.02
PFBS	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	54	0.03	0.1	2%	MDL	0.05	MDL	MDL	MDL
PFHpS	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/g	54	0.03	0.1	100%	0.05	0.67	0.55	0.29	0.3
PFNS	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	54	0.03	0.1	59%	MDL	0.23	0.06	MDL	0.02
PFDoS	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	54	0.1	0.3	9%	MDL	14.2	0	MDL	0.3
8:2 FTS	ng/g	54	0.1	0.3	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	54	0.8	2.4	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	54	0.8	2.4	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/g	54	0.03	0.1	37%	MDL	0.08	0.05	MDL	0.02
N-MeFOSA	ng/g	54	0.03	0.1	22%	MDL	0.06	0.05	MDL	0.01
N-EtFOSA	ng/g	54	0.1	0.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	54	0.03	0.1	6%	MDL	0.06	MDL	MDL	0.003
N-EtFOSAA	ng/g	54	0.03	0.1	76%	MDL	0.42	0.24	0.06	0.1
N-MeFOSE	ng/g	54	0.3	1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	54	0.3	1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	54	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	54	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/g	54	0.1	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	54	0.03	0.1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	54				0.05	15.87	1.66	0.8	1.1

Table A6: 2020 Sediment Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	12	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	12	0.04	0.1	8%	MDL	0.043	MDL	MDL	0.004
PFHpA	ng/g	12	0.04	0.1	8%	MDL	0.045	MDL	MDL	0.004
PFOA	ng/g	12	0.04	0.1	42%	MDL	0.15	0.08	MDL	0.04
PFNA	ng/g	12	0.04	0.1	25%	MDL	0.06	0.05	MDL	0.01
PFDA	ng/g	12	0.04	0.1	42%	MDL	0.06	0.05	MDL	0.02
PFUnA	ng/g	12	0.04	0.1	42%	MDL	0.05	0.04	MDL	0.02
PFDaA	ng/g	12	0.03	0	25%	MDL	0.04	0.04	MDL	0.01
PFTTrDA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFTTeDA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFBS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHpS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/g	12	0.04	0.1	92%	MDL	0.16	0.13	0.11	0.1
PFNS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFDoS	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
8:2 FTS	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	12	0.9	1.3	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	12	0.9	1.3	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	12	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	12	0	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/g	12	0	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	12	0.4	0.5	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	12	0.4	0.5	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	12	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PfMBA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
PfMPA	ng/g	12	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
11CI-PF3OUdS	ng/g	12	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	12	0.04	0.1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	12				MDL	0.33	0.33	0.19	0.2

Table A7: 2021 - 2024 Open Water Concentrations (ng/L)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/L	73	1.6	2.3	38%	MDL	4.9	3.3	MDL	1.1
PFPeA	ng/L	73	0.8	1.1	63%	MDL	5.1	3.5	1.6	1.6
PFHxA	ng/L	73	0.4	0.6	85%	MDL	6.0	4.3	1.6	1.9
PFHpA	ng/L	73	0.4	0.6	53%	MDL	1.9	1.4	0.5	0.5
PFOA	ng/L	73	0.4	0.6	74%	MDL	4.0	2.5	1.3	1.3
PFNA	ng/L	73	0.4	0.6	18%	MDL	0.8	0.5	MDL	0.1
PFDA	ng/L	73	0.4	0.6	11%	MDL	0.7	0.4	MDL	0.1
PFUnA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFDaA	ng/L	73	0.3	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFTTrDA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFTeDA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFBS	ng/L	73	0.4	0.6	55%	MDL	2.7	1.6	0.6	0.6
PFPeS	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/L	73	0.4	0.6	60%	MDL	2.5	2.0	0.7	0.8
PFHpS	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/L	73	0.4	0.6	73%	MDL	5.5	4.2	1.5	1.8
PFNS	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFDoS	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/L	73	1.5	2	14%	MDL	178.0	2.2	MDL	2.9
8:2 FTS	ng/L	73	1.4	2.2	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/L	73	10.2	14.1	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/L	73	10.2	14.1	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/L	73	0.4	0.6	11%	MDL	0.9	0.3	MDL	0.1
N-MeFOSA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/L	73	1.1	1.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/L	73	4.1	5.6	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/L	73	3.2	5.7	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/L	73	0.8	1.1	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/L	73	0.8	1.1	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/L	73	1.6	2.3	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/L	73	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/L	73				MDL	178.6	25.7	7.6	12.4

Table A8: 2021 - 2024 Near Field Water Concentrations (ng/L)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/L	25	1.7	13.3	80%	MDL	26.8	10.5	3.5	5.3
PFPeA	ng/L	25	0.8	6.6	100%	1.9	28.1	11.2	4.5	6.8
PFHxA	ng/L	25	0.4	3.3	100%	2.2	24.9	11.8	5.3	7.2
PFHpA	ng/L	25	0.4	3.3	96%	MDL	11.2	3.8	1.6	2.3
PFOA	ng/L	25	0.4	3.3	100%	1.5	20.2	8.3	3.8	5.4
PFNA	ng/L	25	0.4	3.3	56%	MDL	4.2	1.7	0.5	0.8
PFDA	ng/L	25	0.4	3.3	68%	MDL	5.3	3.0	0.7	1.1
PFUnA	ng/L	25	0.4	3.3	4%	MDL	1.0	MDL	MDL	0.0
PFDaA	ng/L	25	0.3	3.3	8%	MDL	1.5	MDL	MDL	0.1
PFTTrDA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFTeDA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFBS	ng/L	25	0.4	3.3	92%	MDL	7.5	6.9	2.1	3.1
PFPeS	ng/L	25	0.4	3.3	36%	MDL	3.1	2.1	MDL	0.6
PFHxS	ng/L	25	0.4	3.3	92%	MDL	11.3	9.3	1.9	3.7
PFHpS	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/L	25	0.4	3.3	100%	2.0	17.3	12.7	4.6	7.1
PFNS	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFDoS	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/L	25	1.7	13.3	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/L	25	1.5	12	28%	MDL	8.9	2.0	MDL	0.7
8:2 FTS	ng/L	25	1.4	13.3	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/L	25	1.4	13.3	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/L	25	10.4	82.9	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/L	25	10.4	82.9	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSA	ng/L	25	0.4	3.8	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/L	25	1.2	8.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/L	25	0.4	3.3	8%	MDL	0.9	MDL	MDL	0.1
N-MeFOSE	ng/L	25	4.2	33.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/L	25	4.2	24.8	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/L	25	1.7	12.6	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/L	25	1.7	13.3	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/L	25	0.8	6.6	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/L	25	0.8	6.6	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/L	25	1.7	13.3	0%	MDL	MDL	MDL	MDL	MDL
11CI-PF3OUdS	ng/L	25	1.7	13.3	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/L	25	0.4	3.3	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/L	25				10.9	143.0	77.7	31.5	44.5

Table A9: 2019 Sportfish Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	33	0.4	0.8	3%	MDL	0.96	MDL	MDL	0.03
PFPeA	ng/g	33	0.2	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFOA	ng/g	33	0.1	0.2	6%	MDL	0.24	MDL	MDL	0.03
PFNA	ng/g	33	0.1	0.2	15%	MDL	0.31	0.22	MDL	0.04
PFDA	ng/g	33	0.1	0.2	64%	MDL	1.75	0.68	0.17	0.26
PFUnA	ng/g	33	0.1	0.2	82%	MDL	0.88	0.35	0.21	0.21
PFDoA	ng/g	33	0.1	0.2	91%	MDL	1.46	0.76	0.43	0.41
PFTTrDA	ng/g	33	0.1	0.2	88%	MDL	0.74	0.61	0.28	0.3
PFTeDA	ng/g	33	0.1	0.2	88%	MDL	0.79	0.61	0.37	0.36
PFBS	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	33	0.1	0.2	36%	MDL	0.57	0.26	MDL	0.09
PFHpS	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/g	33	0.1	0.6	91%	MDL	10.4	7.94	2.07	2.81
PFNS	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	33	0.1	0.2	48%	MDL	0.64	0.28	MDL	0.12
PFDoS	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	33	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	33	0.3	0.7	3%	MDL	1.22	MDL	MDL	0.04
8:2 FTS	ng/g	33	0.3	0.8	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	33	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	33	2.4	2.5	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	33	2.4	2.5	9%	MDL	3.2	MDL	MDL	0.27
PFOSA	ng/g	33	0.1	0.2	94%	MDL	1.72	1.31	0.28	0.5
N-MeFOSA	ng/g	33	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	33	0.3	0.5	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	33	0.1	0.2	3%	MDL	0.224	MDL	MDL	0.01
N-EtFOSAA	ng/g	33	0.1	0.2	12%	MDL	0.296	0.16	MDL	0.03
N-MeFOSE	ng/g	33	1	2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	33	0.7	1.5	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	33	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	33	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	33	0.2	0.7	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	33	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	33	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/g	33	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/g	33	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	33	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	33				MDL	16.45	12.02	4.76	5.53

Table A10: 2024 Sportfish Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	67	0.2	0.3	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	67	0.1	0.3	4%	MDL	0.76	MDL	MDL	0.02
PFHpA	ng/g	67	0.1	0.2	3%	MDL	0.12	MDL	MDL	0.003
PFOA	ng/g	67	0.1	0.2	21%	MDL	0.34	0.16	MDL	0.04
PFNA	ng/g	67	0.1	0.2	25%	MDL	0.97	0.16	MDL	0.05
PFDA	ng/g	67	0.1	0.2	66%	MDL	1.24	0.39	0.16	0.18
PFUnA	ng/g	67	0.1	0.2	45%	MDL	0.42	0.27	MDL	0.09
PFDoA	ng/g	67	0.07	0.12	78%	MDL	0.83	0.37	0.18	0.2
PFTTrDA	ng/g	67	0.1	0.4	36%	MDL	1.78	0.44	MDL	0.18
PFTeDA	ng/g	67	0.1	0.3	34%	MDL	0.66	0.45	MDL	0.11
PFBS	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	67	0.1	0.2	1%	MDL	0.19	MDL	MDL	0.003
PFHxS	ng/g	67	0.1	0.2	27%	MDL	0.4	0.22	MDL	0.06
PFHpS	ng/g	67	0.1	0.2	1%	MDL	0.15	MDL	MDL	0.002
PFOS	ng/g	67	0.1	0.2	99%	MDL	8.07	3.54	1.15	1.92
PFNS	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	67	0.1	0.2	37%	MDL	0.33	0.17	MDL	0.06
PFDoS	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	67	0.3	0.6	9%	MDL	66.6	MDL	MDL	2.07
8:2 FTS	ng/g	67	0.3	0.5	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	67	2.3	3.9	4%	MDL	3.38	MDL	MDL	0.13
7:3 FTCA	ng/g	67	2.3	3.9	6%	MDL	5.07	MDL	MDL	0.24
PFOSA	ng/g	67	0.1	0.2	76%	MDL	1.35	0.97	0.17	0.35
N-MeFOSA	ng/g	67	0.1	0.2	1%	MDL	0.27	MDL	MDL	0.004
N-EtFOSA	ng/g	67	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/g	67	0.1	0.2	4%	MDL	0.16	MDL	MDL	0.01
N-MeFOSE	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	67	0.9	1.5	10%	MDL	2.17	0.4	0	0.15
HFPO-DA	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	67	0.2	0.3	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	67	0.2	0.3	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/g	67	0.4	0.6	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	67	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	67				0	70.14	10	3.3	5.9

Table A11: 2023 Preyfish Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	39	0.4	0.8	18%	MDL	2.81	1.12	MDL	0.26
PFPeA	ng/g	39	0.2	0.3	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	39	0.1	0.4	8%	MDL	0.48	MDL	MDL	0.02
PFHpA	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOA	ng/g	39	0.1	0.1	46%	MDL	0.19	0.2	MDL	0.06
PFNA	ng/g	39	0.1	0.1	26%	MDL	0.37	0.13	MDL	0.04
PFDA	ng/g	39	0.1	0.1	74%	MDL	0.81	0.55	0.18	0.25
PFUnA	ng/g	39	0.1	0.1	82%	MDL	0.58	0.32	0.16	0.18
PFDoA	ng/g	39	0.1	0.1	90%	MDL	1.06	0.59	0.34	0.36
PFTTrDA	ng/g	39	0.1	0.1	38%	MDL	0.82	0.31	MDL	0.12
PFTeDA	ng/g	39	0.1	0.2	79%	MDL	0.73	0.46	0.25	0.24
PFBS	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	39	0.1	0.1	31%	MDL	0.43	0.21	MDL	0.06
PFHpS	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOS	ng/g	39	0.1	0.1	100%	0.3	10.1	6.2	1.5	2.6
PFNS	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	39	0.1	0.1	67%	MDL	0.27	0.2	0.13	0.11
PFDoS	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	39	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	39	0.4	0.4	21%	MDL	1.36	0.5	MDL	0.14
8:2 FTS	ng/g	39	0.3	0.3	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	39	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	39	2.4	2.5	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	39	2.4	2.7	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/g	39	0.1	0.1	92%	MDL	2.37	1.11	0.25	0.53
N-MeFOSA	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	39	0.3	0.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	39	0.1	0.1	15%	MDL	0.66	0.13	MDL	0.04
N-EtFOSAA	ng/g	39	0.1	0.1	23%	MDL	1.35	0.32	MDL	0.1
N-MeFOSE	ng/g	39	0.1	1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	39	1	1.1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	39	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	39	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	39	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	39	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/g	39	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/g	39	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	39	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
SumPFAS	ng/g	39				0.5	18.8	10.8	3.4	5.1

Table A12: 2018 Bird Egg Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	6	0.5	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	6	0.5	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	6	0.5	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	6	0.5	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFOA	ng/g	6	0.5	0.5	50%	MDL	2.2	2	0.7	0.9
PFNA	ng/g	6	0.5	0.5	100%	0.6	5.1	5	2.7	2.8
PFDA	ng/g	6	0.5	0.5	100%	1	19.9	19.4	8.7	9.9
PFUnA	ng/g	6	0.5	0.5	100%	1.6	8.7	7.9	3.1	4.4
PFDoA	ng/g	6	0.5	0.5	100%	2.5	19.2	17.3	4.4	8.1
PFBS	ng/g	6	1	1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	6	1	1	67%	MDL	7.2	6.8	2.6	3.1
PFOS	ng/g	6	1	1	100%	14.6	288	284	101.5	135
PFOSA	ng/g	6	0.5	0.5	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	6				20.3	345.5	339.9	125.5	164.3

Table A13: 2022 Bird Egg Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	9	0.4	1.2	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	9	0.2	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	9	0.1	0.3	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	9	0.1	0.1	67%	MDL	0.3	0.3	MDL	0.1
PFOA	ng/g	9	0.1	0.2	100%	0.5	6	5.63	1.1	2.3
PFNA	ng/g	9	0.1	0.1	100%	1.38	9.4	7.44	3.5	4.1
PFDA	ng/g	9	0.1	0.1	100%	5.56	24.7	15.74	9.49	10.4
PFUnA	ng/g	9	0.1	0.1	100%	2.3	8.8	5.7	4	4.1
PFDaA	ng/g	9	0.1	0.1	100%	2.8	15.5	8.2	5.4	5.8
PFTTrDA	ng/g	9	0.1	0.1	100%	1.1	4.7	3.1	1.9	2.2
PFTeDA	ng/g	9	0.1	0.1	100%	1.3	5.3	3.1	1.8	2.2
PFBS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	9	0.1	0.1	100%	0.7	3.3	2.8	1.2	1.6
PFHpS	ng/g	9	0.1	0.1	100%	0.19	1.67	1.11	0.55	0.64
PFOS	ng/g	9	0.1	0.3	100%	36.8	251	173.4	78.2	101.5
PFNS	ng/g	9	0.1	0.1	67%	MDL	0.3	0.2	MDL	0.1
PFDS	ng/g	9	0.1	0.1	100%	0.8	4.1	4	1.6	1.9
PFDoS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	9	0.3	0.7	100%	0.6	6.8	5.9	3.7	3.2
8:2 FTS	ng/g	9	0.3	0.3	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	9	2.3	2.5	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	9	2.3	2.5	100%	3.7	63.5	42.8	27.3	27.3
PFOSA	ng/g	9	0.1	0.1	100%	0.11	0.32	0.24	0.19	0.2
N-MeFOSA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	9	0.3	0.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	9	0.1	0.1	11%	MDL	0.15	MDL	MDL	MDL
N-EtFOSAA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	9	0.1	1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	9	0.1	1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	9	0.1	1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	9	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	9	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
9Cl-PF3ONS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
11Cl-PF3OUdS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	9	0.1	1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	9				76.1	399.3	270.5	130.8	167.6

Table A14: 2024 Bird Egg data (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	9	0.4	1.2	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	9	0.2	0.7	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	9	0.1	0.5	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	9	0.1	0.2	11%	MDL	0.157	0.03	MDL	0.02
PFOA	ng/g	9	0.1	0.3	100%	0.37	5.41	3.67	0.92	1.86
PFNA	ng/g	9	0.1	0.1	100%	1.78	5.4	4.66	2.64	3.08
PFDA	ng/g	9	0.1	0.1	100%	5.6	13.8	13.56	7.23	8.55
PFUnA	ng/g	9	0.1	0.1	100%	2.31	4.48	4.31	3.38	3.34
PFDaA	ng/g	9	0.1	0.1	100%	3.16	8.84	8.62	5.61	5.62
PFTTrDA	ng/g	9	0.1	0.1	100%	1	3.36	2.9	2.28	2.17
PFTeDA	ng/g	9	0.1	0.1	100%	1.84	4.25	3.9	2.65	2.96
PFBS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHxS	ng/g	9	0.1	0.1	100%	0.76	3.4	3.37	1.52	1.77
PFHpS	ng/g	9	0.1	0.1	100%	0.14	0.71	0.69	0.37	0.4
PFOS	ng/g	9	0.1	0.1	100%	32.6	165	141	70.3	82
PFNS	ng/g	9	0.1	0.1	33%	MDL	0.25	0.22	MDL	0.06
PFDS	ng/g	9	0.1	0.1	100%	1.13	2.2	2.02	1.4	1.51
PFDoS	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
8:2 FTS	ng/g	9	0.3	0.3	11%	MDL	1.2	0.24	MDL	0.13
3:3 FTCA	ng/g	9	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
5:3 FTCA	ng/g	9	2.5	2.5	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	9	2.5	2.5	100%	10.4	24.8	21.9	19.1	18.6
PFOSA	ng/g	9	0.1	0.1	78%	MDL	0.25	0.21	0.2	0.14
N-MeFOSA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	9	0.3	0.3	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	9	1	0.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	9	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMBA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	9	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/g	9	0.4	0.4	11%	MDL	1.05	0.21	MDL	0.12
11CI-PF3OUdS	ng/g	9	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	9	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	9				76.6	232.7	199.3	114.2	132.5

Table A15: 2023/2024 Harbor Seal Serum Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	6	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
PFPeA	ng/g	6	0.2	0.8	0%	MDL	MDL	MDL	MDL	MDL
PFHxA	ng/g	6	0.1	1.8	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	6	0.1	0.8	17%	MDL	0.13	0.06	MDL	0.02
PFOA	ng/g	6	0.1	13.2	100%	0.13	0.86	0.83	0.65	0.56
PFNA	ng/g	6	0.1	1.8	100%	1.69	6.21	5.37	2.6	3.27
PFDA	ng/g	6	0.1	0.8	100%	1.29	6.16	5.69	1.75	2.95
PFUnA	ng/g	6	0.1	0.8	100%	2.21	5.56	4.22	2.58	3.02
PFDaA	ng/g	6	0.1	0.8	100%	0.56	3.73	3.22	1.57	1.79
PFTTrDA	ng/g	6	0.1	0.2	83%	MDL	1.83	1.81	1.16	1.12
PFTeDA	ng/g	6	0.1	0.2	100%	0.26	1.48	1.25	0.6	0.71
PFBS	ng/g	6	0.1	5	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	6	0.1	0.2	50%	MDL	0.23	0.21	MDL	0.09
PFHxS	ng/g	6	0.1	7.4	100%	2.67	14.1	13.45	9.33	8.68
PFHpS	ng/g	6	0.1	0.2	100%	0.38	1.79	1.73	1.01	1.12
PFOS	ng/g	6	0.1	24.5	100%	31.5	70.4	65.75	51.95	49.9
PFNS	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	6	0.1	0.2	100%	0.45	1.27	1.19	0.62	0.76
PFDoS	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	6	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	6	0.4	0.8	0%	MDL	MDL	MDL	MDL	MDL
8:2 FTS	ng/g	6	0.4	0.7	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	6	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
5:3 FTCA	ng/g	6	2.6	5.3	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	6	2.6	5.3	0%	MDL	MDL	MDL	MDL	MDL
PFOSA	ng/g	6	0.1	2.5	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSA	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	6	0.3	0.6	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	6	1	2.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	6	1	2.1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	6	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	6	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	6	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
PFMBA	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	6	0.2	0.4	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/g	6	0.4	0.9	0%	MDL	0	0	MDL	0
11CI-PF3OUdS	ng/g	6	0.4	0.9	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	6	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	6				47.1	97.1	90.8	78	74

Table A16: 2023/2024 Harbor Seal Liver Concentrations (ng/g ww)

Analytes	Units	Number of Samples	MDLs (Range)		Detection Frequency	Minimum	Max	Q90	Median	Mean1
PFBA	ng/g	7	0.4	0.4	14%	MDL	0.49	0.2	MDL	0.07
PFPeA	ng/g	7	0.2	0.2	43%	MDL	0.99	0.74	MDL	0.27
PFHxA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFHpA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFOA	ng/g	7	0.1	0.1	100%	0.13	0.55	0.46	0.2	0.27
PFNA	ng/g	7	0.1	0.1	100%	0.6	5.46	4.45	0.91	2.06
PFDA	ng/g	7	0.1	0.1	100%	0.46	5.86	5.55	1.88	2.73
PFUnA	ng/g	7	0.1	0.1	100%	0.98	3.66	3.54	1.91	2.11
PFDaA	ng/g	7	0.1	0.1	100%	0.45	2.94	2.42	1.38	1.56
PFTTrDA	ng/g	7	0.1	0.1	100%	0.62	1.87	1.58	1.1	1.18
PFTeDA	ng/g	7	0.1	0.1	100%	0.3	1.42	1.19	1.02	0.91
PFBS	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFPeS	ng/g	7	0.1	0.1	14%	MDL	0.19	0.08	MDL	0.03
PFHxS	ng/g	7	0.1	0.1	100%	0.52	7.29	5.15	3.64	3.17
PFHpS	ng/g	7	0.1	0.1	100%	0.16	0.78	0.61	0.43	0.42
PFOS	ng/g	7	0.1	0.1	100%	27.1	74.1	71.3	36.7	44.9
PFNS	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFDS	ng/g	7	0.1	0.1	100%	0.31	0.92	0.64	0.42	0.48
PFDoS	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
4:2 FTS	ng/g	7	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
6:2 FTS	ng/g	7	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
8:2 FTS	ng/g	7	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
3:3 FTCA	ng/g	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
5:3 FTCA	ng/g	7	2.4	2.6	0%	MDL	MDL	MDL	MDL	MDL
7:3 FTCA	ng/g	7	2.4	2.6	100%	9	116	76.6	25.6	41.4
PFOSA	ng/g	7	0.1	0.1	57%	MDL	0.21	0.18	0.14	0.09
N-MeFOSA	ng/g	7	0.1	0.2	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSA	ng/g	7	0.3	0.4	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSAA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSAA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
N-MeFOSE	ng/g	7	1	1	0%	MDL	MDL	MDL	MDL	MDL
N-EtFOSE	ng/g	7	1	1	0%	MDL	MDL	MDL	MDL	MDL
HFPO-DA	ng/g	7	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
ADONA	ng/g	7	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
NFDHA	ng/g	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
PFMBA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
PFMPA	ng/g	7	0.2	0.2	0%	MDL	MDL	MDL	MDL	MDL
9CI-PF3ONS	ng/g	7	0.4	0.4	14%	MDL	0.46	0.19	MDL	0.07
11CI-PF3OUdS	ng/g	7	0.4	0.4	0%	MDL	MDL	MDL	MDL	MDL
PFEESA	ng/g	7	0.1	0.1	0%	MDL	MDL	MDL	MDL	MDL
Sum of PFAS	ng/g	7				50.8	172.6	153.5	108.1	101.7

Appendix B: Comparisons with Other Regions and Studies by Matrix

Table B1: Sediment

Reference	Location	Sample Size	Sampling Date(s)	PFOS (ng/g)	sumPFAS (ng/g)	Number of Analytes in sumPFAS
Sedlak et al., 2017	SF Bay, California, US	n = 12	2012	median: 1.2 range: ND-2.61	median: 1.2 range: ND-3.9	13 in method, 10 detected
Sedlak et al., 2018	SF Bay, California, US	n = 27	2014	median: 0.31 range: ND-3.4	median: 0.31 range: ND-4.6	13 in method, 7 detected
Mendez et al., 2025	SF Bay, California, US	n = 11 n = 12 n = 54	2017 2020 2023	2017 median: 0.54 range: 0.16-6.9 2020 median: 0.10 range: <MDL-0.17 2023 median: 0.29 range: 0.05-0.67	2017 median: 1.6 range: 0.50-9.9 2020 median: 0.14 range: <MDL-0.33 2023 median: 0.80 range: 0.09-16	40 in all methods, 20 detected in 2017 8 detected in 2020 15 detected in 2023
Ahmadireskety et al., 2021	Pensacola Bay, Florida, US	n = 34	2020	range: 0.01-0.46	range: 0.05-3.9	51 in method, 28 detected
Pulster et al., 2022	Biscayne Bay, Florida, US	n = 17	2022	NA	range: 0.04–3.0	25 in method, 21 detected
Griffin et al., 2023	Estuarine & Coastal Sites, Florida, US	n = 82	2021	median: 0.05 range: <MDL-1.3	median: 0.09 range: <MDL–2.5	92 in method, 15 detected
Meng et al., 2021	Bohai Sea, China	n = 40	2018	median: 0.04 range: 0.01 - 0.12	median: 0.81 range: 0.37–4.2	29 in method, 24 detected

Reference	Location	Sample Size	Sampling Date(s)	PFOS (ng/g)	sumPFAS (ng/g)	Number of Analytes in sumPFAS
Zhao et al., 2020	Bohai Sea, China	n = 30	2017	range: <MDL-0.20	range: 0.70–4.1	39 in method, 13 detected
Codling et al., 2018	Great Lakes, US	n = 77	2011/2012	range: 0.01-2.5	range: 0.01–26	22 in method, 17 detected
Goodrow et al., 2020	Freshwater sites in New Jersey, US	n = 11	2017	median: 0.55 range: <MDL–27	median: 1.3 range: <MDL–31	13 in method, 8 detected
Macorps et al., 2023	Freshwater sites, France	n = 43	2018	median: 0.44 range: <MDL–5.1	median: 1.3 range: <MDL–23	36 in method, 27 detected
Göckener et al., 2023	Freshwater sites, Germany	n = 60	2021	median: 0.26 range: <MDL-49	median: 1.4 range: <MDL–53	41 in method, 17 detected
Guckert et al., 2022	Lower Saxony, Germany	n = 41	2019	median: <MDL range: <MDL - 2.3	range: 0.14 - 44	43 in method, 21 detected
Mendez et al., 2025 (TOP assay)	SF Bay, California, US	n = 31	2017-2023	NA	range of % increase: <MDL - 1950% 12/31 substantial (>35%)	32 in method, 9 detected
Macorps et al., 2023 (TOP assay)	Freshwater sites across France	n = 43	2018	NA	range of % increase: <MDL– 10,000% 27/43 substantial (>35%)	36 in method, 9 detected
Göckener et al., 2023 (TOP assay)	Freshwater sites across Germany	n = 60	2021	NA	range of % increase: <MDL– 200% 44/60 substantial (>35%)	22 in method, 13 detected
Guckert et al., 2022 (TOP assay)	Lower Saxony, Germany	n = 41	2019	NA	range of % increase: <MDL – 1200% 12/41 substantial (>35%)	13 in method, 13 detected

Table B2: Bay Water

Reference	Location	Sample Size	Sampling Date(s)	PFPeA (ng/L)	PFHxA (ng/L)	PFOA (ng/L)	PFOS (ng/L)	sumPFAS (ng/L)	Number of Analytes in sumPFAS
Mendez et al., 2025	San Francisco Bay, California, US	n = 93	2021 - 2024	median: 2.0 range: <MDL–28	median: 2.5 range: <MDL–25	median: 1.8 range: <MDL–20	median: 2.6 range: <MDL–20	median: 15 range: <MDL–179	40 in method, 16 detected
da Silva et al., 2022	Pensacola Bay, Florida, US	n = 45	2020	median: 3.0 range: <MDL–52	median: 0.66 range: 0.22–43	median: 1.0 range: 0.3–19	median: 3.3 range: 0.68–269	range: 3 - 677	51 in method, 21 detected
Ogunbiyi et al., 2024	Biscayne Bay, Florida, US	n = 19	2022	median: 4.9 range: 1.8–11	median: 3.7 range: 1.2–7.8	median: 1.9 range: 0.36–5.0	median: 5.8 range: <2.2–20	median: 29 range: 8.6 - 72	30 in method, 24 detected
Griffin et al., 2023	Estuarine and Coastal Regions, Florida, US	n = 69	2021	median: ND range: <MDL–8.6	median: 1.0 range: <MDL–7.4	median: 0.76 range: <MDL–13	median: 0.75 range: <MDL–20	median: 6.5 range: <MDL–66	92 in method, 14 detected
Mulabagal et al., 2018	Perdido Bay, Alabama	n = 32	2017	N/A	range: 0.45–4.6	range: 0.71–10	range: 0.83–4.3	range: 4.5–25	23 in method, 8 detected
Viticoski et al., 2022	Mobile Bay and tributaries, Alabama	n = 8	2020	average: 11 range: 4.4–15	average: 4.2 range: <MDL–6.5	average: 5.7 range: 2.7–87.5	average: 3.4 range: <MDL–31	average: 25 range: 8.5–57	17 in method, 6 detected
Meng et al., 2021	Bohai Sea, China	n = 17	2018	median: 2.3 range: 1.3–23	median: 1.9 range: 0.64–26	median: 29 range: 5.7–592	median: 0.23 range: 0.08–0.59	median: 80 range: 13–718	29 in method, 23 detected

Reference	Location	Sample Size	Sampling Date(s)	PFPeA (ng/L)	PFHxA (ng/L)	PFOA (ng/L)	PFOS (ng/L)	sumPFAS (ng/L)	Number of Analytes in sumPFAS
Muir & Miaz, 2021 (Review study)	Global (Compared to Highest levels in Bohai & Yellow Seas)	Varied, n = 166 or 232	2000-2019	median range: <MDL- 31	median range: 0.5 - 3.6	median range: 0.42-5.6	median range: 0.15-3.3	Western US Sum PFASs: 3-6	15 analytes reviewed
Mendez et al., 2025 (TOP assay)	San Francisco Bay, California, US	n = 33	2023/24	N/A	N/A	N/A	N/A	range of % increase: <MDL- 24000% 8/33 substantial (>35%)	32 in method, 9 detected
Neuwald et al., 2022 (TOP assay)	Drinking water sources, Germany	n = 46	2020	N/A	N/A	N/A	N/A	median % increase: 27	19 in method, 8 detected
Foddy et al., 2024 (TOP assay)	Rivers in NW England	n = 8	2024	N/A	N/A	N/A	N/A	range of % increase: 1300-5260 8/8 substantial (>35%, 5 from 0)	29 in method, 6 detected
Joerss et al., 2020 (TOP assay)	Rivers in Germany and China	n = 31	2018	N/A	N/A	N/A	N/A	range of % increase: <MDL-360 26/31 substantial (>35%, 3 from 0)	14 in method, 14 detected

Table B3: Fish

Reference	Location	Species	Sample Size	Sampling Date(s)	Tissue(s)	PFOS Concentration (ng/g ww)	Sum of PFAS Concentration (ng/g ww)	Number of Analytes in sumPFAS
This study, Mendez et al., 2025	SF Bay, California, US	Up to 12 fish species (see above)	2014 n (only archive) = 27 2019 n (+ archive) = 33 2024 n = 67 (composites varied from 1-20)	2014 2019 2024	Muscle fillet (skin-off) or whole (head, tail, & viscera removed)	2014 mean: 4.1 range: <MDL-17 2019 mean: 2.7 range: <MDL-10 2024 mean: 1.9 range: <MDL-8.1	2014 range: <MDL-29 2019 range: <MDL-16 2024 Range: <MDL-70	2014 40 in method, 17 detected in archive 2019 40 in method, 10 detected in archive 2024 40 in method, 21 detected
This Study (Prey Fish)	SF Bay, California, US	shiner surfperch (<i>Cymatogaster aggregate</i>) topsmelt (<i>Atherinops affinis</i>)	n = 39 (composites varied from 20-30)	2023	whole (head, tail, & viscera removed)	median: 1.5 range: 0.31-10	range: 0.52-19	40 in method, 16 detected
Sedlak et al., 2017 (Prey Fish)	SF Bay, California, US	yellowfin gobies (<i>Acanthogobius flavimanus</i>) chameleon gobies (<i>Tridentiger trigonocephalus</i>) northern anchovy (<i>Engraulis mordax</i>) shiner surfperch (<i>Cymatogaster aggregate</i>) staghorn sculpin (<i>Leptocottus armatus</i>)	n = 24 (composites varied from 1-20 by species)	2012-2013	whole (head, tail, & viscera removed)	median: 11 range: <MDL-241	range: <MDL-290	13 in method, 8 detected
Hedgespeth et al., 2023	RI, US	14 fish species	n = 260	2006-2014	Muscle fillet (skin-off) or whole	range: <MDL-7.5	range: <MDL-18	24 in method, 19 detected

Reference	Location	Species	Sample Size	Sampling Date(s)	Tissue(s)	PFOS Concentration (ng/g ww)	Sum of PFAS Concentration (ng/g ww)	Number of Analytes in sumPFAS
Fair et al., 2019	Charleston, SC, US	Atlantic croaker (<i>Micropogonias undulatus</i>) red drum (<i>Sciaenops ocellatus</i>) spot (<i>Leiostomus xanthurus</i>) spotted seatrout (<i>Cynoscion nebulosus</i>) striped mullet (<i>Mugil cephalus</i>)	n = 39 whole fish n = 37 fillets	2014	Muscle fillet (skin-on) or whole	Whole fish range: 2.5–66 Fillet range: 0.4–30	Whole fish range: 2.5–85 Fillet range: 1.8–37	11 in method, 5 detected
Pulster et al., 2022	Tampa Bay, FL, US	24 fish species	n = 140	2021–2022	Muscle fillet (skin-off)	range: 0.92–30	range: 0.31–33.6	25 in method, 24 detected
Barbo et al., 2023	Freshwaters across US	44 fish species	Great Lakes n = 152 US Rivers and Streams (R&S) n = 349	2013–2015	Muscle fillet (skin-off)	Great Lakes mean: 17 range: <MDL-64 US R&S mean: 16 range: <MDL–283	Great Lakes range: <MDL-75 US R&S Range: <MDL–287	13 in method, 13 detected
Goodrow et al., 2020	Freshwater sites in NJ, US	12 fish species	n = 96	2015–2016	Muscle fillet (skin-off)	mean: 28 range: <MDL-208	range: <MDL- 230	13 in method, 9 detected
Boatman et al., 2024	Freshwater sites in NC, US	Lepomis spp. sunfish white perch (<i>Morone americana</i>) yellow perch (<i>Perca flavescens</i>) channel catfish (<i>Ictalurus punctatus</i>)	n = 48	2020	Muscle fillet (skin-off)	range: 7.5-124	NA	Non-targeted, 36 detected
Pickard et al., 2022	Merrimack Rivershed in NH, US	8 species	n = 43 (blue gill were composites, 3)	2019	Muscle fillet (skin-off)	range: MDL-52	range: 0.95–60	37 in method, 25 detected

Table B4: Bird Eggs

Reference	Location	Species	Sample Size	Sampling Date(s)	PFOS Concentration (ng/g ww)	sumPFAS Concentration (ng/g ww)	Number of Analytes in sumPFAS
This study	San Francisco Bay, CA, US	double-crested cormorant (<i>Nannopterum auritum</i>)	n = 9, composited 7 individuals each sample year	2018 2022 2024	2018 mean: 135 2022 mean: 101 2024 mean:	2018 mean: 164 2022 mean: 168 2024 mean:	2018 13 in method; 7 detected 2022 40 in method; 17 detected 2024 40 in method; 17 detected
Sedlak et al., 2017	San Francisco Bay, CA, US	double-crested cormorant (<i>Nannopterum auritum</i>)	n = 9, composited 7 individuals	2012	mean: 188	mean: 236	13 in method; 7 detected
Sedlak et al., 2018	San Francisco Bay, CA, US	double-crested cormorant (<i>Nannopterum auritum</i>)	n = 9, composited 7 individuals	2016	mean: 263	mean: 313	13 in method; 8 detected
Colomer-Vidal et al., 2022	Atlantic and Mediterranean Coasts, Spain	yellow-legged gull (<i>Larus michahellis</i>) Audouin gull (<i>Larus audouinii</i>)	n = 105*, 3 composites of 12 at four sites per year n = 30, 3 composites of 12 eggs per year	2009-2018	mean range: 15–104 mean: 70	mean range: 20–148 mean: 100	17 in method; 7 detected
de Wit et al., 2020	Baltic Sea Coast, Sweden	common eider (<i>Somateria mollissima</i>) common guillemot (<i>Uria aalge</i>) white-tailed eagle (<i>Haliaeetus albicilla</i>)	n = 2, composited 4 or 5 individuals	2015 or 2016	mean: 17 mean: 390 mean: 170	mean: 25 mean: 450 mean: 220	18 in method; 14 detected

Reference	Location	Species	Sample Size	Sampling Date(s)	PFOS Concentration (ng/g ww)	sumPFAS Concentration (ng/g ww)	Number of Analytes in sumPFAS
Gebbink et al., 2016	Baltic Sea Coast, Sweden	common guillemot (<i>Uria aalge</i>)	n = 10	2013	mean: 426	mean: 470	32 in method; 18 detected
Grade & Vogel, 2023	New Hampshire, US	common loon (<i>Gavia immer</i>)	n = 144	2017-2022	geometric mean: 99	geometric mean: 306	40 in method, 30 detected
Jouanneau et al., 2022	Svalbard, Norway	black-legged kittiwake (<i>Rissa tridactyla</i>)	n = 25	2019	mean: 25	mean: 90	37 in method, 17 detected
Kesic et al., 2023	Pacific Coast, BC, Canada	double-crested cormorant (<i>Nannopterum auritum</i>) pelagic cormorant (<i>Urile pelagicus</i>) rhinoceros auklet (<i>Cerorhinca monocerata</i>) Leach's storm petrel (<i>Hydrobates leucorhous</i>)	n = 5, composited 3 individuals	2018 or 2019	mean: 5.2 mean: 4.2 mean: 23 mean: 1.6	mean: 12 mean: 13 mean: 62 mean: 15	17 in method; 14 detected
Letcher et al., 2015	Great Lakes, US	herring gull (<i>Larus argentatus</i>)	n = 114	2012 and 2013	range of means: 43–723	N/A	22 in method; 14 detected
Su et al., 2017	Great Lakes, US	caspian tern (<i>Hydroprogne caspia</i>) herring gull (<i>Larus argentatus</i>)	n = 30 n = 50	2012-2014	range of means: 387–1395 range of means: 103–170	N/A	22 in method; 15 detected
Wilkinson et al., 2022	Charleston, South Carolina, US	eastern brown pelican (<i>Pelecanus occidentalis carolinensis</i>)	n = 36	2019	mean: 135	mean: 175	24 in method, 15 detected
Wu et al., 2020	Great Lakes, US	bald eagle (<i>Haliaeetus leucocephalus</i>)	n = 22	2000-2012	median: 106	median: 174	40 in method, 30 detected

Table B5: Marine Mammal

Reference	Location	Species	Sample Size	Sampling Date(s)	Tissue(s)	PFOS (ng/g ww)	sumPFAS (ng/g ww)	Number of Analytes in sumPFAS
2023/2024 RMP monitoring	SF Bay	Pacific harbor seal (<i>Phoca vitulina richardii</i>)	13 (mix of pups and adults)	2023/2024	Serum Liver	median: 52 range: 31–74	median: 78 median: 90	40 in method; 18 detected
Sedlak et al. 2017	SF Bay; Tomaes Bay	Pacific harbor seal (<i>Phoca vitulina richardii</i>)	37 7	2009-2011 2014	Serum	median range: 11.8–1,050 median: 331	not reported	13 in method; 13 detected
Heuvel-Greve et al. 2026	Western Scheldt & Wadden Sea, Netherlands	harbor seal (<i>Phoca vitulina</i>)	8 11	2023	Liver	median: 815 median: 110	median: 962.5 median: 134.8	16 in method; 14 detected
Simola et al. 2026	Lake Saimaa, Finland	Saimaa ringed seal (<i>Pusa saimensis</i>)	12 pups 2 weaned pups 3 adults	2014-2022	Liver	pups median: 23.4 weaned pups median: 26.7 adults median: 11.6	only PFOS measured	n/a
Khalid et al. 2024	St. Lawrence Estuary and Gulf, Quebec, Canada	harbor seal (<i>Phoca vitulina</i>) gray seal (<i>Halichoerus grypus</i>) harp seal (<i>Pagophilus groenlandicus</i>) hooded seal (<i>Cystophora cristata</i>)	5 2 1 1	not reported	Liver	mean: 72 mean: 46 22 45	median: 140 median: 100.75 44.3 126.3	77 in method; 41 detected

Reference	Location	Species	Sample Size	Sampling Date(s)	Tissue(s)	PFOS (ng/g ww)	sumPFAS (ng/g ww)	Number of Analytes in sumPFAS
Hall et al. 2023	New South Wales, Australia	long-nosed fur seal (<i>Arctocephalus forsteri</i>)	16 pups 12 adults	1998-2019	Liver	pups median: 8.4 adults median: 14	pups median: 19 adults median: 24	25 in method; 14 detected
Byrne et al. 2022	Sivuqaq, Alaska	spotted seal (<i>Phocalargha</i>)	1	2016	Liver	5.6	18	13 in method; 6 detected
Spaan et al. 2020, 2021	US Atlantic Coast; Iceland; Sweden US Atlantic Coast; Iceland; Sweden West Greenland; East Greenland; Sweden West Greenland; Iceland	gray seal (<i>Halichoerus grypus</i>) harbor seal (<i>Phoca vitulina</i>) ringed seal (<i>Pusa hispida</i>) harp seal (<i>Pagophilus groenlandicus</i>)	5; 5; 6 5; 5; 5 5; 10; 5 not reported; 5	2000-2004; 2009-2010; 2012-2016; 2016 2000-2008; 2009-2010; 2015 2013; 2012; 2015 2016; 2009-2010	Liver	means: 168.6; <5.8; 327.8; 172.1 means: 50; 37.9; 857.9 means: 28.1; 99.7; 483.4 means: 21.3; 39.1	means: 285.2; 27.8; 442.2; 236.3 means: 273.6; 99.0; 982.5 means: 78.3; 178.5; 806.8 means: 58.5; 68.7	36 in method; 20 detected
Grønnestad et al. 2017	the West Ice, east of Greenland	hooded seal (<i>Cystophora cristata</i>)	15 pups 15 adults	2008	Plasma	pup median: 28.3 adult median: 12.2	pup mean: 66 mother mean: 36	8 in method; 8 detected
Routti et al. 2016	Svalbard, Norway	ringed seal (<i>Pusa hispida</i>)	71 (mix of pups and adults)	1990-2010	Plasma	means range: 18 (in 2007)–76 (in 1990)	means range: 27 (in 2007)–128 (in 2004)	16 in method; 16 detected

Reference	Location	Species	Sample Size	Sampling Date(s)	Tissue(s)	PFOS (ng/g ww)	sumPFAS (ng/g ww)	Number of Analytes in sumPFAS
Routti et al. 2015	McMurdo Sound, Antarctica	Weddell seal (<i>Leptonychotes weddellii</i>)	10	2006	Plasma	median: 0.06	not reported	14 in method; 4 detected
Routti et al. 2014	Svalbard, Norway	harbor seal (<i>Phoca vitulina</i>)	4 pups 8 adults	2009–2010	Plasma	pups median: 38 adults median: 39	pups median: 64 adults median: 63	14 in method; 14 detected
Rigét et al. 2013	Greenland	ringed seal (<i>Pusa hispida</i>)	177	1982–2011	Liver	means range: 12.5 (in 1982)–397 (in 2006)	not reported	7 in method; 7 detected
Rotander et al. 2012	Greenland	ringed seal (<i>Pusa hispida</i>) hooded seal (<i>Cystophora cristata</i>)	9 6	1984–2006 1990–2007	Liver	means: 59–38 means: 48–86	not reported	13 in method; 9 detected
Ahrens et al. 2009b	German Bight	harbor seal (<i>Phoca vitulina</i>)	19 pups; 44 adults	1988–2008	Liver	means range: 472 (in 2006)–1436 (in 2003)	means range: 504 (in 2008)–1502 (in 2003)	40 in method; 17 detected
Ahrens et al. 2009a	German Bight	harbor seal (<i>Phoca vitulina</i>)	2 pups; 2 adults	2007	Liver Whole Blood	mean: 1017 mean: 349	mean: 1071 mean: 381	40 in method; 18 detected
Butt et al. 2008	Canadian Arctic	ringed seal (<i>Pusa hispida</i>)	110	2002–2005	Liver	mean range: 6.5–88.8 ng/g ww	not reported	15 in method; 15 detected

Appendix C: Additional Sport Fish Figures

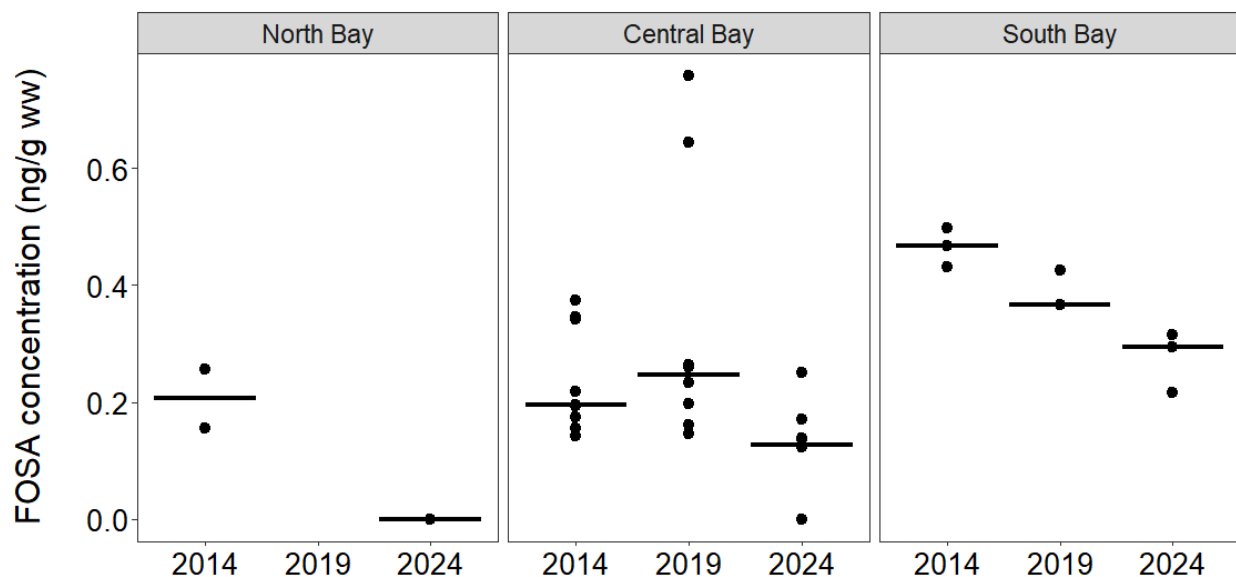


Figure C1. Comparison of concentrations of the PFOS precursor FOSA in shiner surfperch caught in the North Bay (San Pablo Bay station), Central Bay (Berkeley, Oakland, and San Francisco Waterfront stations), and South Bay (Redwood Creek) regions between 2014 and 2024. The bars shown represent the median concentration (non-detect = 0). Samples only include those analyzed in 2019 or later, using the modern analytical method.

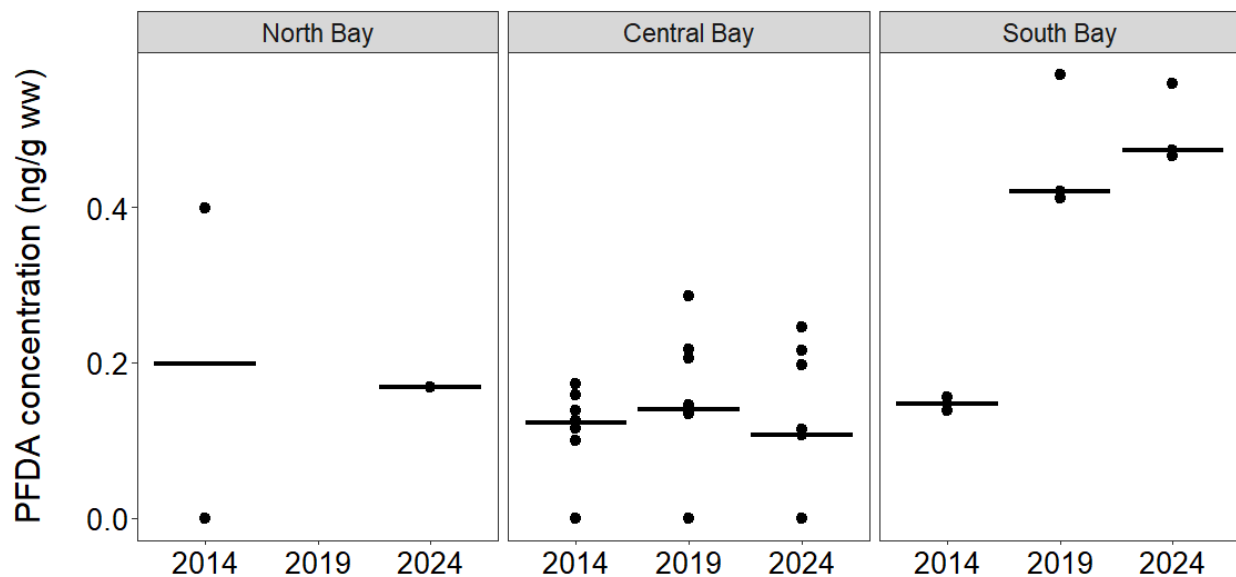


Figure C2. Comparison of concentrations of PFDA, a long-chain PFCA, in shiner surfperch caught in the North Bay (San Pablo Bay station), Central Bay (Berkeley, Oakland, and San Francisco Waterfront stations), and South Bay (Redwood Creek) regions between 2014 and 2024. The bars shown represent the median concentration (non-detect = 0). Samples only include those analyzed in 2019 or later, using the modern analytical method.

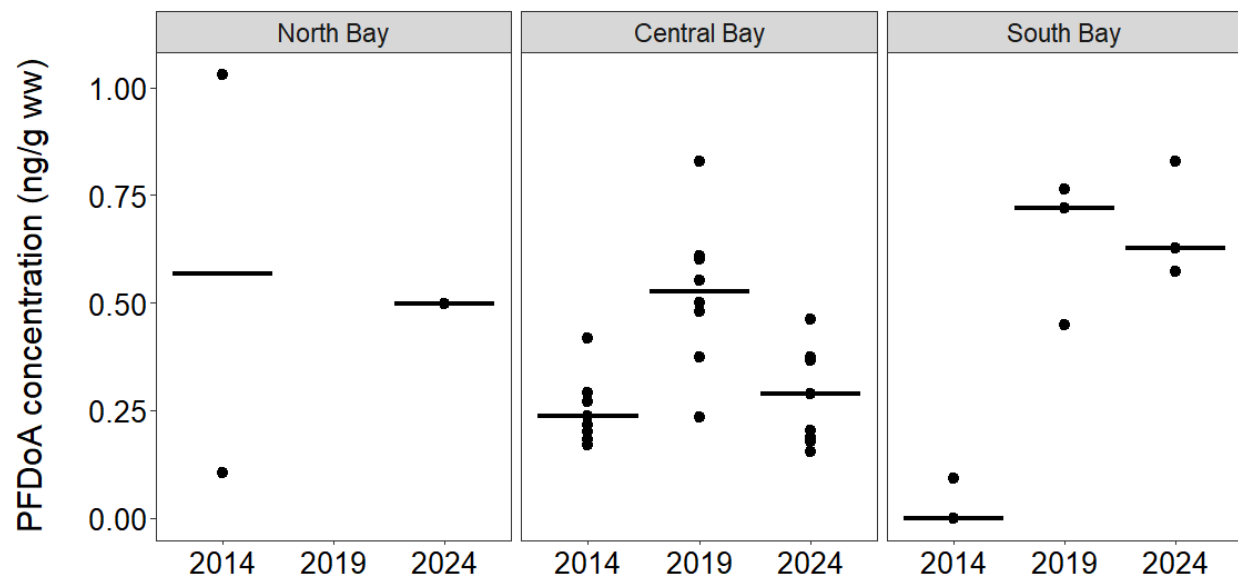


Figure C3. Comparison of concentrations of PFDoA, a long-chain PFCA, in shiner surfperch caught in the North Bay (San Pablo Bay station), Central Bay (Berkeley, Oakland, and San Francisco Waterfront stations), and South Bay (Redwood Creek) regions between 2014 and 2024. The bars shown represent the median concentration (non-detect = 0). Samples only include those analyzed in 2019 or later, using the modern analytical method.

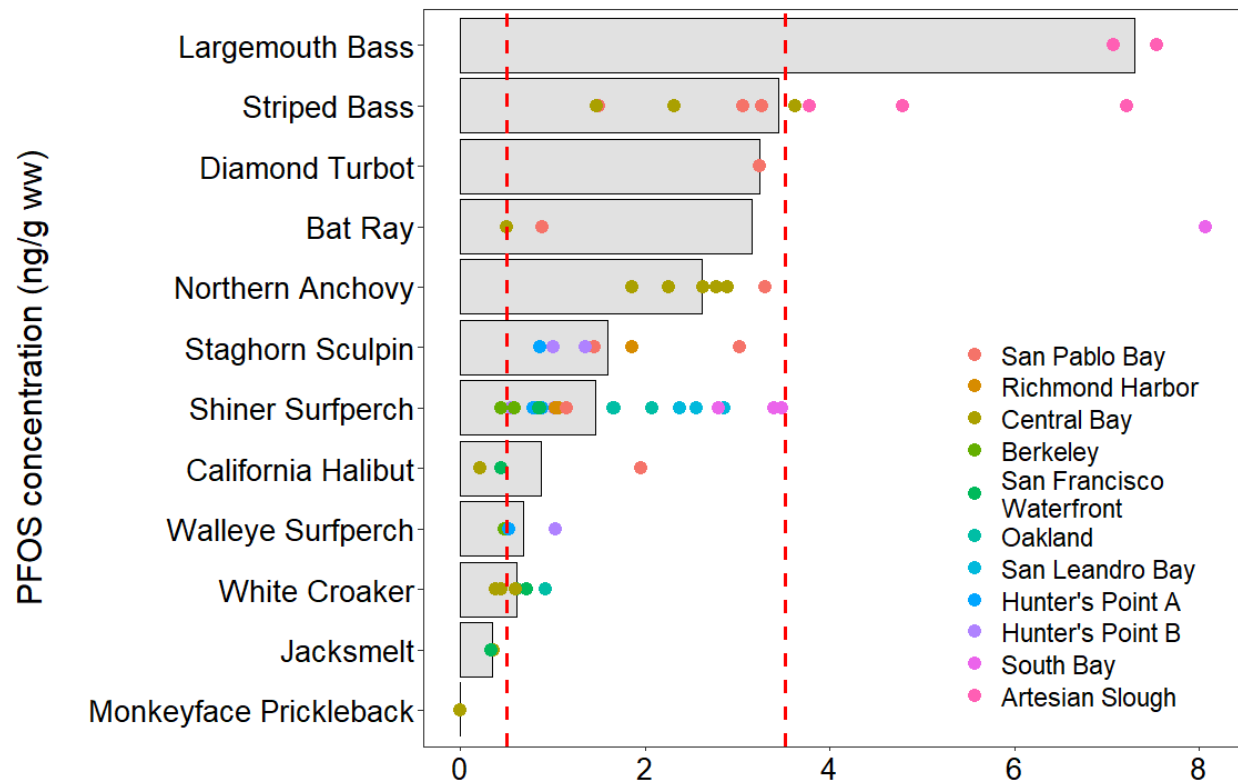


Figure C4. PFOS concentrations (nanograms/gram wet weight) in all San Francisco Bay sport fish, 2024. Bars indicate median concentrations. Points represent individual or composite samples, and colored by sampling location. The red dashed lines represent the Massachusetts one meal per day (0.5 ng/g ww) and one meal per week (3.52 ng/g ww) consumption advisories.