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Cover Photo: View of Lake Washington looking east along the I-90 bridge.

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[1] https://www.ecology.wa.gov/Research-Data/Data-resources/Environmental-Information-Management-database

Greater Lake Washington Watershed PFAS Survey

Final Report

by
Siana Wong and Diane Escobedo

Environmental Assessment Program
Washington State Department of Ecology
Olympia, Washington

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AQP	Air Quality Program
EAP	Environmental Assessment Program
HWTR	Hazardous Waste and Toxics Reduction program
IT	Information Technology
MEL	Manchester Environmental Laboratory
WQP	Water Quality Program

Abstract

Per- and polyfluoroalkyl substances (PFAS) are a group of widely used, environmentally persistent, human-made fluorinated chemicals that can enter aquatic environments through many different point sources and nonpoint pathways. From 2020 to 2023, The Washington State Department of Ecology (Ecology) conducted a study to assess sources and pathways of PFAS to Lake Washington, a large urban freshwater lake. The need for the study arose because of high levels of PFAS found in fish from the lake.

The goal of the study was to characterize, identify, and prioritize the major pathways and sources of PFAS to Lake Washington. Samples collected represented lake and tributary surface water and sediments, groundwater, stormwater, bulk atmospheric deposition, suspended sediments, surface runoff, road dust, biofilm, and aquatic macroinvertebrates (caddisfly larvae).

Total PFAS (sum of 40 analytes) concentration ranges included:

- 11.6 – 23.8 ng/L in Lake Washington surface water
- Non-detect – 139 ng/L in tributary surface water
- Non-detect – 946 ng/L in groundwater
- 13.3 – 115 ng/L in stormwater discharge
- 0.906 – 16.9 ng/g in lake surface sediments
- 0.0518 – 50.6 ng/g in tributary surface sediments
- 0.159 – 10.9 ng/g in tributary suspended sediments
- Non-detect – 11.0 ng/L in surface runoff
- 0.305 – 19.0 ng/g in road dust
- Non-detect – 10.8 ng/L in bulk atmospheric deposition
- Non-detect – 0.852 ng/g in biofilm tissue
- 0.790 – 0.884 ng/g in caddisfly larvae tissue

Results overall showed that PFAS detections were widespread in the watershed, with largely diffuse sources and pathways of PFAS to the lake, and some localized inputs with higher concentrations. Further assessments are needed to better understand the relative importance of different pathways, identify specific land use activities contributing to PFAS in the watershed, and address localized areas where higher PFAS levels were found.

Introduction

Per- and Polyfluoroalkyl Substances (PFAS)

Per- and polyfluoroalkyl substances (PFAS) are a group of thousands of human-made fluorinated chemicals. Their thermostable, stain-resistant, surfactant, and water and oil-repellent properties have made them useful in many industrial and consumer applications since the 1940s. Common uses include: Aqueous film-forming foams (AFFF); carpets, textiles, and upholstery; household and cleaning products; paper and packaging products; surfactants, mist suppressants, fluoropolymer processing aids, and chemical additives used in a wide array of industrial processes and products (ITRC 2023).

By the 2000s, evidence concerning their persistence in the environment and their bioaccumulation and toxicity in humans and organisms had grown. Specific compounds such as perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) began to be voluntarily phased out of manufacture and use in the U.S. However, these compounds are still manufactured in other countries and imported into the U.S. Despite phasing out of certain PFAS compounds, thousands of other PFAS continue to be manufactured and used in the U.S. and internationally.

To organize and manage these chemicals, PFAS are categorized into groups based on their molecular structure (Figure 1). The most common PFAS are perfluoroalkyl acids (PFAAs), which are divided into two major subgroups:

- Perfluoroalkane sulfonic acids (PFSAs); e.g., PFOS
- Perfluoroalkyl carboxylic acids (PFCAs); e.g., PFOA

PFAAs are also categorized by their carbon chain length:

- Long-chain:
  - PFSAs containing at least six carbons (≥C6 PFSAs); e.g., PFOS
  - PFCAs containing at least eight carbons (≥C8 PFCAs); e.g., PFOA
- Short-chain:
  - PFSAs containing fewer than six carbons (<C6 PFSAs); e.g., PFPeS
  - PFCAs containing fewer than eight carbons (<C8 PFCAs); e.g., PFHxA

As long-chain PFAAs have largely become phased out, newer PFAS chemicals continue to be developed and used to replace them. These newer chemicals include:

- Short-chain and ultra short-chain PFAAs
- Perflourooctane sulfonamides and perfluoroalkane sulfonamido substances ("sulfonamides")
- Fluorotelomer substances
- Perfluoroalkyl and polyfluoroalkyl ether acids ("ethers")

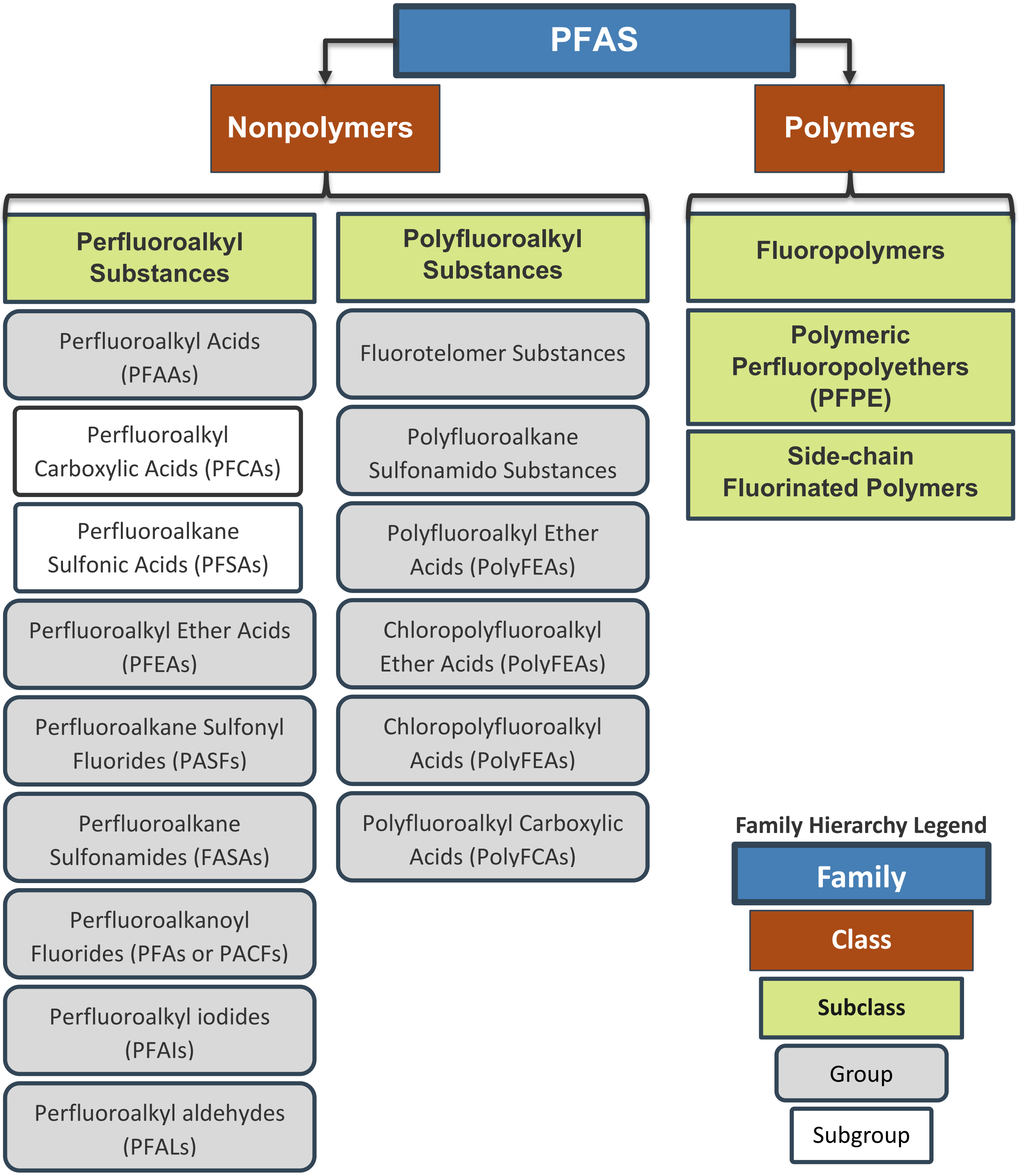

Figure 1. Categorization of per- and polyfluoroalkyl substances (PFAS).
Figure is adapted from ITRC (2023), showing that PFAS are categorized as nonpolymers or polymers. Nonpolymers (analyzed in this study) are further categorized as perfluoroalkyl or polyfluoroalkyl. Perfluoroalkyl substances consist of perfluoroalkyl acids (perfluoroalkyl carboxylic acids and perfluoroalkane sulfonic acids), perfluoroalkyl ether acids, and perfluoroalkane sulfonamides, among others. Polyfluoroalkyl substances consist of fluorotelomers and polyfuoroalkyl ether acids, among others.

However, some of these newer compounds are harmful like those they were meant to replace, and little is known about the health effects of the other thousands of PFAS that are known to exist (Brase et al. 2021). Many newer PFAS compounds are also precursors, which are compounds that can degrade or biotransform into terminal PFAA end products in the environment. For example, perfluoroalkane sulfonamido substances can transform into PFOS, and fluorotelomer substances can transform into PFCAs in the environment.

PFAS in the Environment

Major common releases of PFAS to the environment include industrial facilities that produce, process, or use PFAS; areas where AFFF is or has been used, released, or stored; waste management and wastewater treatment facilities; and discharges linked to surface and floor treatments from commercial facilities (ITRC 2023). Due to their widespread use, PFAS have been found globally in the environment including air, groundwater, surface water, drinking water, rainwater, biofilm, and animals (Munoz et al. 2018; Brase et al. 2021; Xu et al. 2021; Cousins et al. 2022; Waterkeeper Alliance 2022; Andrews et al. 2023; Faust 2023; Smalling et al. 2023).

PFAS are resistant to degradation and, once in the environment, can persist long enough to move through the food chain and accumulate in organisms. Different types of PFAS are more likely to be found in different environmental compartments. For example, short-chain PFAS are more hydrophilic and more likely to be found in water, while long-chain PFAS are more hydrophobic and more likely to be found in sediments and animal tissue.

In 2008, 2016, and 2018 the Washington State Department of Ecology (Ecology) conducted surveys to measure levels of PFAS in freshwater fish collected from waterbodies across the state (Furl and Meredith 2010; Mathieu and McCall 2017; Mathieu 2022). These studies found elevated concentrations of PFOS and several long-chain PFCA compounds in fish collected from urban waterbodies, including Lake Washington, compared to other waterbodies in the state. In 2022, the Washington State Department of Health issued fish consumption advisories for certain fish species from Lake Washington and two other urban lakes, based on PFOS levels in the fish (Christie and Shah 2022).

Goals and Objectives

Our study followed up on recommendations from Mathieu (2022) to investigate sources and pathways of PFAS to urban waterbodies in Washington, with specific focus on Lake Washington. Our study was within the scope of Washington State's strategy to address PFAS by following recommendations in the state's PFAS Chemical Action Plan (Ecology and Health 2022). Specifically, our study assessed sources and pathways of PFAS in the environment where PFAS contamination had been found.

The purpose of our study was to assess how PFAS are getting into Lake Washington. The study goal was to characterize, identify, and prioritize the major pathways and sources of PFAS to Lake Washington.

We completed our study in two phases. The objective of Phase 1 was to characterize PFAS concentrations in the lake and in common contaminant pathways to the lake. These included tributary inflows, stormwater discharges, groundwater discharges, and bulk atmospheric deposition. The main objective of Phase 2 was to further home in on sources and pathways of PFAS to the lake in areas where the highest levels of PFAS were found during Phase 1.

Study Area

Our study took place in the Cedar-Sammamish (Greater Lake Washington) watershed in King and Snohomish Counties (Figure 2). The watershed occupies about 692 square miles and drains into Lake Washington, the second largest natural lake in Washington. The lake is about 34 square miles in surface area, 22 miles long, and 214 ft deep at its deepest point (King County 2015). The average time water resides in the lake is 2.3 years (Edmondson and Lehman 1981). Surface water eventually exits the lake through the Lake Washington Ship Canal and into Puget Sound.

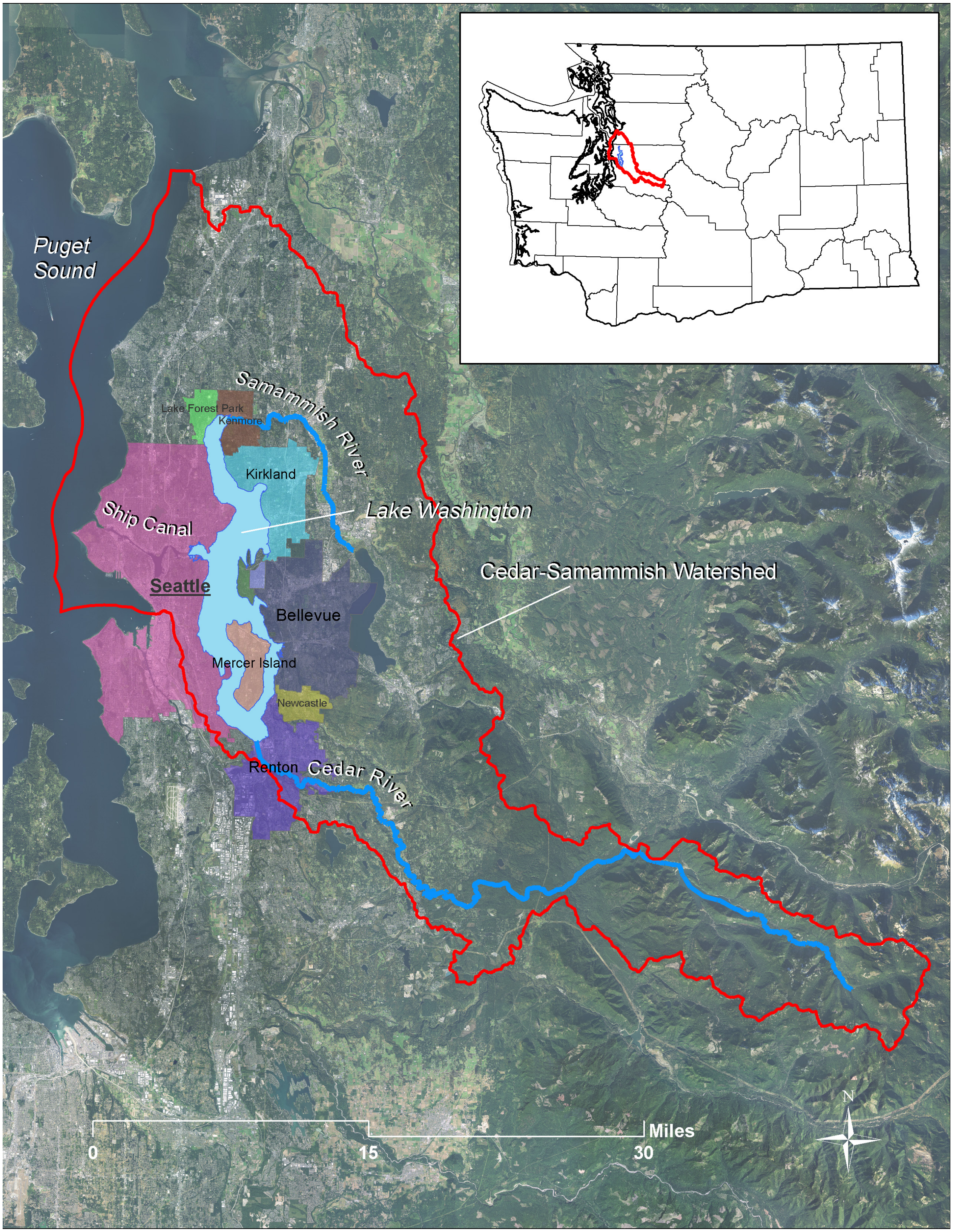

Figure 2. Map showing the Greater Lake Washington watershed study area.
The study boundary, Cedar and Sammamish Rivers, and cities surrounding the lake (Seattle, Bellevue, Renton, Kirkland, Mercer Island, Newcastle, Kenmore, and Lake Forest Park) are highlighted.

Lake Washington has a century-long history of development and associated hydrologic change. One of the biggest changes involved the construction of the Lake Washington Ship Canal in 1916 to provide navigable passage between Lake Washington and the Puget Sound. Historically, waters from Lake Washington emptied into the Puget Sound via the Black/Duwamish River system at the south end of the lake. With the Ship Canal in place, lake water levels were lowered by nine feet, causing the Black River to dry up and become disconnected from the lake (Chrzastowski 1983). Eventually, the Ship Canal became the lake's outlet. Lower lake levels also exacerbated the loss of lake shoreline wetlands, which were increasingly being used for urban development and as landfills (Chrzastowski 1983). Currently, water levels are managed, with a low of about 20 ft in winter and a high of 22 ft in spring.

The Cedar River, which historically emptied into the Black River and was not connected to Lake Washington, became the lake's largest inflow when the river was diverted into the lake to prevent flooding of the surrounding urban area.

With the expansion of metropolitan areas surrounding the lake in the early 1900s, the lake began receiving raw sewage. Eutrophication became a major water quality issue from the 1940s through the early 1960s when high volumes of treated sewage were discharged into the lake (Edmondson 1970). Since the 1960s, when sewage began to be diverted from the lake, water quality issues related to eutrophication have drastically improved (Edmonson 1991). However, some water quality issues in Lake Washington remain, including several toxic chemicals, fecal coliform, and sediment bioassay that are listed on the state's 303(d) list (Washington State Water Quality Assessment[1]).

Stormwater has previously been found to be a major pollutant pathway in the Puget Sound region, including to Lake Washington, largely as surface water runoff from impervious surfaces (Norton et al. 2011; King County 2014). It has been estimated that more than 375,000 acres of land — or about 5% of the Puget Sound watershed — is covered by impervious surfaces (The Nature Conservancy 2023). Stormwater management has been a focus of many groups addressing water quality issues in the region, including all the sub-watersheds sampled in our study.

Groundwater in the study area originates primarily from precipitation infiltrating into the ground, and surface water from lakes and streams discharging to groundwater in upland areas. Infiltration varies based on surficial geology, soils, land use, and vegetation. Shallow groundwater is frequently hydraulically connected to surface water. When the hydraulic head in groundwater is higher than surface water, groundwater will discharge to streams and lakes.

The Puget Sound aquifer system underlies this study area. The aquifer system is composed of alternating fine- and coarse-grained sediment deposited during glacial periods, and warmer interglacial periods of the last approximately 2.6 million years. The coarse grained and alluvial deposits generally form the water bearing units, or aquifers. The coarse-

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[1] https://apps.ecology.wa.gov/ApprovedWQA/ApprovedPages/ApprovedSearch.aspx

grained units generally consist of glacial outwash deposited during glacial advance or retreat, and alluvium deposited in river valleys.

Finer grained deposits create a semi-confining or confining unit where present, providing some protection to the aquifer from contaminants introduced at the land surface (Vaccaro et al. 1998). Semi-confining and confining units generally consist of glacial till (unsorted, compacted glacial sediment deposited by the glacier), lacustrine, glaciomarine, and mudflow deposits (Jones 1999). Discontinuous perched aquifers often occur in the glacial till. In the lowland areas, groundwater movement in this aquifer system is generally from topographic highs to topographic lows, such as from uplands to river valleys and lakes.

Aquifers can be categorized based on the distance between their recharge and discharge areas. Deeper aquifers have longer flow paths and therefore longer residence times, whereas the shallow aquifers have shorter flow paths and residence times. In the Puget Sound lowlands, aquifers with short flow paths and isolated flow systems dominate due to the shallow-confining units within the glacial-interglacial deposits, topographic controls, the location and size of major river valleys, and geometry of basement confining units (Vaccaro et al. 1998).

Methods

Study methods are detailed in the Quality Assurance Project Plan (QAPP) and QAPP addendums for this study (Wong and Mathieu 2020; Escobedo 2021; Wong and Escobedo 2022). Field methods were adapted from Ecology's standard operating procedures (SOPs) and sampling guidance documents for collecting:

- Surface water samples (Urmos-Berry 2022)
- Surface sediment samples (Wong 2023)
- Biofilm samples (Mathieu et al. 2019)
- Stormwater samples (Ecology 2015; Lowe et al. 2024)
- Stormwater solids/suspended sediments (Lubliner et al. 2024)
- Streamflow measurements (Mathieu 2019)
- Measurements using multiparameter sondes (Anderson 2016)
- Groundwater samples (Sinclair and Pitz 2018; Marti 2016; Marti 2024)

All PFAS sampling equipment, except for pre-cleaned high-density polyethylene (HDPE) sample containers provided by the analytical laboratory, were decontaminated following Ecology's SOP for decontaminating field equipment (Friese 2021). This involved scrubbing with soapy water (Liquinox detergent and tap water), followed by a tap water rinse, and a final methanol rinse. We also used PFAS sampling guidance[1] developed by the Michigan PFAS Action Response Team, which includes best practices for avoiding PFAS cross-contamination during sampling. While in the field, PFAS sample containers were placed in Ziploc bags, and all samples were stored in a cooler with wet ice.

Phase 1 Sampling Design & Procedures

In Phase 1, we characterized PFAS in multiple sample media types to understand the levels and types of PFAS getting into the lake from different pathways, and that are found in different matrices within the watershed. We collected surface water samples from Lake Washington and its tributaries, groundwater discharges, stormwater discharges, and bulk atmospheric deposition. Sample matrices also included lake and tributary sediments, and biofilm from tributary locations. All samples were analyzed for 40 target PFAS compounds (Appendix A).

Surface Water

Surface water samples were collected in September 2020 (summer low flow) and April 2021 (spring high flow). We sampled 23 sites in Lake Washington by boat (Figure 3; Appendix B). Three sites were in the middle of the lake at the approximate maximum depths of the north, central, and south basins. Nineteen sites were in the near-shore environment along the lake perimeter, and one site was located at the lake outlet in Montlake Cut.

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[1] https://www.michigan.gov/pfasresponse/investigations/sampling-guidance

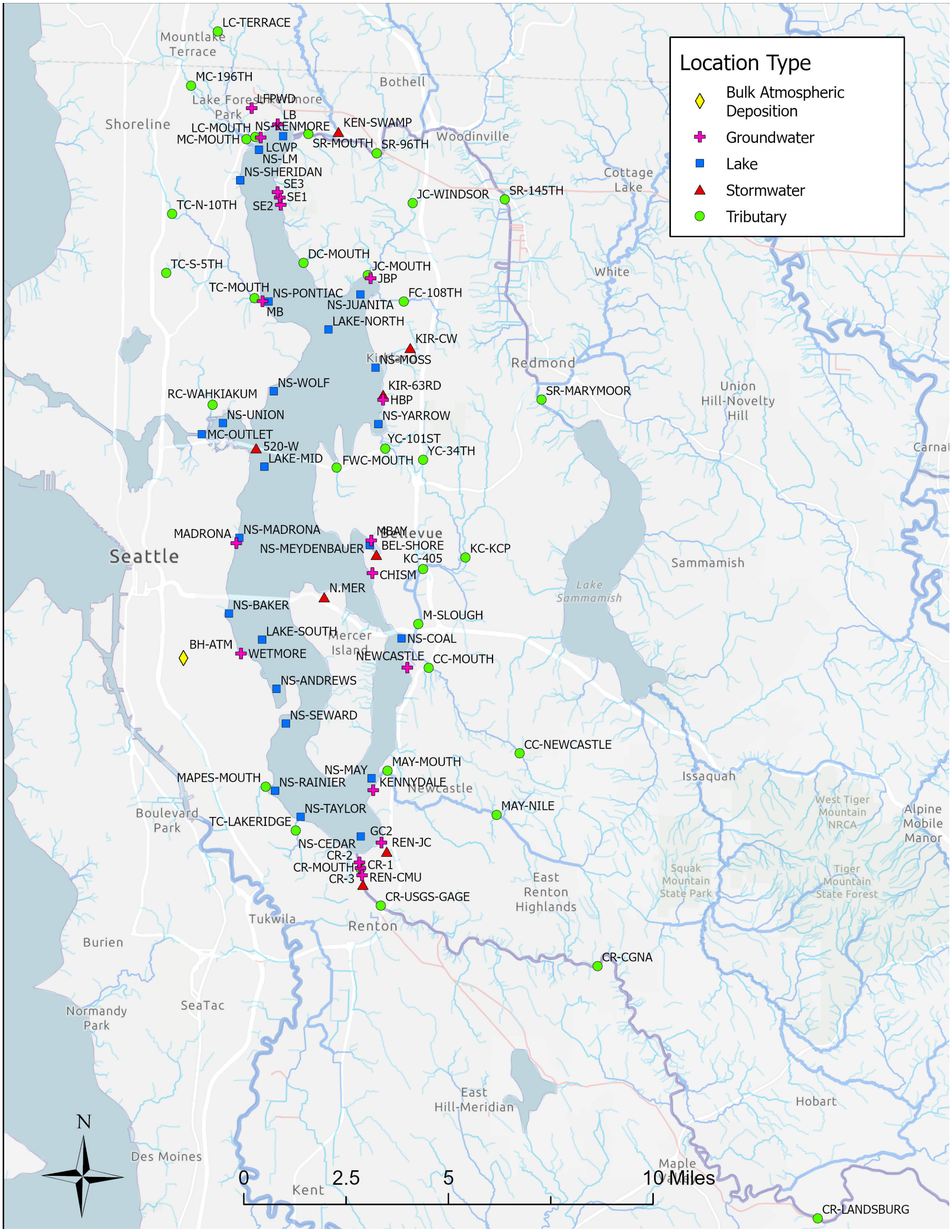

Figure 3. Map of all Phase 1 sampling sites in the Greater Lake Washington watershed.

We sampled 32 tributary sites, representing 14 tributaries that flow into Lake Washington (Appendix B). For each tributary, we sampled one site near the outlet to the lake. In larger tributaries, at least one upstream site was sampled.

Samples for PFAS, total oxidizable precursor (TOP) assay, total organic carbon (TOC), and dissolved organic carbon (DOC) analyses were collected at about 15 – 30 cm below the water surface using bottles connected to a telescopic sampling pole. Sample bottles were triple rinsed with site water prior to sample collection (except for TOC bottles, which were pre-acidified with hydrochloric acid). Water samples for PFAS and TOP assay analyses were collected directly from the water first using the HDPE sample containers attached to the pole. TOC samples were collected using a clean polyethylene transfer bottle, then poured into the polyethylene sample container. DOC samples were collected using a clean polyethylene transfer bottle, filtered using a 0.45-micron polypropylene filter and syringe, and then poured into the polyethylene sample container.

Measurements of water temperature (°C), dissolved oxygen concentration (mg/L), pH, and specific conductance (µS/cm) were also collected at about 15 – 30 cm below the water surface using a calibrated YSI EXO sonde.

Flow was measured using a Marsh McBirney Flo-Mate flow meter at each tributary outlet site during the summer and spring sampling events, except for Ravenna Creek, where we were unable to measure flow. Flow measurements were compared to those from King County and United States Geological Survey (USGS) continuous stream gages where possible to ensure accuracy. Flow measurements were used to calculate and estimate instantaneous loads (Total PFAS Concentration x Flow) from the tributaries during summer low and spring high flow.

Surface Sediments

Surface sediment samples were collected at surface water sites where feasible (21 of the 23 lake sites, and 16 of the 32 tributary sites) during the summer event.

Lake sediment samples were collected using a stainless-steel petite Ponar grab sampler. At each site, we composited sediments from three grabs collected within an approximate 10 m radius. Excess water was siphoned from each Ponar grab, and the top 2 cm of sediment was scooped into a stainless-steel bowl using a stainless-steel spoon. The composited sediment was mixed and then first scooped into HDPE sample containers for PFAS and TOP assay analyses, then a polyethylene container for sediment grain size, and glass jar for TOC analyses.

Tributary sediment samples were collected from the top 2 cm of the sediment surface using stainless-steel scoops. Sediment was composited from grabs collected within an approximate 10 m radius of the site. The composite was mixed in a stainless-steel bowl and then scooped into sample containers for PFAS, TOP assay, grain size, and TOC analyses.

Biofilm

Biofilm samples were collected at tributary surface water sites where feasible (6 of the 32 sites). Submerged cobbles with a layer of brownish, flocculent biofilm attached were collected at each site. A blade was used to scrape the biofilm from each cobble into a stainless-steel bowl. Biofilm was composited and mixed using a stainless-steel spoon, then scooped into an HDPE sampling container for PFAS analysis. Samples were decanted prior to freezing. To estimate biomass at each site, a separate composite sample was collected for analysis of ash-free dry weight. Biofilm surface area measurements were obtained by using imaging software to digitize aluminum foil cutouts of the cobble surfaces on which biofilm was growing (Dudley et al. 2011).

Groundwater

We sampled 19 groundwater sites during the spring, which included 15 Lake Washington shoreline locations, 3 locations near the mouth of the Cedar River, and 1 shallow well field at the north end of the lake. Site selection was based on a temperature reconnaissance survey to identify areas of groundwater discharge and a desktop review of available information on local geology/hydrogeology (literature and technical reports). An attempt was also made to get a wide distribution of sample locations around the lake. Sample locations were limited by accessibility to the shoreline, shoreline structures (e.g., bulkheads and docks) and substrate that was not conducive to the sampling method.

Groundwater samples were collected using a PushPoint sampler. The PushPoint sampler is 6.35-mm diameter stainless-steel tube with a machined point. A 4-cm-long slotted screen with approximately 20% open area is located at the end with the machined point. The PushPoint sampler comes in multiple lengths. The 91-cm and 122-cm samplers were used for this project. The sampler includes an internal guard rod which adds structural support during insertion.

The device is manually inserted into the subsurface (lake or stream bed). Once inserted, the guard rod was removed, HDPE tubing was attached to the sampling port and silicone tubing was attached to the HDPE and inserted at the pump head. Tubing was attached to a flow-through cell with a Hydrolab sonde to collect field parameters prior to sampling. Low-flow sampling methods were used (Sinclair and Pitz 2018). Pumping rates used for purging and sampling were less than 500 mL/min. Measurements of groundwater temperature (°C), dissolved oxygen (mg/L), pH, oxidation-reduction potential (mV), turbidity (NTU), and specific conductance (µS/cm) were recorded for both purge and surface water to ensure dilution of porewater with surface water was not occurring. A Hach turbidity meter was used to measure turbidity. If a sufficient contrast between surface water and groundwater parameters existed and parameters stabilized, PFAS and TOP assay samples were collected.

Stormwater

We collected one grab sample from each of eight stormwater discharge sites representing various land uses that drain into Lake Washington (Table 1). These included residential, commercial (airport), transportation, and mixed land uses. Our criteria for selecting sites were: (1) The site represented average to large size stormwater outflow to the lake, or the site represented a land use type of interest; and (2) sites were publicly accessible, relatively easy to access, and/or we had permission to access the site.

Table 1. Description of Phase 1 stormwater sampling sites surrounding Lake Washington.

Location ID	Stormwater Structure	Primary Land Uses	Events Sampled
520-W	Bridge scupper	Transportation/Communication/Utilities	Storms 2–4
BEL-SHORE	Outfall	Residential	Storm 1
KEN-SWAMP	Outfall	Residential, Open Space & Other, Commercial	Storm 1
KIR-63RD	Outfall	Residential, Open Space & Other	Storm 1
KIR-CW	Outfall	Residential, Commercial	Storms 1–5
N.MER	Drainage ditch	Residential	Storm 1
REN-CMU	Catch Basin	Commercial (Airport)	Storm 1
REN-JC	Outfall	Commercial, Open Space & Other, Residential, Transportation/Communication/Utilities	Storms 1–5

We sampled during five qualifying stormwater events between November 2020 and April 2021:

- Storm 1: November 3, 2020
- Storm 2: December 15, 2020
- Storm 3: February 25, 2021
- Storm 4: March 18, 2021
- Storm 5: April 24, 2021

A qualifying storm event was defined as at least 0.2 in of rain following an antecedent period of <0.05 in rainfall in the last 48 hours. We collected a sample if stormwater discharge was observed to be flowing—we did not collect standing water from previous storms. All sites were sampled during the first storm event, except for the 520 bridge (520-W), where we were unable to collect a sample.

To assess among-storm temporal variability, three sites were sampled during four additional storm events (520-W, KIR-CW, and REN-JC). To assess within-storm temporal variability, three samples were collected at the REN-JC site: (1) grab sample at the beginning of the storm sampling period; (2) grab sample at the end of the storm sampling period; and (3) composite of the beginning and end samples.

During all five storm events, we also sampled five tributaries (Cedar River, Sammamish River, Thornton Creek, Juanita Creek, and May Creek) at the outlet sites to compare concentrations and instantaneous loads among storm, summer low flow, and spring high flow events. The five tributaries represent the largest flows to the lake with continuous flow gages.

Stormwater and tributary grab samples were collected for analyses of PFAS, TOP assay, TOC, DOC, and total suspended solids (TSS). A sampling pole was used if necessary. PFAS and TOP assay samples were collected first: HDPE bottles were triple rinsed with site water and then used to collect water directly from the site. TSS sample bottles were also triple rinsed with site water, then used to collect water directly from the site. TOC samples were collected using a clean polyethylene transfer bottle, then poured into the polyethylene sample container. DOC samples were collected using a clean polyethylene transfer bottle, filtered using a 0.45-micron polypropylene filter and syringe, and then poured into the polyethylene sample container. YSI EXO sonde measurements for water temperature (°C), dissolved oxygen concentration (mg/L), pH, and specific conductance (µS/cm) were only collected at the tributary sites.

Bulk Atmospheric Deposition

We sampled bulk (wet + dry) atmospheric deposition at one site, located at Ecology's Beacon Hill air quality monitoring station, during four deployment events:

- Deployment 1: December 7 – 29, 2020
- Deployment 2: January 21 – February 3, 2021
- Deployment 3: February 26 – March 16, 2021
- Deployment 4: April 2 – 26, 2021

The length of deployment depended on the amount of precipitation received on site. We tried to time sample retrievals so that collection containers did not overfill; however, the containers likely overfilled during Deployments 1 and 2.

Our bulk atmospheric deposition sampler consisted of a 31 cm-diameter stainless-steel funnel and 4 L amber glass collection container connected by HDPE tubing, housed inside a 5.5-ft tall wooden and aluminum frame (Figure 4).

During sample retrieval, the sample collection container was disconnected from the tubing and shaken vigorously to mix the sample water. Water was then poured from each container into separate HDPE sample bottles for PFAS analysis.

The volume of sample water collected during the deployment period was estimated by weighing the collection container before deployment (empty) and after deployment (full), where we assumed the sample volume in liters is equal to the sample weight in kilograms. We did not estimate sample collection volumes for the first two deployments because the collection containers likely overfilled.

During each sample retrieval, we collected one sample and three quality control samples for PFAS analysis (described in the Quality Assurance/Quality Control: Field Samples section of this report).

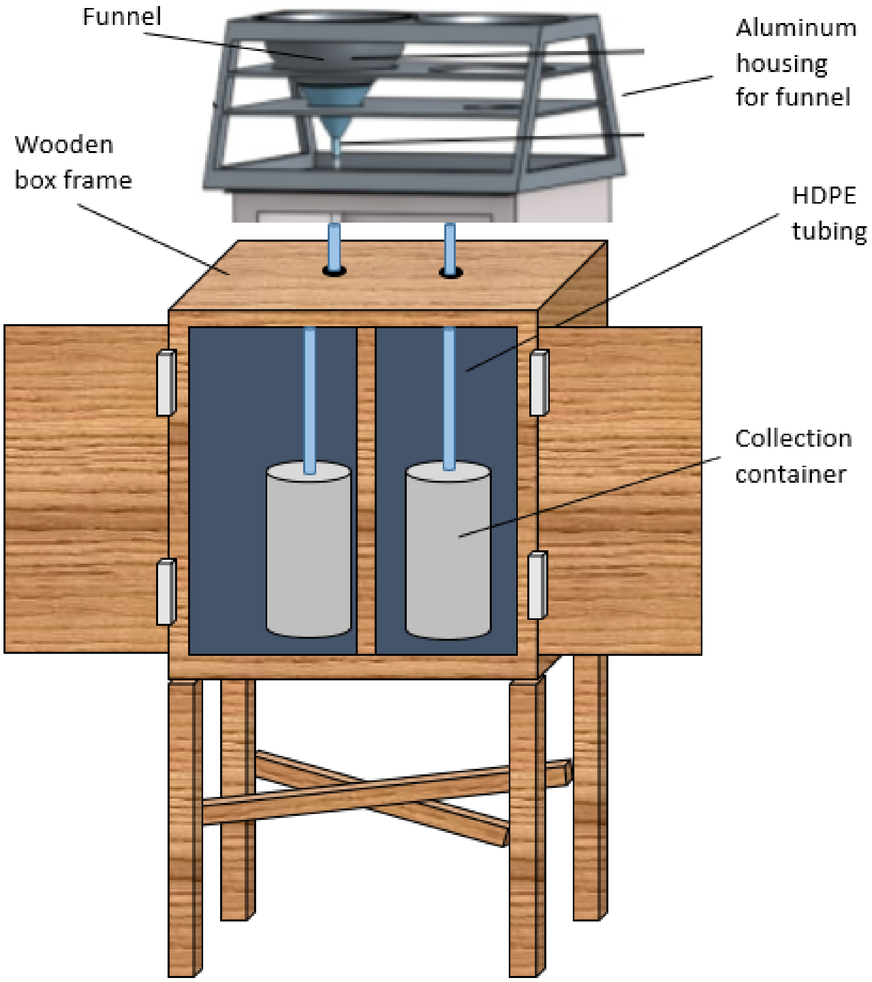

Figure 4. Schematic of bulk atmospheric deposition sampler.

Phase 2 Sampling Design & Procedures

Our Phase 2 objective was to home in on potential sources by focusing sampling in areas where the highest PFAS levels (concentrations and/or instantaneous loads) were found during Phase 1. Our sampling design was therefore largely based on results from Phase 1. Since Phase 1 results indicated PFAS were widespread within the lower watershed, we were also interested in characterizing PFAS in diffuse sources. Our Phase 2 sampling design consisted of three components: (1) Phase 1 follow-up sampling; (2) diffuse source characterization; and (3) opportunistic sampling.

Phase 1 Follow-Up Sampling

Focus Subbasins

We focused on three tributary subbasins where the highest PFAS levels were found during Phase 1 to conduct intensive follow-up sampling: Cedar River, Juanita Creek, and Thornton Creek subbasins. Although the Sammamish River had the highest tributary loads to Lake Washington during our Phase 1 sampling, we did not include the subbasin in Phase 2 because of its large geographic size and complexity. We recommend a separate study be conducted to more adequately assess sources within the subbasin.

Table 2 shows watershed characteristics for the subbasins. Brief descriptions of the focus subbasins are provided below.

Table 2. Watershed characteristics for the seven tributary subbasins in the Greater Lake Washington watershed sampled during Phase 2.

Land use information was sourced from King County's Stream Reports[1].

Land use characteristics were not available for Ravenna Creek or Fairweather Creek.

Subbasin	Approximate Subbasin Area (acres)	Agriculture	Developed	Forest	Scrub	Wetlands	Other
Cedar River	120,320	<1%	13% (Commercial, industrial, residential in the lower watershed)	70% (mostly upstream)	10%	2%	>4%
Juanita Creek	4,000	0%	88% (Residential, Commercial)	10%	<1%	1%	<1%
Thornton Creek	7,402	0%	96% (N. Branch – mostly residential; S. Branch – mostly commercial upstream, mostly residential downstream)	3%	<1%	<1%	<1%
Fairweather Creek	600	no data	no data (Commercial, residential)	no data	no data	no data	no data
Kelsey Creek	10,870	0%	86% (Commercial, residential)	12%	<1%	<2%	<1%
McAleer Creek	5,700	0%	92% (Residential)	6%	<1%	<1%	<2%
Ravenna Creek	1,294	no data	no data (Commercial, residential)	no data	no data	no data	no data

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[1] https://green2.kingcounty.gov/streamsdata/WaterShedInfo.aspx

Subbasin Descriptions

Cedar River

The Cedar River originates in the Cascade Mountains and flows for about 45 miles to its outlet to Lake Washington in Renton. It has the second largest drainage area to Lake Washington after the Sammamish River subbasin. The Upper Cedar River watershed is mostly forested and protected. Land uses become increasingly developed (mostly residential and commercial) towards Renton. The Renton Airport and aerospace industrial land uses lie adjacent to the river near the outlet.

The Renton Municipal Airport and aerospace facility are underlain by artificial fill material and native river and lake deposits. The native deposits are comprised of silt, sand, and gravel interbedded with peat (Roy F. Weston, Inc. 2001). Shallow groundwater near the mouth either flows toward the Cedar River or Lake Washington.

Renton Municipal Airport has been in operation since the end of World War II. A fire training area operated at the airport from the late 1950s until 1970 (SEACOR 1993). The fire training area was located approximately 0.4 miles south of Lake Washington, adjacent to the west bank of the river (SEACOR 1993).

The approximately 277-acre aerospace manufacturing facility has operated adjacent to the river near the mouth since 1941. Manufacturing operations include parts preparation, metal finishing and cleaning, mechanical assembly, painting, testing, and other support operations.

Hazardous substance releases to soil and groundwater have been confirmed at various locations within the adjacent aerospace facility. Release investigations and remediation and cleanup activities to address confirmed releases have occurred (WSP 2022). However, the regulatory framework for addressing PFAS releases did not exist until recently. Therefore, characterization of these releases has not included PFAS analysis.

According to satellite images (Google Earth Historical Imagery) an area used for parking of Aircraft Rescue and Firefighting (ARFF) vehicles is located in the approximate upgradient direction of Phase 2 groundwater sample locations along the east bank of the river, to the south of Phase 1 locations. The additional locations were added near the mouth of the river to increase spatial distribution of groundwater samples along this industrial reach of the river, where groundwater is known to discharge to the Cedar River.

Industrial land use areas also exist further upstream within the Larry Phillips Natural Area. The plateau to the east of the Cedar River is currently occupied by a composting facility. Historical industrial land use includes the Queen City Farms Superfund Site (listed in 1984). Release areas within Queen City Farms included a pig farm that used municipal solid waste as feed, a former plating facility, unlined liquid waste ponds (period of use 1955 to late 1960s; waste included paint, petroleum products, organic solvents, and oils), buried hazardous waste drums, a solvent recycling and reformulating facility (1966-1986), gravel mining (mid 1970s-1992), and a general aviation airport (EPA 2020). Currently, 26 acres of the former Queen City Farms is occupied by the composting facility.

Three aquifers underlying Queen City Farms were characterized as part of site investigations (EPA 2008). The uppermost aquifer is composed of sand and gravel (recessional glacial outwash deposits) that is perched on top of a leaky layer of clayey silt (aquitard). This aquifer recharges the aquifers beneath it through the leaky aquitard, and small springs in the southern portion of the site. Underlying the uppermost aquifer is a second aquifer composed of coarse sand and gravel in the upper portions, and fine sand, silt, and gravel in the lower portion. The deepest (third) aquifer is also composed of fine sand, silt, and gravel. The two deeper aquifers serve as a drinking water source to nearby residences.

The plateau is in an area of aquifer recharge. Precipitation seeps into the ground to refill the aquifer resulting in radial groundwater flow in the two deeper aquifers. There is a downward vertical gradient, with a component of flow towards the Cedar River. Based on the local hydrogeology (Landau Associates, 2020) and topography, the likely source of the springs that discharge from the hillside above the right (northeast) bank of the Cedar River is an aquifer in the coarse-grained glacial unit (advance outwash), or older/deeper deposits.

Queen City Farms is bordered to the north by Cedar Hills Regional Landfill. Regional groundwater flow beneath the landfill is to the north, towards McDonald Creek, within the Issaquah Creek basin. A 7-mile-long pipeline transports leachate from the landfill, paralleling 228th Avenue SE (east of Queen City Farms), Cedar Grove Road SE (south of Queen City Farms), and the Maple Valley Highway, which parallels the Cedar River. The pipeline transports the leachate to the King County South Treatment Plant in Renton (Jacobs 2021a). Three releases along the pipeline occurred in the fall of 2020 (Jacobs 2021a, 2021b, 2021c). Documentation of the spills and cleanup was submitted to Ecology. Ecology determined cleanup was sufficient and all three releases received a No Further Action determination.

This study also included sampling wells that are part of the City of Renton's network of resource protection wells. The resource protection wells (also called monitoring wells) are used solely for monitoring purposes. The wells are used to help identify and manage known or potential impacts to the aquifer from spills or releases of hazardous substances. Most of these monitoring wells draw water from the Cedar Valley Aquifer.

The Cedar Valley Aquifer underlies the lower end of the Cedar River valley. The aquifer is shallow, approximately 23 ft below ground surface, making it prone to contamination. An aquifer protection zone was established in 1992, which included limiting storage and handling of hazardous materials. Approximately 87 percent of the Renton's drinking water is supplied by the Cedar Valley aquifer. The drinking water wells (also called production wells) are in Liberty and Cedar River parks.

Ten of the twelve monitoring wells sampled for this study are located around the downtown production wells, within the aquifer protection zone. Two monitoring wells are located outside of the urban downtown area, above the Maplewood Aquifer. The Maplewood Aquifer, which underlies the vicinity of the Maplewood Golf Course in Renton, is also used for drinking water by the City. Monitoring well (MW33) is located in an 11-acre park (Riverview Park), above where the Maplewood Aquifer thins, and groundwater flows upward from below (PGG 2012).

The other monitoring well, MW36, is located adjacent to a golf course, within a shallow aquifer overlying the deep Maplewood Aquifer (135-345 ft below ground surface).

Sampling of the production wells to test for PFAS was completed in March 2023 by the City as part of required monitoring of unregulated contaminants (Safe Drinking Water Act Fifth Unregulated Contaminated Monitoring Rule). PFAS were not detected in downtown production wells or the Maplewood well field (City of Renton, 2024).

Juanita Creek

Juanita Creek originates from wetlands and springs east of I-405 and flows for about five miles to its outlet to Lake Washington in Juanita Beach Park in Kirkland. Several main tributaries flow into Juanita Creek including Upper West (Simonds Tributary), Lower West (Billy Creek), and Lower East tributaries (Totem Lake Tributary). Juanita Creek subbasin is heavily developed (mostly residential and commercial), with about 68% of land cover classified as impervious surfaces such as pavement or roofs (King County 2012). Residential development along the stream banks has made Juanita Creek highly confined (City of Kirkland 2023).

Thornton Creek

The north branch of Thornton Creek originates at Ronald Bog adjacent to I-5, and the south branch originates from wetlands west of I-5. Corliss landfill, which operated from 1946 to 1959, extends from North 167th to North 163rd, and is bounded to the west by the housing development (east of Meridian Avenue), and by 1st Avenue Northeast to the east (PHSKC 1985). The depth of fill ranges from 2 to 37 ft. Coarse-grained glacial (advance) outwash deposits underlies the waste in the northern portion of the former landfill (Booth et al. 2009). Currently the Shoreline Recycling and Transfer Station is located on the eastern portion of the former landfill.

The two branches join just upstream of Meadowbrook Pond, a stormwater detention and flood control pond. During the wet months, overflow of Thornton Creek is directed into the ponds. This slows the water flow and sediments settle out before water re-enters Thornton Creek. Sediment accumulation is monitored, and removal occurs as needed. The Thornton Creek system flows for 18 miles to its outlet to Lake Washington at Matthews Beach Park in Seattle. Thornton Creek subbasin is heavily developed (mostly residential and commercial), with about half of the watershed covered by impervious surfaces (Thornton Creek Watershed Management Committee 2000).

Sampling Procedures

Surface Water

In each focus subbasin, tributary surface water grab samples were collected during one summer low flow event in August/September 2022 and one spring high flow event in March/April 2023. We sampled 5 – 10 sites upstream to downstream in each subbasin (Appendices C and D). Surface water grab samples were collected during both flow events for analyses of PFAS, TOC, DOC, and TSS using the same procedures as in Phase 1. YSI EXO sonde measurements for water temperature (°C), dissolved oxygen (mg/L), pH, and specific conductance (µS/cm) were also collected.

During three storm events (October 25, 2022, November 30, 2022, and February 7, 2023) we collected surface water grab samples at the outlet sites of each tributary to compare and assess PFAS concentrations and instantaneous loads during storm events. YSI EXO sonde measurements for water temperature (°C), dissolved oxygen (mg/L), pH, and specific conductance (µS/cm) were also collected.

Groundwater

Groundwater samples were collected during the summer 2022 and spring 2023 sampling events. Groundwater samples were analyzed for PFAS, DOC, sulfate, and chloride. A YSI EXO sonde was used for Phase 2 to collect measurements for water temperature (°C), dissolved oxygen (mg/L), pH, specific conductance (µS/cm), oxidation-reduction potential (mV), and turbidity (FNU — Formazin Nephelometric Units). PFAS samples were collected using HDPE sample containers. DOC samples were collected using a clean polyethylene transfer bottle, filtered using a 0.45-micron polypropylene filter and syringe, and discharged into the polyethylene sample container. Sulfate and chloride samples were collected using polyethylene sample containers.

Groundwater samples collected at the groundwater-surface water interface were collected using the PushPoint sampler, using the same procedures described for Phase 1. When possible, groundwater sites were co-located with a subset of the surface water sites, otherwise alternative sites were sampled for groundwater discharge.

In addition to the co-located Cedar River subbasin locations, 12 City of Renton monitoring wells were sampled in the urban lower basin of Cedar River during summer 2022 and spring 2023 (MW3 through MW5, MW10, MW11, MW25S, MW30S, MW31S, MW33, MW36S, and MW42). This network of resource protection wells was installed to protect the City's water supply wells. We used Pacific Groundwater Group's well prioritization report (2012) to guide well selection. Only shallow monitoring wells were sampled. All wells were screened below the water table. Monitoring well details and a risk assessment based on potential sources of contamination are discussed in the well prioritization report (PGG, 2012).

Dedicated pumps and tubing were removed from all wells two months prior to sampling. All monitoring wells except for one (MW33) were sampled using a QED Sample Pro bladder pump with a low-density polyethylene bladder, and HDPE tubing. Monitoring well MW33 was sampled using a submersible Mini-Typhoon pump with low-flow sampling controller and new Tru-Poly tubing. The same low-flow sampling techniques described for Phase 1 were used for monitoring well sampling.

Springs

Water samples were collected at two springs located adjacent to Cedar River during the summer 2022 and spring 2023. One area of discharge is located in the Larry Phillips Natural Area, and one is in the Landsburg Reach Natural Area. Discharge from springs generally occurred over a broad area, on a hillside and/or at the base of a hillside rather than a distinct point. Samples were collected as close to the point of discharge as possible, where sufficient flow allowed for filling of the bottle. Stagnant pools or seeps were not sampled. The same procedure used to collect tributary grab water samples was used. Flow from the springs was not sufficient to submerge the sample bottle 15 cm below the surface.

Samples collected from springs were analyzed for PFAS, DOC, sulfate, and chloride. A YSI EXO sonde was used for Phase 2 to collect measurements for water temperature (°C), dissolved oxygen (mg/L), pH, specific conductance (µS/cm), oxidation-reduction potential (mV), and turbidity (FNU).

Suspended Sediments

Suspended sediment samples were collected at one suitable downstream location in each tributary during the summer and spring events. Suspended sediments were collected at each site using a Hamlin trap deployed for about four weeks. Upon retrieval, sediment in the trap was scooped into a stainless-steel bowl and mixed using a stainless-steel spoon. The mixed sediment was then scooped into separate sample containers for PFAS, TOC, and grain size analyses.

Biofilm and Macroinvertebrates

As an exploratory part of our study design, six biofilm and aquatic larval macroinvertebrate (caddisfly) samples were collected concurrently at six surface water sites on the Cedar River during summer to provide information on bioaccumulation of PFAS in aquatic organisms that may serve as food sources for fish. Biofilm samples were collected using the same procedures described in Phase 1.

Caddisfly larvae samples were collected individually from rocks or the streambed. The casing of each larva was removed using stainless-steel tweezers, and a scale was used to weigh at least 2 g of soft tissue sample for PFAS analysis. Tissue was stored in an HDPE sample container until further processing. Back at Lacey headquarters, the caddisfly tissue was homogenized using a mortar and pestle, then scooped into an HDPE sample container for PFAS analysis using a stainless-steel spoon and stored frozen at -20°C.

Additional Subbasins

We collected samples in four additional tributary subbasins where among the highest PFAS levels were found during Phase 1: Fairweather, Kelsey, McAleer, and Ravenna Creek. Table 2 shows watershed characteristics for the four subbasins. Brief descriptions of each subbasin are provided below.

Subbasin Descriptions

Fairweather Creek

Fairweather Creek lies within a small drainage, flowing for about 1.4 miles before emptying into Lake Washington in Hunts Point. A majority of the stormwater runoff from an adjacent mostly low-density residential community is directed into two detention ponds on a golf course and is eventually discharged into Fairweather Creek. Surrounding land uses are predominantly residential but also include the golf course and the 520 bridge (520-W) overlying the creek near the outlet.

Kelsey Creek

Kelsey Creek originates from wetlands between Phantom Lake and Larsen Lake (a small peat bog lake), and flows for about 10.7 miles before emptying into Mercer Slough underneath I-90, adjacent to Lake Washington in Bellevue. Historically, Phantom Lake served as the headwaters for Kelsey Creek, but the outlet of Phantom Lake was redirected to Lake Sammamish in the late 1800s. Kelsey Creek receives flows from several smaller tributaries including Richards, Sunset, West, Goff, and Valley Creeks. Kelsey Creek subbasin is heavily developed (mostly commercial and residential), with about 41% of land cover classified as impervious (Jacobs Engineering and Herrera Environmental Consultants 2021).

West Tributary flows through an industrial area of the Kelsey Creek subbasin. The area is underlain by glacial till, both advance and recessional glacial outwash deposits (Minard 1983). Artificial fill and wetland deposits were observed in the immediate vicinity of West Tributary (Enviros 1992). Shallow perched groundwater occurs in the glacial till. The regional aquifer occurs at deeper depths in the outwash deposits. Depth to shallow groundwater ranges 4 –16 ft below ground surface. Groundwater flows toward and discharges to West Tributary at the bus base (Paramatrix 2021). Groundwater flow direction is uncertain in areas outside of the bus base but likely flow towards West Tributary.

McAleer Creek

McAleer Creek lies within a heavily developed (mostly residential), medium-sized drainage. McAleer Creek originates from Lake Ballinger, which receives surface waters from Hall Creek, and flows for about six miles to its outlet to Lake Washington in Lake Forest Park. Hall Creek parallels a highly urbanized corridor along Highway 99, and originates from Hall Lake, a small kettle lake adjacent to I-5. Hall Lake receives stormwater runoff from I-5, and all the subdrainages receive stormwater runoff from surrounding local jurisdictions.

Ravenna Creek

Ravenna Creek is a small drainage originating from springs in Ravenna Park that eventually empties into Union Bay in Seattle. About 0.7 miles of the creek is currently daylighted in Ravenna Park. Downstream from the park, the creek flows into an underground pipe through mostly commercial and residential land uses. The creek resurfaces downstream of Montlake Boulevard (SR-513), where it then flows through Union Bay Natural Area and into Union Bay.

Union Bay Natural Area was once the site of the Montlake Dump, which operated from 1926 to 1966. The dump was used by the City of Seattle for household waste and by local industries and individuals who hauled their own waste to the site. Ecology records indicate the area on the west side of Ravenna Creek, north of NE Wahkiakum Lane (currently a playfield and track field) and south of NE Clark Road may contain industrial waste. However, it is only recorded in meeting notes with no details regarding the source of the information.

Depth of garbage fill from the Montlake Dump varies from 6 to 40 ft. Peat, varying in thickness from a few ft to approximately 40 ft, underlies the garbage, and silt and clay underlie the peat (PHSKC 1984, 1999). Dikes of timber and rubble were constructed to create cells for fill and to stabilize the peat. The channel through which Ravenna Creek flows (formerly known as the central canal and the University Slough) was originally a drainage canal excavated through landfill cap and fill material. Ravenna Creek was disconnected from the sewage system and directed to discharge to the central canal in 2005.

After the dump was closed and capped in 1966 (prior to the establishment of Washington State closure standards for landfills in 1973), the area became a site of environmental restoration and education, protection of wildlife habitat, recreational walking trails, and birdwatching. Current use west of the channel includes sports fields, parking lots, and a driving range. Current use upgradient of the dump includes grocery and retail stores, a golf driving range, and a sports field.

This study includes sampling of two monitoring wells located in the Union Bay Natural Area, within the footprint of the dump (UW-MW1 and UW-MW2). The wells are located on a 1.5-acre urban organic farm and educational facility. Irrigation on the farm varies by season. Soil amendments depend on the season, crop, and soil conditions. The farm follows National Organic Program and Good Agricultural Practices, which includes standards for compost. A review of soil amendments and composting practices at the farm was not conducted as part of this study.

The wells were installed as part of a 2016 investigation evaluating potential contamination associated with the landfill (GeoEngineers 2017). A total of six borings were advanced to 10-12 ft below ground surface (bgs). Groundwater was encountered in two of the six borings at 5 and 9 ft bgs during a December event. Landfill debris was only encountered at UW-MW2 at a depth of 9 ft below ground surface. The two monitoring wells installed during the investigation were not sufficient to determine groundwater flow direction.

Sampling Procedures

In each additional subbasin, we sampled 5 – 10 sites along the upstream to downstream gradient, during one summer low flow event in August/September 2022. Surface water grab samples were collected for PFAS analysis only. In Fairweather Creek subbasin, we only sampled two tributary surface water locations because of the small size of the creek.

In Ravenna Creek subbasin, we included collection of summer (August/September 2022) and spring (April/May 2023) groundwater samples, co-located with Ravenna Creek surface water grab sample locations. Groundwater samples were collected for analyses of PFAS, DOC, chloride, and sulfate using the PushPoint sampler procedure described for Phase 1. PFAS samples were collected using HDPE sample containers. DOC samples were collected using a clean polyethylene transfer bottle, filtered using a 0.45-micron polypropylene filter and syringe, and discharged into the polyethylene sample container. Sulfate and chloride samples were collected using polyethylene sample containers.

In addition to the co-located sample locations, groundwater samples were collected from two monitoring wells at the northern end of the Union Bay Natural Area in the Ravenna Creek subbasin. Samples were collected using a peristaltic pump and new HDPE and silicone tubing. Insufficient water was available for sampling during the summer event. An attempt was made to collect samples during the summer event. Both wells went dry immediately upon purging and the field schedule did not permit enough time for the wells to recover. During the spring event, both monitoring wells went dry shortly after purging began (less than a liter of purge water was generated). Both wells were slow to recover and so were left overnight. Samples were collected without purging the following day. Collection of PFAS samples was prioritized due to the low volume of water and slow recovery. PFAS and DOC samples were collected at well UW-UW1 and one PFAS sample was collected from well UW-MW2.

Lake-Shoreline Groundwater Locations

We revisited two Phase 1 Lake Washington shoreline locations, MADRONA and WETMORE.

The MADRONA sample, located on the west shore of the lake near the Seattle neighborhood of Madrona, had the highest Phase 1 total PFAS concentration detected in groundwater. Phase 2 sampling included two events (summer 2022 and spring 2023). Sample locations included groundwater and springs. Groundwater sampling included resampling at the Madrona location and adding locations along the shoreline to the north and south (HARRISON and KING ST respectively). Spring sampling included two areas above the shoreline where springs discharge along the hillside forming small creeks – Madrona Creek upland of the Madrona sample location, and Frink Creek upland of the KING ST sample location. Spring samples (FRINK-SP and MADRONA-SP) were collected as close to the headwaters of the creeks as possible.

The Genesee Landfill, formerly the Wetmore Slough, is located upgradient of the Wetmore area sample locations. The landfill was in operation from 1947 until approximately 1968 and was used for municipal waste, commercial waste, and served portions of the industrial area south of downtown Seattle (PHSKC 1984). The garbage ranges from 5 to 30 feet thick. Parts of the surrounding neighborhood, including Rainier Playfield, were filled with debris at an earlier date. The landfill area is now a park with playfields.

The WETMORE sample had the highest Phase 1 PFOS concentration detected in groundwater. Additional groundwater samples were collected at the Wetmore location and along the shoreline to the north (BAKER), during the summer 2022 and spring 2023 Phase 2 events. A surface water grab sample was collected from Lake Washington at the Wetmore location during the spring event. A sample location along the shoreline to the south was planned but conditions were not conducive to using the PushPoint sampler (guarded shoreline and nearshore cobble size sediment).

In addition to the Madrona and Wetmore sample locations, we revisited groundwater sample locations along the shoreline that were located within two of the focus subbasins. We resampled at the Matthew’s Beach sample location (MB), located within the Thornton Creek subbasin. We also resampled at Juanita Beach Park (JBP and JBP2), located within the Juanita Creek subbasin.

Lake Washington

One surface water grab sample for PFAS analysis was collected from each of three sites in the middle of Lake Washington (LAKE-NORTH, LAKE-MID, and LAKE-SOUTH) during the summer event. YSI EXO sonde measurements for water temperature (°C), dissolved oxygen (mg/L), pH, and specific conductance (µS/cm) were also collected. Sites were accessed by boat and sampled using the same procedures as in Phase 1.

Diffuse Source Characterization

Surface Runoff and Road Dust

As an exploratory part of our Phase 2 study design, samples were collected to characterize PFAS in runoff and particulates from paved surfaces. We collected four runoff and four road dust samples in each of the three focus subbasins in publicly accessible areas representing residential, commercial, open space, or industrial land uses (Table 3). The sites sampled were either paved parking lots or road surfaces. We also collected one runoff and road dust sample from the 520 bridge to represent transportation land use.

Table 3. Descriptions of surface runoff and road dust sites in the Cedar River, Juanita Creek, and Thornton Creek subbasins sampled during Phase 2.

Location ID	Subbasin	Land Use Type	Site Description
CR-COMM	Cedar	Commercial	Parking lot of shopping mall area
CR-IND	Cedar	Industrial	Streets surrounding aviation/aerospace industrial land uses
CR-OPEN	Cedar	Open	Parking lot of Landsburg Park
CR-RES	Cedar	Residential	Streets within low density residential area
JC-COMM1	Juanita	Commercial	Streets surrounding auto service land uses
JC-COMM2-COMP	Juanita	Commercial	Streets surrounding food service/retail land uses
JC-COMM2-GAS	Juanita	Commercial	Paved area surrounding gas station
JC-COMM2-WASH	Juanita	Commercial	Street down gradient of car wash
JC-RES1	Juanita	Residential	Streets within low density residential area
JC-RES2	Juanita	Residential	Streets within high density residential area
TC-COMM1	Thornton	Commercial	Parking lot of shopping mall area
TC-COMM2	Thornton	Commercial	Streets surrounding auto service land uses
TC-OPEN	Thornton	Open	Parking lot of Matthews Beach Park
TC-RES	Thornton	Residential	Streets within high density residential area
520-W	520 Bridge	Transportation	Streets/scuppers along 520 bridge, west-bound

Road dust samples were collected during one sampling event (September 2022) in which 0.6 mm of rainfall was recorded nine days prior to sampling. Samples were collected using field methods adapted from Van Metre et al. (2008). Briefly, a polypropylene-bristle hardwood handle brush was used to sweep particulates into a Reynolds® heavy duty aluminum foil lined dustpan. A new brush and foil-lined pan was used for each site. At each site, a 4 m string was used to outline sampling spots of approximately 1 m^2. Sub-samples were swept and composited from several spots within the sampling area. The number of sub-samples used to create composites depended on the size of the area and amount of particulates present at each site. On average, we swept about 10 m^2 per site.

The composited sweepings were poured through a 0.5 mm stainless-steel mesh sieve into a stainless-steel bowl to remove coarse particulates. The composite sample was mixed using a stainless-steel spoon, then scooped into separate sample containers for PFAS and TOC analyses.

Surface runoff samples were collected from the same sites as road dust samples. Most sites were sampled during a storm event on January 12, 2023, while the remaining sites were sampled during a storm event on January 18, 2023. Samples were collected primarily as sheet runoff. A Reynolds® heavy duty aluminum foil lined dustpan was used to capture runoff water before draining into catch basins within the sampling area. A new foil-lined pan was used for each site. As with road dust samples, runoff samples were composited from several spots within the sampling area depending on size of the area. Water from the pan was poured into separate sample containers for analyses of PFAS, TOC, DOC, and TSS.

Bulk Atmospheric Deposition

Bulk atmospheric deposition samples were collected at two sites: Ecology's air monitoring stations at Beacon Hill in Seattle (BH-ATM) and in North Bend (NB-ATM). We added the North Bend site to compare PFAS concentrations in samples collected from outside the urban growth boundary surrounding Lake Washington. North Bend's urban growth area of about six square miles (City of North Bend 2015) is a fraction of King County's total urban growth area of about 460 square miles (King County 2021), most of which includes urban areas surrounding Lake Washington and along the I-5 and 405 corridors.

Prevailing winds in Seattle are south/southwest during the wet season and northwest during the dry season. In North Bend, prevailing winds are variable throughout the year: West from April to September; east from September to October and December to February; and south from February to April and October to December.

Bulk atmospheric deposition samples were collected at both sites during five deployment events:

- Deployment 1: September 22 – November 3, 2022
- Deployment 2: November 3 – November 17, 2022
- Deployment 3: November 17, 2022 – November 30, 2022
- Deployment 4: November 30, 2022 – December 16, 2022
- Deployment 5: December 16, 2022 – January 9, 2023

We used the same methodologies to sample bulk atmospheric deposition as in Phase 1. One modification was replacing the 4-liter amber glass collection containers with 8-liter stainless-steel collection containers.

Opportunistic Sampling

"Opportunistic" samples were collected to assess impacts from potential sources as we acquired knowledge about them, or interest from project collaborators to have them sampled. In addition, two stormwater sites from Phase 1 were resampled as follow-up. Opportunistic samples included the following:

Airport Stormwater

Stormwater from the Renton Airport was sampled during a storm event on November 30, 2022, as follow-up to sampling conducted during Phase 1 at the same site (REN-CMU). The sample was collected from a storm conveyance system that collects runoff from the airport runway and surrounding area and drains into the Cedar River near its outlet to Lake Washington. The site was accessed with permission and escort by City of Renton staff. One PFAS grab sample was collected from the site using an HDPE sample bottle attached to a telescopic sampling pole.

Residential Stormwater

Stormwater was sampled from a public stormwater outfall draining a small (<5 acre) residential area within a single-family zoning district as follow-up to Phase 1 sampling conducted at the same site (BEL-SHORE). The outfall is a 12-inch corrugated metal gravity main that drains directly to Lake Washington in Meydenbauer Bay. The sample was collected during a storm event on November 4, 2022. One PFAS grab sample was collected using an HDPE bottle to catch water discharging from the outfall pipe.

I-90 Bridge Stormwater

Stormwater from the I-90 bridge was sampled during a storm event on November 22, 2022. The sample was collected from a discharge point located on the eastern side of the bridge that drains the floating bridge section from the abutment. Permission to sample was approved by Washington State Department of Transportation (WSDOT), and the site was accessed with escort by WSDOT staff. One PFAS grab sample was collected using an HDPE bottle attached to a telescopic sampling pole.

Stormwater from Turf and Grass Athletic Fields

Stormwater from four neighboring athletic fields in Seattle were sampled during a storm event on February 7, 2023. These samples were collected to assess whether fields composed of artificial turf (which may contain PFAS) contribute to PFAS in runoff. Two fields were artificial turf (FieldTurf® and AstroTurf®), and the other two were grass. The AstroTurf® field was installed in 2021. The age of the FieldTurf® field is unknown but is older than the AstroTurf® field.

One grab sample for PFAS was collected from each of the four drainage systems carrying stormwater from the fields to Lake Washington. For the AstroTurf® and FieldTurf® fields, a control sample was collected from an on-site drainage system that did not capture stormwater from the fields.

Horse Creek

Two surface water grab samples were collected from Horse Creek to assess potential impacts from historic AFFF use during fire training and contaminated sites in the area. Horse Creek empties into the Sammamish River in Bothell. One PFAS grab sample was collected upgradient from the historic fire training/contaminated sites area, and another was collected downgradient of the area. Samples were collected during a storm event on November 22, 2022.

West Tributary

Two surface water grab samples and two groundwater samples were collected from West Tributary, in Bellevue, to assess potential impacts from historic AFFF use during firefighter trainings and to assess PFAS concentrations and composition in an industrial area. Sample locations were in the inferred downgradient direction of the fire training area and multiple current and historic industrial facilities. Surface water grab samples were collected for PFAS, TOC, and DOC analyses using the same procedures described for lake and tributary surface waters samples. Groundwater samples were collected using the Push Point sampler using the same procedure described for Phase 1. YSI EXO sonde measurements for water temperature (°C), dissolved oxygen (mg/L), pH, and specific conductance (µS/cm) were also collected at each site for surface water and groundwater. Samples were collected in the spring of 2023.

Laboratory Methods

PFAS samples were stored on ice during the sampling day and transferred to -20°C or less within 48 hours of collection. Frozen PFAS samples were then shipped overnight to the analytical laboratory, and samples were analyzed within 90 days from collection.

During Phase 1, PFAS samples were analyzed by SGS AXYS Laboratory in Sydney, B.C., Canada. All samples were analyzed for 40 target PFAS analytes using liquid chromatography-tandem mass spectrometry with isotopic dilution following the laboratory's procedure, *MLA-110: Analytical Procedure for the Analysis of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous Samples, Solids, Tissues, AFFF Products, Blood/Serums and Solvent Extracts by LC-MS/MS*. A list of the 40 target PFAS analytes is provided in Appendix A.

To assess the presence of unknown PFAS precursors, Phase 1 water and sediment samples were also analyzed using the TOP assay method following the laboratory's procedure, *MLA-111: Analytical Procedure for the Analysis of Total Oxidizable Precursors (TOP) in Aqueous and Solid Matrices.* The TOP assay converts oxidizable PFAS precursors into PFCA end products through hydroxyl radical based oxidation reactions.

During Phase 2, water samples were analyzed for 40 target PFAS analytes by Manchester Environmental Laboratory (MEL) in Port Orchard, WA using first draft Method EPA 1633 (EPA 2021). Sediment, road dust, and tissue samples were analyzed for 40 target PFAS analytes by SGS AXYS Laboratory using first draft method EPA 1633. Draft Method EPA 1633 was first published in August 2021—after Phase 1 field sampling was completed, and before Phase 2 sampling began. The draft method is equivalent to SGS AXYS's MLA-110 procedure, and therefore we expected results between Phase 1 and 2 and the analytical laboratories to be comparable. For quality assurance purposes, 10% of water samples were collected as split samples to be analyzed by both laboratories.

TOC, DOC, and TSS samples were refrigerated at 4°C or less after field sampling and delivered within 24 hours to MEL in Port Orchard, WA. TOC and DOC water samples were analyzed using SM5310B. TOC-sediment samples were analyzed using EPA 440.0 and biofilm samples were analyzed for ash-free dry weight using SM10300C.

Sediment grain size was analyzed by ALS Environmental (Kelso, WA) using PSEP (1986). Samples were stored at 4°C or less prior to overnight shipping to the laboratory.

Quality Assurance/Quality Control

Field Samples

Field Duplicates

Field duplicates were collected during each sampling event at 10% of the total number of samples per matrix and sample type to assess precision of the field collection procedure and sample representativeness. For surface water and stormwater samples, field duplicates were collected by filling a second HDPE sample bottle with site water. For solid/sediment and tissue samples, field duplicates were collected by filling a second HDPE sample container with solid/sediment or tissue from the composited mixture. For groundwater samples, field duplicates were collected by filling a second HDPE sample bottle.

Equipment Blanks

Equipment blanks were also collected during each event at 10% of the total number of water samples collected for each sample type to assess contamination of field sampling equipment. For surface water and stormwater samples, equipment blanks were collected by filling an HDPE sample bottle with laboratory-grade PFAS-free water provided by the analytical laboratory.

Equipment blanks for surface runoff samples were collected by pouring laboratory-grade PFAS-free water into a clean foil-lined pan, and subsequently into an HDPE sample bottle. For road dust sample collection, equipment blanks were collected by pouring laboratory-grade PFAS-free water through a clean brush, into a clean foil-lined pan, and subsequently into an HDPE sample bottle.

Groundwater equipment blanks were collected for the PushPoint sampler, bladder pump, tubing, and water level meter. PushPoint sampler equipment blanks were collected by connecting the decontaminated device to the peristaltic pump. New silicone tubing was inserted into the pump head and new HDPE tubing was attached to the sampler. Laboratory-provided PFAS-free water was run through the tubing and discharged directly into the HDPE bottle. Approximately two liters of water was through the device and tubing prior to collecting the equipment blank.

Equipment blanks for the bladder pump involved the same method of hooking up clean HDPE tubing to the pump and running laboratory-grade PFAS-free water through the tubing. The bladder pump is designed so that water also comes in contact with the stainless-steel pump parts, buna-n nitrile o-rings, and a low-density polyethylene bladder. In addition, a decontaminated glass graduated cylinder was used to keep the pump upright, and a higher volume (approximately one gallon) of PFAS-free water was pumped through the tubing and pump prior to sample collection.

The water level meter equipment blank was collected by submerging approximately four feet of tape in PFAS-free water and agitating for approximately one-minute. The water was then poured over the length of tape into a clean, HDPE PFAS-free sample bottle.

Bulk Atmospheric Deposition Quality Control Samples

For Phase 1 and 2 bulk atmospheric deposition sampling deployment, we collected three quality control (QC) samples: A field duplicate, rinsate blank, and equipment blank. A separate funnel-container system was used for the field duplicate, and the sample and field duplicate were set up side-by-side in the same wooden box frame.

To collect the rinsate blank, water remaining in the sample collection container was dumped out, and the container was reconnected to the funnel system. 500 mL of PFAS-free laboratory blank water was squirted onto all sides of the funnel, drained through the tubing, and into the collection container to rinse any remaining PFAS from the funnel system. The collection container was then disconnected from the tubing, shaken vigorously, and water was poured from the container into an HDPE sample bottle.

To collect the equipment blank, a new system was set up consisting of a separate decontaminated funnel, tubing, and collection container. Similar to the rinsate blank procedure, 500 mL of PFAS-free laboratory blank water was used to rinse the funnel system, and the rinsate was poured into an HDPE sample bottle.

Data Validation

Phase 1 PFAS data were validated by Validata Chemical Services, Inc., an independent third-party data validation service. Phase 1 data were validated according to *Data Validation Guidelines Module 3: Data Validation Procedure for Per- and Polyfluoroalkyl Substances Analysis by QSM Table B-15*.

Phase 2 PFAS data were validated by an independent data validator at MEL. Phase 2 PFAS data were validated according to criteria from first draft method EPA 1633 (EPA 2021).

Data quality assessments were provided by the data validators in the form of case narratives. A summary of the data quality is provided below.

Data Quality Summary

Field Duplicates

Field duplicates met project measurement quality objectives (MQOs) for field duplicate pairs. Exceptions included:

- One Phase 2 suspended sediment sample (#2209083-26) and its duplicate in which the relative percent difference (RPD) in PFOS concentration exceeded 40% between samples. In this sample, concentrations of other detected PFAS analytes exceeded RPD criteria, although the concentrations were generally low.
- One Phase 2 water sample (#2208096-27) and its duplicate exceeded RPD criteria for PFOS and PFDA.
- Field duplicates for one Phase 2 bulk atmospheric deposition sample (#2212042-02RE1) were noticeably different in that one sample had no detected PFAS analytes, while the other had estimated detections of several PFCA analytes.

There were some RPD exceedances for detected PFAS analytes in other duplicate sample pairs; however, the analyte concentrations were generally low (J-qualified as estimated values).

Equipment Blanks

Equipment blanks overall met MQOs. There were several exceptions in which PFAS analytes were detected at mostly estimated (J-qualified) values. These included:

- Detections of several PFCAs in some bulk atmospheric deposition equipment blanks (samples #2101029-04, #2102029-16, #2103040-17, #2104043-17, #2211045-04, and #2211045-17).
- Detection of 6:2 FTS in two field blanks (#2104044-65 and #2304034-23) collected during surface water sampling.
- Detection of N-MeFOSE in groundwater field blank sample #2208081-05.
- Detection of PFHxS in groundwater field blank sample #2303004-08.

Laboratory Duplicates

Laboratory duplicates overall met MQOs, with the following exceptions:

- 6:2 FTS in one Phase 1 sediment sample (#2009066-53) and water sample (#2104043-06) and their lab duplicates exceeded RPD criteria of 40%.
- PFOS in one Phase 2 surface water sample (#2208093-03) and its duplicate exceeded RPD criteria.

Method Blanks

PFAS analytes that were detected in samples at less than five times the detected analyte concentration in the associated method blank were treated as non-detect (U-qualified) due to method blank contamination. These results were not included in total PFAS calculations.

Among water samples, method blank contamination resulted in U-qualification of primarily four analytes (PFBA, 6:2 FTS, PFOA, and PFOSA) in several Phase 1 samples. Method blank contamination also resulted in U-qualification of PFDoS and PFTrDA in three Phase 2 groundwater samples.

Among solid/sediment samples, method blank contamination resulted in U-qualification of four analytes (6:2 FTS, PFHxA, PFNA, and PFOSA) in several Phase 1 sediment samples. Among Phase 2 solid/sediment samples, method blank contamination resulted in U-qualification of PFBA in 10 road dust samples, and PFOSA in 5 road dust and 3 suspended sediment samples.

Among tissue samples, method blank contamination was primarily due to several analytes (PFOS, PFBA, PFOA, PFDA, and PFUnA). As a result, PFOS was U-qualified in 4 of 5 Phase 1 and 5 of 6 Phase 2 biofilm samples, as well as 3 of 6 Phase 2 macroinvertebrate samples. PFBA was U-qualified in all six Phase 1 biofilm samples and one Phase 2 biofilm and macroinvertebrate sample. PFOA was U-qualified in all Phase 1 biofilm samples. PFDA was U-qualified in one Phase 2 macroinvertebrate sample, and PFUnA was U-qualified in one Phase 1 biofilm sample.

Ongoing Precision and Recovery/Laboratory Control Samples

Ongoing precision and recovery and laboratory control samples for both Phase 1 and 2 datasets met MQOs for all detected analytes.

Extracted Internal Standards (EIS)

EIS recovery exceedances (outside recovery limits of 50 – 150%) resulted in J-qualification and potential bias for several analytes in multiple samples. The most consistent exceedances in Phase 1 water samples included high surrogate recoveries of 4:2 FTS (150 – 297%), 6:2 FTS (150 – 311%), and 8:2 FTS (150 – 219%), indicating potential high bias for those analytes, and low surrogate recovery for PFBA (3 – 30%) indicating potential low bias for PFBA. Among Phase 2 water samples, high surrogate recoveries included N-MeFOSAA (154 – 216%), N-EtFOSAA (151 – 220%), 4:2 FTS (151 – 414%), 6:2 FTS (151 – 426%), 8:2 FTS (152 – 398%), and PFHpA (151 – 201%) in some samples. Low surrogate recoveries included PFBA (4 – 30%), PFPeA (10 – 50%), N-MeFOSE (30 – 50%), N-EtFOSE (30 – 50%) in some samples.

Among Phase 1 sediment samples, high surrogate recoveries in some samples included 4:2 FTS (154 – 200%), 6:2 FTS (170 – 177%), and 8:2 FTS (161 – 208%), N-MeFOSAA (153 – 155%), and N-EtFOSAA (151 – 193%). Low surrogate recoveries in some Phase 1 and 2 solid/sediment samples included PFBA (8 – 48%), MeFOSA (9 – 50%), MeFOSE (10 – 50 %), and EtFOSE (12 – 50%).

Among Phase 1 biofilm samples, high surrogate recoveries of 4:2 FTS (159 – 204%) and N-EtFOSAA (196 – 240%) occurred in 3 of the 6 samples collected. Among Phase 2 tissue samples, EIS recoveries for 24 surrogate compounds were below recovery limits in some samples. Only 4 PFAS analytes (PFOS, PFBA, 6:2 FTS, and PFMBA) were detected among the Phase 2 tissue samples.

Matrix Spike/Matrix Spike Duplicates

Matrix Spike/Matrix Spike Duplicates (MS/MSDs) overall met MQOs (50 – 150% recovery). Exceptions included:

- Low recovery of 3:3 FTCA (10 – 49%) in some Phase 1 and 2 MS/MSD water samples.
- Low recovery of PFBA (28 – 43%) in some Phase 2 MS/MSD water samples.
- High recoveries of N-EtFOSAA (167 – 176%), N-MeFOSAA (229 – 232%), PFMBA (155 – 167%), and PFMPA (150 – 227%) in some Phase 2 MS/MSD water samples.
- High recoveries of 7:3 FTCA (153%), 5:3 FTCA (166%), NFDHA (362%), and PFOS (154%) in the Phase 2 MS/MSD biofilm sample.

Laboratory Split Samples

Water samples split between MEL and SGS AXYS for laboratory analysis during Phase 2 were overall comparable. The RPD was less than 40% for approximately 97% of analyte results. For results that exceeded 40% RPD, analyte concentrations were generally low. Reporting limits between the two laboratories were generally within five times of each other. Differences in reporting limits between the two laboratories did not seem to consistently affect analyte detections in one laboratory versus the other.

Data Reporting and Analysis

In this study, the total PFAS concentration of a sample was defined as the sum of 40 target PFAS analyte concentrations in the sample. PFAS data qualified as U (non-detect) were treated as 0s for analysis. Detected analytes and analytes qualified as J (estimate) and NJ (tentatively identified and estimated) were included in total PFAS calculations.

We used exploratory approaches in our data analyses. Graphs and maps were used to compare PFAS concentrations and look for spatial and temporal patterns in the data. Scatterplots were used to explore relationships between PFAS analytes and water chemistry variables. Relationships between non-parametric variables were tested using Spearman's rank order correlation (rho; ρ) with a predetermined significance level of $p<0.05$. Linear relationships between parametric variables were tested using Pearson correlation (R) with a predetermined significance level of $p<0.05$, applying transformations to normalize the data if necessary. Principal component analysis (PCA) was used as a dimensionality reduction tool to explore variations in PFAS composition among samples, using PFAS analyte concentrations relative to the total concentration.

We also conducted desktop research on current land uses and site history to assess potential PFAS source areas. Analysis using Geographic Information Systems was performed to estimate the percentage of impervious land cover within the tributary subbasins. The Coastal Change Analysis Program's 2015 – 2017 30-meter impervious surface land cover layer (Office for Coastal Management 2024), and USGS's National Hydrography Dataset Plus catchment datasets (USGS 2024) were used to perform the analysis. The impervious land cover estimates were used to analyze the relationship between subbasin imperviousness and PFAS concentration.

Results

Phase 1 Results

PFAS were detected in 224 of 226 total samples collected during Phase 1. The two non-detect samples included one bulk atmospheric deposition sample and one surface water sample collected from our most upstream site on the Cedar River (CR-LANDSBURG).

Surface Water

Lake Washington

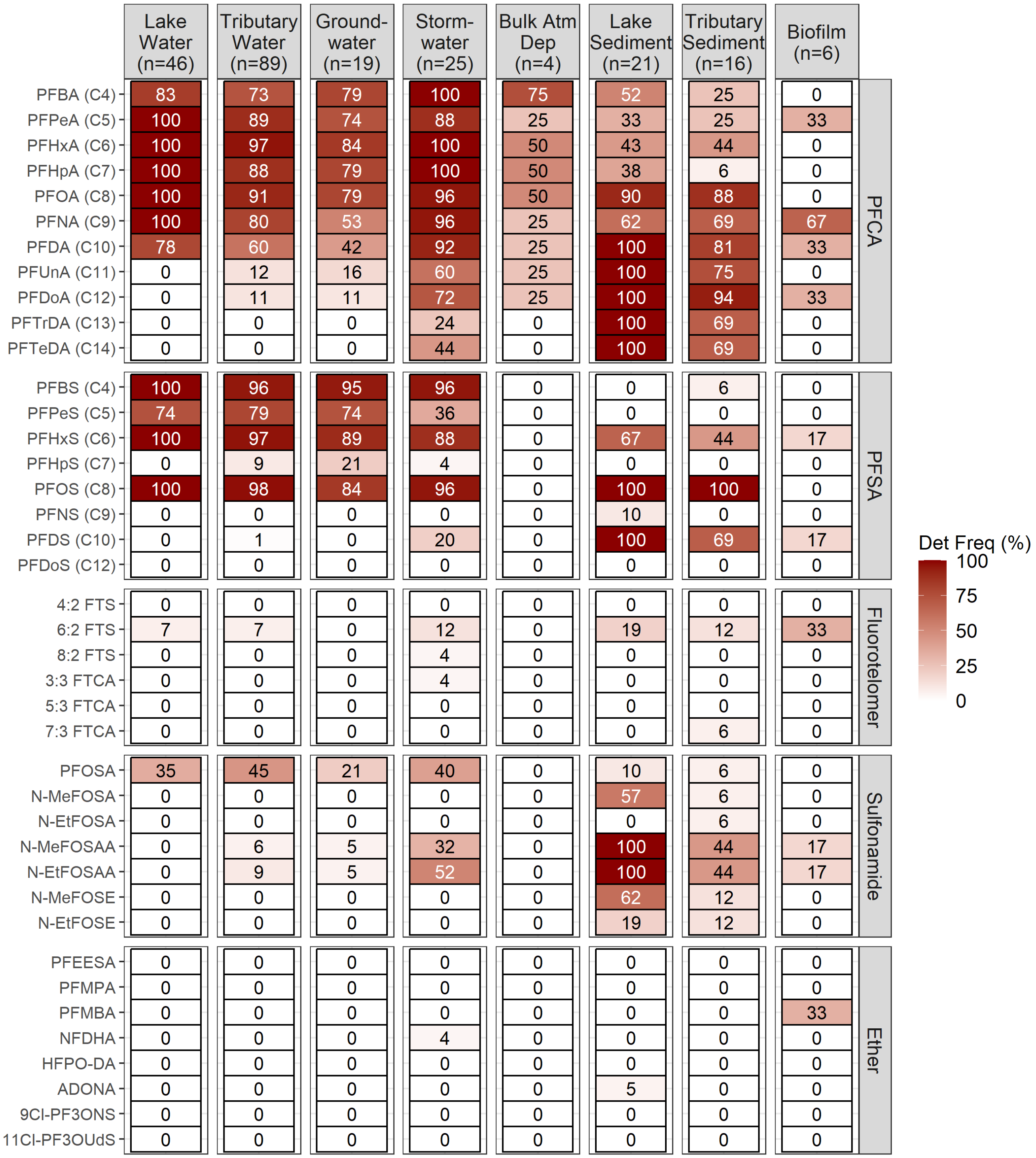
Total PFAS concentrations in lake surface water samples were 12.3 – 20.8 ng/L (Table 4). The most frequently detected analytes were PFAAs. Of these, eight analytes (C5 – C9 PFCAs and C4, C6, and C8 PFSAs) were detected in 100% of the samples collected (Figure 5). Several long-chain PFAS (C11 – C14 PFCAs and C9, C10 and C12 PFSAs) were not detected in lake surface water samples. Apart from detection of 6:2 FTS and PFOSA in some samples, fluorotelomers, sulfonamides, and ethers were also not detected.

PFOS (C8 PFSA) had the highest mean concentration in the lake surface water samples, representing 12% – 25% of the total PFAS. Concentrations of C4 – C6 and C8 PFCAs, and C4 and C6 PFSAs were also dominant (Figure 6).

Table 4. Minimum – Maximum (Median) total PFAS and PFOS concentrations by sample matrix and location type for all Phase 1 samples collected in the Greater Lake Washington watershed.

Sample Matrix	Location Type (Number of Samples)	Total PFAS	PFOS
Water (ng/L)	Lake (n=46)	12.3 – 20.8 (15.1)	2.29 – 3.58 (2.90)
Water (ng/L)	Tributary (n=89)	ND – 134 (23.2)	ND – 15.5 (3.60)
Water (ng/L)	Groundwater (n=19)	0.227 – 105 (28.4)	ND – 14.1 (3.16)
Water (ng/L)	Stormwater (n=25)	13.3 – 115 (30.8)	ND – 22.2 (3.62)
Water (ng/L)	Bulk Atmospheric Deposition (n=4)	ND – 10.8 (2.03)	ND
Sediment (ng/g)	Lake (n=21)	0.906 – 16.9 (4.71)	0.273 – 3.55 (0.694)
Sediment (ng/g)	Tributary (n=16)	0.0518 – 50.6 (0.910)	0.0307 – 0.997 (0.266)
Biofilm (ng/g)	Tributary (n=6)	0.0596 – 0.852 (0.318)	ND

n = number of samples; ND = Non-Detect;
PFAS = Per- and polyfluoroalkyl substances;
PFOS = Perfluorooctane sulfonic acid.



	Lake Water (n=46)	Tributary Water (n=89)	Ground-water (n=19)	Storm-water (n=25)	Bulk Atm Dep (n=4)	Lake Sediment (n=21)	Tributary Sediment (n=16)	Biofilm (n=6)	
PFBA (C4)	83	73	79	100	75	52	25	0	PFCA
PFPeA (C5)	100	89	74	88	25	33	25	33	
PFHxA (C6)	100	97	84	100	50	43	44	0	
PFHpA (C7)	100	88	79	100	50	38	6	0	
PFOA (C8)	100	91	79	96	50	90	88	0	
PFNA (C9)	100	80	53	96	25	62	69	67	
PFDA (C10)	78	60	42	92	25	100	81	33	
PFUnA (C11)	0	12	16	60	25	100	75	0	
PFDoA (C12)	0	11	11	72	25	100	94	33	
PFTrDA (C13)	0	0	0	24	0	100	69	0	
PFTeDA (C14)	0	0	0	44	0	100	69	0	
PFBS (C4)	100	96	95	96	0	0	6	0	PFSA
PFPeS (C5)	74	79	74	36	0	0	0	0	
PFHxS (C6)	100	97	89	88	0	67	44	17	
PFHpS (C7)	0	9	21	4	0	0	0	0	
PFOS (C8)	100	98	84	96	0	100	100	0	
PFNS (C9)	0	0	0	0	0	10	0	0	
PFDS (C10)	0	1	0	20	0	100	69	17	
PFDoS (C12)	0	0	0	0	0	0	0	0	
4:2 FTS	0	0	0	0	0	0	0	0	Fluorotelomer
6:2 FTS	7	7	0	12	0	19	12	33	
8:2 FTS	0	0	0	4	0	0	0	0	
3:3 FTCA	0	0	0	4	0	0	0	0	
5:3 FTCA	0	0	0	0	0	0	0	0	
7:3 FTCA	0	0	0	0	0	0	6	0	
PFOSA	35	45	21	40	0	10	6	0	Sulfonamide
N-MeFOSA	0	0	0	0	0	57	6	0	
N-EtFOSA	0	0	0	0	0	0	6	0	
N-MeFOSAA	0	6	5	32	0	100	44	17	
N-EtFOSAA	0	9	5	52	0	100	44	17	
N-MeFOSE	0	0	0	0	0	62	12	0	
N-EtFOSE	0	0	0	0	0	19	12	0	
PFEESA	0	0	0	0	0	0	0	0	Ether
PFMPA	0	0	0	0	0	0	0	0	
PFMBA	0	0	0	0	0	0	0	33	
NFDHA	0	0	0	4	0	0	0	0	
HFPO-DA	0	0	0	0	0	0	0	0	
ADONA	0	0	0	0	0	5	0	0	
9Cl-PF3ONS	0	0	0	0	0	0	0	0	
11Cl-PF3OUdS	0	0	0	0	0	0	0	0	

Figure 5. Detection frequencies (%) of 40 target PFAS analytes by location type and sample matrix for Phase 1 samples collected in the Greater Lake Washington watershed.

Higher detection frequencies are symbolized by higher numbers and darker shade.

Analytes are grouped by PFAS category.

PFAS = per- and polyfluoroalkyl substances.

PFCA = perfluoroalkyl carboxylic acid.

PFSA = perfluoroalkane sulfonic acid.

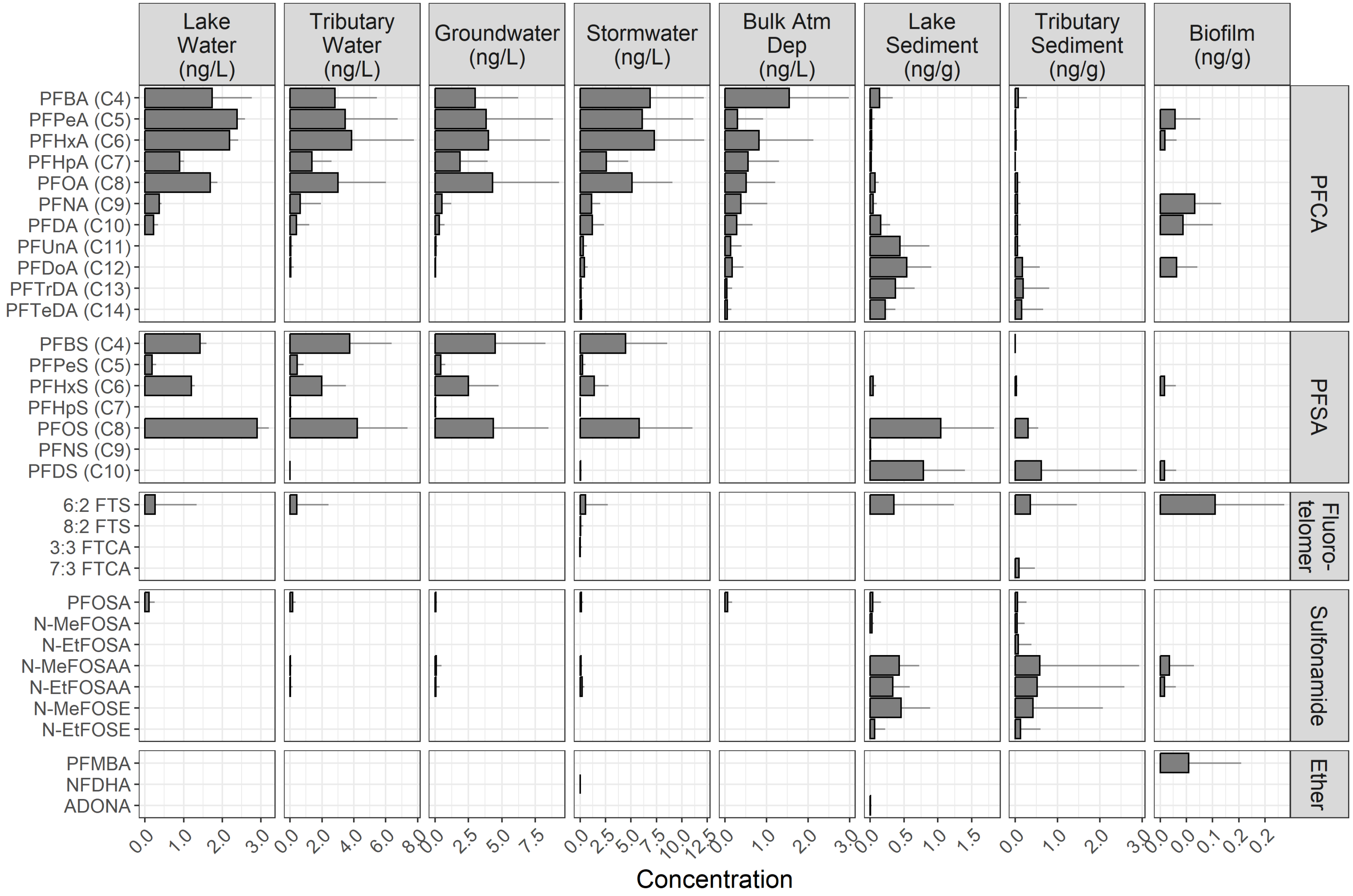


Figure 6. PFAS analyte concentrations (mean + standard error) by location type and sample matrix for Phase 1 samples collected in the Greater Lake Washington watershed.

Note different concentration scales. Non-detected analytes are not shown.

PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Tributaries

Median total PFAS concentrations were approximately 1.5 times higher in tributary surface water samples (23.2 ng/L) compared to those collected from the lake (15.1 ng/L; Table 4). Total PFAS concentrations in tributary samples ranged from non-detect to 134 ng/L, more variable than in the lake (Table 4). The highest total PFAS concentration was observed in Fairweather Creek (FWC-MOUTH), with on average four times higher concentration than all other sites (Figure 7).

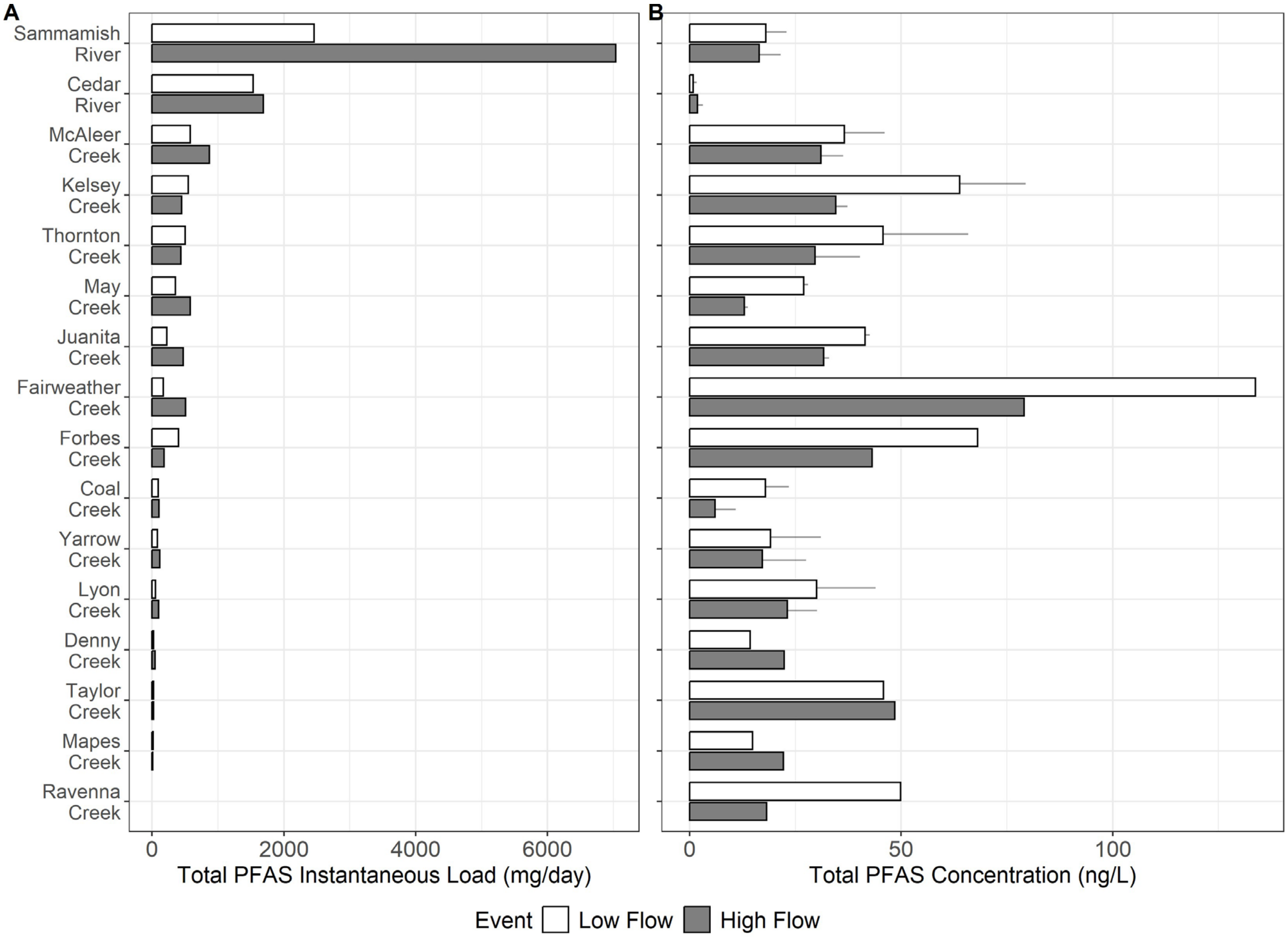


Figure 7. (A) Instantaneous loads (mg/day) and (B) surface water concentrations (ng/L; mean ± standard error) of total per- and polyfluoroalkyl substance (PFAS) in the sampled tributaries of the Greater Lake Washington watershed during Phase 1.

Loads were not calculated for Ravenna Creek because flow was not measured.

Although the Sammamish and Cedar Rivers had relatively low total PFAS concentrations, they represented the largest instantaneous loads to the lake (Figure 7). On average, the Sammamish and Cedar Rivers represented about 45% and 18% of the total PFAS instantaneous load from the sampled tributaries, respectively, while the small tributaries together represented approximately 37% of the instantaneous load. Instantaneous load was positively correlated with stream flow (R>0.8, p<0.05; Figure 8).

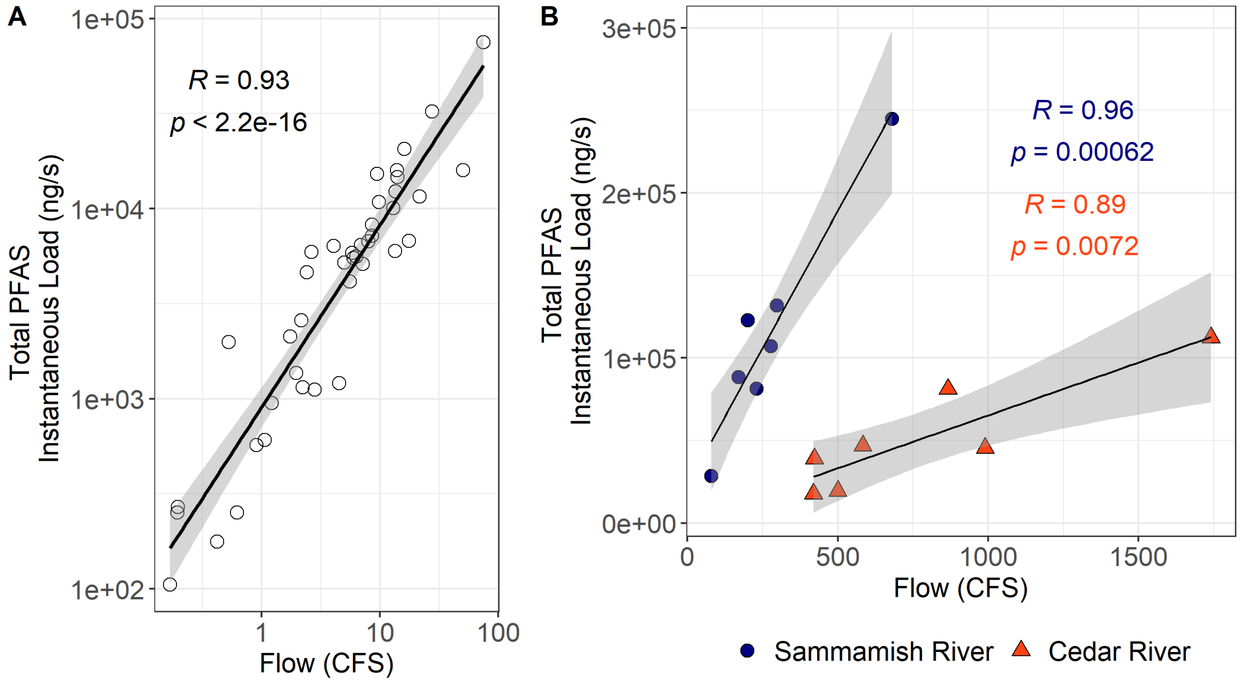


Figure 8. Correlation between PFAS instantaneous load (ng/s) and flow (CFS) for (A) the small creeks, and (B) two large rivers of the Greater Lake Washington watershed sampled during Phase 1.

The Pearson correlation coefficient (R), significance value (p), and 95% confidence interval (gray shade) are shown for the small creeks, and for the Cedar and Sammamish Rivers. Log10 transformation was applied to the x and y axis scales for Figure 8A.

PFAS = per- and polyfluoroalkyl substances
CFS = Cubic feet per second

Among tributary locations, total PFAS concentration was on average about 53.7% higher during summer low flow than spring high flow. This was supported by a negative correlation between total PFAS concentration and flow (R=-0.64, $p=1.9\times10^{-7}$; Figure 9).

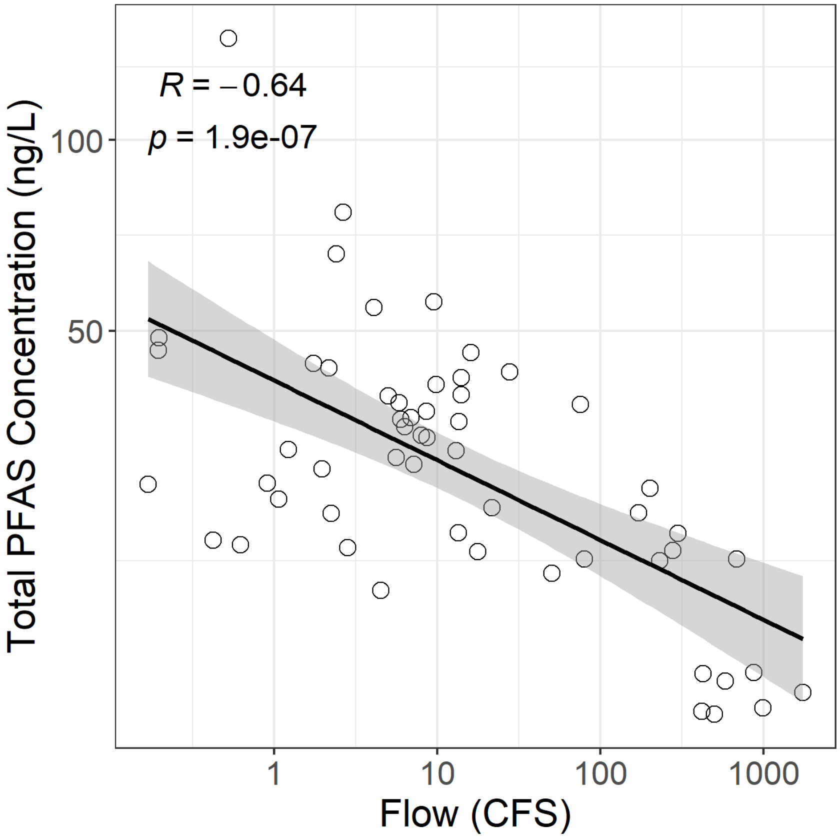


Figure 9. Correlation between PFAS concentration (ng/L) and flow (CFS) for Phase 1 tributary samples collected in the Greater Lake Washington watershed.

The Pearson correlation coefficient (R), significance value (p), and 95% confidence interval (gray shade) are shown. Log10 transformation was applied to the x axis, and square root transformation was applied to the y axis scales.

PFAS = per- and polyfluoroalkyl substances
CFS = Cubic feet per second

Upstream and downstream total PFAS concentrations were similar in most tributaries where upstream samples were collected, and sometimes higher at the upstream sites (Figure 10).

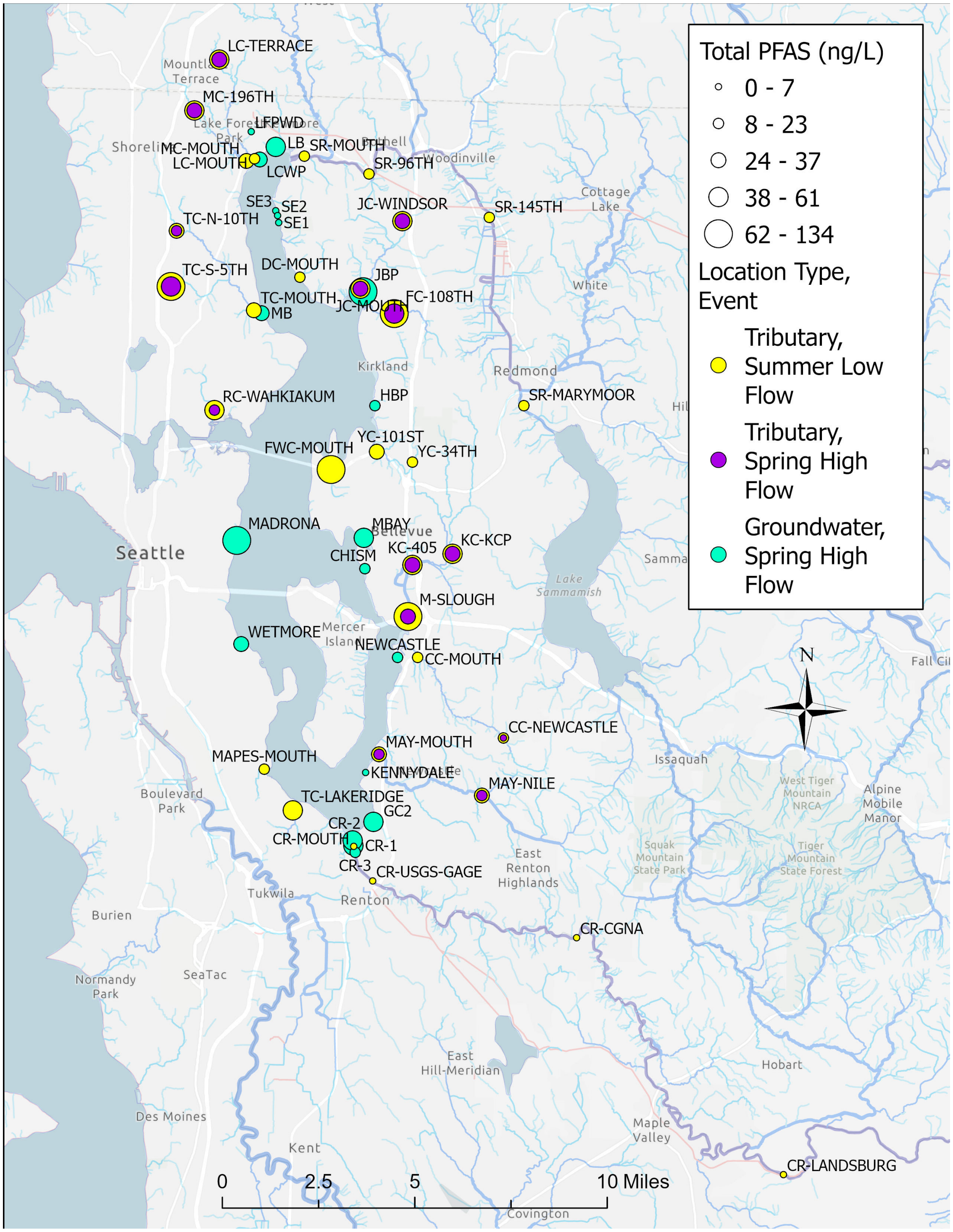


Figure 10. Concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) at Phase 1 tributary surface water and lake shoreline groundwater sampling sites in the Greater Lake Washington watershed.

Tributary sites further from Lake Washington represent upstream sites. Tributary sites closer to the lake represent downstream sites.

Among tributary outlet sites, there was a positive correlation between total PFAS concentration and imperviousness of the subbasin (R = 0.66; p = 0.0057) (Figure 11).

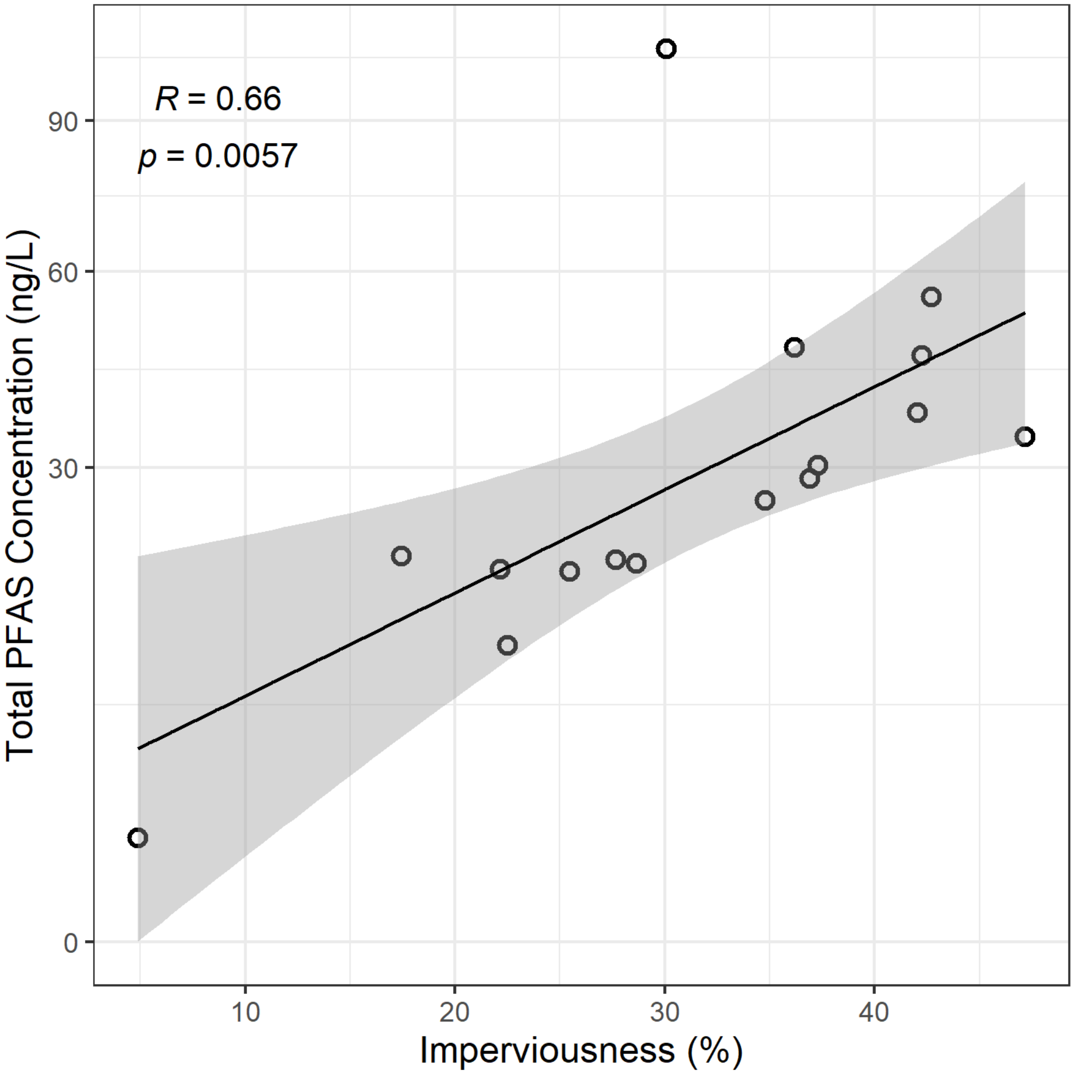


Figure 11. Correlation between mean concentrations of total per- and polyfluoroalkyl substances (PFAS; ng/L) at the most downstream site of each tributary subbasin and the percent (%) impervious land cover in the tributary subbasin of the Greater Lake Washington watershed.

The Pearson correlation coefficient (R), significance value (p), and 95% confidence interval (gray shade) are shown. Square-root transformation was applied to the y-axis scale.

Similar to Lake Washington, PFAAs were most frequently detected in tributary surface water samples, particularly C4 – C10 PFCAs and C4 – C6 and C8 PFSAs (Figures 5 & 6). Few detections of fluorotelomers, sulfonamides, and ethers were found in tributary samples, including PFOSA, which was detected in approximately 50% of samples, and several detections of 6:2 FTS, N-MeFOSAA, and N-EtFOSAA.

In lake and tributary surface water samples, PFAAs were positively correlated with one another, particularly C4 – C10 PFCAs and C4 and C8 PFSAs ($\rho \geq 0.6$, $p < 0.05$; Figure 12). TOC and DOC were correlated with PFAAs, most strongly with C5 – C9 PFCAs and C8 PFSA (PFOS) ($\rho = 0.7$, $p < 0.05$). Specific conductance was positively correlated with most PFAAs ($\rho \geq 0.2$, $p < 0.05$), except for C9, C11, and C12 PFCAs and C7 and C10 PFSA. Water temperature was weakly positively correlated with C4 – C8 and C10 PFCAs, PFBS, PFOS, PFOSA, an N-EtFOSAA ($\rho \leq 0.5$, $p < 0.05$).

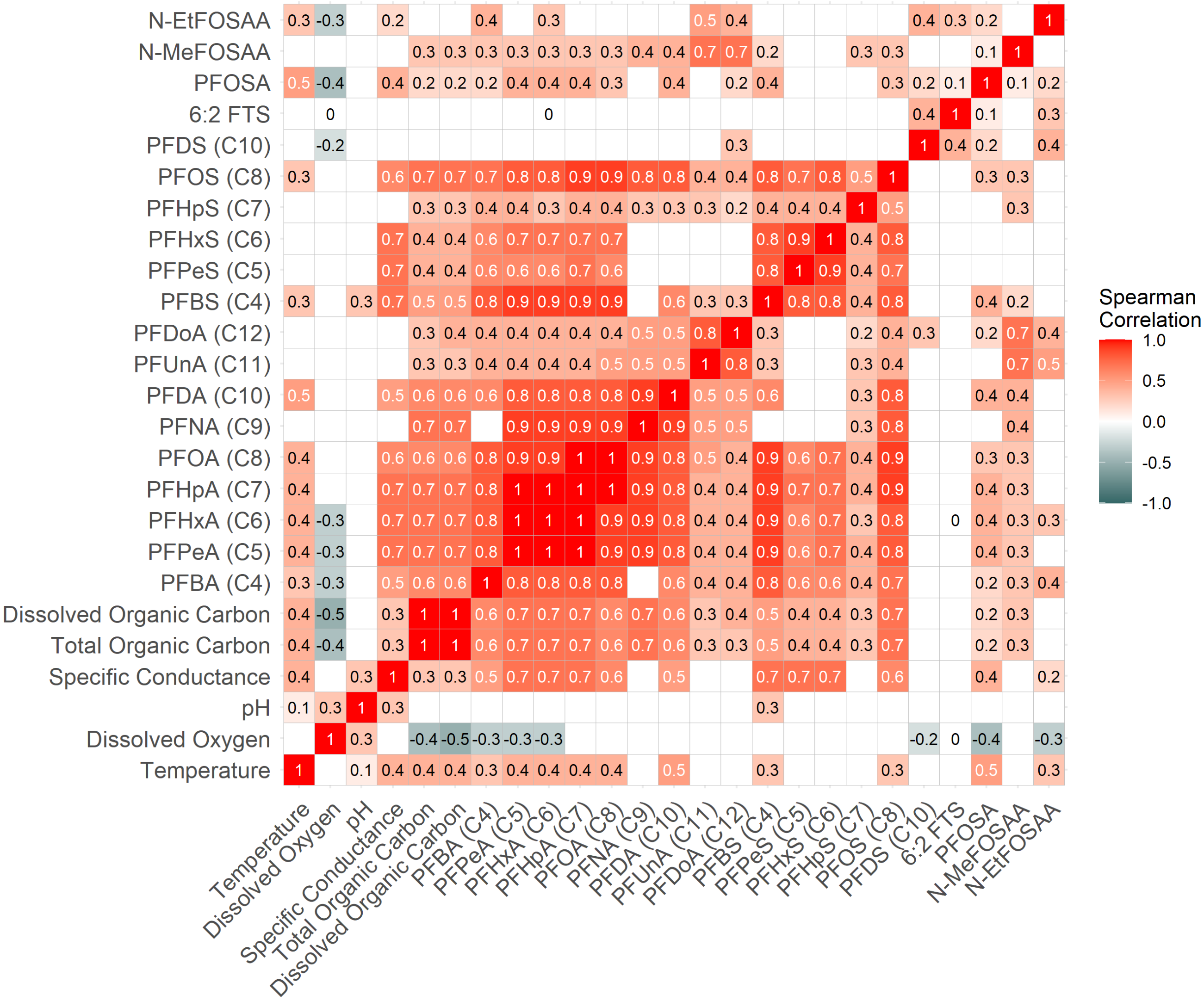


Figure 12. Matrix showing significant ($p<0.05$) Spearman rank correlation values between detected per- and polyfluoroalkyl substances (PFAS) and water chemistry variables for Phase 1 surface water samples collected in the Greater Lake Washington watershed.

Sediment

Lake Washington

Total PFAS concentrations in lake sediment samples were 0.906 – 16.9 ng/g, with the highest concentrations observed in the deep water sites of the south and central basins (Table 4). Total PFAS concentration in lake and tributary sediments was positively correlated with the percentage of fine particulates (clay and silt) in the sample (ρ>0.7, p<0.05; Figure 13). Total PFAS concentration was not significantly positively correlated with particulates larger than silt (Figure 13). Sediment TOC was positvely correlated (ρ≥0.2, p<0.05) with all PFAS analytes except for C5 and C7 PFCAs and 6:2 FTS. Sediment TOC was also correlated with the percentage of fine particulates in the sample (ρ>0.6, p<0.05; Appendix E)

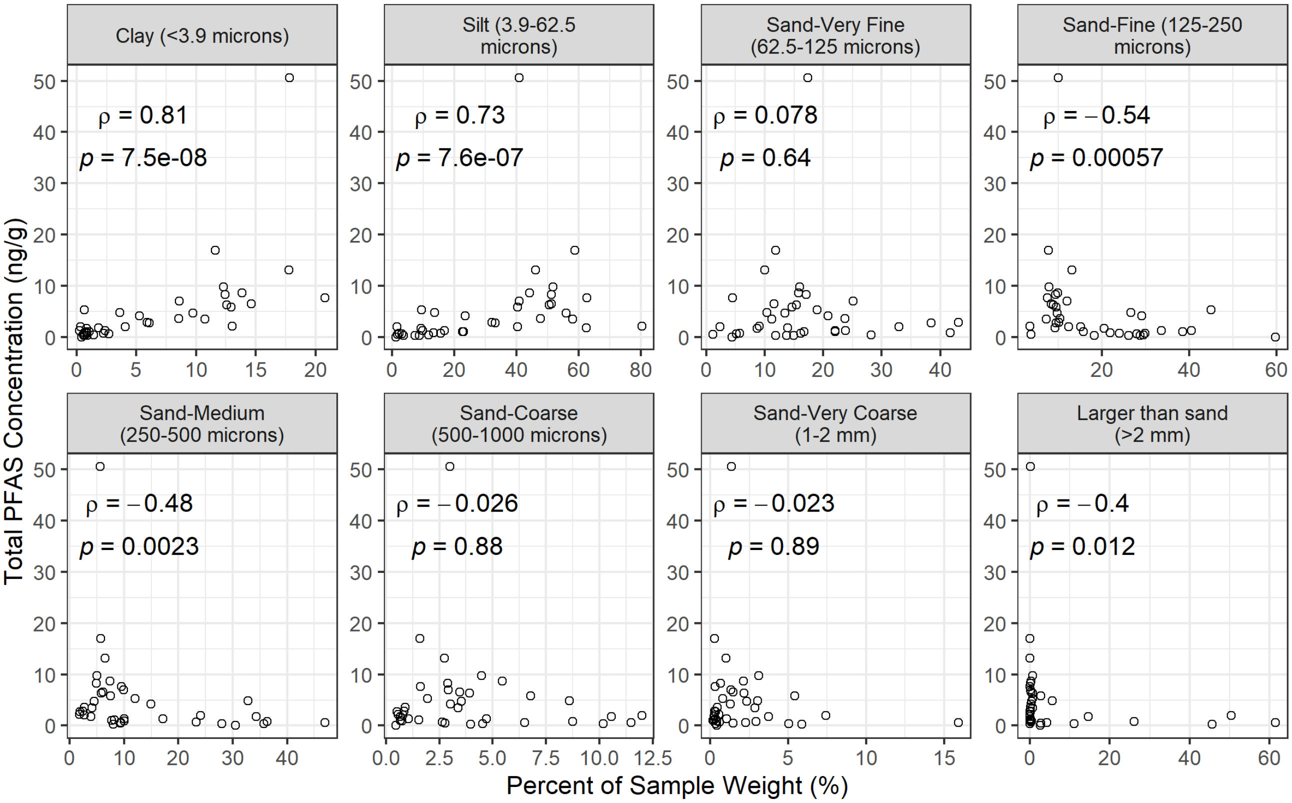


Figure 13. Correlations between total PFAS concentration and percentage of each grain size class in lake and tributary sediment samples collected in the Greater Lake Washington watershed during Phase 1.

Spearman correlation (ρ) and significance values (p) are shown. Grain size classes are arranged small to large from top left to bottom right. PFAS = per- and polyfluoroalkyl substances.

Unlike surface water samples, long-chain PFAS (particularly C10 – C14 PFCAs, and C8 and C10 PFSAs) and sulfonamides (N-MeFOSAA and N-EtFOSAA) were the most dominant analytes in lake sediment samples, collectively representing about 91% of the total PFAS on average among the samples (Figure 6). Nine analytes (C10 – C14 PFCAs, C8 and C10 PFSAs, N-MeFOSAA, and N-EtFOSAA) were detected in 100% of the samples (Figure 5). On average, the highest analyte concentration in the lake sediments was PFOS (1.050 ± 0.171 ng/g), followed by PFDS (0.801 ± 0.133 ng/g), long-chain PFCAs (0.268 ± 0.0250 ng/g), and several sulfonamides (0.198 ± 0.0243 ng/g) (Figure 6).

Tributaries

The median total PFAS concentration in tributary sediment samples (0.910 ng/g) was lower than in lake sediment samples (4.71 ng/g) (Table 4). Tributary sediment samples also tended to be composed of coarser grain sizes than lake sediments (Appendix F). Similar to lake sediments, tributary sediments were dominated by long-chain PFAS and sulfonamides, on average comprising 84% of the total PFAS (Figure 6).

Total PFAS concentrations in the tributary sediments were mostly less than 2 ng/g, apart from one sample. Sediment from Ravenna Creek (RC-WAHKIAKUM) had a total PFAS concentration (50.6 ng/g) that was an order of magnitude greater than all other tributary sediment samples and represented the highest sediment concentration among all lake and tributary sediment samples collected. Approximately 70% of the total PFAS in the Ravenna Creek sample was composed of four analytes: N-MeFOSAA, PFDS, N-EtFOSAA, and N-MeFOSE. Almost 60% of this sample was composed of clay and silt, different than the other tributary sediment samples, which were mostly composed of particulates larger than silt.

Groundwater

Total PFAS concentrations in groundwater were 0.227 – 105 ng/L, with a median concentration of 28.4 ng/L (Table 4). The highest total PFAS concentration (105 ng/L) was observed at a site located along the western shore of Lake Washington (MADRONA). The highest PFOA concentration was observed at Juanita Beach Park, located on the east side of Lake Washington. The highest PFOS concentration was observed at Wetmore Slough (WETMORE), located on the west side of Lake Washington, at the lake shore edge of the former Wetmore Slough, which was mostly drained following the 1917 opening of the Ship canal. In total, 17 of 40 target PFAS were detected in groundwater.

PFAAs with a carbon chain length of C8 or less were the most frequently detected compounds in groundwater samples (Figure 5). PFBS (C4), detected at 95% of the sample locations, was the most frequently detected PFSA, followed by PFHxS (C6; 89%), PFOS (C8; 84%), and PFPeS (C5; 74%). PFHxA (C6), detected at 84% of the sample locations, was the most frequently detected PFCA, followed by PFBA (C4; 79%), PFHpA (C7; 79%), PFOA (C8; 79%), and PFPeA (C5; 74%). Although PFPeS and PFHpA were among the most frequently detected, mean concentrations were much lower than other frequently detected PFAAs (Figure 6).

Long-chain PFSAs C9, C10, and C12 were not detected at any of the Phase 1 groundwater locations. Long-chain PFCAs C9 and C10 were detected in approximately half the samples. Three locations (WETMORE, MB, and MADRONA) had detections of the long-chain PFCA, PFUnA (C11), and two of these sites (WETMORE and MB) also had a detection of PFDoA (C12). PFCAs C13 and C14 were not detected.

Sulfonamide compounds were detected infrequently at low concentrations. PFOSA was detected at 4 of 19 sites, at concentrations less than 0.30 ng/L. N-MeFOSAA and N-EtFOSAA were detected at one site on the west side of the lake (MB) at a concentration of less than 2 ng/L. Fluorotelomer and ether compounds were not detected in any of the Phase 1 groundwater samples.

Stormwater

Total PFAS concentrations in stormwater samples were 13.3 – 115 ng/L, with median concentration approximately two times higher than in Lake Washington (Table 4). All but two PFAAs (C9 and C12 PFSA) were detected in the stormwater samples (Figure 5). Stormwater samples had more detections of long-chain PFAAs and sulfonamides (N-MeFOSAA and N-EtFOSAA) compared to surface water samples.

TOC was positively correlated with several PFCAs (C4, C6, C9, and C14 PFCAs) and PFOSA ($\rho \leq 0.5$, $p<0.05$). DOC was positively correlated with C4 and C6 PFCAs ($\rho \leq 0.4$, $p<0.05$). TSS was not significantly correlated with any PFAS analytes ($-0.4 \leq \rho \leq 0.4$, $p>0.05$).

The highest total PFAS concentration among stormwater samples (115 ng/L) was observed at a site that drains runoff from a small residential area on the eastern shore directly into Lake Washington in Meydenbauer Bay (BEL-SHORE; Figure 14). The sample was composed mostly of PFOS and C5 – C8 PFCAs (Figure 15).

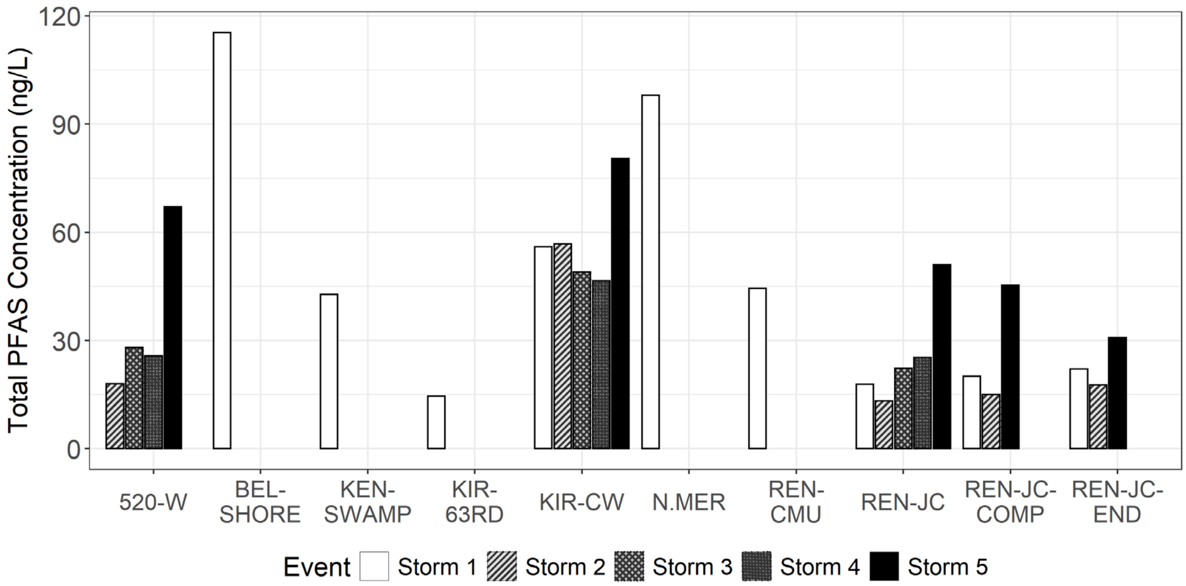


Figure 14. Concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in Phase 1 stormwater samples collected during five storm events from 8 sites that discharge to Lake Washington.

BEL-SHORE, KEN-SWAMP, KIR-63RD, N.MER, and REN-CMU were only sampled during storm 1. No samples were collected at 520-W during storm 1. REN-JC samples were time-composited: Samples from REN-JC were collected at the beginning of each storm event; REN-JC-COMP samples represent time composites of the beginning and end of each storm event; REN-JC-END samples represent the end of each storm event. REN-JC-COMP and REN-JC-END samples were not collected during storms 3 and 4.

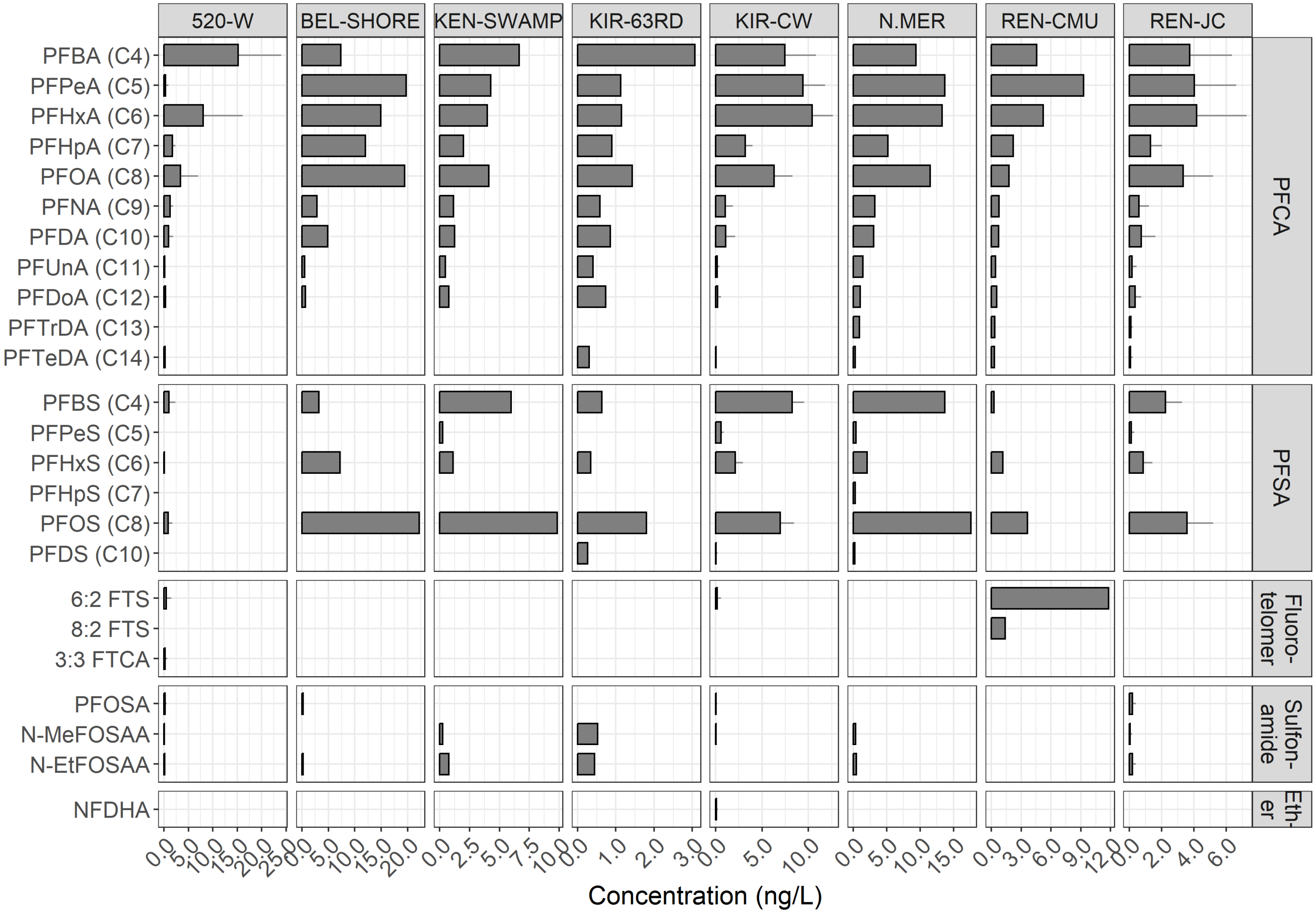


Figure 15. Mean (± standard error) PFAS analyte concentrations (ng/L) in Phase 1 stormwater samples collected from 8 sites that discharge to Lake Washington.

N = 1 for BEL-SHORE, KEN-SWAMP, KIR-63RD, N.MER, and REN-CMU. N = 4 for 520-W. N = 5 for KIR-CW and REN-JC. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Stormwater collected from the 520 bridge (520-W) was composed almost entirely of PFCAs, with PFBA being the dominant analyte (Figure 15).

At the site draining stormwater from the Renton Airport into the lower Cedar River (REN-CMU), total PFAS was composed mostly of 6:2 FTS and C4 – C6 PFCAs (Figure 15).

Samples collected from outfalls draining mixed land use types (KEN-SWAMP, KIR-63RD, KIR-CW, and REN-JC) were composed of a mix of PFAAs, predominantly C4 – C6 and C8 PFCAs and C4 and C8 PFSAs, with low concentrations of long-chain PFAS (Figure 15).

At sites where samples were collected during multiple storm events (520-W, KIR-CW, and RENJC), total PFAS concentrations during the fifth storm event were approximately 1.4 – 3.8 times higher than storms 1 – 4 (Figure 14).

At the REN-JC site, where samples were time-composited during each storm event, total PFAS concentrations in beginning (REN-JC), end (REN-JC-END), and time-composited (REN-JC-COMP) samples were comparable during storms 1 and 2 (Figure 14). During storm 5, total PFAS concentration in the end sample (REN-JC-END) was approximately 1.7 times higher than the beginning sample (REN-JC).

Juanita and May Creeks showed variability in total PFAS concentrations among storm events, while total PFAS concentrations in the Cedar River, Sammamish River, and Thornton Creeks were more comparable among storm events (Figure 16). Total PFAS instantaneous loads were 26% – 61% higher during storm events than during the summer low and spring high flow events in all five tributaries.

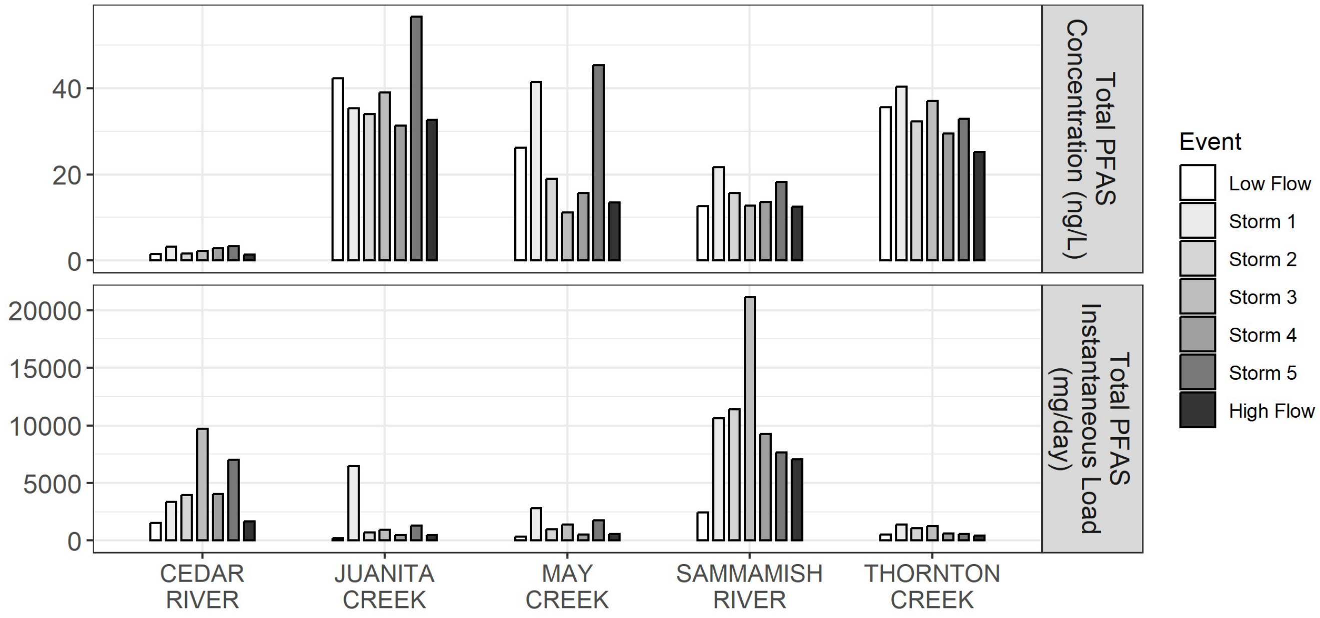


Figure 16. Surface water concentrations (ng/L) and instantaneous loads (mg/day) of total per- and polyfluoroalkyl substances (PFAS) in tributaries sampled within the Greater Lake Washington watershed during Phase 1 seasonal and storm events.

Bulk Atmospheric Deposition

Total PFAS concentrations in bulk atmospheric deposition samples were low relative to other water sample types, ranging from non-detect to 10.8 ng/L (Table 4). Samples were composed entirely of PFCAs (Figure 5). Total PFAS concentrations varied by deployment period, in which the highest total concentrations were detected during the spring deployment (April 2021) and lowest in winter (December 2020) (Figure 17).

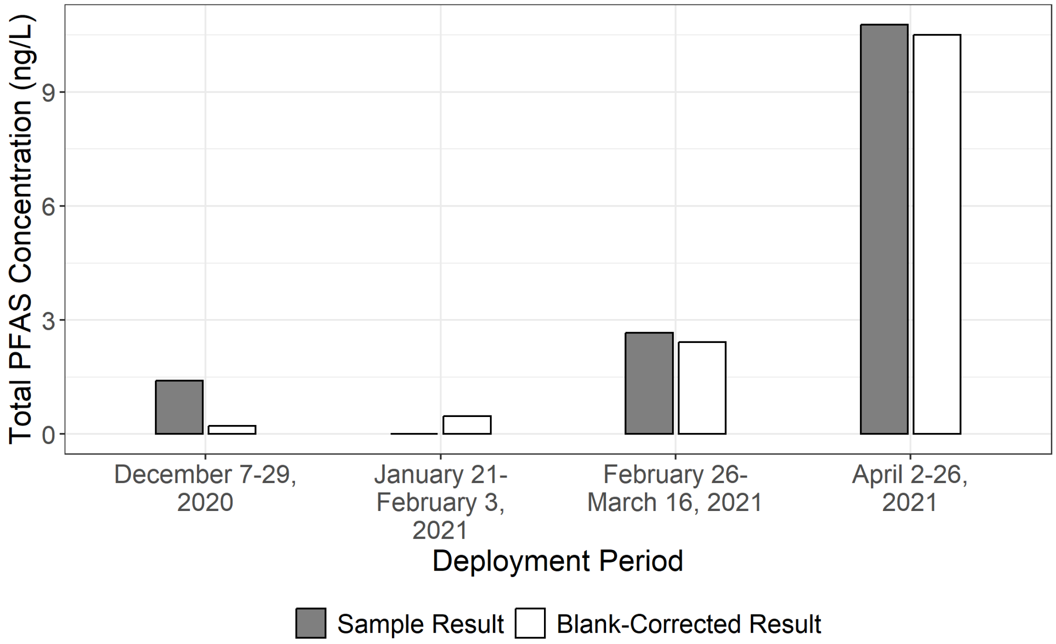


Figure 17. Actual and blank-corrected concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in four bulk atmospheric deposition samples collected in Seattle during four deployment periods of Phase 1.

When blank corrections were applied to sample results (subtraction of analyte mass in the equipment blank, and addition of analyte mass in the rinsate blank), blank-corrected and actual total PFAS concentrations were comparable (RPD<10%) when sample concentrations were higher during the February and April deployments (Figure 17).

Biofilm

Total PFAS concentrations in the six biofilm samples were less than 1 ng/g (Table 4). The lowest total PFAS concentration (0.0596 ng/g) was observed at the upmost site on the Cedar River (CR-LANDSBURG), and the highest (0.852 ng/g) near the outlet of Juanita Creek (JC-MOUTH).

Among the six samples, there were 10 analytes detected, with PFNA having the highest detection frequency (67%; Figure 5). Method blank contamination resulted in the qualifications of PFOS, PFOA, and PFBA in all the samples, with the exception of PFOS in the CR-LANDSBURG sample, in which PFOS was not detected.

TOP Assay

Among sediment samples, post-oxidation total PFAS concentrations were 4.8 times higher on average than target analysis concentrations (termed here as “pre-oxidation” results; Figure 18). This trend was more pronounced at higher concentrations. For most sediment samples, increases in C4 – C12 PFCAs were observed pre- to post-oxidation (Figure 19). PFSA concentrations remained mostly unchanged because precursors are typically only converted to carboxylic end products (PFCAs) through the TOP assay oxidation process.

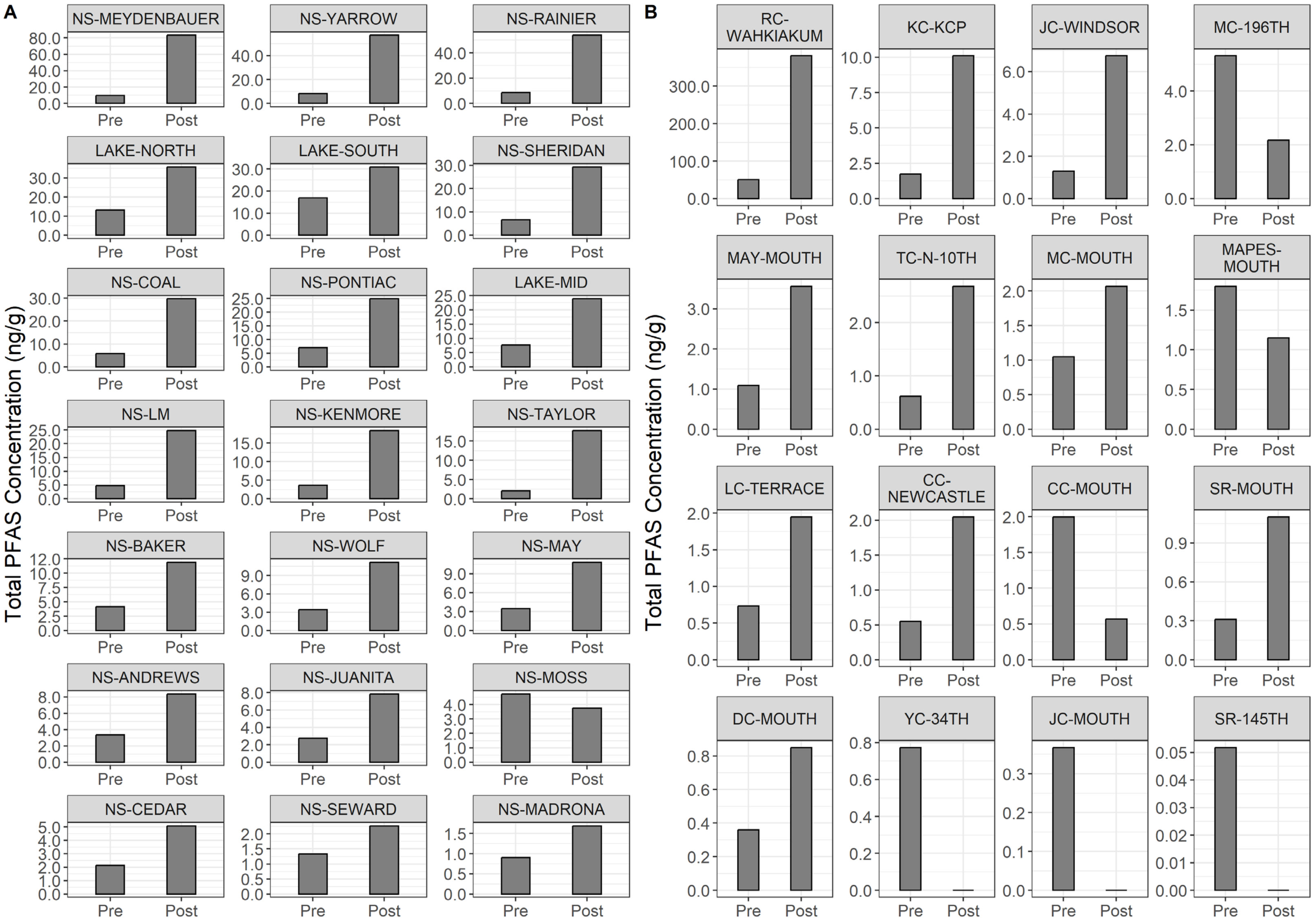


Figure 18. Pre- and post-oxidation concentrations (ng/g) of total per- and polyfluoroalkyl substances (PFAS) in Phase 1 (A) lake sediment and (B) tributary sediment samples collected in the Greater Lake Washington watershed. Note different concentration scales. Plots are arranged in order of decreasing concentration from top left to bottom right.

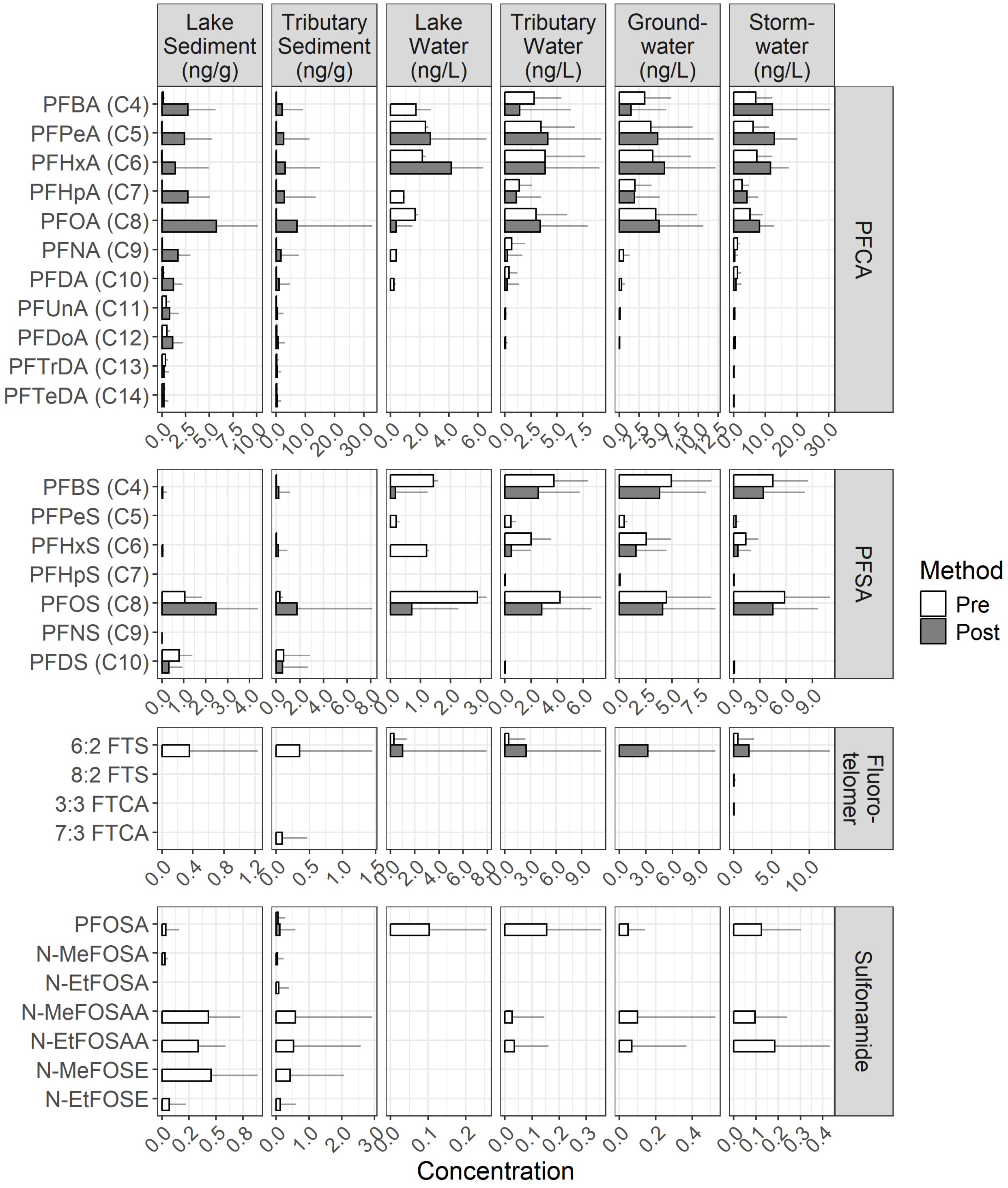


Figure 19. Mean (± standard error) pre- and post-oxidation PFAS analyte concentrations (ng/L) by location type and matrix in Phase 1 samples collected in the Greater Lake Washington watershed.

Note different concentration scales. Non-detected analytes are not shown. Ether compounds were not analyzed in the TOP assay, and are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Surface water samples did not yield post-oxidation total PFAS concentrations consistently higher than pre-oxidation results (Figure 20). However, increases in PFPeA, PFHxA, and 6:2 FTS concentrations were often observed pre- to post-oxidation in the surface water (Figure 19). In some samples, especially those with lower analyte concentrations, the "pre-oxidation" concentration was higher than the post-oxidation concentration (Figure 20). This could be explained by the difference in reporting limits between the target ("pre-oxidation") and TOP assay analyses: The reporting limit for target analysis was about 10 times lower than for the TOP assay, potentially resulting in more detections through the target analysis.

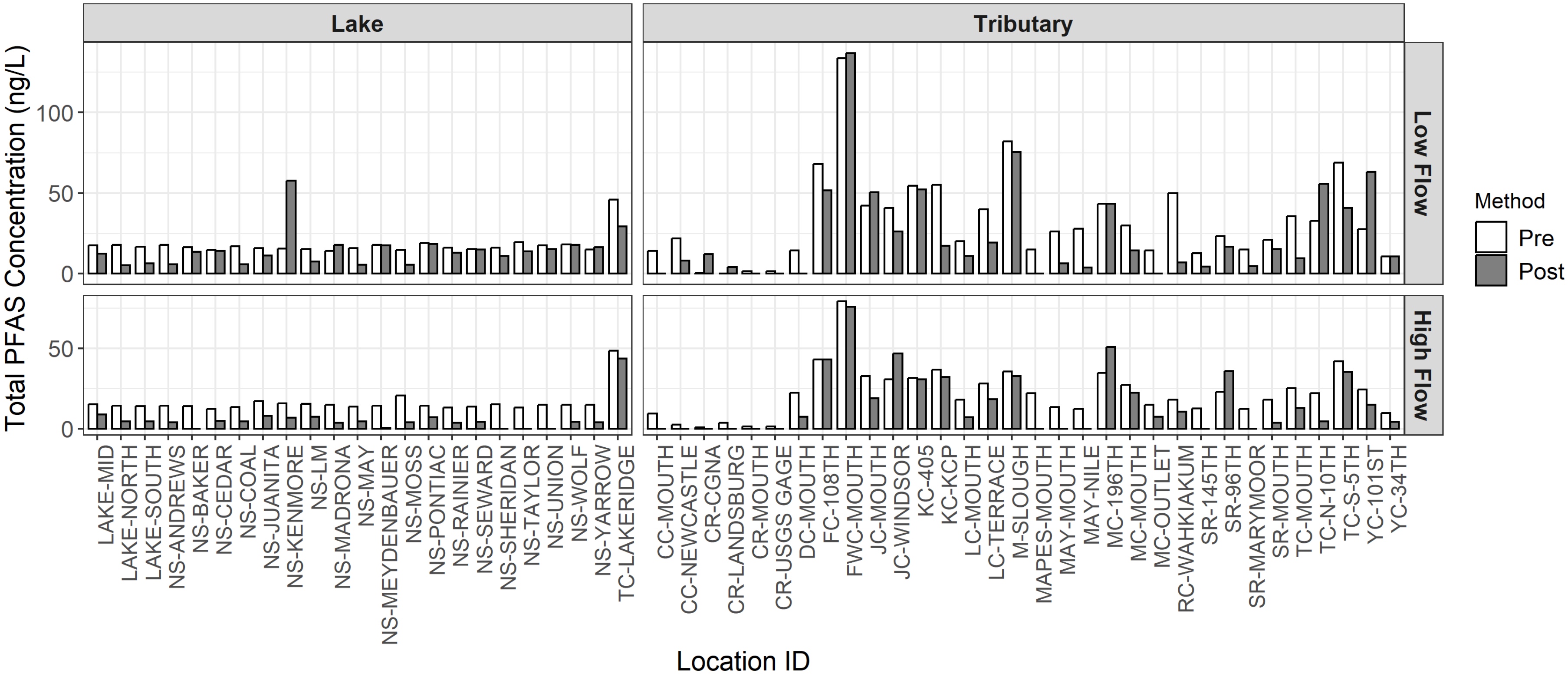


Figure 20. Pre- and post-oxidation concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in Phase 1 lake and tributary surface water samples collected in the Greater Lake Washington watershed.

Groundwater samples also did not consistently yield post-oxidation total PFAS concentrations greater than pre-oxidation results (Figure 21). At the MBAY site, increases in C4 – C6 PFCAs were observed. At MADRONA, WETMORE, and MB, the higher post-oxidation results were due to the presence of 6:2 FTS in the post-oxidation sample, and its absence in the pre-oxidation sample.

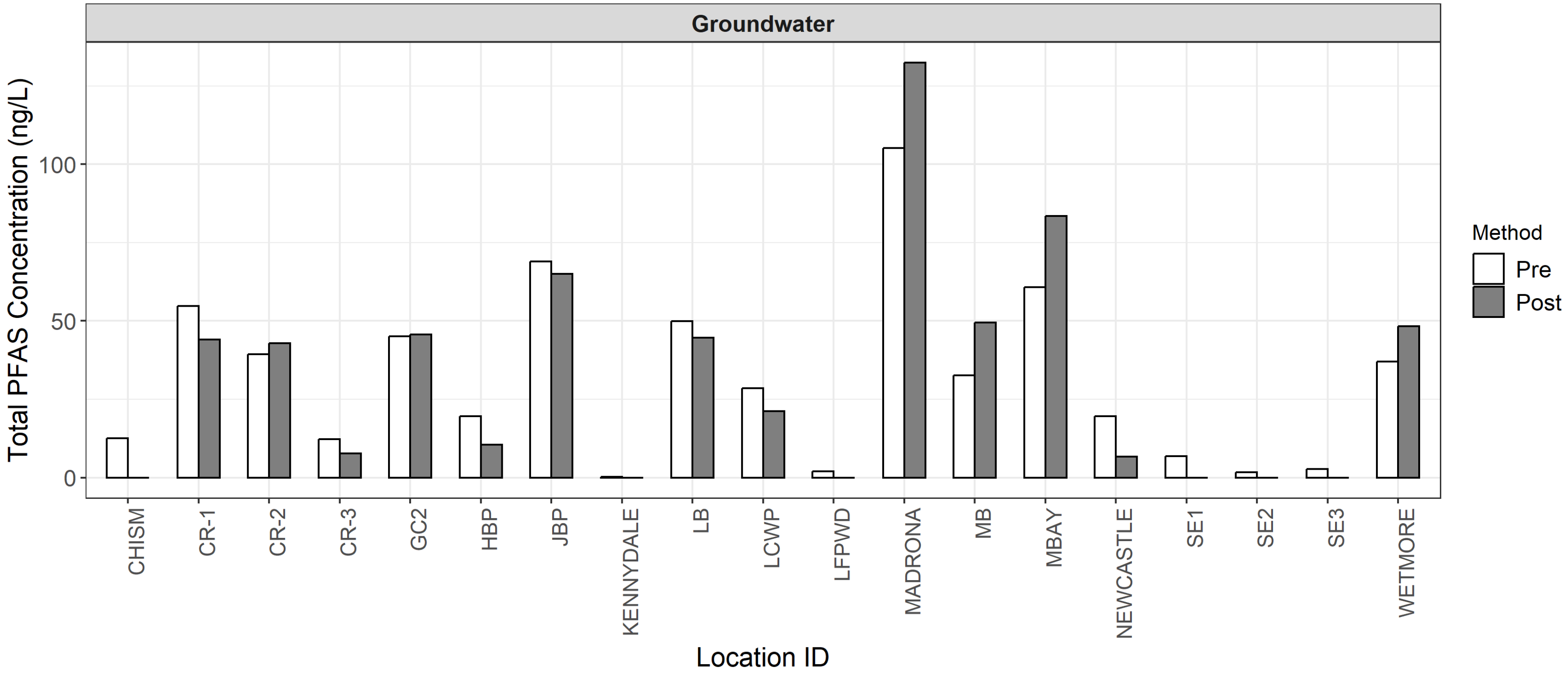


Figure 21. Pre- and post-oxidation concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in Phase 1 groundwater samples collected along the shoreline of Lake Washington.

Stormwater samples yielded post-oxidation total PFAS concentrations greater than pre-oxidation results at some sites and during some events (Figure 22). Specifically, post-oxidation total PFAS concentrations were 61% – 83% higher than pre-oxidation concentrations during the fifth storm event at all three repeated sampling sites (520-W, KIR-CW, and REN-JC). Post-oxidation total PFAS concentrations were also consistently higher by 52% – 170% at the 520-W site across storm events two through five. In most samples, increases in total PFAS concentration pre- to post-oxidation were due to increases in C4 – C8 PFCAs, particularly PFPeA and PFHxA.

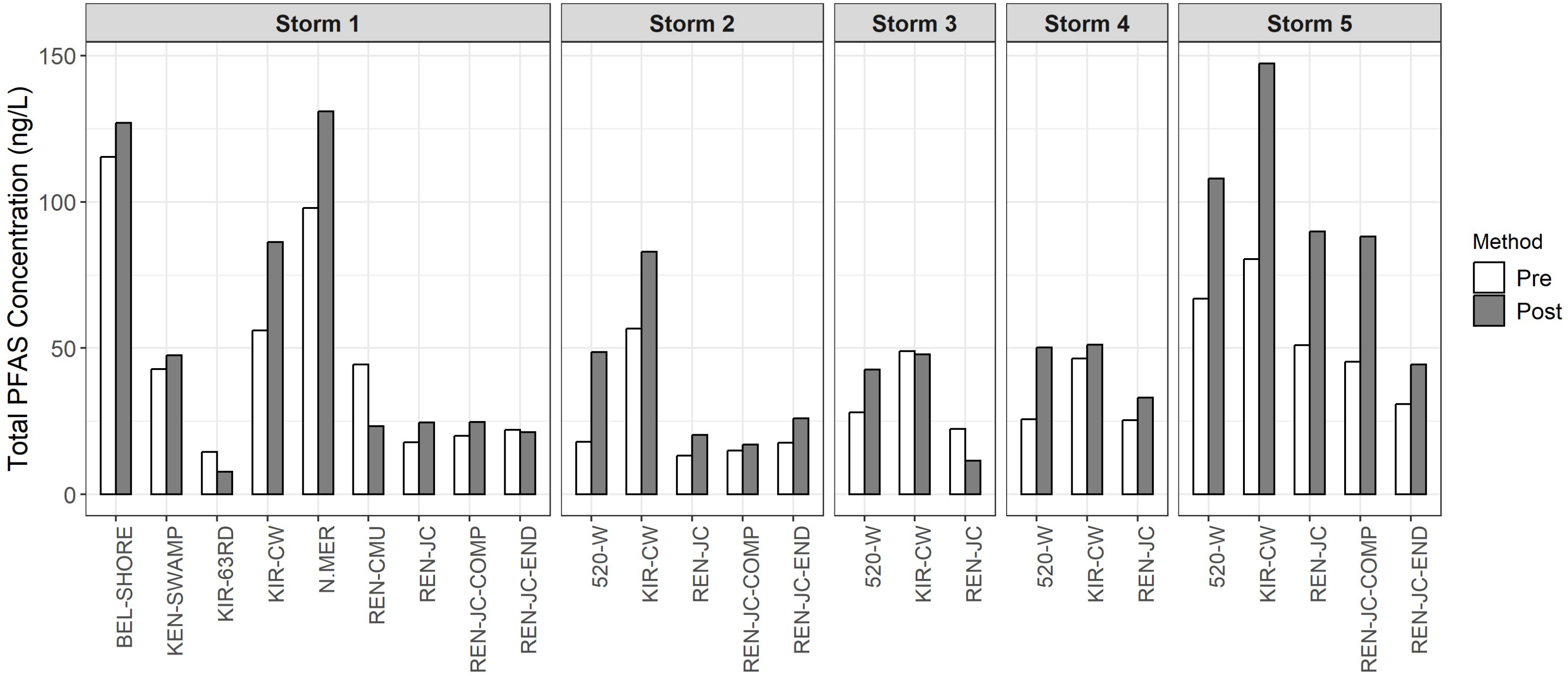


Figure 22. Pre- and post-oxidation concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in Phase 1 stormwater samples collected from 8 sites that discharge to Lake Washington

BEL-SHORE, KEN-SWAMP, KIR-63RD, N.MER, and REN-CMU were only sampled during storm 1. No samples were collected at 520-W during storm 1. REN-JC samples were time-composited: Samples from REN-JC were collected at the beginning of each storm event; REN-JC-COMP samples represent time composites of the beginning and end of each storm event; REN-JC-END samples represent the end of each storm event. REN-JC-COMP and REN-JC-END samples were not collected during storms 3 and 4.

Phase 2 Results

PFAS were detected in 243 of 253 samples collected during Phase 2. Total PFAS concentrations in Lake Washington surface water samples were similar to concentrations in Phase 1 (Figure 23). Total PFAS concentrations at tributary sites that were resampled in Phase 2 were on average 34% lower than Phase 1 during low flow, and 27% higher during high flow. Total PFAS concentrations at resampled groundwater sites were comparable, except for MADRONA, in which the Phase 2 concentration was about 98% lower than Phase 1.

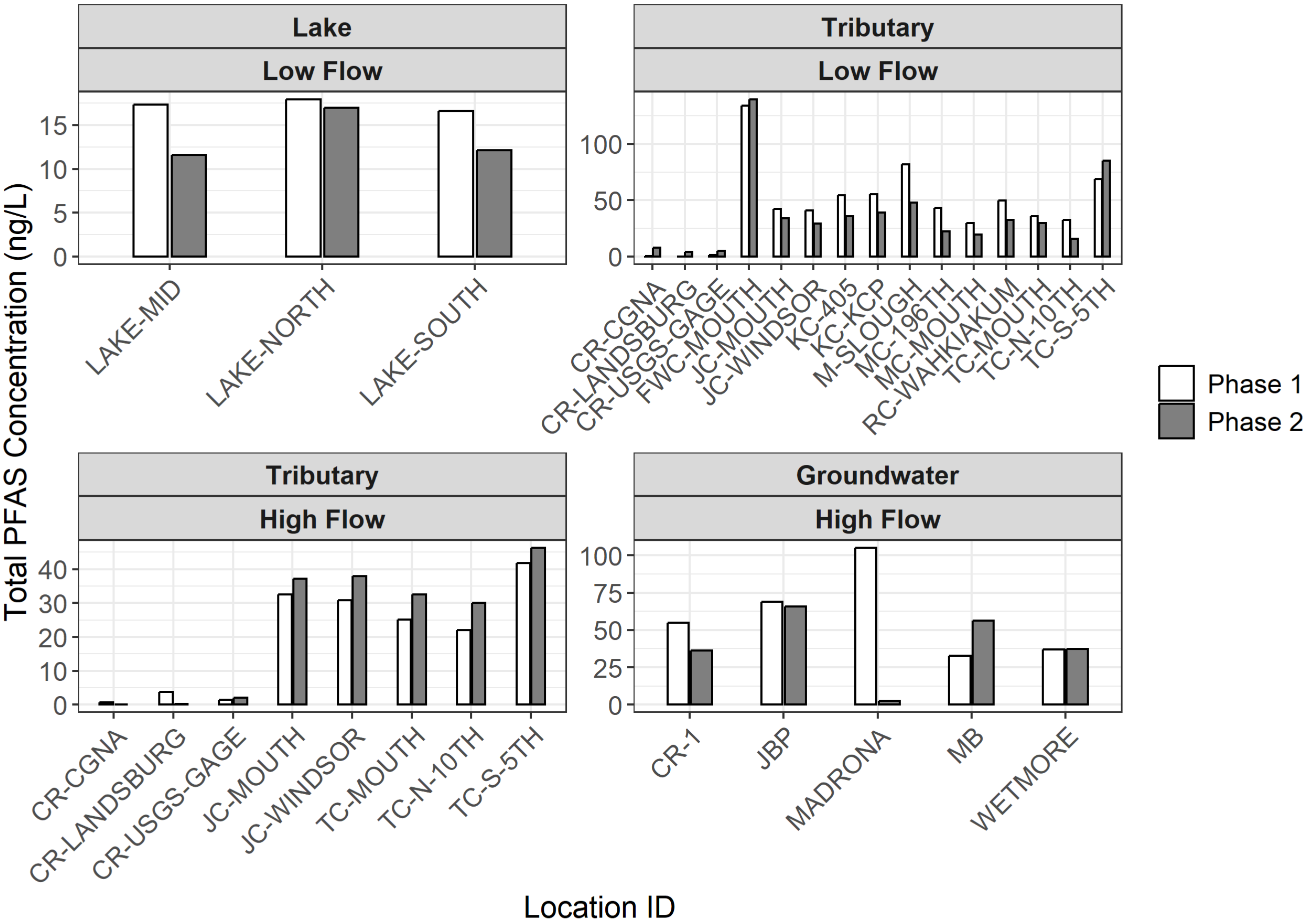


Figure 23. Comparison of total PFAS concentrations (ng/L) between Phase 1 and Phase 2 samples collected at revisited sites during summer low flow (top panels) and spring high flow (bottom panels) in the Greater Lake Washington watershed.
Note different concentration scales. PFAS = per- and polyfluoroalkyl substances.

Phase 1 Follow-Up Sampling

Focus Subbasins

Cedar River

Surface Water

Median total PFAS concentrations in surface water samples from the Cedar River were approximately 6 – 54 times lower than median concentrations in all other tributaries (Table 5).

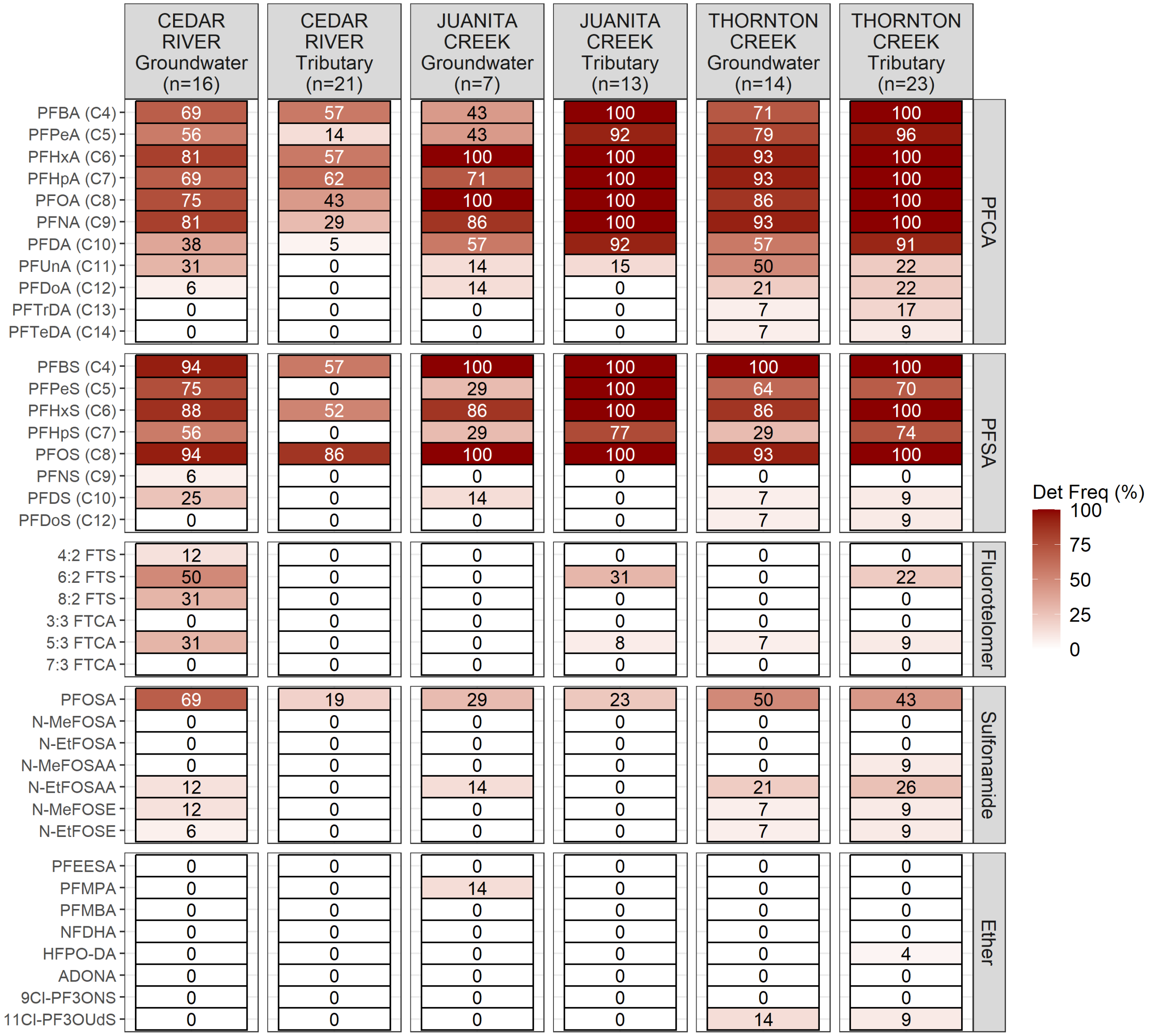
Table 5. Minimum – Maximum (Median) total PFAS and PFOS concentrations by sample matrix and location type for Phase 2 samples collected in the Greater Lake Washington watershed.

Sample Type	Location Name (Number of Samples)	Total PFAS	PFOS
Surface Water (ng/L)	Lake Washington (N=5)	11.6 - 23.8 (17.0)	2.44 - 6.19 (3.34)
Surface Water (ng/L)	Cedar River (N=21)	ND - 7.97 (2.50)	ND - 0.871 (0.356)
Surface Water (ng/L)	Juanita Creek (N=10)	29.1 - 46.3 (38.3)	3.26 - 12.3 (6.14)
Surface Water (ng/L)	Thornton Creek (N=20)	15.9 - 85.0 (38.2)	2.13 - 7.42 (4.65)
Surface Water (ng/L)	Fairweather Creek (N=2)	87.5 - 139 (113)	9.71 - 10.4 (10.1)
Surface Water (ng/L)	Kelsey Creek (N=7)	26.0 - 53.8 (38.9)	4.07 - 6.68 (5.18)
Surface Water (ng/L)	Valley Creek (N=3)	17.6 - 92.0 (18.8)	0.536 - 9.66 (2.58)
Surface Water (ng/L)	McAleer Creek (N=3)	19.5 - 22.8 (22.4)	2.58 - 3.40 (2.87)
Surface Water (ng/L)	Hall Creek (N=2)	42.3 - 53.1 (47.7)	6.30 - 10.2 (8.25)
Surface Water (ng/L)	Ravenna Creek (N=6)	7.64 - 54.0 (30.3)	0.402 - 9.67 (2.95)
Groundwater (ng/L)	Cedar River (N=20)	ND - 946 (31.9)	ND - 225 (4.80)
Groundwater (ng/L)	Juanita Creek (N=7)	4.53 - 53.7 (13.4)	1.16 - 10.7 (4.89)
Groundwater (ng/L)	Thornton Creek (N=16)	0.979 - 103 (47.1)	ND - 17.1 (4.83)
Groundwater (ng/L)	Ravenna Creek (N=8)	9.84 - 203 (54.8)	0.0203 - 91.4 (13.6)
Bulk Atmospheric Deposition (ng/L)	Seattle, North Bend (N=10)	ND - 7.04 (1.39)	ND
Suspended Sediment (ng/g)	Cedar River, Juanita Creek, Thornton Creek (N=6)	0.159 - 10.9 (1.9)	0.131 - 5.80 (0.919)
Surface Runoff (ng/L)	Cedar, Juanita, Thornton subbasins, 520-W bridge (N=15)	ND - 11.0 (2.36)	ND - 5.17 (0.188)
Road Dust (ng/g)	Cedar, Juanita, Thornton subbasins, 520-W bridge (N=14)	0.305 - 19.0 (3.05)	0.0406 - 1.68 (0.295)
Biofilm (ng/g)	Cedar River (N=6)	ND - 0.128 (ND)	ND
Caddisfly larvae (ng/g)	Cedar River (N=6)	0.790 - 0.884 (0.837)	0.790 - 0.884 (0.837)

ND = Non-Detect. PFAS = Per and polyfluoroalkyl substances. PFOS = Perfluorooctane sulfonic acid

Eleven of the forty target PFAS analytes were detected among Cedar River surface water samples, with PFOS being the most frequently detected (Figure 24). PFAS were non-detect in two surface water samples from the Cedar River: One sample collected from the Upper Cedar River watershed, upstream of Chester Morse Lake (CR-CML); and one sample collected from Cedar Grove Natural Area (CR-CGNA) in spring. The highest surface water total PFAS concentrations were observed in the lower Cedar River watershed (Figure 25; Appendix D-Figure D1).

At all sites except for CR-LPNA (located in the middle reach of the Cedar River), total PFAS concentrations during summer low flow were higher than during spring high flow. At the outlet of the Cedar River (CR-USGS-GAGE), average total PFAS concentration among the three storm events in October, November, and February was 4.57 ng/L, about 1.3 times higher than average summer low and spring high flow concentrations at that site.



	CEDAR RIVER Groundwater (n=16)	CEDAR RIVER Tributary (n=21)	JUANITA CREEK Groundwater (n=7)	JUANITA CREEK Tributary (n=13)	THORNTON CREEK Groundwater (n=14)	THORNTON CREEK Tributary (n=23)	
PFBA (C4)	69	57	43	100	71	100	PFCA
PFPeA (C5)	56	14	43	92	79	96	
PFHxA (C6)	81	57	100	100	93	100	
PFHpA (C7)	69	62	71	100	93	100	
PFOA (C8)	75	43	100	100	86	100	
PFNA (C9)	81	29	86	100	93	100	
PFDA (C10)	38	5	57	92	57	91	
PFUnA (C11)	31	0	14	15	50	22	
PFDoA (C12)	6	0	14	0	21	22	
PFTrDA (C13)	0	0	0	0	7	17	
PFTeDA (C14)	0	0	0	0	7	9	
PFBS (C4)	94	57	100	100	100	100	PFSA
PFPeS (C5)	75	0	29	100	64	70	
PFHxS (C6)	88	52	86	100	86	100	
PFHpS (C7)	56	0	29	77	29	74	
PFOS (C8)	94	86	100	100	93	100	
PFNS (C9)	6	0	0	0	0	0	
PFDS (C10)	25	0	14	0	7	9	
PFDoS (C12)	0	0	0	0	7	9	
4:2 FTS	12	0	0	0	0	0	Fluorotelomer
6:2 FTS	50	0	0	31	0	22	
8:2 FTS	31	0	0	0	0	0	
3:3 FTCA	0	0	0	0	0	0	
5:3 FTCA	31	0	0	8	7	9	
7:3 FTCA	0	0	0	0	0	0	
PFOSA	69	19	29	23	50	43	Sulfonamide
N-MeFOSA	0	0	0	0	0	0	
N-EtFOSA	0	0	0	0	0	0	
N-MeFOSAA	0	0	0	0	0	9	
N-EtFOSAA	12	0	14	0	21	26	
N-MeFOSE	12	0	0	0	7	9	
N-EtFOSE	6	0	0	0	7	9	
PFEESA	0	0	0	0	0	0	Ether
PFMPA	0	0	14	0	0	0	
PFMBA	0	0	0	0	0	0	
NFDHA	0	0	0	0	0	0	
HFPO-DA	0	0	0	0	0	4	
ADONA	0	0	0	0	0	0	
9Cl-PF3ONS	0	0	0	0	0	0	
11Cl-PF3OUdS	0	0	0	0	14	9	

Figure 24. Detection frequencies (%) of 40 target PFAS analytes in Phase 2 groundwater and tributary samples by focus subbasin in the Greater Lake Washington watershed.

Higher detection frequencies are symbolized by higher numbers and darker shade. Analytes are grouped by PFAS category PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid

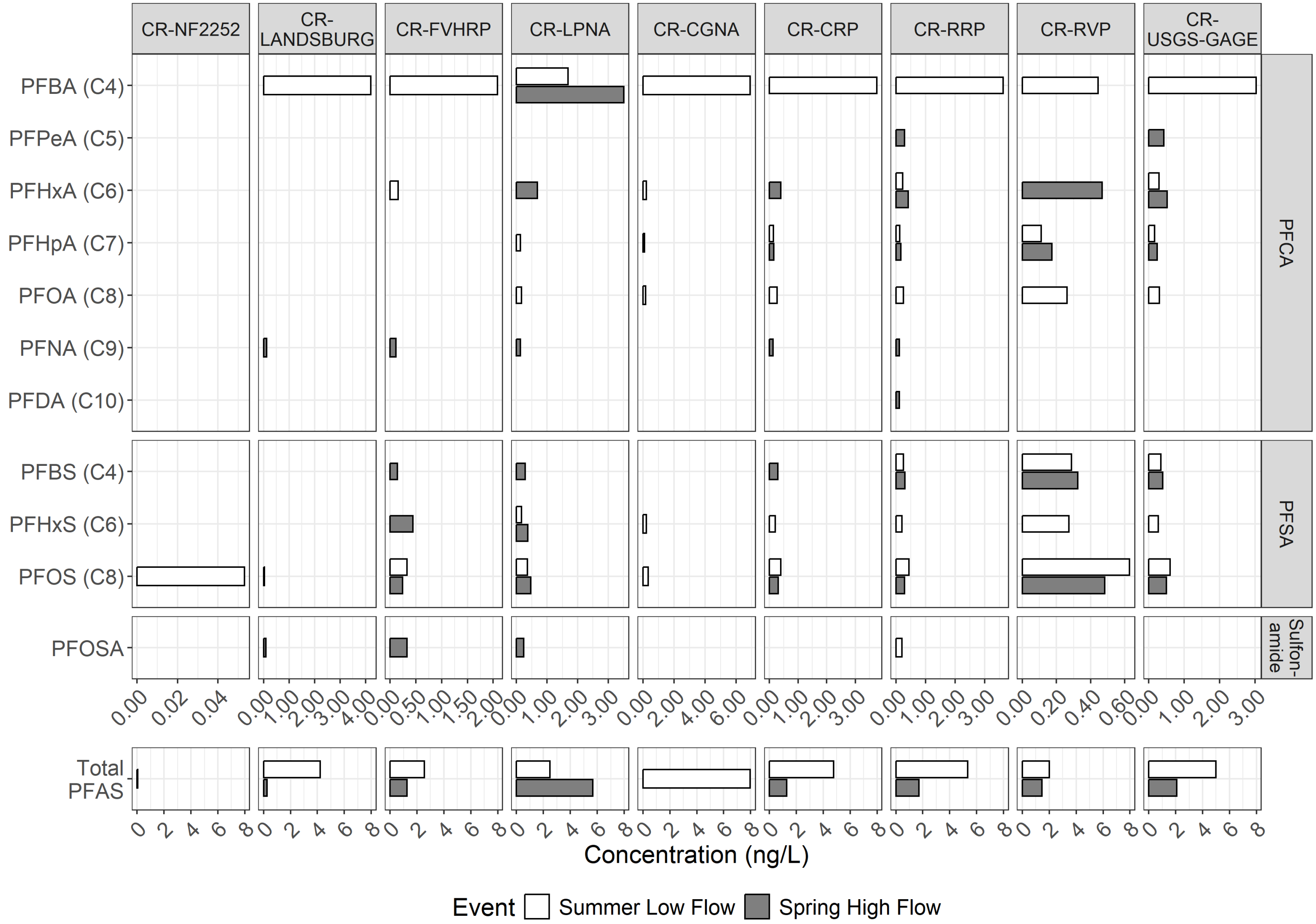


Figure 25. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 Cedar River sites, arranged upstream to downstream.

Note different concentration scales. Non-detected analytes are not shown. CR-CML is not shown because there were no detections. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

PCA results for Cedar River surface water samples showed PFAS composition was influenced by sampling event (Figure 26). This was primarily due to the influence of PFBA on summer low flow samples. PFBA was the dominant analyte in seven of the eight summer low flow samples, and was non-detect in seven out of eight spring high flow samples.

At site CR-LPNA, PFBA was the dominant analyte during both flow regimes, with 2.3 times higher total PFAS concentration during spring high flow than summer low flow—different than most of the other Cedar River sites (Figure 25). At site CR-RVP in the lower Cedar River, PFOS was the dominant analyte during both summer low and spring high flow, which differed from other Cedar River sites. During spring high flow, PFAS composition primarily consisted of PFOS and a mix of other short-chain PFAAs. Within the subbasin, sulfonamide and fluorotelomer compounds were not detected, with exception of PFOSA in several samples.

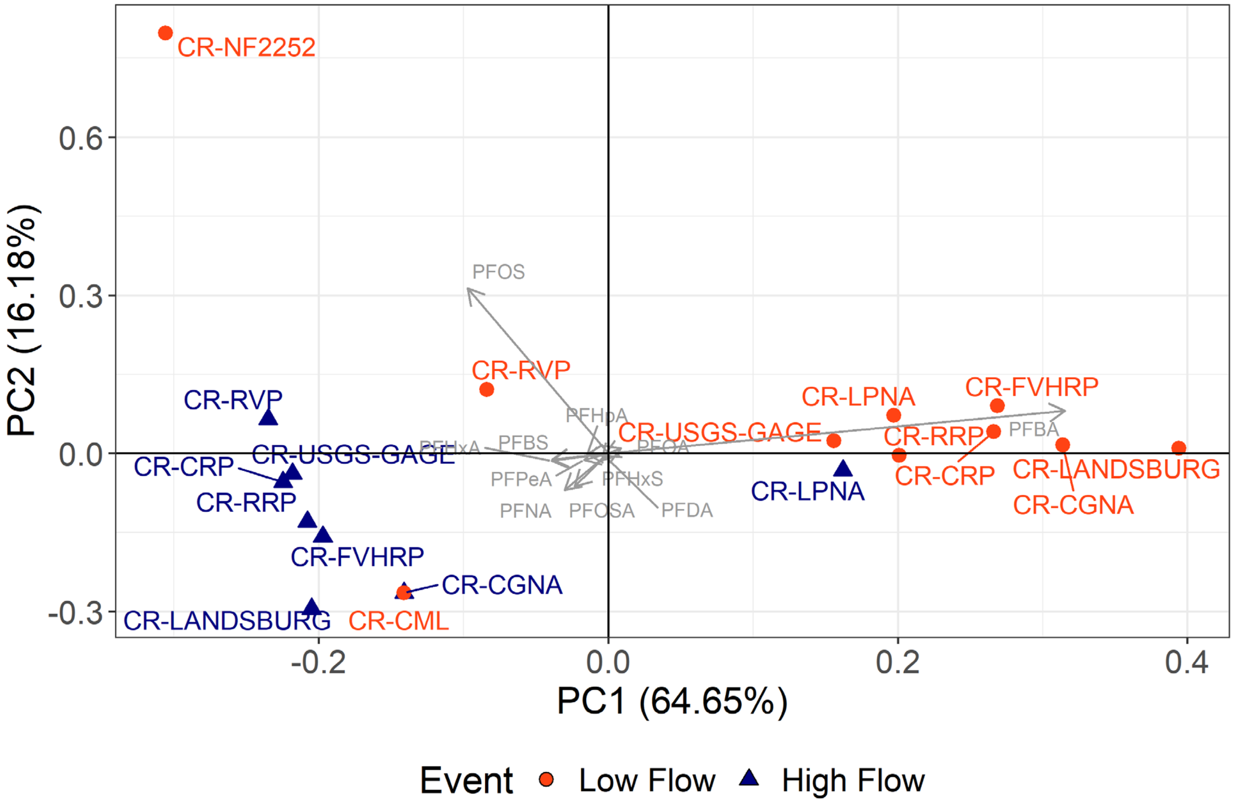


Figure 26. Principal components analysis scores and loadings plot for Phase 2 Cedar River surface water PFAS samples by flow regime.

The first two principal components, explaining 81% of the variance, are plotted. Dots (scores) represent samples. Vectors (loadings) represent PFAS analytes. PFAS = per and polyfluoroalkyl substances.

Groundwater

Total PFAS concentrations in all Cedar River subbasin groundwater samples, ranged from non-detect to 946 ng/L, with a median concentration of 31.9 ng/L (Table 5). The median total PFAS concentration of Cedar River subbasin samples collected at the groundwater-surface water interface (36.2 ng/L) was higher than in surface water (2.50 ng/L). The median PFAS concentration of groundwater samples collected from the Renton monitoring well network was lower (13.3 ng/L) than in samples collected at the groundwater-surface water interface.

PFAS were detected in all groundwater samples in the subbasin except for the sample collected at the most upstream site, CR-LANDSBURG-GW, during the spring event (Figure 27). Generally, total PFAS concentrations followed the same trend as surface water samples in which total concentrations were higher in the urban lower reach of the Cedar River (Figure 27; Appendix D-Figure D1). Total PFAS concentrations were higher during the spring event apart from one sample location (CR-5) near the mouth (Figure 27). Sample locations CR-6 through CR-8 were only sampled during the spring event.

PFBS and PFOS were the most frequently detected compounds in groundwater (94% of the samples; Figure 24). Additionally, of the PFSAs, detection frequencies of PFPeS (C5; 75%) and PFHxS (C6; 88%) were high. PFCAs with high detection frequencies included PFBA (C4; 69%), PFHxA (C6; 81%), PFHpA (C7; 69%), PFOA (C8; 75%), and PFNA (C9; 81%). Detection frequencies of PFAAs were greater during the summer, however, seasonal differences in concentrations varied by location. PFCAs C10 and C11 were detected more frequently during the summer event. PFSAs detections of C4 – C6 and C8 were detected in all samples during the summer event. PFCAs C12-C14 and PFSAs C9 and C12 were not detected.

Fluorotelomers 4:2 FTS, 6:2 FTS, 8:2 FTS and 5:3 FTCA were detected less frequently than PFAAs (50% or less of samples). PFOSA was the most frequently detected sulfonamide (69%). N-EtFOSAA, N-MeFOSE, and N-EtFOSE were detected during the summer event. Ethers were not detected in any of the Cedar River subbasin groundwater samples.

Two areas with high PFAS concentrations and distinct compositions were observed: Groundwater samples collected near the mouth of the Cedar River and groundwater and spring samples collected in the Larry Phillips Natural Area.

Phase 2 sampling included resampling at the CR-1 location (CR-1-PH2) and additional locations south of Phase 1 location CR-3. During the summer 2022 Phase 2 event, PFAS was detected at much higher concentration at one of the additional sample locations, CR-5 (946 ng/L; n=1), compared to other locations near the mouth of the Cedar River (Figure 27). To better understand the extent of the high PFAS concentrations along this reach of the river additional sampling was conducted at locations to the south of CR-5 (i.e., CR-6 through CR-8) during the spring 2023 Phase 2 event.

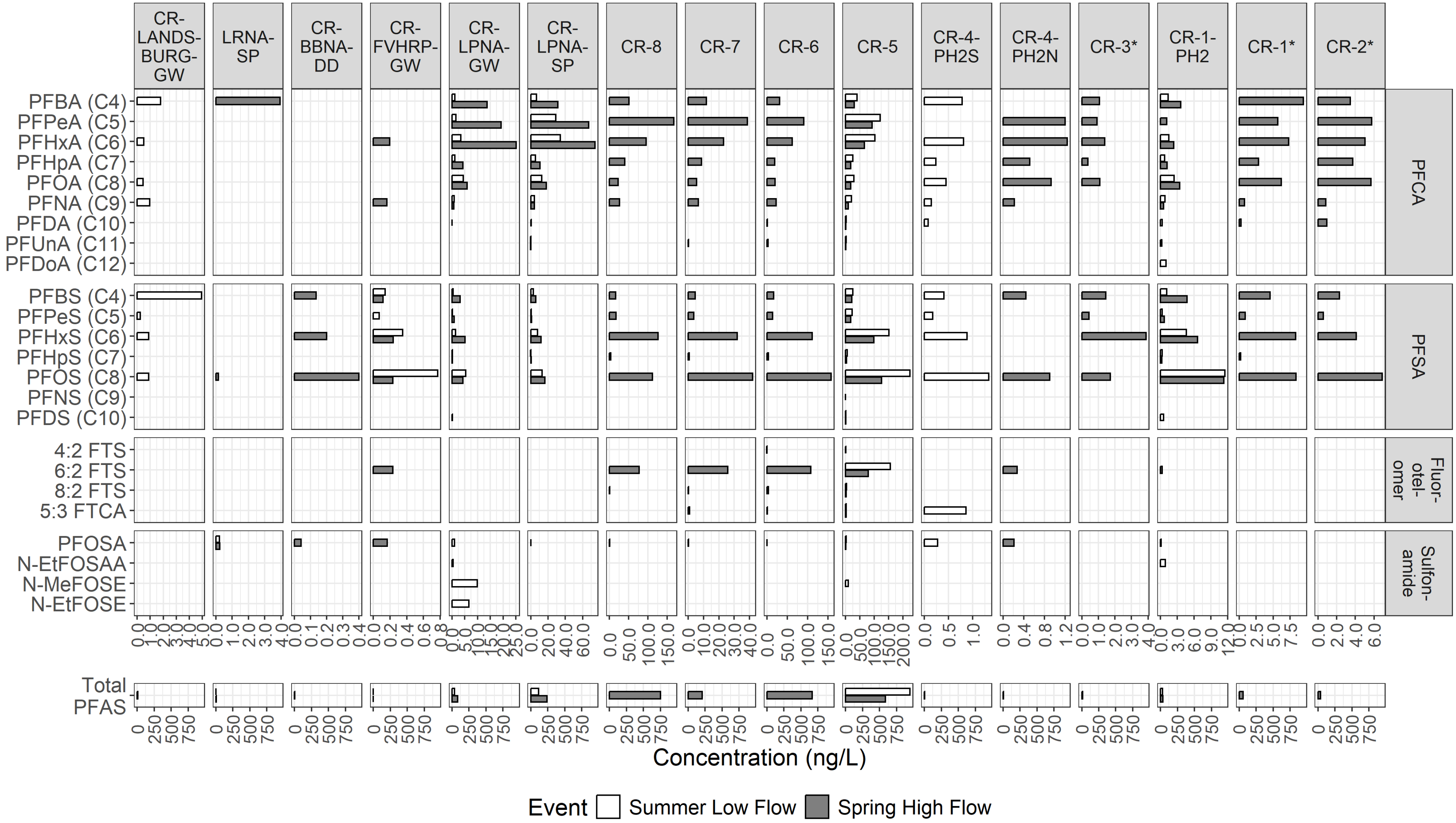


Figure 27. Analyte and total PFAS concentrations (ng/L) in Phase 1 and 2 Cedar River groundwater and springs samples, arranged upstream to downstream.

* Phase 1 sample. CR-BBNA-DD, CR-8, CR-7, CR-6, CR-4-PH2N, CR-3, CR2, and CR-1 were only sampled during the spring high flow event. CR-4-PH2S was only sampled during the Phase 2 summer low flow event. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

High PFAS concentrations ranging from 206 – 753 ng/L were detected at CR-5 through CR-8 during the spring 2023 event. This PFAS plume has a distinct composition dominated by C5 (PFPeA) and C6 (PFHxA) PFCAs, C6 (PFHxS) and C8 (PFOS) PFSAs, and 6:2 FTS (Figure 27). The high concentrations and similar composition indicate a similar source. Phase 1 locations CR-1 through CR-3 were also primarily composed of PFAAs, but dominant compounds varied by location, concentrations were much lower, and 6:2 FTS was not detected.

The total PFAS concentration decreased dramatically to the north of CR-5 (CR-4-PH2S total PFAS concentration <7.00 ng/L). This may indicate the northern extent of the plume. However, it may be due to the presence of a retaining wall located along the walking path within approximately 50 feet of the bank or a 350 ft sheet pile wall approximately 150 feet northeast of CR-4-PH2S. The sheet pile wall, which is part of a contaminated site remediation, impedes shallow contaminated groundwater flow to the river. The wall does not hydraulically isolate shallow groundwater from the Cedar River but likely alters the location of groundwater discharging to the river (Roy F. Weston, Inc.2001). Therefore, the portion of the plume that discharges to the river may extend further north than CR-4-PH2S suggests. The concentration of CR-8 was the highest concentration detected in the plume during the spring 2023 (Figure 27). Based on this high concentration, the plume likely extends further south. Additional samples would be needed to delineate the full extent of the plume.

Upstream of the lower Cedar River, higher total PFAS concentrations were also detected in the Larry Phillips Natural Area (CR-LPNA-SP and CR-LPNA-GW; Figure 27). Total PFAS concentrations in groundwater along the right bank were 38.1 ng/L (summer) and 82.5 ng/L (spring). Total PFAS concentrations collected from springs in the Larry Phillips Natural Area (CR-LPNA-SP) were 118 ng/L (summer) and 242 ng/L (spring). Both sites were dominated by short-chain PFCAs (C4-C6; Figure 27).

Total PFAS concentrations in groundwater samples collected from the monitoring well network range from 0.585 – 42.1 ng/L, with a median concentration of 13.3 ng/L (Figure 28). The highest PFAS concentrations were observed in downtown monitoring wells, north of I-405 nearest the right (east) bank of the Cedar River (MW11, MW30S, and MW3). Groundwater samples from the monitoring wells were dominated by PFAAs. PFSAs C4, C6, and C8 were the most frequently detected compounds (88 – 92%).

The sulfonamide PFOSA was observed at low concentrations (<1.00 ng/L) in three wells during the summer event and one well during the spring event, whereas N-MeFOSE was observed at one location during the summer event (MW10). Fluorotelomers and ethers were not observed in any of the Renton monitoring wells.

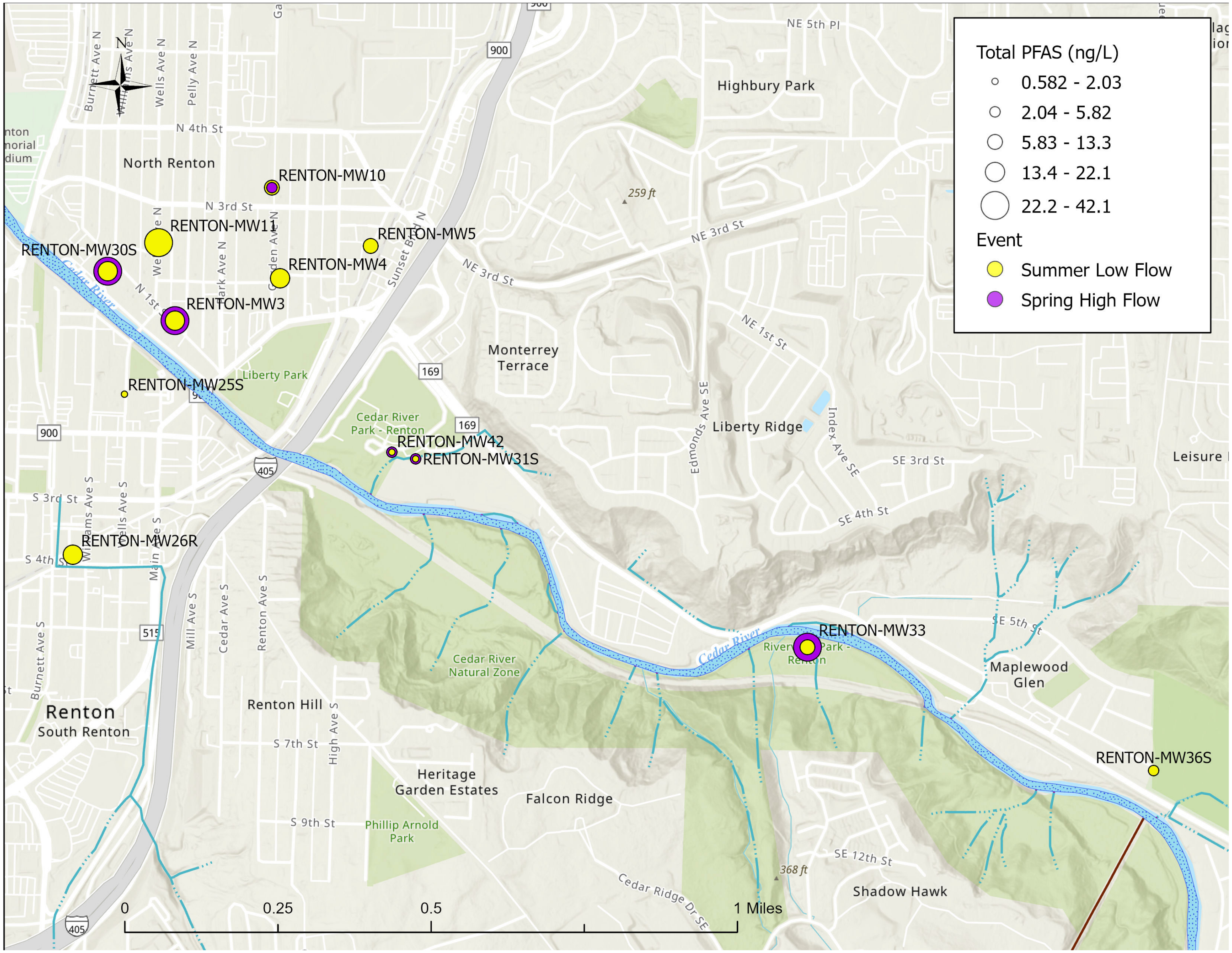


Figure 28. Map of total PFAS concentrations at Renton, WA monitoring well sites, sampled during Phase 2.

PFAS = per- and polyfluoroalkyl substances.

Juanita Creek

Surface Water

Total PFAS concentrations in Juanita Creek surface water samples were 29.1 – 46.3 ng/L, with median concentration more than two times higher than in the lake (Table 5). There was little variation in total PFAS concentrations upstream to downstream, and between summer low and spring high flow sampling events (Figure 29; Appendix D-Figure D2). Among the three storm events in October, November, and February, total PFAS concentrations at the outlet (JC-MOUTH) were 38.0 – 60.1 ng/L, with the highest concentration observed during the October storm event.

Detection frequencies of C4 – C10 PFCAs and C4 – C8 PFSAs were at or near 100% (Figure 24). In each sample, C4 – C6 and C8 PFCAs and C4, C6, and C8 PFSAs were the dominant analytes (Figure 29). 6:2 FTS, 5:3 FTCA, and PFOSA were detected in some samples (8 – 31% detection frequency) at low concentrations, mostly during summer low flow.

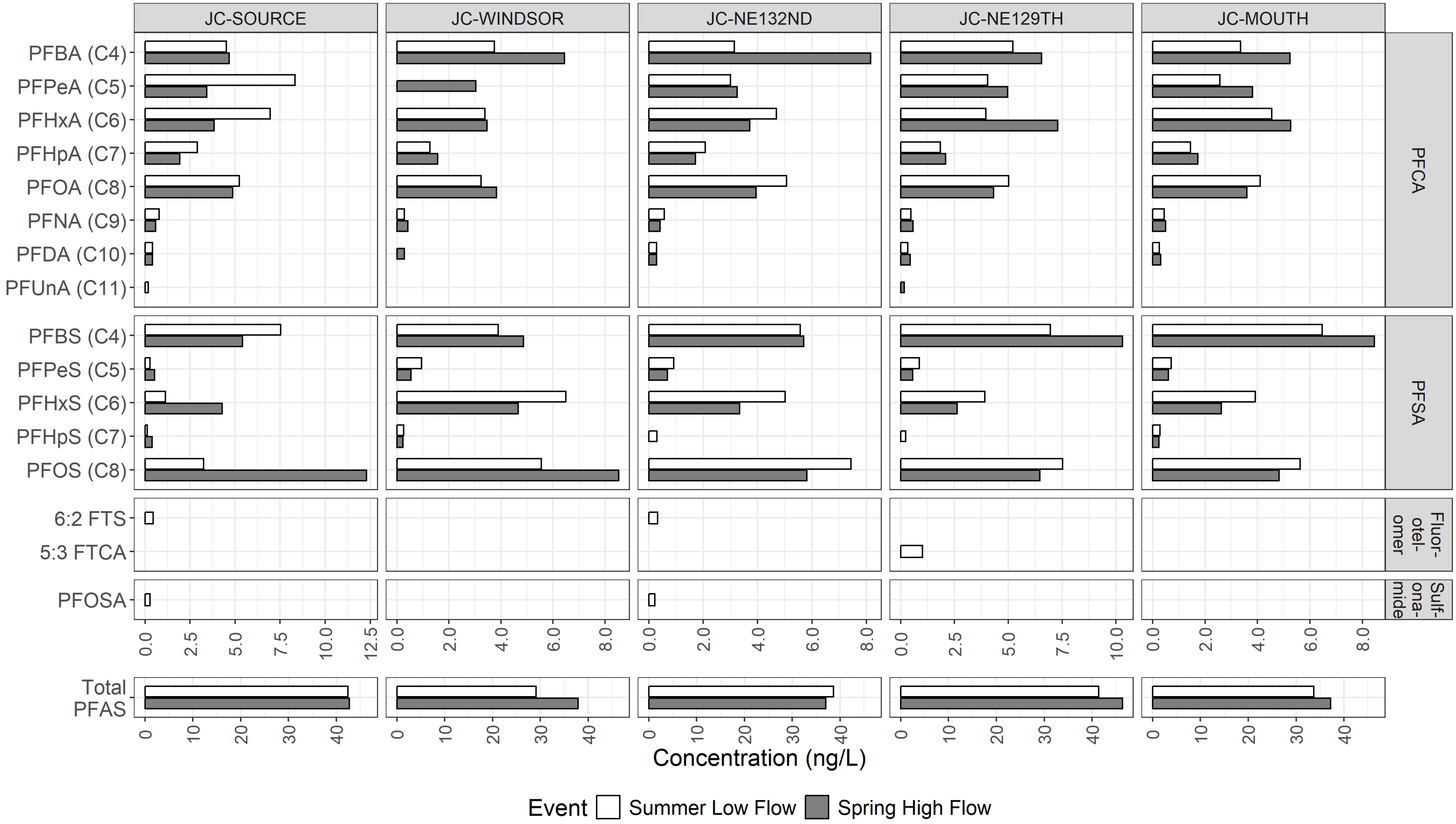


Figure 29. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 Juanita Creek sites, arranged upstream to downstream.

Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

PCA results for Juanita Creek surface water samples showed that total PFAS concentrations at the most upstream sites (JC-SOURCE and JC-WINDSOR) were influenced by specific PFAS analytes (Figure 30). PFOS had the most influence on total PFAS concentration in the two upstream samples during spring high flow. During summer low flow JC-SOURCE was most influenced by PFPeA, and JC-WINDSOR was most influenced by PFHxS. Samples from the mid and downstream sites (JC-NE129TH, JC-NE132ND, and JC-MOUTH) were less influenced by specific analytes and sampling events.

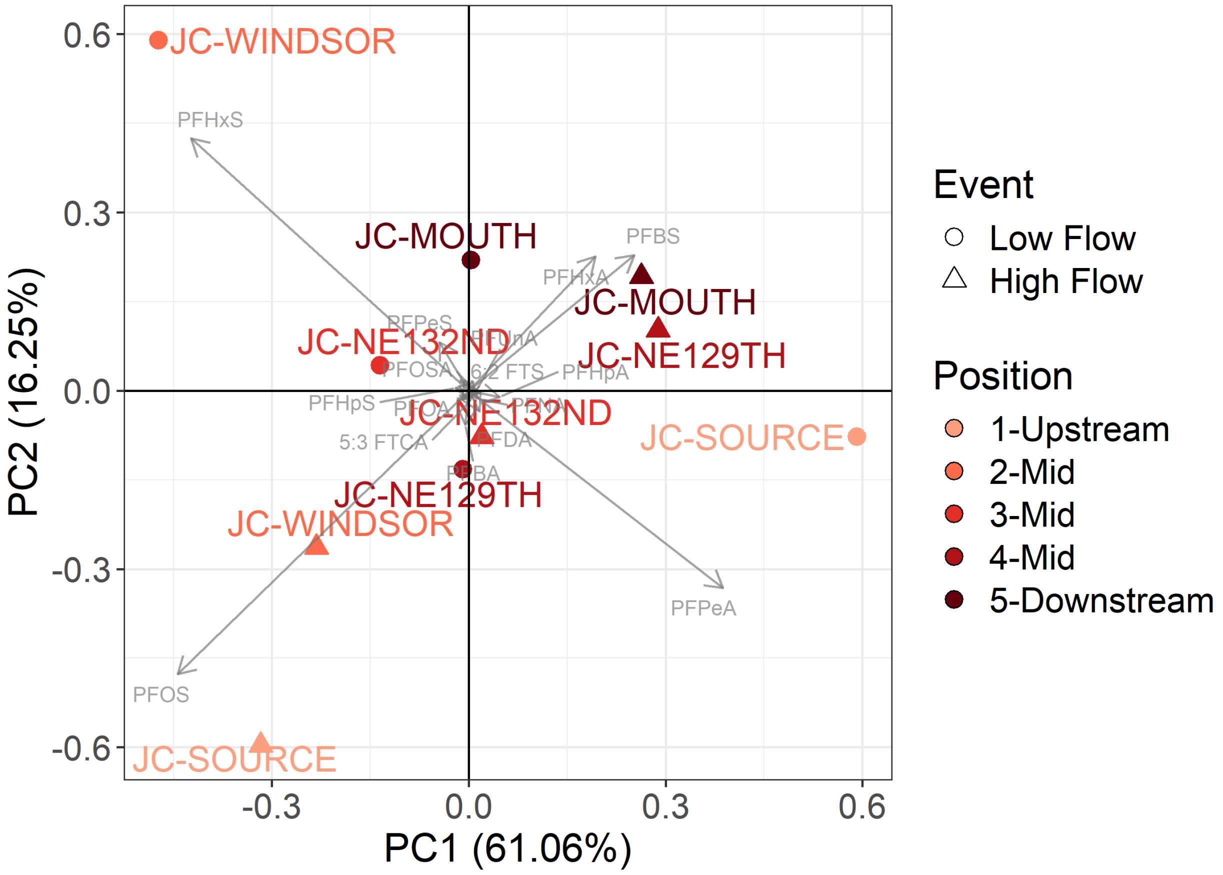


Figure 30. Principal components analysis scores and loadings plot for Phase 2 Juanita Creek surface water PFAS samples by flow regime and stream positioning.

The first two principal components, explaining 77% of the variance, are plotted.
Symbols (scores) represent samples. Vectors (loadings) represent PFAS analytes. PFAS = per- and polyfluoroalkyl substances.

Groundwater

Total PFAS concentrations in Juanita Creek groundwater samples were 4.53 – 53.7 ng/L, with median concentration of 13.4 ng/L. Groundwater samples were only obtained at three out of five surface water locations. Due to limitations of the sampling device (difficulty inserting due to coarse substrate) we were unable to collect samples between JC-WINDSOR and JC-MOUTH.

We had difficulty obtaining a sample from the summer Windsor Vista Park location (JC-WINDSOR-GW) due to seasonal changes in the substrate and variability in groundwater discharge location. A gravel bar was noted near the sample location, which may have influenced the local gradient. A second location (JC-WINDSOR-GW2) was selected on the left (south) bank. Pore water parameters (specific conductance, dissolved oxygen, pH and oxidation reduction potential [ORP]) at JC-WINDSOR-GW2 varied from JC-WINDSOR-GW, and so both samples were submitted for analysis.

Total PFAS concentrations in groundwater decreased from the upstream to the downstream sample locations (Figure 31; Appendix D-Figure D2), unlike surface water concentrations which had little spatial variation. The highest concentration was observed during the spring event at the location nearest the headwaters (JC-SOURCE-GW, 53.7 ng/L).

C6 and C8 PFCAs and C4 and C8 PFSAs were detected in all groundwater samples (Figure 24). PFOS (C8) was the dominant compound at the two upstream locations, whereas PFBS (C4) was the dominant compound at the mouth (Figure 31). C10 through C12 PFCAs were observed during the summer event at JC-SOURCE-GW. Ether PFMPA was observed at a low concentration (0.102 ng/L) at JC-MOUTH-GW during the spring event. Low concentrations (<1.30 ng/L) of sulfonamides PFOSA and N-EtFOSAA were detected in three samples. Fluorotelomers were not observed at any of the Juanita Creek groundwater locations.

Total PFAS concentrations in groundwater were similar (12.8 and 13.4 ng/L) at JC-WINDSOR-GW and JC-WINDSOR-GW2. Groundwater concentrations were noticeably lower than the surface water concentration (37.9 ng/L) at this location. The two locations showed some variation in composition. JC-WINDSOR-GW was dominated by PFSAs and JC-WINDSOR-GW2 was dominated by PFOA and PFOS.

The groundwater total PFAS concentration at the mouth of Juanita Creek was slightly higher during the spring sampling event (8.16 ng/L) than during summer (4.53 ng/L), which was consistent with our surface water results from this site. Groundwater concentrations at the mouth were much lower than concentrations observed in surface water (34.9 - 38.0 ng/L). The difference in concentration between co-located groundwater and surface water samples may indicate a lack of upgradient PFAS sources near the mouth. However, groundwater could be the source of PFAS somewhere upstream of the mouth. Alternatively, a non-point source to surface water is located nearby.

Nearby lake shore samples, JBP and JBP2, ranged in concentration from 56.1 – 65.7 ng/L (Phase 2), indicating a source is located in the upgradient direction. The lake shore samples were dominated by PFOA with a relatively low concentration of PFOS (Figure 31). PFOS was more dominant in groundwater samples collected along the creek. JC-MOUTH-GW was collected along the right (west) bank of the creek. A sample collected along the left (east) bank of the creek may have yielded different results, representing groundwater flow inputs from upgradient of the left bank.

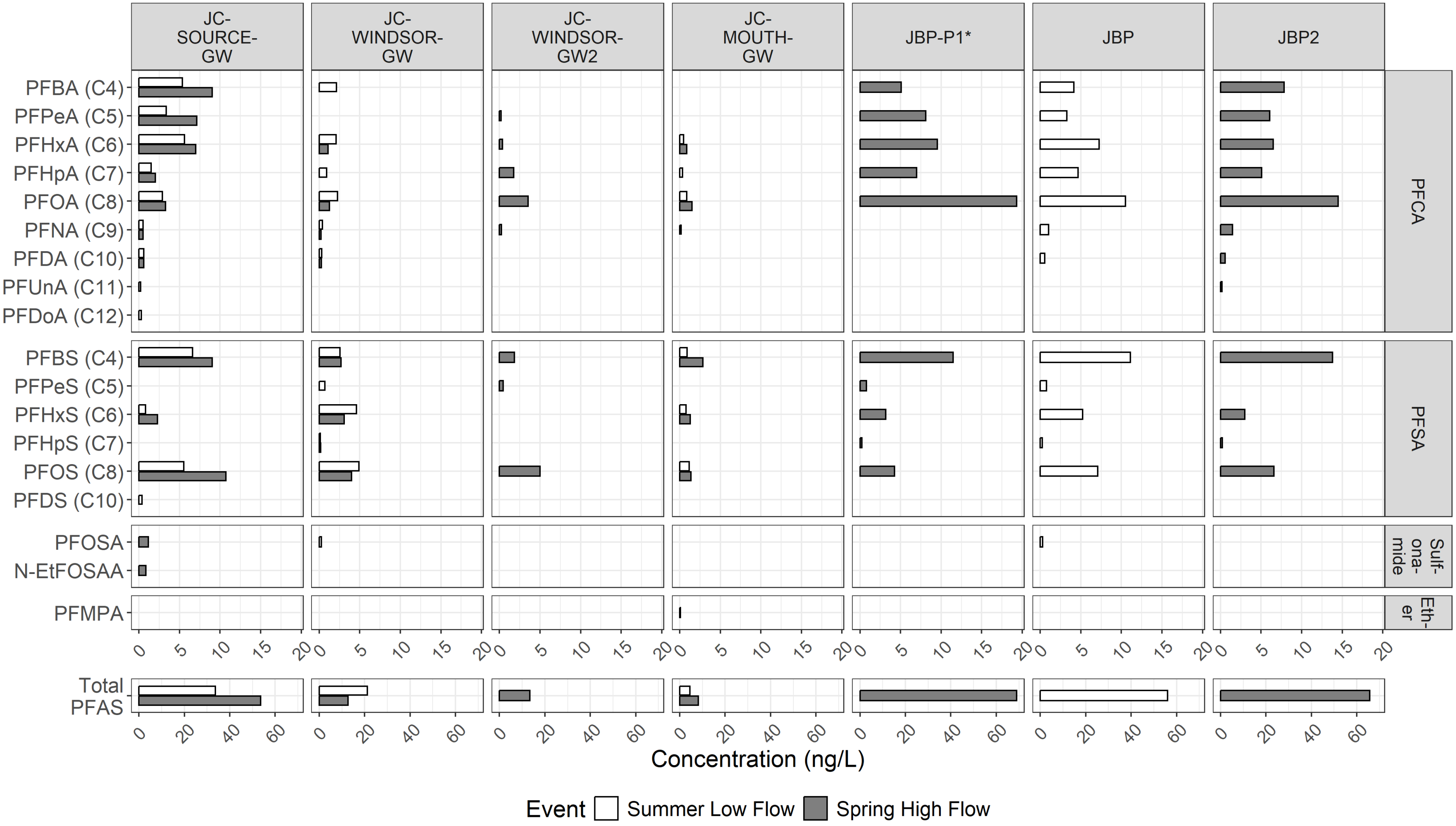


Figure 31. Analyte and total PFAS concentrations (ng/L) in Phase 2 Juanita Creek groundwater samples and Phase 1 and 2 Lake Washington shoreline groundwater samples (JBP and JBP2), arranged upstream to downstream.

* JBP-P1 shows Phase 1 results for the JBP site. JC_WINDSOR-GW2, JBP-P1, and JBP2 were only sampled during the spring high flow event. JBP was only sampled during the Phase 2 summer low flow event. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Thornton Creek

Surface Water

Total PFAS concentrations in Thornton Creek surface water samples were 15.9 – 85.0 ng/L, with median concentration more than two times higher than in the lake (Table 5). The highest total PFAS concentration was observed during summer low flow in the south branch of Thornton Creek at TC-S-5TH (Figure 32). The lowest total PFAS concentration was observed during summer low flow in the north branch at TC-N-10TH, which lies within Thornton Creek Natural Area Park.

Total PFAS concentrations were 1.5 times higher on average at the most upstream sites of both the north and south branches (TC-N-RONALD and TC-S-BARTON) than at the outlet (TC-MOUTH) (Figure 32; Appendix D-Figure D3). With exception of the TC-S-5TH site, total PFAS concentrations during summer low versus spring high flow samples were comparable. Total PFAS concentrations at the outlet of Thornton Creek (TC-MOUTH) during the three storm events were 31.7 – 40.1 ng/L, with the highest concentration observed during Storm 1.

Detection frequencies of C4 – C10 PFCAs and C4, C6, and C8 PFSAs were at or near 100% (Figure 24). Compared to the Cedar River and Juanita Creek, there were more detections of long-chain PFAAs and sulfonamides (N-EtFOSAA, N-MeFOSE, and N-EtFOSE), especially during summer low flow (Figure 32). Fluorotelomer compounds (6:2 FTS and 5:3 FTCA) and PFOSA were also detected in some samples.

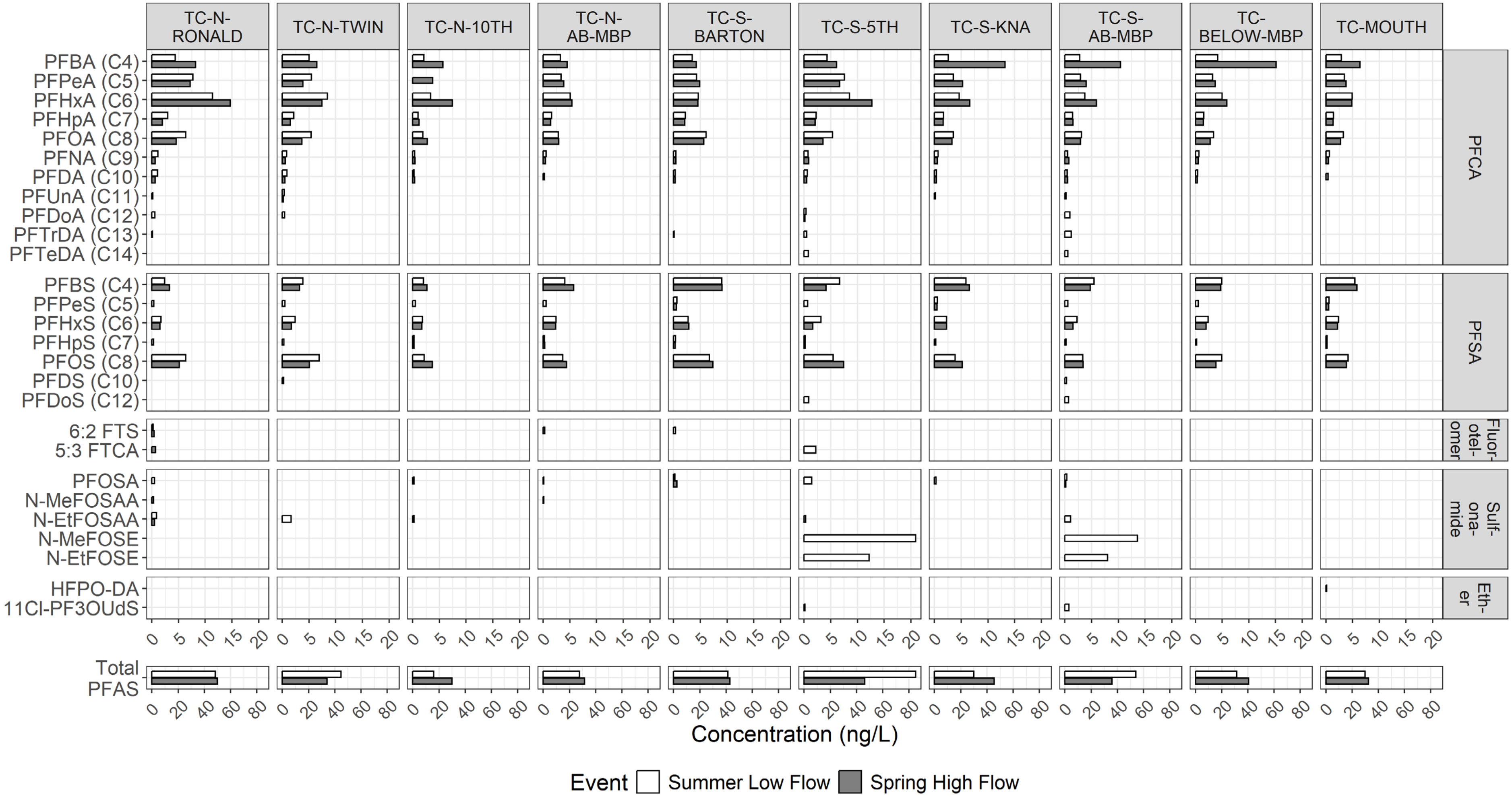


Figure 32. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 Thornton Creek sites, arranged upstream to downstream.

Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

PCA results for Thornton Creek surface water samples showed that PFAS composition was influenced by sampling event (Figure 33). This was primarily due to the dominance of two sulfonamide compounds (N-MeFOSE and N-EtFOSE) in the summer low flow samples from two sites (TC-S-5TH and TC-S-AB-MBP), and the influence of PFBA on spring high flow samples. At most Thornton Creek sites during spring high flow, PFBA made up a higher proportion of the PFAS composition and was often the dominant analyte (Figure 32).

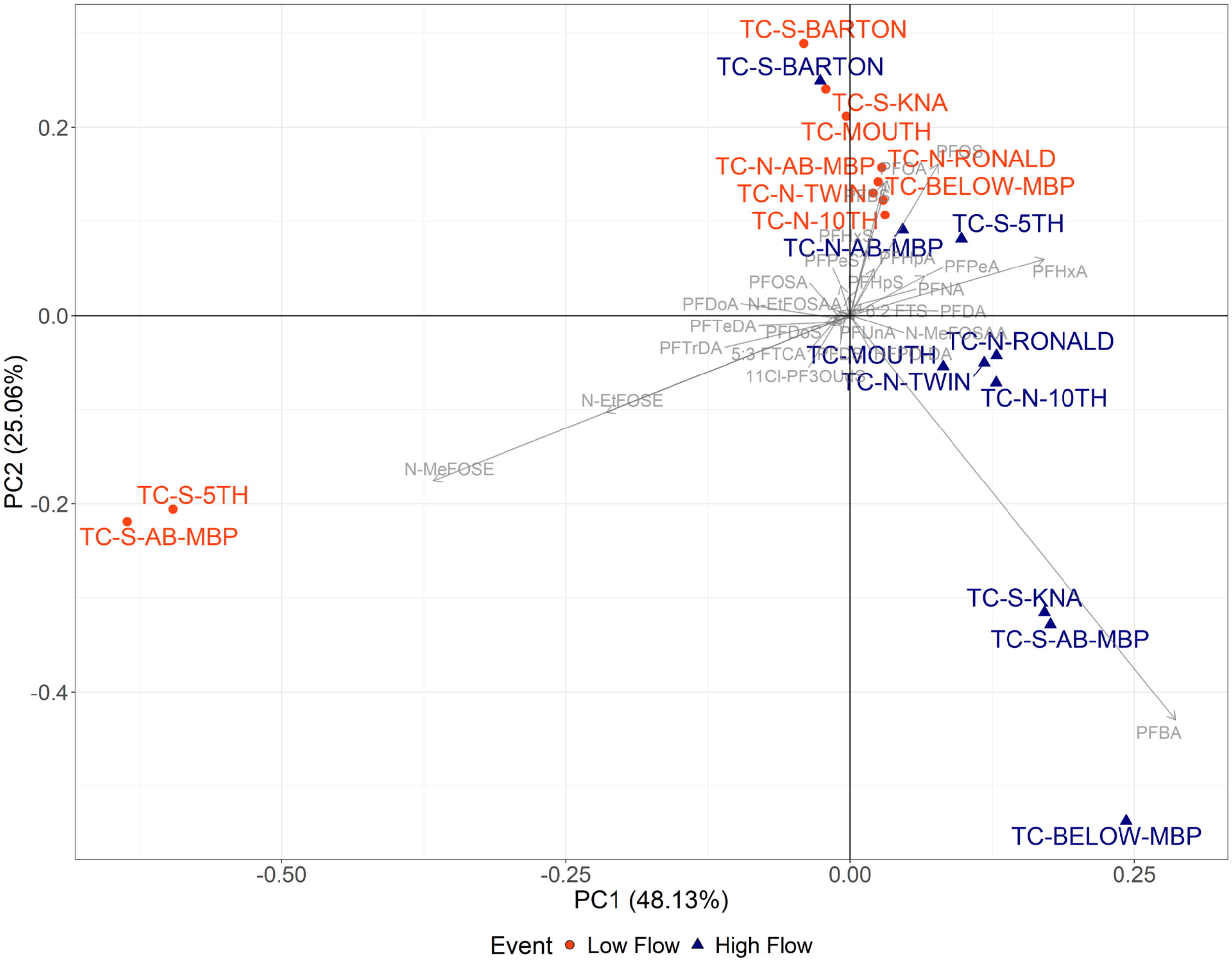


Figure 33. Principal components analysis scores and loadings plot for Phase 2 Thornton Creek surface water PFAS samples by flow regime.

The first two principal components, explaining 73% of the variance, are plotted. Symbols (scores) represent samples. Vectors (loadings) represent PFAS analytes. PFAS = per- and polyfluoroalkyl substances.

Groundwater

Total PFAS concentrations in Thornton Creek groundwater samples were 0.979 – 103 ng/L, with a median concentration of 47.1 ng/L (Table 5) Thornton Creek had the highest median total PFAS concentration of the three focus subbasins. High total PFAS concentrations were observed on the north and south branch, with the lowest concentrations observed within two natural areas. The highest total PFAS concentration was detected at the sample location downstream of Meadowbrook Pond (TC-BELOW-MBP-GW) (Figure 34). The lowest total PFAS concentration was observed on the north branch within the Thornton Creek Natural Area Park (TC-N-10th-GW), which is also the surface water sample location with the lowest concentration.

Low total PFAS concentrations (9.65 and 16.0 ng/L, summer and spring respectively) were also detected in springs discharging from the hillside above Thornton Creek within the Kingfisher Natural Area (TC-S-KNA-SP). Surface water samples collected from Thornton Creek at this location (TC-S-KNA) were higher (29.9 – 45.4 ng/L) than samples collected from the springs. The higher concentration corresponds to the spring event.

The most frequently detected PFAS were PFCAs C4 – C9 and PFSAs C4, C6, and C8 (Figure 24). PFBS was detected in all samples (Figure 24). Ether compound 11Cl-PF3OUdS, which was not detected in the other subbasins, was detected at low concentrations (less than 1 ng/L) at two Thornton Creek sample locations (TC-BELOW-MBP-GW and TC-N-167THST-GW). Fluorotelomer 5:3 FTCA was also detected at one location (TC-S-AB-MBP-GW). Sulfonamides N-EtFOSAA, N-EtFOSE, N-MeFOSE, and PFOSA were detected at low concentrations (less than 2 ng/L), except for N-EtFOSE (9.56 ng/L) and N-MeFOSE (20.8 ng/L). The higher concentrations were observed during the summer event at the location below Meadowbrook Pond (TC-BELOW-MBP-GW) (Figure 34). This is the highest N-MeFOSE concentration detected in any of the groundwater samples in the study.

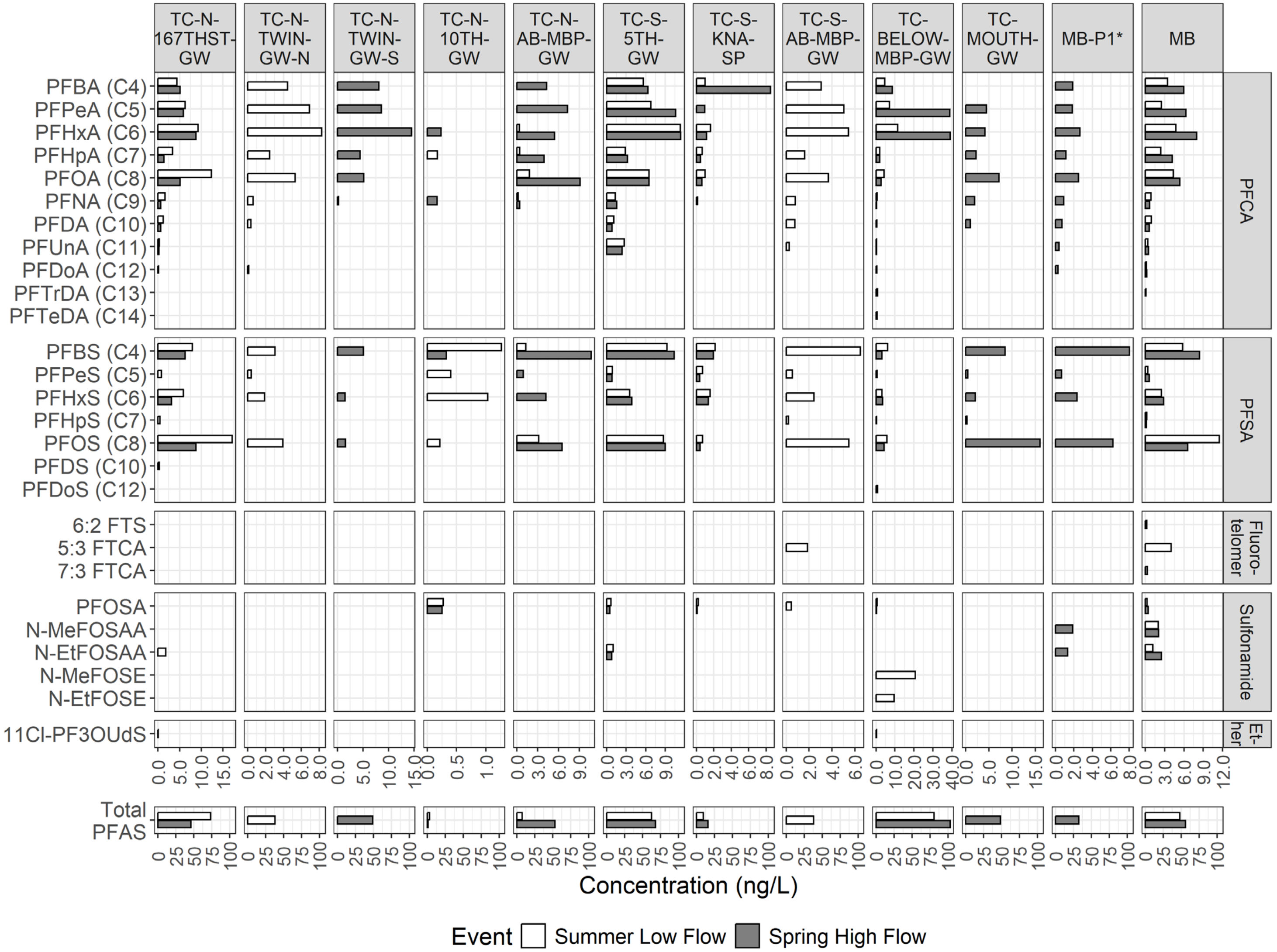


Figure 34. Analyte and total PFAS concentrations (ng/L) in Phase 2 Thornton Creek groundwater samples.

* MB-P1 shows Phase 1 results for the MB site. TC-N-TWIN-GW-S, TC-MOUTH-GW, AND MB-P1 were only sampled during the spring high flow event. TC-N-TWIN-GW-N and TC-S-AB-MBP-GW were only sampled during the Phase 2 summer low flow event. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Additional Subbasins

Fairweather Creek

Total PFAS concentrations in Fairweather Creek surface water samples were 87.5 ng/L at the most upstream site (FWC-NE24TH) and 139 ng/L at the outlet (FWC-MOUTH), which was the highest total PFAS concentration observed among surface water samples in the study (Figure 35; Appendix D-Figure D4). Phase 1 and Phase 2 total PFAS concentrations and PFAS compositions at the outlet were similar.

At both sites, PFAS composition was primarily C5, C6, and C8 PFCAs, with lower concentrations of other PFAA analytes (Figure 35). Compared to other tributary surface water sites, Fairweather Creek had relatively high concentrations of long-chain PFAS, including higher PFDA concentrations (1.64 – 9.40 ng/L) than in all other tributaries (non-detect – 1.44 ng/L), and among the highest PFOS (8.08 – 14.4 ng/L) and PFOA (11.4 – 25.6 ng/L) concentrations.

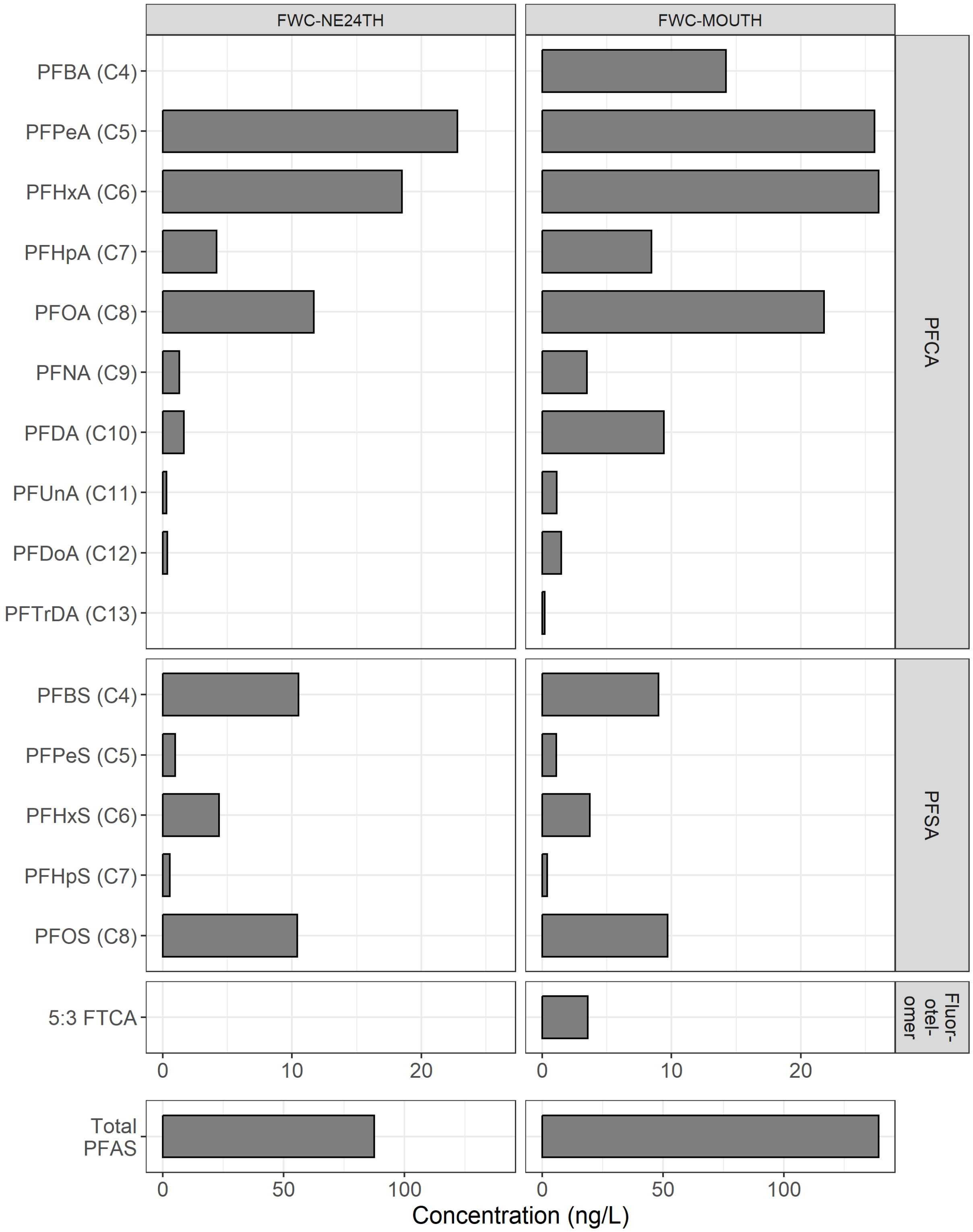


Figure 35. Analyte and total PFAS concentrations (ng/L) in Phase 2 Fairweather Creek surface water samples, arranged upstream to downstream.

Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances.
PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Kelsey Creek

Surface water total PFAS concentrations were 26.0 – 53.8 ng/L in the mainstem of Kelsey Creek (Table 5) and varied from upstream to downstream (Figure 36; Appendix D-Figure D5). The highest concentration was observed at the outlet of Larsen Lake (KC-LARSEN), near the headwaters of Kelsey Creek. The lowest concentration was observed at KC-ABOVE-VC, downstream from KC-LARSEN, and upstream from Valley Creek and other major tributary inflows to Kelsey Creek.

PFAS composition in Kelsey Creek was primarily C4 – C8 PFCAs and C4, C6, and C8 PFSAs (Figure 36). PFNA, PFDA, and PFPeS were present in all samples at low concentrations. At some sites, 6:2 FTS, N-EtFOSAA, and N-EtFOSE were detected at low concentrations.

Surface water total PFAS concentrations in Valley Creek, a tributary to Kelsey Creek, were 17.6 – 92.0 ng/L. The lowest total PFAS concentrations in Valley Creek were observed at the two upstream-most sites near the headwaters (VC-ABOVE-BGC and VC-BELOW-BGC), and the highest near its outlet to Kelsey Creek (VC-MOUTH). Surface water total PFAS concentration at VC-MOUTH was among the highest observed during Phase 2, which also included concentrations in West Tributary, another sub-drainage in the Kelsey Creek subbasin (see Opportunistic Sampling section).

VC-MOUTH was composed of a mixture of PFAS analytes. The sample had the highest concentrations of long-chain (C11 – C14) PFAS among all tributary surface water samples—different than the two upstream sites. In particular, PFDoA concentration (18.0 ng/L) was an order of magnitude higher than in all other water samples. PFDS, 5:3 FTCA, and N-EtFOSAA (intermediary transformation product of N-EtFOSE) were also analytes not commonly found in the tributary surface water samples but detected at relatively high concentrations in the VC-MOUTH sample.

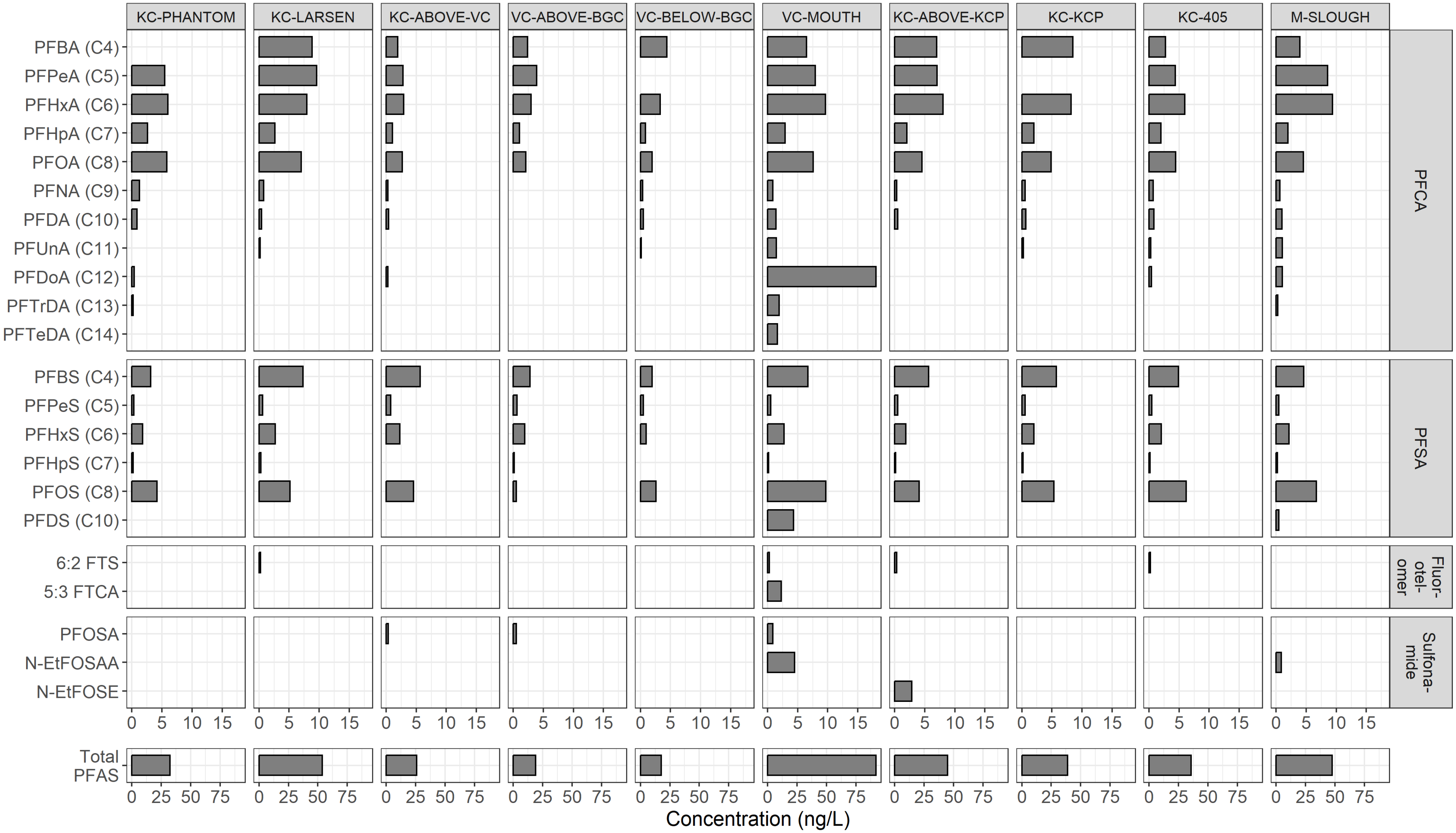


Figure 36. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 Kelsey Creek subbasin sites, arranged upstream to downstream.

PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.
Non-detected analytes are not shown.

McAleer Creek

Total PFAS concentrations in McAleer Creek surface water samples were 19.5 – 22.8 ng/L (Table 5). Total PFAS concentrations in Hall Creek, the main tributary to Lake Ballinger (the source waters of McAleer Creek), were higher than in McAleer Creek: 53.1 ng/L at the upstream site (HC-HALL-LK-OUT) and 42.3 ng/L at the downstream site (HC-BALL-IN). PFAS concentrations declined towards the McAleer Creek outlet (MC-MOUTH) (Figure 37; Appendix D-Figure D6).

PFAS composition in McAleer and Hall Creek samples was primarily C4 – C8 PFCAs and C4, C6, and C8 PFSAs (Figure 37). Fluorotelomer and sulfonamide compounds were not detected, apart from detections of PFOSA and 6:2 FTS at low concentrations in some samples.

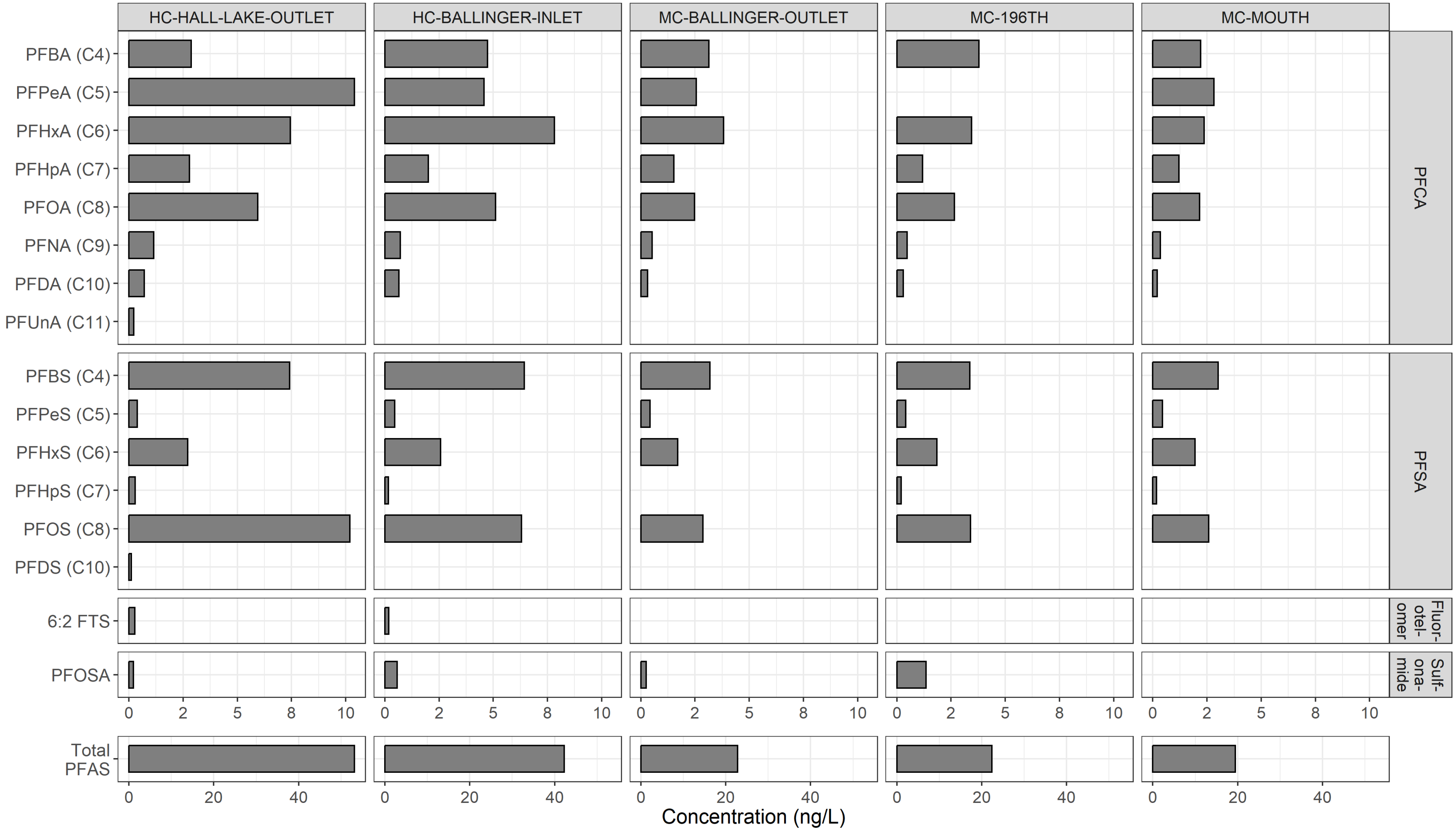


Figure 37. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 McAleer Creek subbasin sites, arranged upstream to downstream.

PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.
Non-detected analytes are not shown.

Ravenna Creek

Surface Water

Total PFAS concentrations in Ravenna Creek surface water samples were 7.64 – 35.6 ng/L (Table 5). The highest concentrations were observed at the downstream-most sites in Union Bay Natural Area, while the lowest concentration was observed at the upstream-most site in Ravenna Park (Figure 38; Appendix D-Figure D7).

Samples were also collected in two ponds within the Union Bay Natural Area. Total PFAS concentration was 53.6 ng/L in Carp Pond (RC-CARP) and 28.2 ng/L in Central Pond (RC-CENTRAL).

PFAS composition in the upstream sample (RC-RP) was primarily C4, C6, and C8 PFCAs and C4, C6, and C8 PFSAs, with PFBS being the dominant analyte (Figure 38). In the downstream samples, C4 – C6 PFCAs were more dominant, and detections of other PFAS analytes were also more frequent. In the RC-MONTLAKE and RC-WAHKIAKUM samples, 5:3 FTCA made up a considerable proportion of the total PFAS (% composition). PFOS was not the dominant analyte in any of the Ravenna Creek samples, however it was one of the dominant analytes in the two pond samples (the others including C5, C6, and C8 PFCAs).

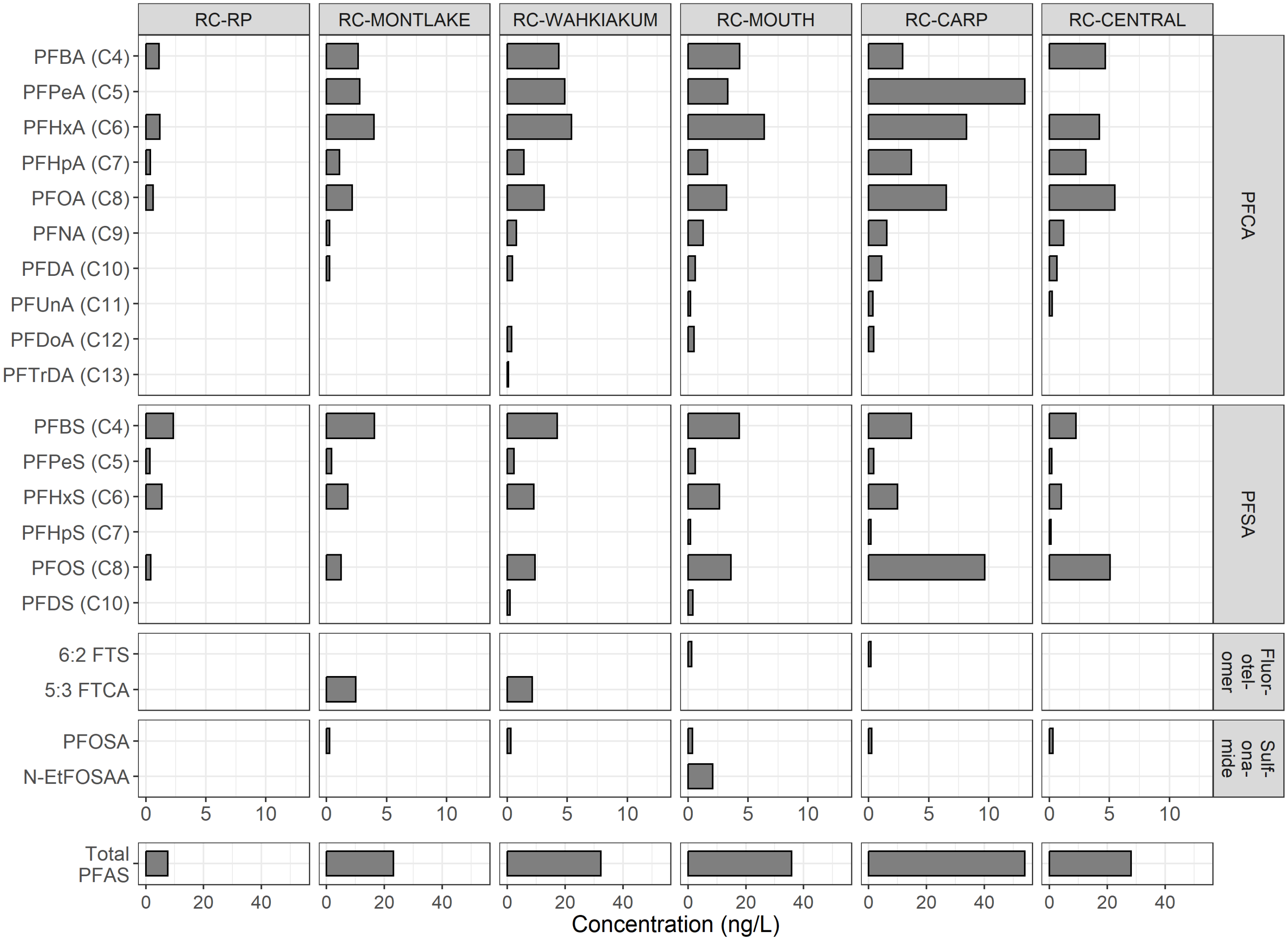


Figure 38. Analyte and total PFAS concentrations (ng/L) in surface water samples collected from Phase 2 Ravenna Creek subbasin sites, arranged upstream to downstream.

PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid. Non-detected analytes are not shown.

Groundwater

Total PFAS concentrations in Ravenna Creek groundwater samples were 9.84 – 203 ng/L. The highest concentration was observed during the summer event, at RC-MONTLAKE-GW (Figure 39). This sample was located where the creek resurfaces after travelling through an underground pipe. The nearest downstream sample location to the south, RC-WAHKIAKUM-GW, also had one of the highest total PFAS groundwater concentrations detected (168 ng/L). This location coincides with the highest sediment total PFAS concentration (50.6 ng/g). The total PFAS concentration at RC-MONTLAKE-GW was nearly 6.5 times higher during the summer event (203 ng/L) compared to the spring event (31.5 ng/L). The lowest total PFAS concentration in groundwater was observed at the upstream-most site in Ravenna Park (RC-RP-GW, 9.84 ng/L, spring event).

PFAS detections in the sample collected nearest the headwaters (RC-RP-GW) was primarily composed of PFAAs (Figure 39). Short chain PFCAs, C4 through C7, were detected at low concentrations (<1.50 ng/L). PFBS (C4) was the dominant PFSA, with low concentrations (<1.50 ng/L) of C5, C6, and C8 detected. Sulfonamide PFOSA was detected during the spring event at a low concentration (0.238 ng/L). Other sulfonamides, fluorotelomers, and ethers were not detected at this location.

PFAS composition of groundwater samples collected downstream of Montlake Boulevard was primarily C4 through C10 PFCAs and C4, C6, C8, and C10 PFSAs. Longer chain PFCAs, C11 through C14, were also observed at low concentrations (less than 1.00 ng/L). PFDoA (C12) was observed at all three locations but only detected during one of the two events. PFTrDA (C14) was only observed during the spring event at RC-WAHKIAKUM-GW.

RC-MONTLAKE-GW was predominantly composed of PFOS and had concentrations up to 91.4 ng/L, which was among the highest observed in any of the groundwater samples (Table 5; Figure 39). Elevated PFOS concentrations were also identified in the plume near the mouth of the Cedar River (maximum concentration = 225 ng/L) and were associated with much higher total PFAS concentrations compared to Ravenna Creek. Although the total PFAS concentration at RC-MONTLAKE-GW was lower during the spring event, the sample was still dominated by PFOS.

Fluorotelomers 6:2 FTS, 7:3 FTCA, and 5:3 FTCA were observed at locations downstream of Montlake Boulevard. PFOSA was the most frequently detected sulfonamide (75% detection frequency; n=8). N-EtFOSAA and N-MeFOSAA were detected at RC-MONTLAKE-GW and RC-WAHKIAKUM-GW. Ethers were not detected in any samples Ravenna Creek subbasin samples.

Two shallow monitoring wells were sampled within the Ravenna Creek subbasin during the spring event (UW-MW1 and UW-MW2). Total PFAS concentrations were 793 (UW-MW1) and 41.7 ng/L (UW-MW2). Both samples were primarily composed of PFAAs. UW-MW1 was dominated by C4 through C9 PFCAs and had the highest PFOA concentration detected (112 ng/L) of any of the Phase 1 or Phase 2 water samples. The next highest PFOA concentration of 29.3 ng/L was observed in groundwater samples in the lower Cedar River. UW-MW2 had a similar composition but much lower concentrations. Fluorotelomer 6:2 FTS was observed at a low concentration in both wells (UW-MW1=0.943; UW-MW2=1.27 ng/L).

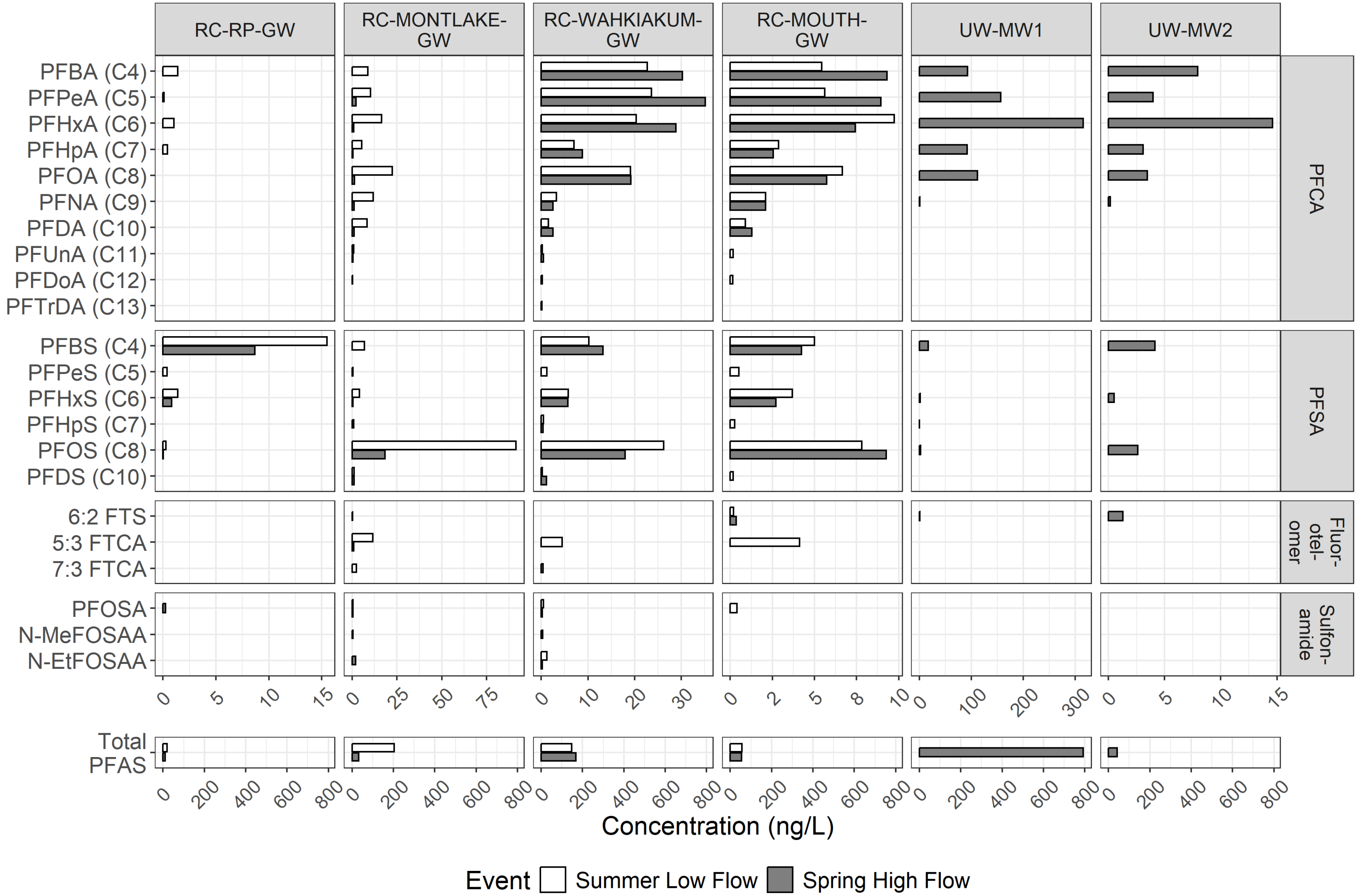


Figure 39. Analyte and total PFAS concentrations (ng/L) in Phase 2 Ravenna Creek subbasin groundwater samples, arranged upstream to downstream.

UW-MW1 and UW-MW2 were only sampled during spring high flow. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Lake Shoreline Groundwater Sampling

Madrona Beach

Total PFAS concentrations in groundwater, surface water, and springs in the Madrona area ranged from 1.02 to 95.3 ng/L. The Madrona Beach shoreline groundwater location (MADRONA), which had the highest total PFAS concentration (105 ng/L) from Phase 1, was revisited. Interestingly, Phase 2 MADRONA groundwater samples had fewer detections and much lower total PFAS concentrations compared to Phase 1 groundwater samples at this site (Figure 40). A surface water sample (MADRONA-SW) was also collected at this location during the spring event and the total PFAS concentration was 21.9 ng/L.

Upland of the Madrona shoreline sample, surface water samples were collected from springs (MADRONA-SP) emanating from the hillside that form Madrona Creek. Samples were collected from as close to the headwaters of Madrona Creek as possible (Appendix D-Figure D8). Total PFAS concentrations were 49.3 and 56.1 ng/L (summer and spring events respectively).

Samples were collected along the shoreline approximately 4200 ft to the south of the Madrona site (KING-ST) and from upland springs (FRINK-SP). The springs discharge from the hillside above the King Street shoreline location, forming Frink Creek. Samples were collected as close to the headwaters of Frink Creek as possible. Total PFAS concentrations ranged from 8.05 to 10.9 ng/L in the southern shoreline groundwater and spring samples. Samples were also collected along the shoreline approximately 4800 ft north of the Madrona site (HARRISON). Higher total PFAS concentrations of 82.5 and 95.3 ng/L (summer and spring events respectively) were observed at the Harrison Street location.

PFAS composition was primarily PFAAs, with variation of the dominant compounds for each area (FRINK-SP/KING ST, MADRONA/MADRONA-SP, and HARRISON) (Figure 40). The Phase 1 composition of the shoreline groundwater sample at Madrona Beach (MADRONA-P1) closely resembled the samples collected from Madrona springs (MADRONA-SP) (Figure 40). The composition of the MADRONA-P1 and MADRONA-SP samples were dominated by short chain PFCAs. However, PFOA and PFOS were more dominant in the Phase 1 MADRONA sample than the MADRONA-SP samples. The shoreline groundwater sample collected at KING-ST and the sample collected from Frink springs (FRINK-SP) were similar in composition. Both locations were primarily composed of C4 through C8 PFCAs and C4, C6, and C8 PFSAs. To the north, at the Harrison location, PFCA C6 is the dominant compound.

Fluorotelomers 5:3 FTCA and 6:2 FTS were detected during the spring event at low concentrations (<0.600 ng/L) at MADRONA-SW (Lake Washington surface water). PFOSA was the only sulfonamide detected and was detected at more sample locations during the spring (5/7 sites; ~71% detection frequency) than summer (3/7 sites; ~43% detection frequency). Ethers were not detected in any of the Madrona area samples.

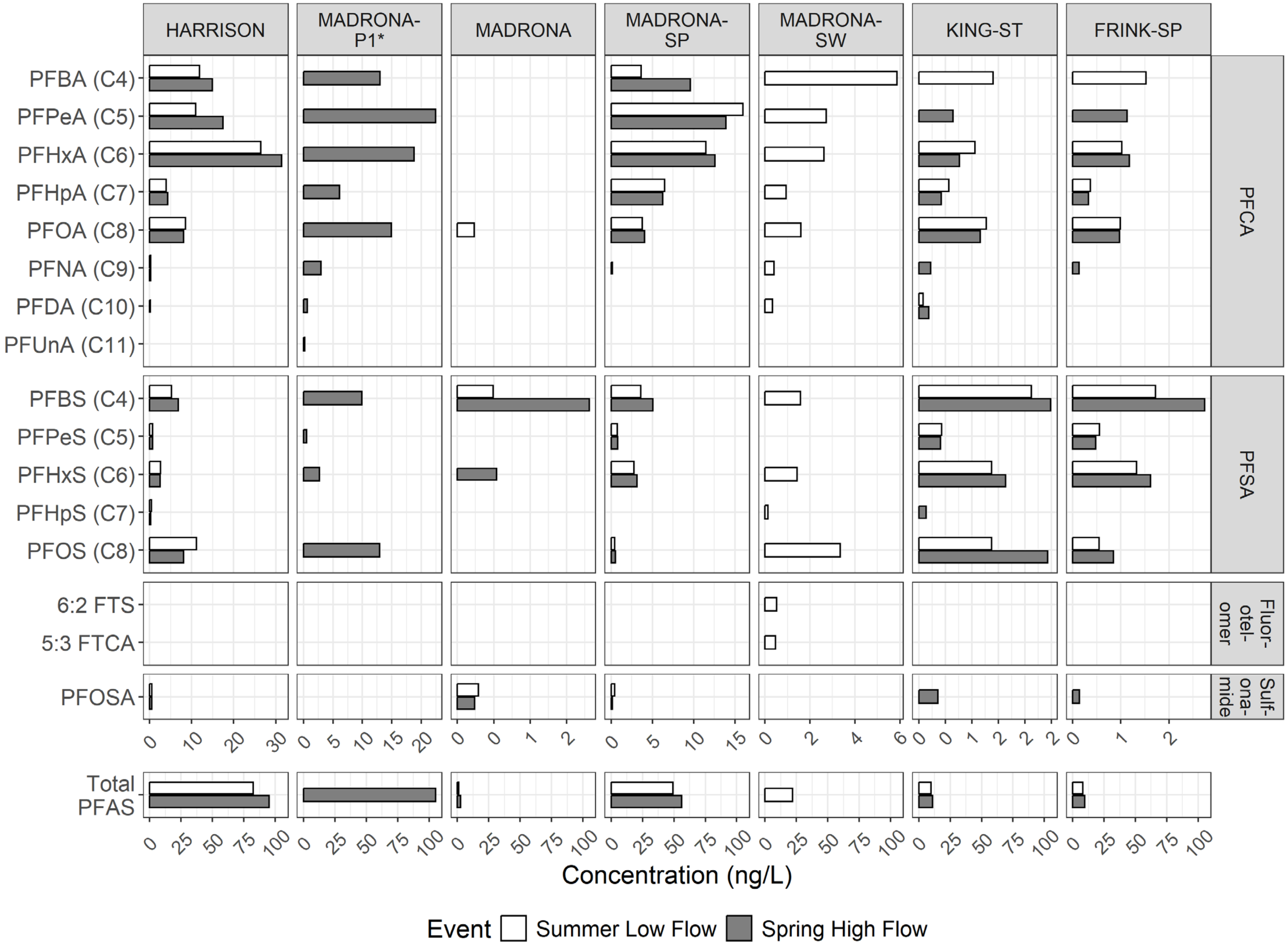


Figure 40. Analyte and total PFAS concentrations (ng/L) in Phase 1 and 2 Madrona area surface water, groundwater, and springs samples.

* MADORNA-P1 shows Phase 1 results for the MADRONA site. MADRONA-P1 was only sampled during the spring high flow event. MADRONA-SW was only sampled during the Phase 2 summer low flow event. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Wetmore Slough

The highest Phase 1 PFOS concentration (14.1 ng/L) was detected downgradient of the former Genesee landfill (Figure 41). During Phase 2, an attempt was made to resample the shoreline groundwater location, Lake Washington surface water, and areas to the north and south. The goal was to better understand whether the elevated PFOS concentration was localized or was observed up and down shore from this location, outside of the area that may be influenced by the landfill.

Figure 41. Map of Phase 1 and 2 surface water sampling sites in the Wetmore Slough area along the southwestern side of the Lake Washington shoreline.
Approximate outline of the Genesee Landfill in red.

We were unable to insert the PushPoint sampler deep enough to collect Phase 2 groundwater samples from the Phase 1 location. However, Phase 2 summer and spring groundwater samples (WETMORE-PH2-S and WETMORE-PH2-N respectively) were collected as close as possible to the Phase 1 location and were within 100 ft of the original site (Figure 41).

Phase 2 total PFAS concentrations in Wetmore slough area (groundwater, surface water, and springs) was 23.8-94.5 ng/L. The highest concentration was observed in a groundwater sample collected during the summer event (WETMORE-PH2-S), when an area of seasonal discharge created a ponded area that was upland of the shore of Lake Washington (Figure 42). The sample was primarily composed of PFAAs, with PFOS (32.5 ng/L) and PFOA (14.3 ng/L) as the dominant analytes. Sulfonamides N-EtFOSAA (0.540 ng/L) and N-MeFOSE (8.55 ng/L) were also observed at this location.

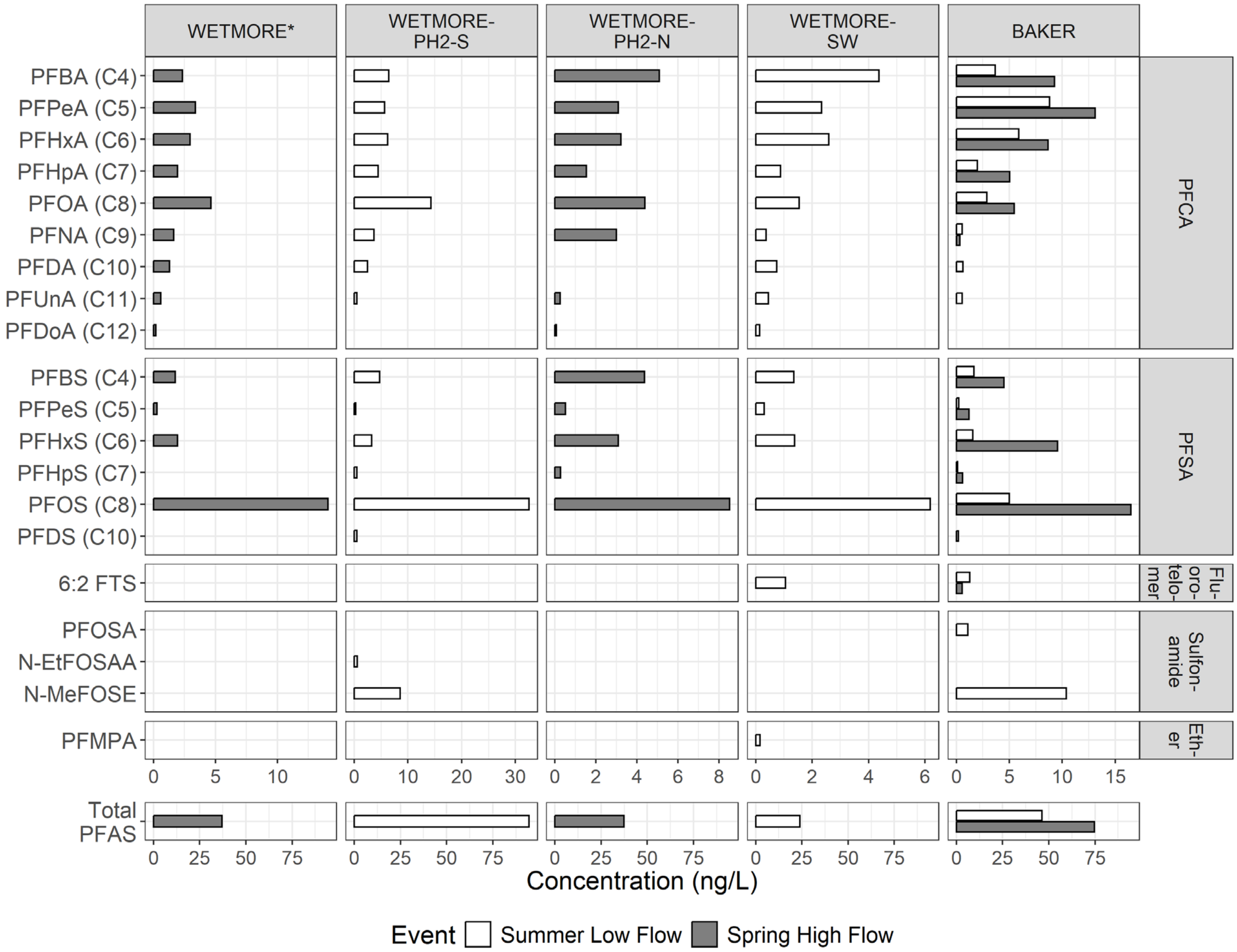


Figure 42. Analyte and total PFAS concentrations (ng/L) in Phase 1 and 2 groundwater and surface water samples collected in the Wetmore Slough area.

* Phase 1 sample. WETMORE-SW = surface water sample.
WETMORE and WETMORE-PH-2 were only sampled during the spring high flow event.
WETMORE-PH2-S and WETMORE-SW were only sampled during the Phase 2 summer low flow event.
PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid. Note different concentration scales. Non-detected analytes are not shown

A lower total PFAS concentration of 37.3 ng/L was observed during the spring event at WETMORE-PH2-N, which was similar to the total PFAS concentration observed during Phase 1 sampling (37.0 ng/L). This location was closer to the shoreline than WETMORE-PH2-S, like the Phase 1 location (WETMORE), and was composed of only of PFAAs. WETMORE-PH2-N was also dominated by PFOS, at a concentration of 8.51 ng/L.

A surface water sample was collected from Lake Washington at the shoreline (WETMORE-SW) and had the lowest total PFAS concentration (23.8 ng/L). Like the other Wetmore Slough area samples, this surface water sample was primarily composed of PFAAs, with PFOS (6.19 ng/L) as the dominant analyte. Fluorotelomer 6:2 FTS and ether PFMPA were also observed in this sample.

Approximately 1-mile north at BAKER, PFOS was also the dominant analyte at a concentration of 16.5 ng/L, during the spring event. A high total PFAS concentration of 74.7 ng/L was detected at this location. In addition to PFAAs, low concentrations (<1.30 ng/L) of fluorotelomer 6:2 FTS were observed during both sampling events and sulfonamides, N-MeFOSE and PFOSA, were detected during the summer event.

Suspended Sediments

Total PFAS concentrations were 0.159 – 10.9 ng/g among six suspended sediment samples collected from the Cedar River (CR-RVP), Juanita Creek (JC-NE129TH), and Thornton Creek (TC-BELOW-MBP) during summer low and spring high flow (Table 5). Summer low flow total PFAS concentrations in suspended sediments were 1.6 – 22 times higher than spring high flow suspended sediment concentrations (Figure 43).

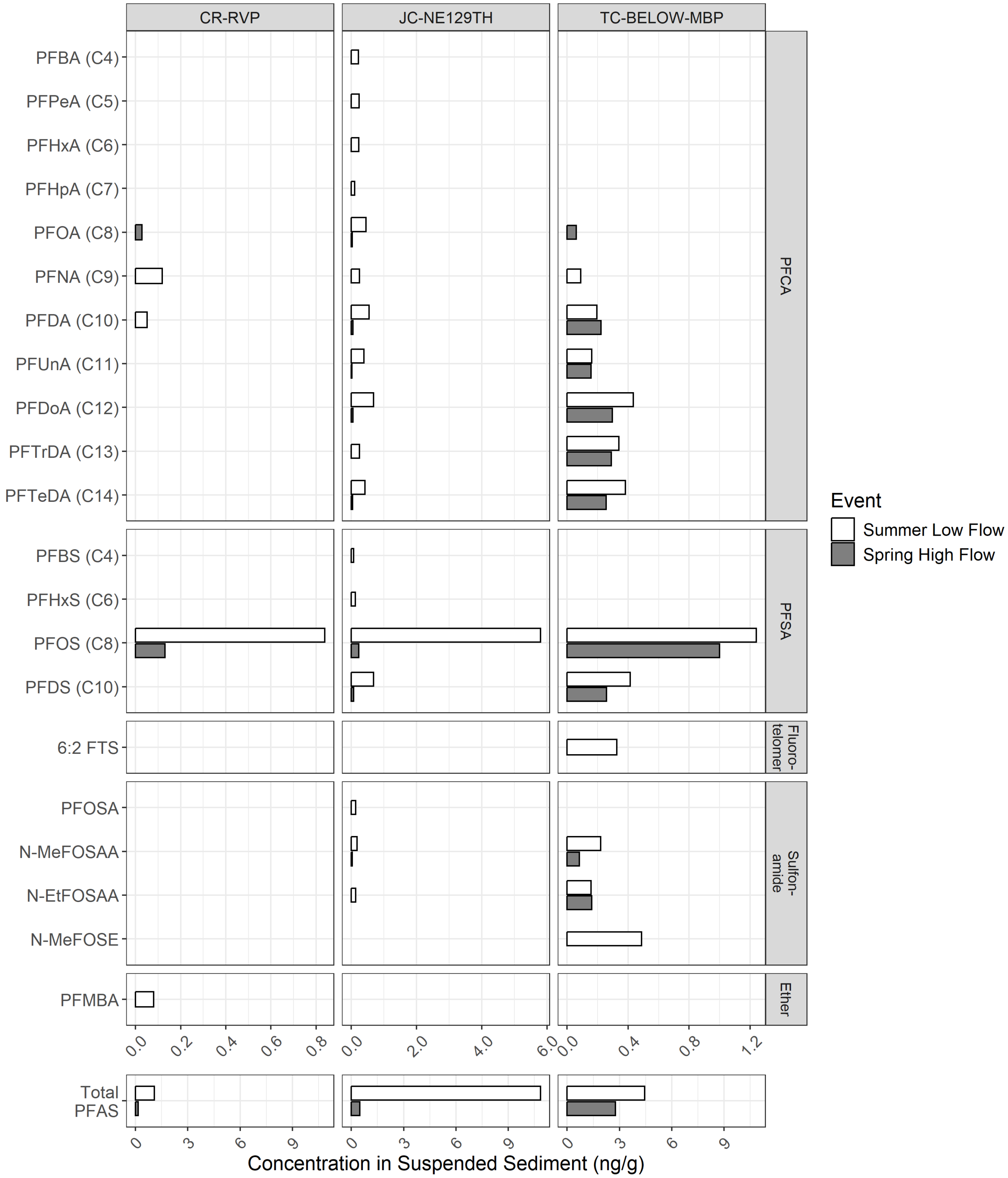


Figure 43. Analyte and total PFAS concentrations (ng/g) in suspended sediments samples collected from the Cedar River (CR-RVP), Juanita Creek (JC-NE129TH), and Thornton Creek (TC-BELOW-MBP) during Phase 2.

Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

PFOS was the dominant analyte in all samples, comprising 28% – 82% of the total PFAS (Figure 43). Long-chain PFAAs were prominent in the samples from Thornton and Juanita Creeks. Detections and concentrations of short-chain PFAAs were relatively low in all samples. Sulfonamides (N-MeFOSE, N-EtFOSAA, N-MeFOSA) were prominent in the Thornton Creek sample, and were detected at low concentrations in the Juanita Creek sample.

Sediment grain size was primarily clay and silt during summer low flow (89% of the sample weight), and fine to coarse sands during spring high flow (76% of the sample weight) (Appendix G). A significant negative correlation was found between total PFAS concentration and the amount of very coarse sand in the sample (ρ=-0.89, p=0.033). No other significant correlations between total PFAS concentration and sediment grain size were found (-0.83≤ρ≤0.6, p>0.05) (Figure 44). No significant correlations were found between TOC concentration and PFAS analytes (-0.1≤ρ≤0.9, p>0.05).

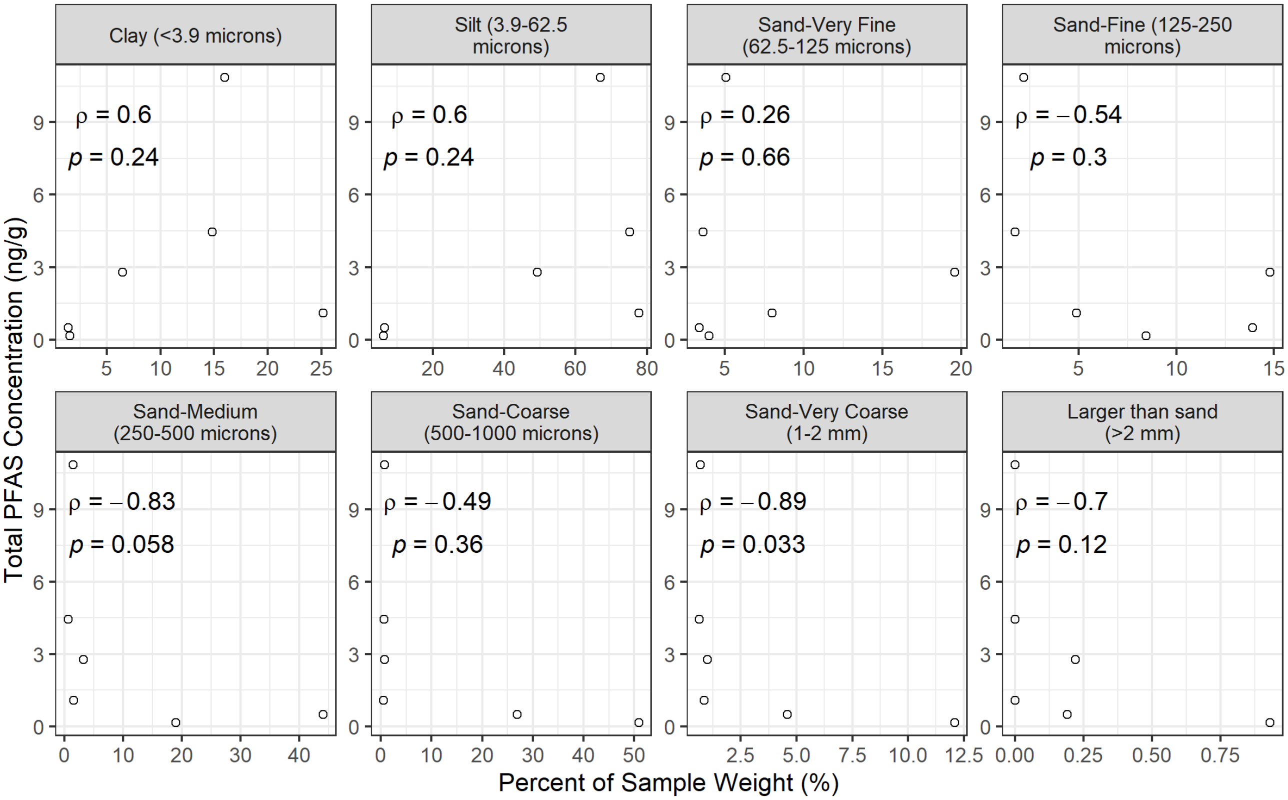


Figure 44. Correlations between total PFAS concentration (ng/g) and percentage of each grain size class in suspended sediment samples collected from the Cedar River, Juanita Creek, and Thornton Creek subbasins during Phase 2.

Spearman correlation (ρ) and significance values (p) are shown. Grain size classes are arranged small to large from top left to bottom right. PFAS = per- and polyfluoroalkyl substances.

Surface Runoff & Road Dust

Total PFAS concentrations among 13 surface runoff samples collected from the three focus subbasins and the 520 bridge ranged from non-detect to 11.0 ng/L. The highest total PFAS concentrations were observed in the 520 bridge runoff sample and the industrial runoff sample collected from the Cedar River subbasin (Figure 45). The lowest concentration was found in the open space land use sample from the Cedar River subbasin. Among the four residential runoff samples, total PFAS concentrations in the two high-density residential runoff samples (JC-RES2 and TC-RES) were more than two times higher than in the low-density residential runoff samples (CR-RES and JC-RES1).

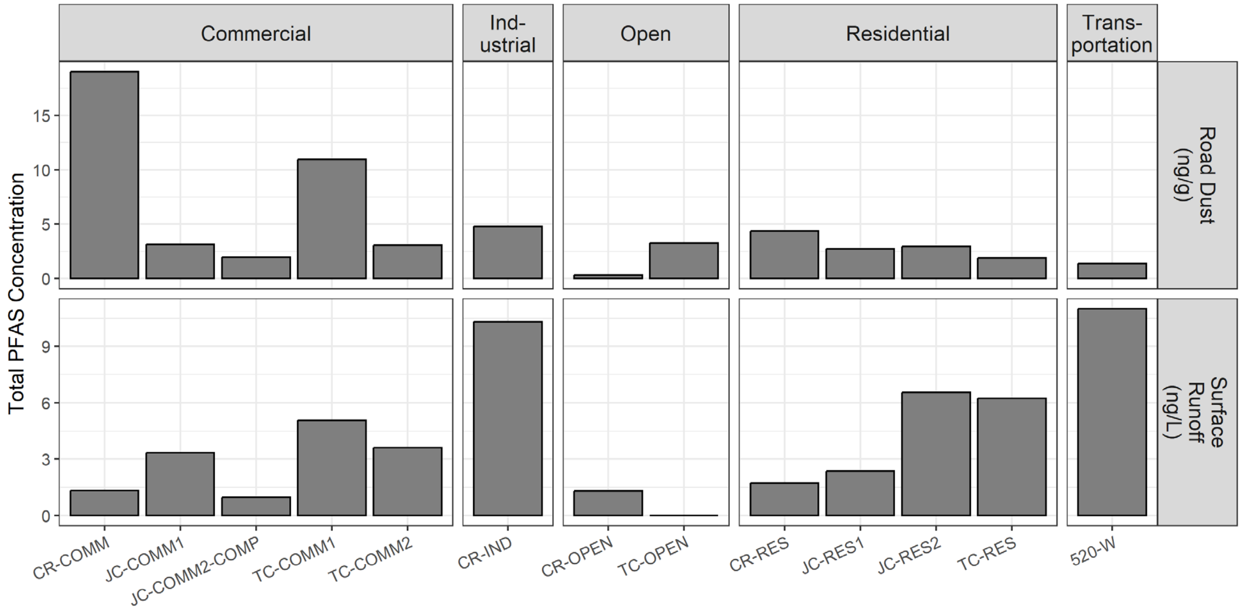


Figure 45. Concentrations of total per- and polyfluoroalkyl substances (PFAS) in surface runoff (ng/L) and road dust samples (ng/g) collected from different land use types in the Cedar River, Juanita Creek, and Thornton Creek subbasins, and the 520 bridge during Phase 2 Sampling.

The most frequently detected analytes in the surface runoff samples were C4 and C6 – C10 PFCAs, and PFOS (Figure 46). PFPeA was not detected in any surface runoff samples, although it was frequently detected in stormwater and all other water sample types.

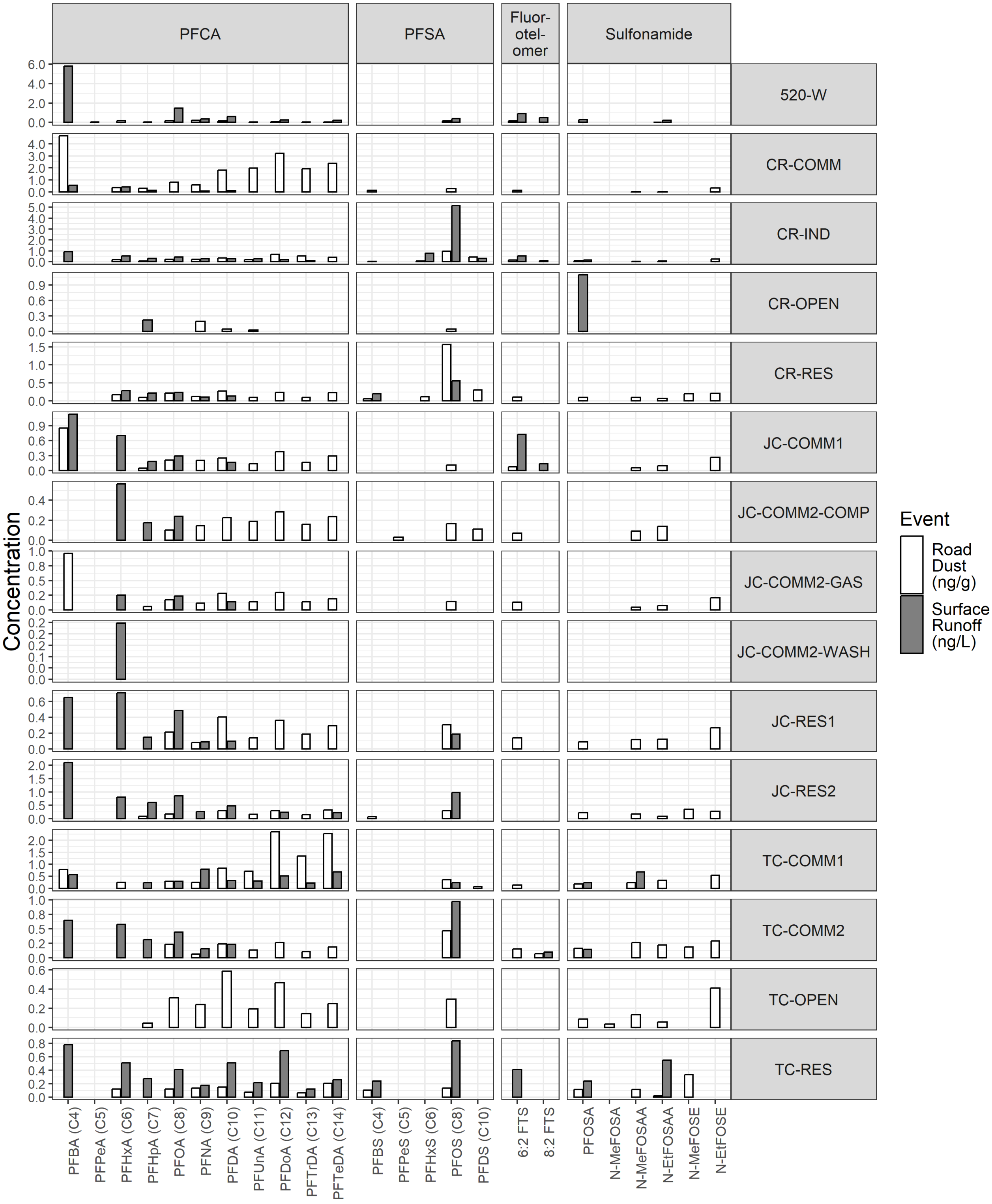


Figure 46. PFAS analyte concentrations in Phase 2 surface runoff (ng/L) and road dust (ng/g) samples collected from different land use types in the Cedar River, Juanita Creek, and Thornton Creek subbasins, and the 520 bridge during Phase 2.

Note different concentration scales. Non-detected analytes are not shown.

PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid.

PFSA = perfluoroalkane sulfonic acid.

PCA results for surface runoff samples showed that PFAS composition was influenced by subbasin (Figure 47). Samples from Juanita Creek subbasin were most influenced by shorter-chain (C4, C6, C8) PFCAs. Samples from Thornton Creek subbasin were most influenced by PFOS, long-chain (C11 – C14) PFCAs, and sulfonamides. Within the Cedar River subbasin, PFAS composition was influenced by land use type: The open space runoff sample was influenced most by PFOSA; the commercial runoff sample by PFBA; and the industrial and residential samples by PFOS.

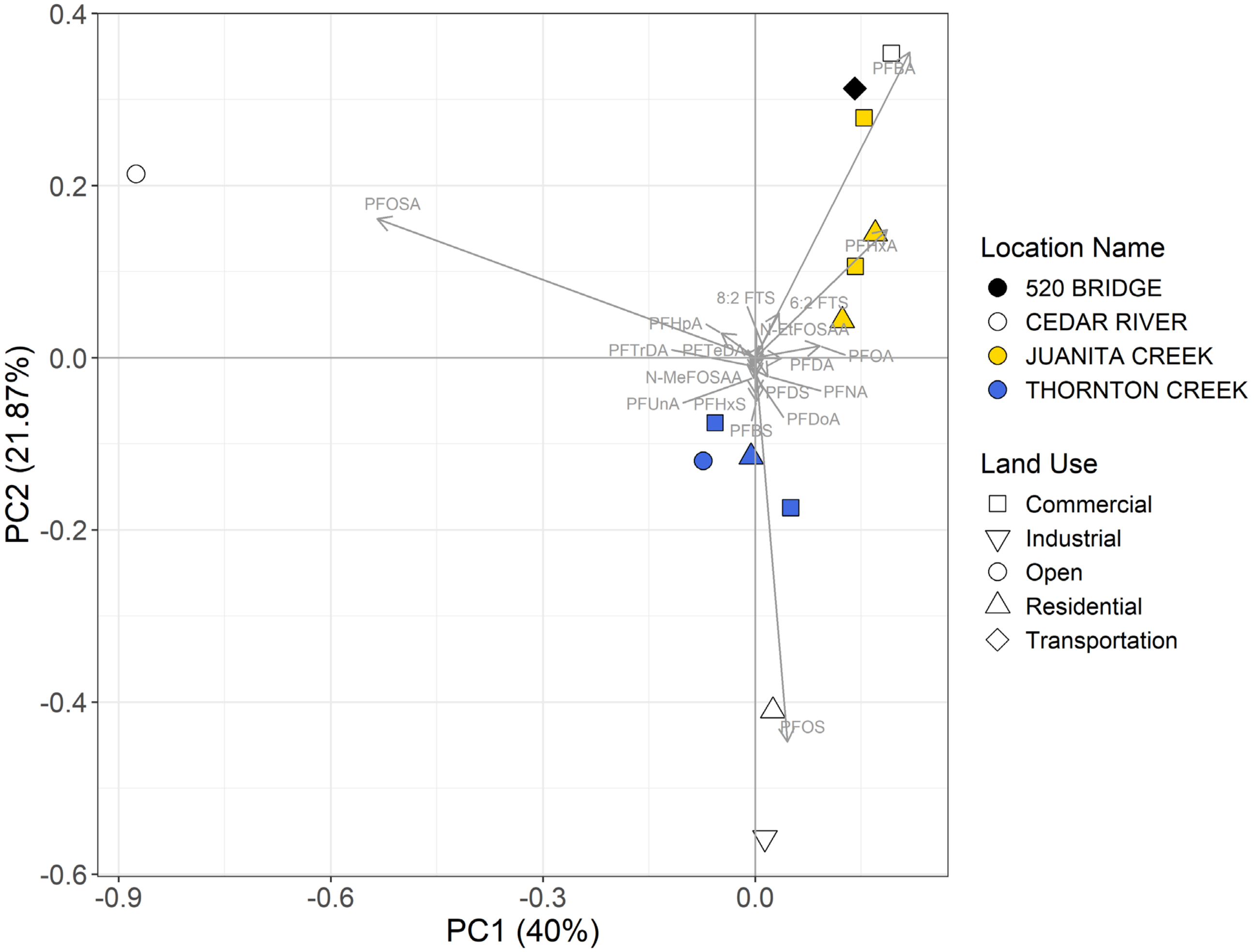


Figure 47. Principal components analysis scores and loadings plot for surface runoff PFAS samples by land use type and tributary subbasin.
The first two principal components, explaining 62% of the variance, are plotted.
Symbols (scores) represent samples. Vectors (loadings) represent PFAS analytes.
PFAS = per- and polyfluoroalkyl substances.

Total PFAS concentrations among the 13 road dust samples ranged from 0.305 to 19.0 ng/g, with the highest concentration found in the commercial sample from the Cedar River subbasin (Figure 45). Although the industrial and 520 bridge samples had relatively high surface runoff concentrations, they both had relatively low road dust concentrations. The lowest road dust concentration was found in the open space land use sample from Cedar River subbasin.

The most frequently detected analytes in road dust samples were C8 – C14 PFCAs, PFOS, N-EtFOSAA, and N-MeFOSAA. PFPeA was detected in one road dust sample (Figure 46).

PCA results for road dust samples showed that land use type influenced PFAS composition, particularly between three commercial samples and all other samples (Figure 48). The three commercial road dust samples (CR-COMM, JC-COMM1, and TC-COMM1) were influenced more by long-chain (C11 – C14) PFCAs and PFBA.

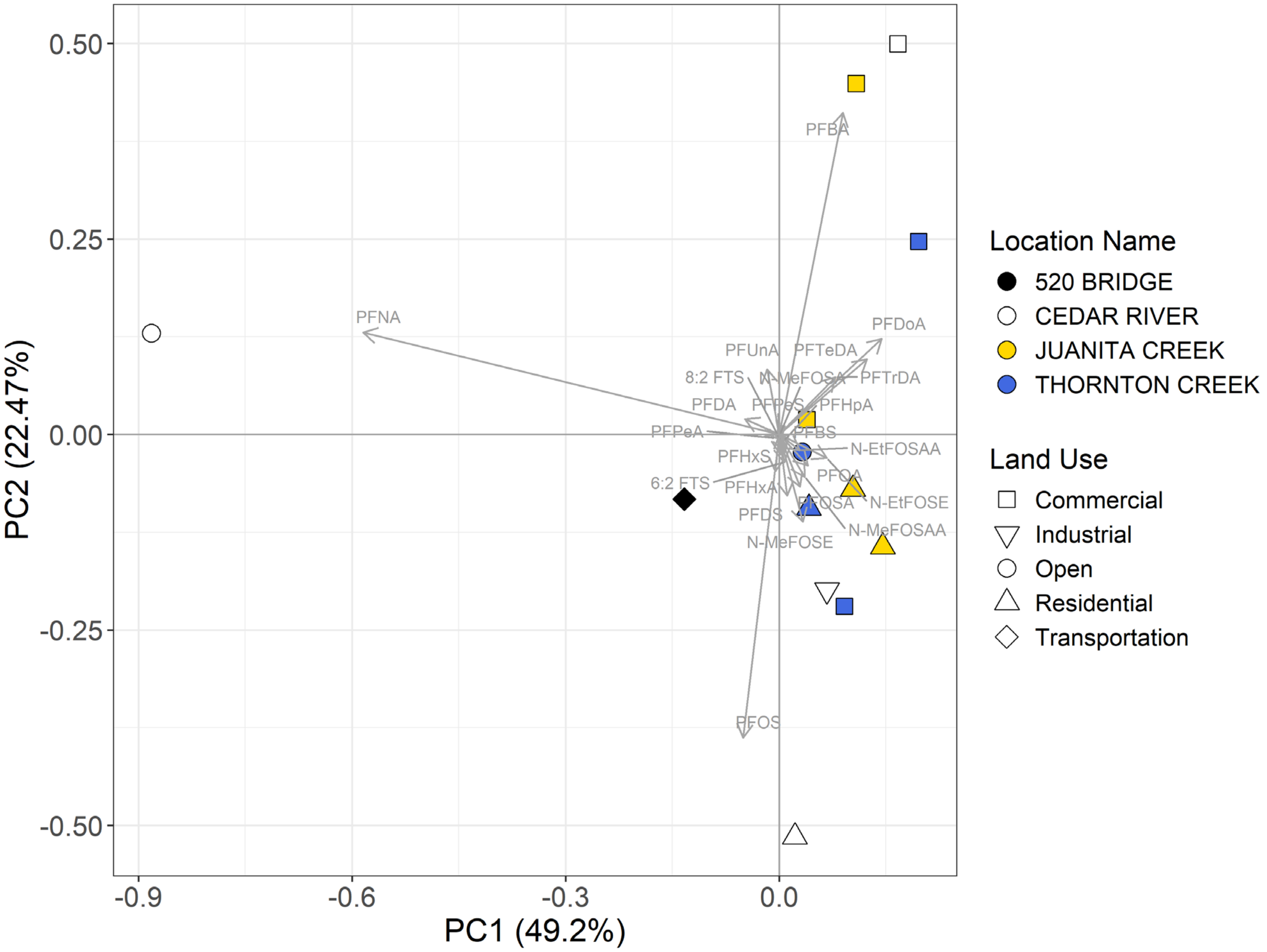


Figure 48. Principal components analysis scores and loadings plot for road dust PFAS samples by land use type and tributary subbasin.

The first two principal components, explaining 72% of the variance, are plotted. Symbols (scores) represent samples. Vectors (loadings) represent analytes. PFAS = per- and polyfluoroalkyl substances.

The mean concentrations of C10 – C14 PFCAs in road dust from commercial land use areas were overall higher than in open space and residential land use areas, and higher than in lake and tributary sediments (Figure 49).

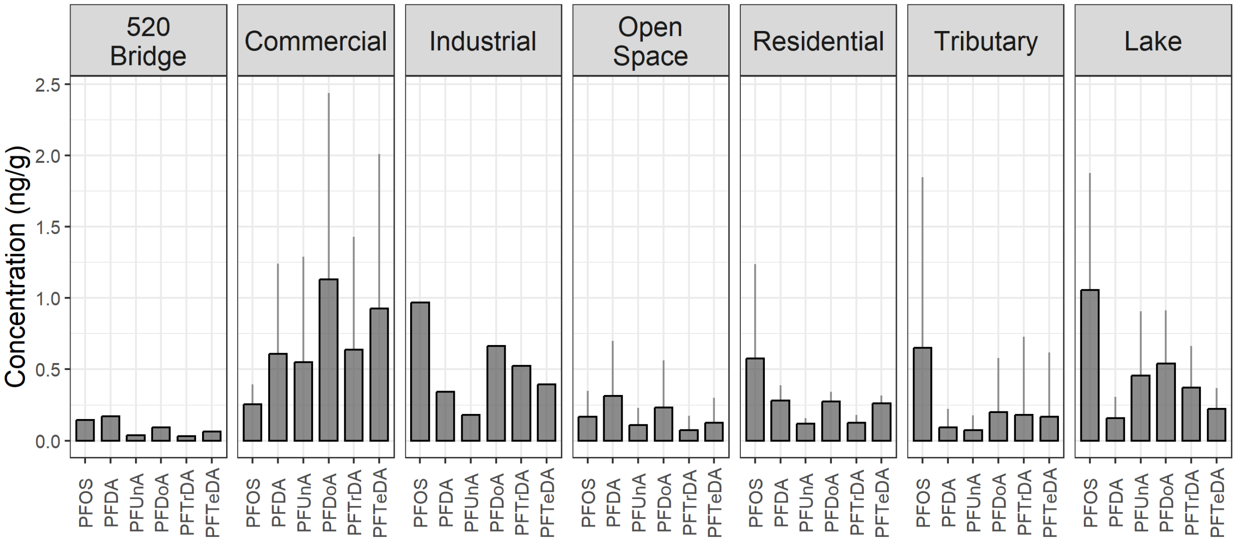


Figure 49. Mean (± standard error) concentrations (ng/g) of PFOS and long-chain PFCAs in road dust samples collected from the 520 Bridge (N = 1), commercial (N = 5), industrial (N = 1), open space (N = 2), and residential (N = 4) land use types in the Phase 2 focus subbasins, and in Phase 1 tributary and lake sediments (N = 16 and N = 27, respectively).
PFCA = perfluoroalkyl carboxylic acid; PFOS = Perfluorooctane sulfonic acid

In surface runoff samples, TOC and DOC were positively correlated with C4, C8, and C10 PFCAs, 6:2 FTS, and 8:2 FTS ($\rho \geq 0.3$, $p<0.05$). TSS was not significantly correlated with any PFAS analytes except for PFOSA ($\rho=0.5$, $p<0.05$). In road dust samples, TOC was not significantly correlated with any PFAS analytes ($-0.3 \leq \rho \leq 0.6$, $p>0.05$).

Bulk Atmospheric Deposition

Total PFAS concentrations in bulk atmospheric deposition samples ranged from non-detect to 7.04 ng/L.

Total PFAS concentrations at the North Bend site were low during all five deployments. Total PFAS concentrations at the Seattle site were 2.8 – 6.3 times higher than at the North Bend site during the first two deployments (September – November 2022) but were more comparable during the last three deployments (November 2022 – January 2023) (Figure 50).

Blank-corrected total PFAS results were comparable with uncorrected sample results (approximately 14% RPD on average). An exception was the Seattle duplicate sample during the December 2–16 deployment, in which the blank-corrected total PFAS concentration was about two times higher than the uncorrected result. PFAS in the first sample of the duplicate pair was non-detect (Figure 50).

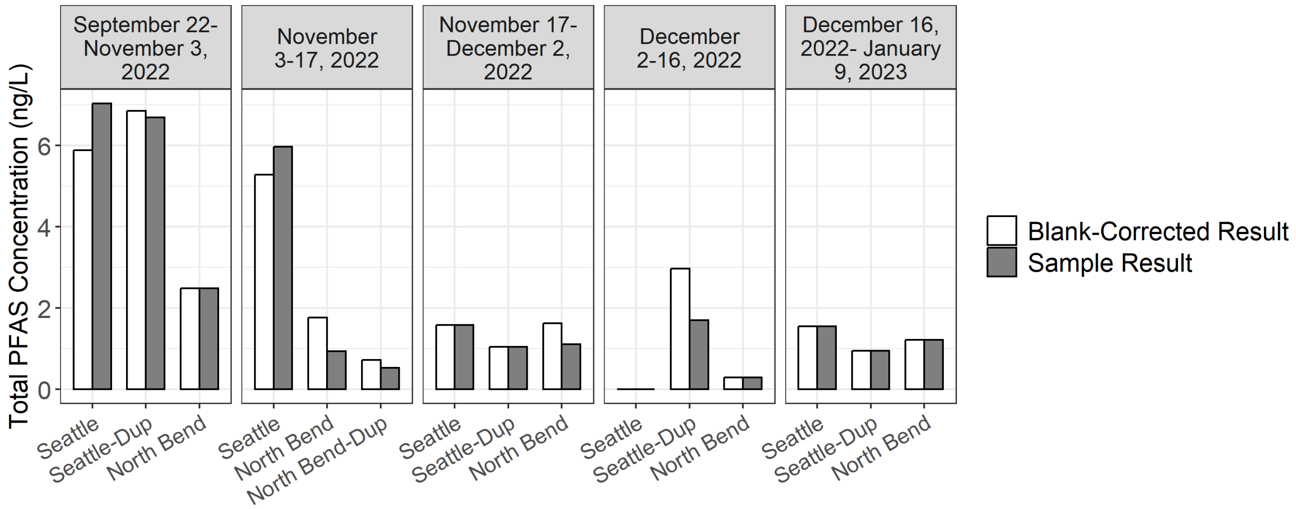


Figure 50. Concentrations (ng/L) of total per- and polyfluoroalkyl substances (PFAS) in Phase 2 bulk atmospheric deposition samples collected during five deployments in Seattle and North Bend.

Among Phase 1 and 2 bulk atmospheric deposition samples collected from the Seattle site, total PFAS concentration was positively correlated with the length of the dry period during deployment (R=0.86, p=0.0031; Figure 51). Total PFAS concentration was not correlated with the amount of precipitation received at the location during deployment.

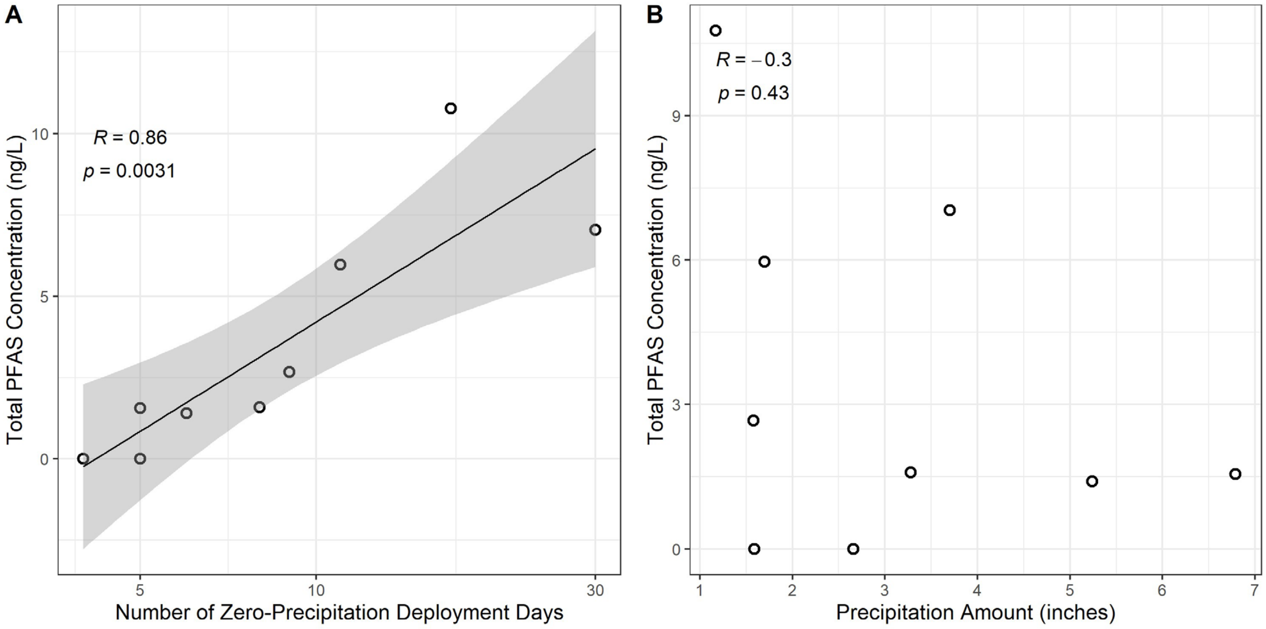


Figure 51. Correlation between total PFAS concentration (ng/L) and (A) number of zero-precipitation days during deployment at the Seattle site, and (B) amount of precipitation (inches) received on-site.

The Pearson correlation coefficient (R), significance value (p), and 95% confidence interval (gray shade) are shown. Log10 transformation was applied to the x-axis scale in Figure 51A. PFAS = per- and polyfluoroalkyl substances.

Samples were comprised entirely of PFCAs, except for the presence of 6:2 FTS in four samples from the Seattle site. Similar to surface runoff samples, PFPeA was not detected in any Phase 2 bulk atmospheric deposition samples.

PCA results showed that samples from the North Bend site were influenced more by short-chain PFCAs (PFBA and PFHxA), while samples from the Seattle site were influenced more by long-chain PFCAs (primarily PFOA) and 6:2 FTS (Figure 52). PCA results also showed that the longer-chain PFCAs tended to have more influence on samples during the earlier deployments (September–November) than the later deployments (November–January).

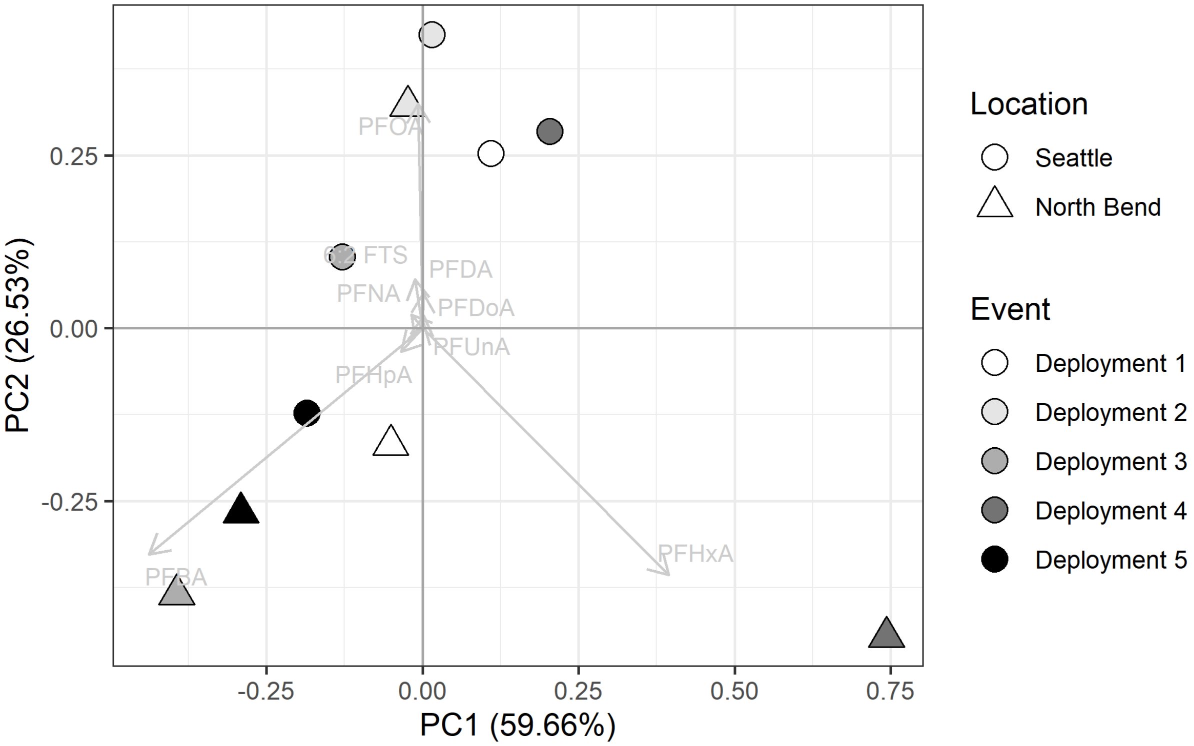


Figure 52. Principal components analysis scores and loadings plot for bulk atmospheric deposition PFAS samples by location and sampling event.

The first two principal components, explaining 86% of the variance, are plotted. Symbols (scores) represent samples. Vectors (loadings) represent PFAS analytes. PFAS = per- and polyfluoroalkyl substance.

Opportunistic Sampling

Airport Stormwater

Total PFAS concentration in the Phase 2 stormwater sample from the Renton Airport (REN-CMU) was 98.5 ng/L. Similar to the sample collected during Phase 1, the sample was composed primarily of C4 – C6 PFCAs and 6:2 FTS (Figure 53). Different than the Phase 1 sample, the Phase 2 sample had a higher proportion of PFHxS, and a total PFAS concentration more than two times higher than the Phase 1 sample.

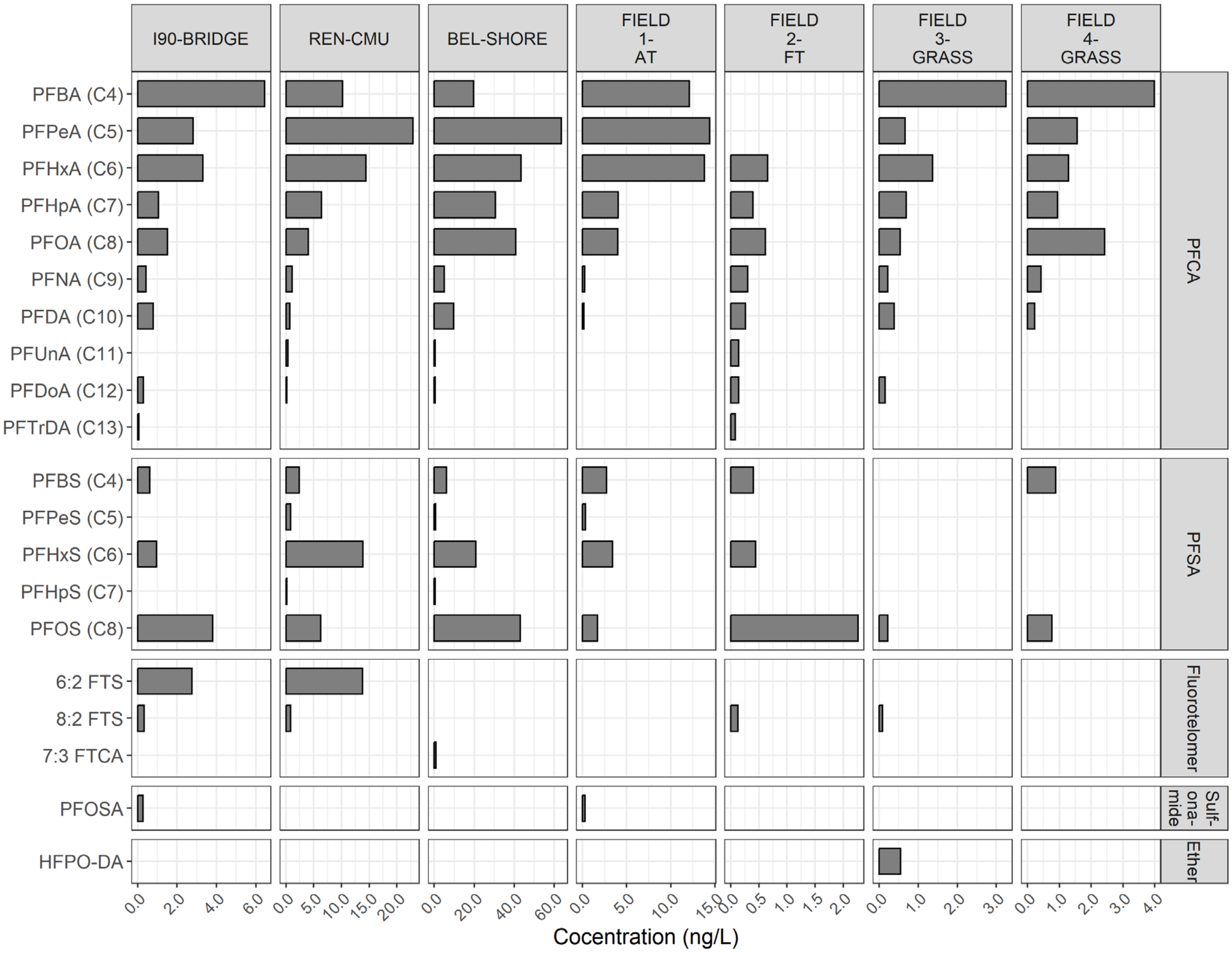


Figure 53. PFAS analyte concentrations (ng/L) in Phase 2 opportunistic stormwater samples. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Residential Stormwater

Total PFAS concentration in the stormwater sample from the small residential area (BEL-SHORE) was 286 ng/L, more than two times higher than the Phase 1 sample. PFAS composition in the Phase 2 sample was nearly identical to the Phase 1 sample and composed primarily of C5 – C8 PFCAs and PFOS (Figure 53).

I-90 Bridge Stormwater

Total PFAS concentration in the stormwater sample from the I-90 bridge was 25.5 ng/L. The sample was composed predominantly of C4 – C6 PFCAs, PFOS, and 6:2 FTS, with lower concentrations of C7 – C10 PFCAs, and C4 and C6 PFSAs (Figure 53).

Stormwater from Turf and Grass Athletic Fields

Among the four athletic fields sampled, the AstroTurf ® field (FIELD 1-AT) had the highest total PFAS concentration, more than four times higher than the other three fields, and more than five times higher than the control sample (Table 6). Total PFAS concentration in the control sample for the grass field (FIELD 3-GRASS-C) was comparable to the FIELD 3-GRASS stormwater sample.

Table 6. Concentrations of total per- and polyfluoroalkyl substances (PFAS) in stormwater samples (ng/L) collected from four athletic fields and two control samples.

Location ID (Field Type)	Total PFAS Concentration (ng/L)
FIELD 1-AT (AstroTurf)	57.1
FIELD 2-FT (FieldTurf)	5.90
FIELD 3-GRASS (Grass)	8.10
FIELD 4-GRASS (Grass)	12.5
Control Sample (FIELD 1-AT-C)	8.30
Control Sample (FIELD 3-GRASS-C)	11.9

The AstroTurf ® stormwater sample was composed predominantly of C4 – C6 PFCAs. PFOS and PFOA were detected but were not dominant in the sample (1.71 and 3.98 ng/L, respectively).

Stormwater from FIELD 3-GRASS was composed predominantly of PFCAs (40% PFBA, and lower relative concentrations of C5 – C12 PFCAs). PFOS was also detected in the sample but was not dominant (2.7% of the total PFAS). The control sample for field 3 (FIELD 3-GRASS-C) had similar PFAS composition as field 3, except the control sample had slightly higher concentrations of long-chain (C11 – C14) PFCAs. In both the Field 3 and Field 3 control samples, HFPO-DA was detected at 0.553 and 0.866 ng/L, respectively. This represented two of three detections of HFPO-DA among all Phase 1 and 2 samples collected.

Different than the other three fields, FieldTurf ® (FIELD 2-FT) stormwater was composed mostly of PFOS, with lower concentrations of C6 – C13 PFCAs and C4 and C6 PFSAs. However, the sample also had the lowest total PFAS concentration.

Stormwater from FIELD 4-GRASS was composed mostly of C4 – C10 PFCAs, with PFBA being the dominant analyte, followed by PFOA. PFOS and PFBS were also detected in the sample, but at lower concentrations (<1 ng/L).

Horse Creek

Surface water samples were collected in the upstream and downstream portion of Horse Creek to assess potential effects from historic AFFF use during fire training and contaminated sites in the area. Total PFAS concentrations were 30.3 ng/L in the upstream sample (HC-1), and 33.0 ng/L in the downstream sample (HC-2). The composition of both samples was similar: Primarily C4 – C8 PFCAs, and C8 and C4 PFSAs. The only sulfonamide detected was PFOSA (1.37 ng/L) at the downstream site. Fluorotelomers (often associated with AFFF) and sulfonamides were not detected.

West Tributary

Two surface water and two groundwater samples were collected in the West Tributary to assess concentrations and compositions associated with historic AFFF use and an industrial area. Total PFAS in surface water were slightly higher at the northern location (WESTTRIBN) than the southern location (WESTTRIBS). The surface waters samples were predominantly composed of short chain PFCAs C4 through C6 (Figure 54).

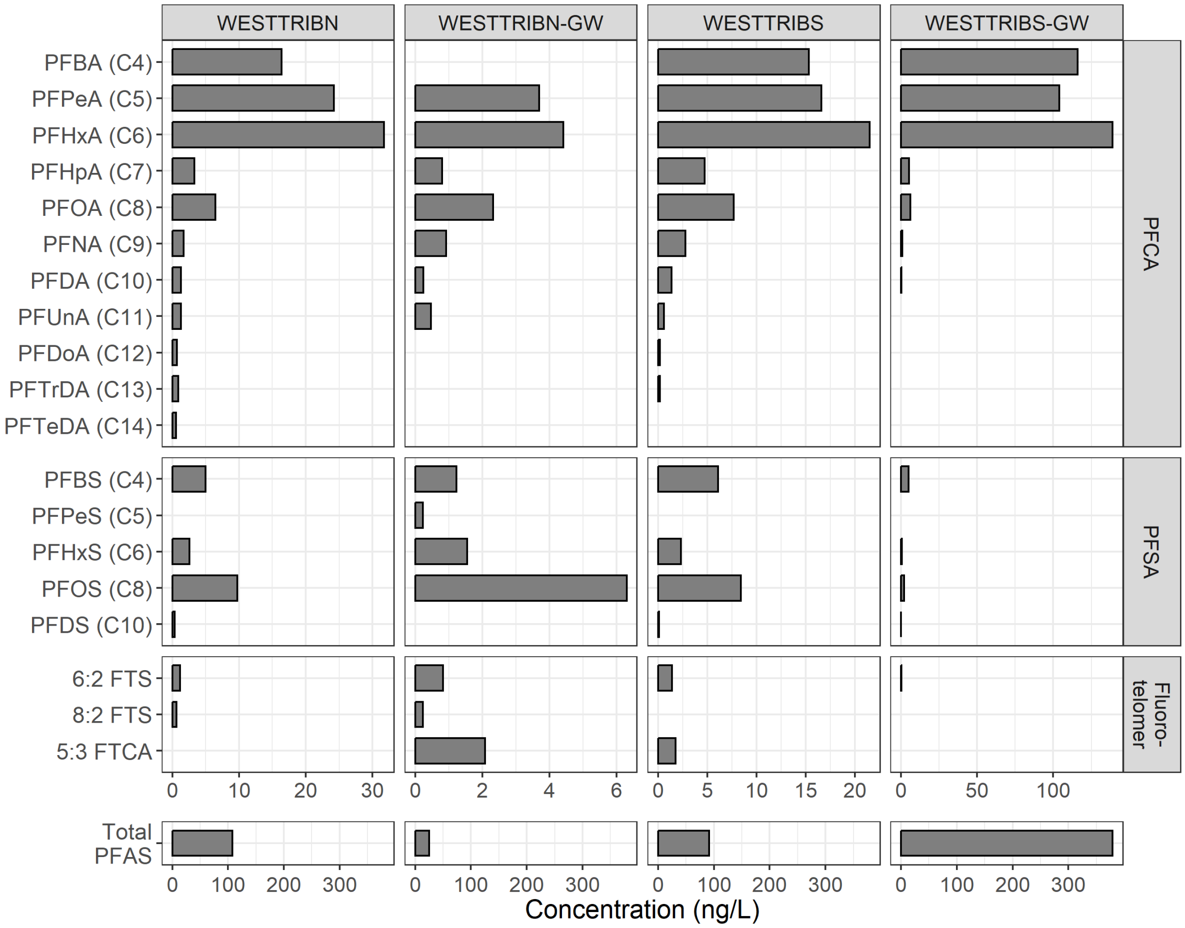


Figure 54. Analyte and total PFAS concentrations (ng/L) in Phase 2 groundwater and surface water samples from West Tributary.

WESTTRIBN and WESTRIBS represent surface water samples. WESTTRIBN-GW and WESTRBS-GW represent groundwater samples. Note different concentration scales. Non-detected analytes are not shown. PFAS = per- and polyfluoroalkyl substances. PFCA = perfluoroalkyl carboxylic acid. PFSA = perfluoroalkane sulfonic acid.

Total PFAS in groundwater were 25.3 and 379.2 ng/L. The higher concentration was detected at the southern location, WESTTRIBS-GW. More than 95% of the composition was short-chain PFCAs (C4 through C6). PFCAs C7 through C10, PFSA C4, C6, C8, and C10 and fluorotelomer 6:2 FTS were also detected. PFOS was the dominant analyte at the northern location. PFCAs C5 through C11 PFCAs and C4 through C6 and C8 PFSA, and fluorotelomers 6:2 FTS, 8:2 FTS, and 5:3 FTS were also detected at the northern location.

Both groundwater samples had high turbidity. Turbidity at the northern location was 33 FNU at the time the sample was collected and 17 FNU at the southern location. Ideally, groundwater samples are not collected until turbidity falls below 10 FNU. This may have influenced PFAS concentrations and composition at these locations.

Groundwater and surface water temperatures were higher at the southern location (11.5°C and 15.2°C respectively) than the northern location (10.0°C and 12.9°C). The lower surface water temperature may indicate greater groundwater flux at the northern location. In addition, other surface water parameters were closer to porewater parameters at this location and more typical of porewater (low pH, high specific conductance, negative ORP).

Biofilm & Macroinvertebrate Tissue

Among the six biofilm tissue samples collected from the Cedar River during Phase 2, three analytes were detected: PFOS (5 detections), PFBA (1 detection), and PFMBA (2 detections). One biofilm sample (CR-LANDSBURG) had zero detections of any analytes. Except for PFOS in one sample, detected result concentrations were low (J-qualified as estimated values). Apart from PFMBA in one sample, all detected results were qualified as non-detect due to method blank contamination. Due to low detections and method blank contamination, we did not further analyze biofilm data from Phase 2.

Among the six caddisfly larvae tissue samples collected, five analytes were detected: PFOS (5 detections), PFBA (5 detections), PFDA (1 detection), PFTeDA (1 detection), and 6:2 FTS (1 detection). Of the 13 detected results, five were qualified as non-detect due to method blank contamination, including PFOS in three samples, PFBA in one sample, and PFDA in one sample. The two non-qualified detections were PFOS in the downstream-most samples collected from the Cedar River (0.884 ng/g at CR-RVP and 0.790 ng/g CR-USGS-GAGE).

Discussion

PFAS in the Greater Lake Washington Watershed

During the 2020 – 2023 study period, PFAS detections were widespread in the Greater Lake Washington watershed: At least one compound was detected in 467 of 479 (97.5%) samples and 100% of all media types sampled (i.e., surface water, groundwater, stormwater discharge, bulk atmospheric deposition, surface runoff, road dust, and sediment). This is consistent with previous studies that also reported a high frequency of detections among similar media types (Mendez et al. 2022; Llewellyn et al. 2024). Overall, results from the present study, in conjunction with previous research, indicate the diffuse nature of PFAS sources and pathways in the watershed.

Despite being phased out of production and use since the early 2000s, PFOS was the most frequently detected analyte (425 of 479 samples, 88.7%) and had the highest median relative concentration (15.2% of the total PFAS) among all sample types collected in our study, except for bulk atmospheric deposition in which it was not detected. These results show that sources of PFOS to the lake are diffuse — they enter the lake via tributary inflows and their associated runoff, stormwater discharges, and groundwater discharges throughout the watershed. Once in the lake, PFOS accumulates in the sediments and in organisms that live in the lake. As is common with water pollution in urban environments, addressing PFOS contamination in fish from Lake Washington will partly involve continuing to address nonpoint pollution.

High levels of PFOS in some groundwater discharge samples indicated more localized inputs to the receiving surface waters, for example near the mouth of the Cedar River where PFOS concentrations were above 100 ng/L. Further assessment of these areas would help to identify and address sources.

PFAS concentrations in surface water samples collected from Lake Washington and its tributaries in this 2020–2023 study were relatively high compared to surface water samples collected in 2008 and 2016 from the lake and other large freshwater bodies in Washington (Figure 55; Furl and Meredith 2010; Mathieu and McCall 2017). At the high end of this range were waterbodies influenced by wastewater treatment plant effluents, and urban lakes not known to be directly influenced by wastewater treatment plant effluents, including Lake Washington.

Total PFAS concentrations in Lake Washington surface water samples (11.6 – 23.8 ng/L) fit within the ranges found in large watersheds throughout the U.S. For example, total PFAS concentrations (sum of 40 target compounds) ranged from non-detect to 29 ng/L in the San Francisco Bay (n=22; Mendez et al. 2022), 9.49 – 35.9 ng/L (sum of 12 target compounds) from two urban lakes in New York (n = 11; Kim & Kannan 2007), and 0.52 – 47 ng/L (sum of 41 compounds) in surface water samples from the Great Lakes (Table S17 in Xia et al. 2024).

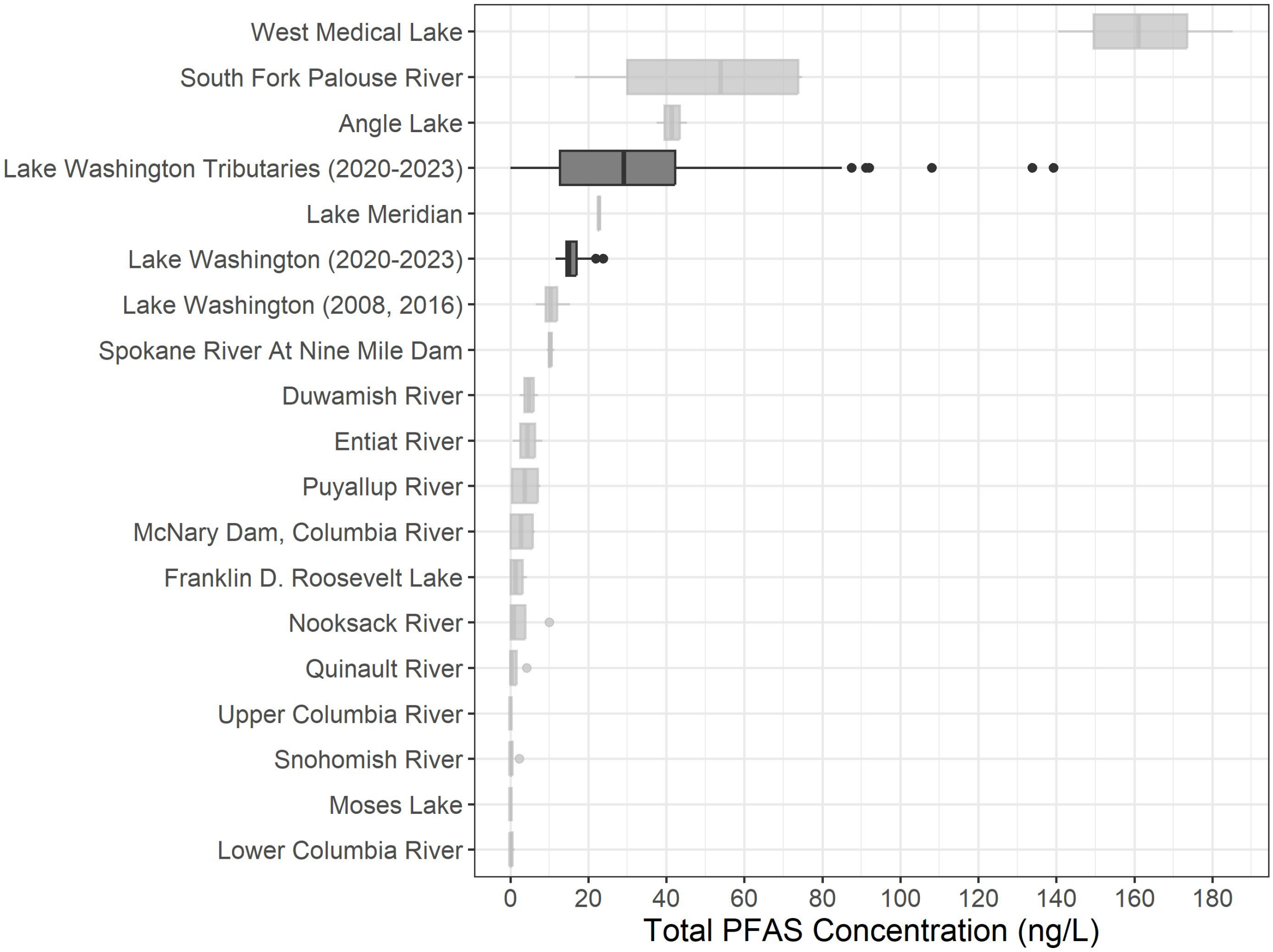


Figure 55. Box plots of total PFAS concentrations (ng/L) in surface water samples collected from Washington State water bodies.

Bolded box plots represent Lake Washington and tributary samples collected during this study. Light box plots represent data from Furl and Meredith (2010) and Mathieu and McCall (2017). PFAS = per- and polyfluoroalkyl substances.

A previous study focused on persistent organic pollutants — including polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs), in the Greater Lake Washington watershed showed widespread contaminant detections among various pathway types, similar to the present study (King County 2014). The highest concentrations of PCBs and PBDEs were found in combined sewer overflows (which were not sampled for PFAS in the current study) and stormwater, and the highest loadings were from tributaries (along with their stormwater runoff) to the lake (King County 2014).

In our study, the highest maximum total PFAS water concentration was observed in groundwater samples collected from an area with suspected historical or current AFFF use. The highest ambient surface water total PFAS concentrations were observed in the small tributaries, while the highest tributary loadings were observed in the two large rivers, the Sammamish and Cedar Rivers.

PFAS Fate & Transport

All sampled pathway types contributed PFAS to Lake Washington. Among Phase 1 and 2 water samples, median total PFAS concentrations were greatest in order of: Tributary surface water, stormwater discharge, groundwater > lake surface water > surface runoff, bulk atmospheric deposition (with ">" representing a significant difference based on Kruskal-Wallis rank sum test followed by Wilcoxon rank sum test). Median PFOS concentrations were greatest in order of: Groundwater, tributary surface water, stormwater discharge, lake surface water > surface runoff > bulk atmospheric deposition.

In terms of PFAS load contribution to the lake, the relative importance of the different pathway types was difficult to quantify. However, sampling of the different pathways provided greater understanding about the mechanisms by which PFAS are transported through the environment and eventually enter into Lake Washington.

Lake and Tributary Surface Water

Total PFAS concentrations and compositions were similar among lake surface water samples across seasons, sampling years, and locations indicating that surface water in the lake was well-mixed. Alternatively, PFAS composition, detection of precursor compounds, and the presence of long-chain PFAAs varied by tributary surface water site, suggesting localized inputs to tributaries. Overall, PFAS composition in tributary surface water samples were similar to the lake and provide evidence that these tributaries serve as a source of PFAS to Lake Washington. Moreover, strong positive correlations among PFAA analytes in the tributary surface water suggest similarities in origins or release and uptake pathways in the environment (Ahmadireskety et al. 2021).

Flow

Flow played an important role in PFAS concentrations at tributary sites. Results showed that total PFAS concentration had a strong, negative relationship with flow, and that there were differences in PFAS concentration and composition between summer low and spring high flow. Lower spring high flow concentrations may be a result of dilution from higher spring flows or differences in point sources or pathways during summer and spring sampling events. It is unclear whether differences in composition were due to unique contributions from runoff, unidentified local point sources, seasonal influence from other pathways such as groundwater discharge or bulk atmospheric deposition, or some combination of these factors.

Furl et al. (2011) also found season-specific PFAS profiles and lower total PFAS concentrations during spring high flow versus fall low flow in surface water samples collected from four Washington state rivers downstream from dense industrial/urban areas. Compositional differences in these rivers were attributed to influences from runoff during spring high flow and wastewater treatment plant effluent during fall low flow.

Flow also played an important role in PFAS instantaneous loads from the tributaries. Total PFAS instantaneous loads were positively correlated with flow, with the highest load observed in the Sammamish River, followed by the Cedar River. While these two large rivers represented 63% of the total instantaneous load to the lake, the small tributaries collectively represented 37% of the load, demonstrating that both large and small tributaries played an important role in contributing PFAS to the lake.

While the Cedar River on average had twice the flow of the Sammamish, the Sammamish had on average 2.6 times the load. The reason for this difference is unclear but could be due to differences in land use activities within the subbasins, hydrogeological differences, or other subbasin characteristics.

Furthermore, instantaneous load calculations for the Sammamish River were likely an underestimate because they were calculated from flows gaged by the USGS upstream from three tributaries that discharge to the Sammamish River (Swamp, North, and Little Bear Creeks). We used the USGS gage location because backflows from Lake Washington can extend upstream past Woodinville during the summer as lake levels rise. While PFAS load contributions from Swamp, North, and Little Bear Creeks were not assessed in this study, the results still suggest large load contributions from the Sammamish upstream from the three tributaries. Additional assessment is needed to better understand PFAS sources and pathways to the Sammamish River.

Upstream to Downstream Gradient

The lowest surface water PFAS concentrations in the Greater Lake Washington watershed occurred in the protected area of the upper Cedar River subbasin, where PFAS were non-detect to near non-detect. Further downstream towards more developed areas of the lower subbasin, higher total PFAS concentrations were found. This finding is analogous to Kim and Kannan (2007), which found a gradual increase in PFOA and several other PFAS analyte concentrations along a rural to urban gradient.

In other tributary subbasins that we sampled, however, total PFAS concentrations were higher at the most upstream sites, or sustained from upstream to downstream. This result can broadly be explained by current or historical land use activities associated with urban development that impact water quality.

In the Greater Lake Washington watershed, one important factor was the conversion of natural vegetation to impervious surfaces (e.g., paved roads, parking lots, and buildings), which is commonly associated with urban development. For example, total land uses in the Thornton, Kelsey, Juanita, and McAleer Creek subbasins—where total PFAS concentrations were relatively high throughout the subbasin (or higher upstream)—were almost entirely (86%–96%) developed. These subbasins also had among the largest percentages of impervious surfaces (37%–42%) within the Greater Lake Washington watershed.

The importance of runoff from impervious surfaces contributing PFAS to receiving waters in the Greater Lake Washington watershed was further demonstrated by a strong positive correlation between total PFAS concentration and subbasin imperviousness. The role of impervious surfaces contributing to impaired water quality has been well-documented and reviewed in the literature (Klein 1979; Brabec 2002; Kim et al. 2016).

In more recent PFAS literature, Peter et al. (2024) found higher concentrations of PFAS and other contaminants in stormwater from urban watersheds (average 64% impervious area) compared to non-urban watersheds (average 4% impervious area) in the San Francisco Bay area. Furthermore, Thai et al. (2022) simulated succeeding rainfall events on AFFF-impacted concrete surfaces and found recurring releases of PFAS to the runoff, possibly due to a "wicking" effect. The study concluded that AFFF-impacted concrete may present an ongoing source of PFAS to stormwater runoff.

Other site-specific factors likely affected upstream to downstream PFAS concentrations within the subbasins. These include stormwater discharges to receiving upstream surface waters; presence of specific land use activities commonly associated with PFAS (e.g., airport and aerospace manufacturing); historical dumping of fill material or other garbage to the surface waters or surrounding area; and surface and subsurface hydrology (e.g., tributary inflows and groundwater discharges to the mainstems).

Lake and Tributary Sediments

PFAS composition in lake and tributary sediments differed from surface water at their respective sites, with more detections and higher concentrations of long-chain PFAAs and sulfonamides in sediments. This is primarily due to their greater hydrophobicity, in which the compounds tend to sorb onto particulates rather than dissolve in the water column (Qi et al. 2016; Munoz et al. 2018; Langberg et al. 2020; Bai and Son 2021).

There was also a positive relationship between total PFAS concentration in lake sediments and fine particulates (clay and silt; 1–62 µm). This relationship was also apparent in suspended tributary sediments; however, the relationship was not significant due to low sample size (n=6). In contrast, there were significant negative correlations between total PFAS concentration in both sediment types and larger grain sizes. A similar finding was observed in Hubert et al. (2023), which found negative correlations between PFAS soil-water partitioning values and soil grain size. As in Hubert et al. (2023), the inverse relationship in our study could be due in part to higher organic carbon content in the finer grain size particulates.

Groundwater and Springs

Groundwater samples had the highest total PFAS concentrations among sample types. Many studies have observed high groundwater PFAS concentrations associated with point sources such as landfills (Hepburn et al. 2019), fire training areas (Weber et al. 2017), dry cleaners (Barnes et al. 2021) and other industrial facilities (Hu et al. 2021). PFAS contaminated groundwater has also been shown to contaminate nearby surface water bodies when a point source is located nearby (Briggs et al. 2020; Pétré et al. 2021; Tokranov et al. 2021). The highest total PFAS concentrations in groundwater were observed at locations where likely point sources associated with current and/or historical land use could be identified. However, elevated PFAS concentrations were also observed in areas where a likely source could not be identified. Further investigation in areas of high PFAS concentrations could help us identify common sources in an urban environment that have not yet been considered.

The groundwater portion of our study primarily focused on groundwater discharging to surface water. The groundwater-surface water transition zone is associated with biogeochemical fluctuations (e.g. oxygen content) that have been shown to impact PFAS concentrations and composition (Tokranov et al. 2021). Similarly, we observed seasonal fluctuations in PFAS concentration and composition, but not a consistent seasonal relationship. This may in part be due to the limited scope of groundwater sampling. Most of our groundwater locations were sampled twice, once in spring and once in summer. In addition, we had a small number of groundwater locations distributed throughout a large watershed, we did not have co-located upland groundwater samples at transition zone locations, and we had a limited number of co-located groundwater-surface water sites. These factors limited our ability to assess the role of the transition zone on the observed seasonal variability at transition zone locations.

Another important consideration for samples collected at the groundwater-surface water interface is the mass of PFAS discharging to a surface water from groundwater (Pétré et al. 2021). PFAS flux can be calculated by first measuring the rate at which groundwater is flowing through a section of streambed at each sample location (flux) and multiplying by the PFAS concentration. Pétré et al. (2021) found that some points of high PFAS concentration had low PFAS flux due to low groundwater flux. Although we did not quantify groundwater flux for this study, at some sample locations a low volume of water was discharged during sampling, especially during the dry summer months. It is plausible some of the higher concentrations observed in our study are in areas of low groundwater flux and so have low PFAS flux. Determining the discharge rate of PFAS from groundwater to surface water would help us better understand the magnitude of contribution to the overall load, which in turn could help prioritize source control strategies.

In addition to factors that may influence concentrations within the groundwater-surface water transition zone, upland effects may result in the mobilization of PFAS within the unsaturated zone. An upland seasonal rise in the groundwater table due to increased precipitation may increase leaching of PFAS near the surface where releases originate. Zeng et al. (2024) found that water table fluctuations can lead to seasonal variations in PFAS concentrations and increase leaching of contamination in the unsaturated zone. Many studies have demonstrated that the unsaturated zone acts as a long-term source of PFAS for decades after their release, with the potential to migrate to groundwater and surface water (Brusseau et al. 2020, Weber et al. 2017, Xiao et al. 2015, Anderson et al. 2019).

In the Puget Sound Lowlands, the water table tends to be highest during the spring and lowest in the summer. We observed higher total PFAS concentrations in the spring at some but not all locations. A high-water table and increased leaching may play a dominant role in contaminant transport in areas where we observed higher concentrations during the spring.

Contaminants in groundwater move at a much slower rate than surface water, minimizing dilution and dispersion. This means contaminants in groundwater move in concentrated plumes. When a plume discharges to surface water, dilution and dispersion spread the contamination at a much faster rate. The volume of water and rate of flow of the surface water body determine how quickly this occurs.

At some of our co-located groundwater-surface water locations, high total PFAS concentrations in groundwater were associated with low PFAS concentrations in surface water. On the Cedar River, where groundwater with high PFAS concentrations discharged to the river, dilution likely played a significant role in the relatively low concentrations observed in surface water samples. However, low surface water concentrations relative to groundwater was also observed at some co-located groundwater-surface water locations in the smaller subbasins. In the smaller subbasins dilution likely plays a smaller role in concentration differences between groundwater and surface water.

The groundwater portion of our study primarily focused on sampling within the dense urban portion of the watershed. The high detection frequency and concentrations we observed in groundwater are consistent with other studies that associate urban land use (McMahon et al. 2022, Silver et al. 2023) and industrial sources (Hu et al. 2021) with increased detections and higher concentrations. At the few locations we had outside the urban corridor and in natural areas within urban areas, analyte detections were fewer and concentrations lower. The prevalence of PFAS detections and high concentrations we observed in this study throughout the urban portion of the watershed and near areas of industrial land use highlights the importance of identifying potential sources and better understanding the relationship between urbanization and PFAS occurrence.

Stormwater

Stormwater samples overall had higher detection frequencies and concentrations of long-chain PFAAs than surface water, suggesting stormwater may serve as a conduit for these compounds to enter receiving water bodies. This finding is similar to Kali et al. (2025), which found increases in long-chain PFCAs in small urban streams in Sweden following runoff events, and Zushi and Masunaga (2009), which found increases in long-chain PFCAs in an urban Japanese River following a first-flush storm event.

In the present study, PFAS composition among stormwater samples was largely site-specific, indicating differences in PFAS origins. For example, stormwater samples from sites draining mixed (commercial, residential, and open space) land uses were mainly composed of a mixture of PFAAs—similar to the lake surface water samples but with more detections of long-chain

PFAAs—suggesting PFAS in stormwater at these sites could have multiple origins. Given the widespread use of PFAS, the possible origins are extensive (e.g., see Table 2-6 in ITRC 2023).

Stormwater samples from residential areas had among the highest total PFAS concentrations and were also composed of a mixture of PFAAs, with highest relative concentrations of PFOS, PFPeA, PFHxA, and PFOA. In residential areas, possible origins could be related to uses or releases of household cleaning products, biocides and fertilizers, and other household consumer products, automotive maintenance, construction and building materials, as well as leaching from residential waste.

Masoner et al. (2019) found household chemicals to be frequently detected contaminants in urban stormwater discharges to surface waters and groundwater across the U.S. Although that study did not include PFAS, PFAS have been documented in a number of aforementioned items. For example, Ecology's Consumer Products Database[1] contains documented results of PFAS in household items such as carpet cleaners, sealers, floor polishes, automotive fabric protectors, food wrapping and packaging, and others. Additionally, in a study of PFAS in leachate from residential and commercial waste collection vehicles, Liu et al. (2021) concluded that components from the basic waste streams of homes and businesses contribute significantly to PFAS loading at municipal landfills.

Some conventional pesticides, which may be commonly used for gardens, lawns, and in public spaces are also known to contain PFAS. In a data compilation study, Donley et al. (2024) found that the biggest contributor to PFAS in U.S.-based conventional pesticides was active ingredients and their metabolites, as well as leaching from fluorinated pesticide containers. Included in their list of PFAS-based pesticides were chemicals that have been previously detected in urban streams in King County (Frans 2004).

Other stormwater samples, such as those collected from the airport had more unique PFAS analyte profiles. Stormwater from the Renton Airport was composed mostly of 6:2 FTS, PFHxA, PFPeA and PFBA, a pattern characteristic of modern fluorotelomer C6 AFFF formulations (ITRC 2023; National Academies of Sciences, Engineering, and Medicine 2023). In addition to AFFF-related releases, other common sources of PFAS contamination at airports include hydraulic fluid releases from maintenance facilities and releases from industrial or manufacturing activities that use PFAS in their production or processes (National Academies of Sciences, Engineering, and Medicine 2023). The consistent PFAS analyte profiles and concentrations in the 2020 and 2022 stormwater samples from the airport in our study suggests a persistent source either from ongoing processes, or from residual contamination related to historic uses or in the conveyance system itself.

Stormwater from the AstroTurf® field had higher total PFAS concentration (57.1 ng/L) than the FieldTurf® and grass athletic fields (5.9 – 12.5 ng/L), and were composed predominantly of three compounds (PFBA, PFPeA, and PFHxA)—different from the grass and FieldTurf®

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[1] https://ecology.wa.gov/research-data/monitoring-assessment/consumer-products-testing

fields, other stormwater sample types, and bulk atmospheric deposition samples. It has been speculated that PFAS are added to artificial turf during the manufacturing process. In studies of fluorine in artificial turf, Lauria et al. (2022) and Whitehead (2023) detected low concentrations of PFAS compounds in the turf samples; however, both studies concluded that most of the fluorine in the turf samples likely came from non-extractable fluoropolymers. In addition, AstroTurf's® website states that its turf has not been manufactured with any added PFAS since 2020.

Additional assessment is needed to evaluate whether PFAS levels in stormwater from the AstroTurf® field are persistent, and the origins of the PFAS. For example, PFAS may be present within the turf matrix itself (synthetic grass fibers, infill, or backing), or detections may be specific to the location sampled (e.g., how the field is maintained, residual in the stormwater drainage system, construction activities related to installing the new field, age of the field).

Stormwater samples from the 520 and I-90 bridges were composed mostly of PFCAs—primarily PFBA, PFHxA, and PFOA. Total PFAS concentrations (11.0 – 67.0 ng/L) in the bridge samples were within range of other Phase 1 stormwater samples (13 – 115 ng/L). Atmospheric deposition is one possible contribution of PFAS to stormwater from the bridge. Similar to the bridge samples, bulk atmospheric deposition samples from the Seattle area were composed almost entirely of PFCAs (predominantly PFBA, with lower concentrations of long-chain PFCAs). However, bulk atmospheric deposition samples were not collected at this specific 520-W bridge location, so results are not directly comparable. The sample collected from the I-90 bridge had detections of PFOS and 6:2 FTS at high relative concentrations, however additional sampling from the I-90 bridge is needed to assess whether this is an ongoing occurrence related to non-atmospheric sources.

In our study, road dust collected from the 520-W bridge site had lower total PFAS concentrations compared to the stormwater from the bridge site, suggesting there are other mechanisms by which PFAS enters bridge stormwater, in addition to road dust. PFAS used in automotive-related products (e.g., waxes, wiper fluids, interior upholstery, electronics) as well as materials related to building and construction (e.g., cement additives, cable and wire insulation, water-resistant paints and coatings) may serve as additional inputs to stormwater from the bridge, and more generally from urban areas (ITRC 2023). In a study of 18 automotive lubricant oils, Zhu and Kannan (2020) found that about 75% of the total PFAS (sum of 13 PFAAs) in the lubricants was composed of several long-chain PFCAs (C8 – C12), with a large presence of unknown short and long-chain precursors.

The extent to which atmospheric or transportation-related sources may have been contributing to PFAS in the bridge samples was not assessed. Nevertheless, it is unknown what other sources could be contributing PFAS to stormwater from the bridge.

Results from our study showed that antecedent dry period was another factor affecting total PFAS concentrations in stormwater samples. For example, higher total PFAS concentrations were detected in samples from Storm 5 at repeated Phase 1 stormwater sites. The last major rainfall (greater than 1 mm) occurred 13 days before Storm 5, which differed from the first four storms, in which at least 1 mm rainfall occurred between one and four days prior to sampling. Increased pollutant concentrations in stormwater following longer antecedent dry periods have been documented in other studies (Chui 1997; Lee et al. 2004), but may not always be the case, as in Kali et al. (2025), which found lower PFAS concentrations following longer dry periods.

Surface Runoff and Road Dust

Total PFAS concentrations in surface runoff samples (non-detect to 11.0 ng/L) were relatively low compared to surface water, groundwater, and other stormwater sample types collected during this study, and to other studies which measured PFAS in surface runoff. Kim and Kannan (2007) found total PFAS (sum of 9 PFAAs) concentrations ranging from 1.11 to 91.7 ng/L in surface runoff samples collected from main streets, parking lots, and residential areas in urban areas of New York. Xiao et al. (2012) found total PFAS (sum of 6 PFAAs) concentrations ranging from 14.3 to 96 ng/L in a heavily trafficked commercial area, and <30 ng/L in residential areas.

In the Greater Lake Washington watershed, total PFAS concentrations in surface runoff samples had similar concentration ranges to bulk atmospheric deposition samples (non-detect to 10.8 ng/L). PFCAs were also frequently detected analytes in surface runoff and bulk atmospheric deposition samples, and PFPeA was interestingly not detected in either sample type. However, based on differing PFAS analyte profiles, PFAS in surface runoff could not be attributed to bulk atmospheric deposition alone. For example, surface runoff samples from Thornton Creek subbasin had a higher percent composition of PFOS, several long-chain PFCAs, and sulfonamides—compounds not detected or infrequently detected in bulk atmospheric deposition samples. Runoff samples from the industrial and residential land use areas in the Cedar River subbasin were also influenced more by PFOS.

PCA results showed differentiation in PFAS composition of surface runoff based on subbasin. It is unclear whether differences could be attributed to variations in subbasin hydro-geomorphology, watershed management, or any other subbasin characteristics. Relatively low concentrations among the samples, low sample size, and lack of comparable studies in the literature make it difficult to assess implications of the result.

Total PFAS concentrations in road dust samples (0.305 – 19.0 ng/g) were comparable to concentrations in lake sediment samples from this study (0.906 - 16.9 ng/g). Long-chain PFCAs, PFOS, and sulfonamides were the most frequently detected analytes in road dust, similar to sediment. However, concentrations of several long-chain PFCAs (C10 – C14) tended to be higher in road dust from busy commercial land use areas compared to lake and tributary sediments, and to road dust in residential and open space land uses.

These findings suggest that long-chain PFAS sorbed to fine particulates and carried via surface runoff into the stormwater system may be a mechanism by which these compounds enter receiving waterbodies, particularly in more heavily-trafficked land use areas.

PCA results for road dust samples showed differentiation in PFAS composition between samples collected from three commercial areas (two shopping mall parking lots, and one busy street in an automotive services area) and samples collected from all other areas. The two shopping mall parking lots also had the highest total PFAS concentrations in road dust.

Murakami and Takada (2008) found higher PFAS levels in fine street dust of heavily-trafficked areas versus residential areas but also found composition to differ only among particle size fractions (fine [<63 µm] versus coarse [63–2000 µm]), not location (heavily-trafficked versus residential). Murakami and Takada (2008) concluded that fine particulates related to traffic contributed to PFAS in road dust and likely served as an origin of PFAS in street runoff and receiving waterbodies. In our study, the higher concentrations of total PFAS and long-chain PFCAs (C10 – C14) in commercial land uses versus open space and residential land uses supports findings from Murakami and Takada (2008). However, it remains uncertain whether particle size fraction played a role in PFAS composition in the road dust samples because we did not measure particle size fractions for road dust. All particulates in the road dust samples were filtered through 500 µm sieves, overlapping the categories in Murakami and Takada (2008).

Bulk Atmospheric Deposition

In our study, total PFAS concentrations among Phases 1 and 2 bulk atmospheric deposition samples ranged from non-detect to 10.8 ng/L. This is comparable to Kim and Kannan (2007), which found total PFAS (sum of 9 PFAAs) concentrations of 0.91 – 13.2 ng/L in rainwater, and Pfotenhauer et al. (2022), which found total PFAS (sum of 34 analytes) concentrations of 0.7 – 6.1 ng/L in rainwater. Bulk atmospheric deposition samples from our study were composed mostly of PFCAs, similar to findings from Ge et al. (2017) and Pfotenhauer et al. (2022). PFOS was not detected in any bulk atmospheric deposition samples in our study, in contrast to Shimizu et al. (2021), which found frequent detections of PFOS in wet and dry deposition from North Carolina.

The higher total PFAS concentrations in Seattle samples compared to North Bend samples during the September – November deployments could be due to Seattle's closer proximity to PFAS source emissions and/or higher levels of PFAS emitted into the air from sources. In a study of atmospheric deposition of chemicals of concern in the Lower Duwamish and Green River watershed, King County (2013) found that atmospheric deposition rates of metals and organic contaminants were related to the degree of urbanization at the sampling locations.

The longer-chain PFCAs in Seattle samples compared to North Bend samples seem to indicate PFAS sources nearer to Seattle. This is consistent with Thackray et al. 2020, which predicted a strong decreasing gradient in long-chain to short-chain PFCA ratio moving from more-polluted to less-polluted areas. Studies of PCBs in bulk atmospheric deposition have also found that contribution of lighter-weight congeners increased, and heavier-weight congeners decreased with increasing distance from heavily urbanized areas (King County 2013; Era-Miller et al. 2019).

Higher total PFAS concentrations in the spring and fall samples than in the winter samples could be related to antecedent rainfall conditions, which was supported by a positive correlation between length of dry period and total PFAS concentrations at the Seattle site. King County (2013) similarly observed declines in trace metal fluxes from summer to early winter. The longer the dry period, the more contaminants and particulates that bind them accumulate in the atmosphere, and the more that are precipitated during the next rainfall. This process of removal of airborne materials via precipitation scavenging has been well-documented in the literature (Engelmann 1965; Cheng et al. 2021; Li et al. 2024).

Biofilm and Macroinvertebrates

High levels of contamination in the method blank and relatively low levels (total PFAS less than 1 ng/g) present in biofilm and caddisfly larvae tissue samples resulted in qualification of PFAS compounds expected to be present at high relative concentrations in the tissue samples. For example, in Wong (2023), PFOS was detected in 11 of 11 biofilm samples (total PFAS 0.11 – 3.6 ng/g) collected from the South Fork Palouse River in Washington, and detection rates of PFOA, PFDA, PFDoA, and PFMBA were over 80%. Munoz et al. (2018) also found high detection frequencies (80 – 100%) of long-chain PFCAs and PFOS in 11 biofilm samples (total PFAS 4.3 – 33 ng/g) collected from the Seine River in France. In macroinvertebrate samples, Bynes et al. (2024) found long-chain PFCAs and PFOS to be the most frequently detected compounds and highest relative concentrations in aquatic macroinvertebrate samples collected from small rivers and streams in Belgium.

Bioaccumulation and biomagnification of PFAS (particularly long-chain PFCAs and PFOS) in biofilms, aquatic macroinvertebrates, and fish have been suggested in findings from other studies (Babut et al. 2017; Munoz et al. 2018; Penland et al. 2020; Zhang et al. 2022). In the present study, we did not estimate bioaccumulation factors for these PFAS compounds because of the large number of qualifications and low detections in the tissue samples. In addition, our sampling size was limited and did not necessarily reflect the food web of Lake Washington fish, but that of surrounding tributaries to the lake.

However, based on findings from Mathieu (2022), which documented relatively high concentrations of PFOS (up to 99.9 ng/g) and long-chain PFCAs (up to 11.3 ng/g) in high trophic level fish from Lake Washington (particularly smallmouth and largemouth bass), the potential for biomagnification from lower trophic level organisms is high. A more robust study that includes sampling the varied diet organisms of high and low trophic level organisms in the food web would provide useful information on bioaccumulation and biomagnification of PFAS in Lake Washington fish.

TOP Assay

TOP assay results showed a large presence of unknown PFAS precursors in the lake and tributary sediment samples. This was not the case for surface water, groundwater, and stormwater samples, for which pre- and post-oxidation concentrations were more variable.

One explanation is that the sediments can act as a sink for pollutants over time — including hydrophobic PFAS precursors—accumulating chemicals that more readily adsorb to sediments than remain in the water column (Langberg et al. 2020). The large presence of hydrophobic precursors in the sediments and their subsequent transformation serves as a potential mechanism for PFAA accumulation in benthic macroinvertebrates and fish (Langberg et al. 2020).

Phase 1 Follow-Up Sampling

Cedar River

Surface water total PFAS concentrations in the Cedar River were low and instantaneous loads were relatively high compared to the smaller tributary subbasins. Both are likely owing to the higher flows in the Cedar River, in which larger volumes of water may dilute concentrations, but yield higher mass loads.

The highest groundwater total PFAS concentrations were identified in two areas of the Cedar River with current and historic industrial activities—the lower Cedar River near the mouth, and the middle reach of the river in Larry Phillips Natural Area. Based on the high total PFAS concentrations observed in these two areas, point source contamination related to land use activities are suspected.

Total PFAS concentrations observed in groundwater near the mouth along the east bank of the river are among the highest concentrations observed in the study. Concentrations approaching one µg/L suggests a nearby point source(s) is likely located in the upgradient direction (to the east-southeast, PGG 2012 and Roy F. Weston, Inc. 2001) of the sample locations.

The composition within the plume was dominated by PFPeA, PFHxA, PFHxS, PFOS, and 6:2 FTS. Several studies (Backe et al. 2013, Schultz et al. 2004, Houtz et al. 2013, and Weber et al. 2017), observed high concentrations of PFSAs, PFCAs, and fluorotelomers in groundwater at AFFF impacted sites, likely due to precursor transformation post-release. These studies suggest precursors retained in the unsaturated zone transform into more mobile PFAAs. This is consistent with the high concentration of short chain PFAAs we observed near the mouth of the river.

The PFAS plume we observed in this study is likely related to the long history of aerospace manufacturing adjacent to the mouth of the river. The high concentrations of PFAAs and detections of fluorotelomer compounds detected in the plume in addition to the adjacent land use, suggests releases of AFFF may be a component of PFAS observed in this plume. A detailed review of previous and current uses and practices is needed before any conclusions can be drawn regarding the source(s) of PFAS in this plume. Airport emergency response or equipment testing may also have resulted in AFFF releases near the mouth, potentially resulting in transport of PFAS to the Cedar River via the groundwater pathway.

High total PFAS concentrations observed the Larry Phillips Natural Area may be associated with current and former industrial land use adjacent to the Cedar River. The sample locations in this area are located adjacent to the plateau where the former Queen City Plating facility operated, and a composting facility is currently located.

Total PFAS concentrations at both the groundwater and spring locations were more than two times higher during the spring event than the summer event. The seasonal difference in concentration is consistent with concentrations observed in Cedar River surface water samples collected just upstream of the groundwater and spring sites. Higher concentrations during spring high flow events were not typically observed for surface water in this study and may indicate influence from groundwater discharge to the river at this location.

The composition of the springs was dominated by PFPeA and PFHxA. The groundwater showed the same dominant compounds during the spring flow event but was dominated by sulfonamides N-MeFOSE and N-EtFOSE during the summer. The increased concentrations and variability in composition may be indicative of seasonal variations such as groundwater flow direction relative to the source(s), mobilization of retained PFAS from the unsaturated zone due to water table fluctuations (Zeng et al. 2024), biologically induced sorption/desorption processes within the groundwater-surface water transition zone (Tokranov et al. 2021) or unknown seasonal inputs.

Contamination from the former industrial activities (Queen City Plating) in the Larry Philips Natural Area have been observed in all three aquifers. PFAS were not regulated as a hazardous substance under the Comprehensive Environmental Response, Compensation, and Liability Act at the time the Queen City Plating site investigation and cleanup activities took place and so have not been assessed. Additional assessment would be needed to determine whether historic or current industrial activities are contributing to the higher PFAS concentrations we observed adjacent to river. Previous releases have contaminated all three aquifers, therefore if PFAS releases occurred, transport to all three aquifers is plausible. The high PFAS concentration in the springs discharging from the hillside above the river suggests a point source(s) located above the right bank of the river.

The three releases that occurred along the Cedar Hills leachate pipeline in the fall of 2020 are located in close proximity to Larry Phillips Natural Area sample locations. Based on cleanup activities for the releases and sample locations for this study, it is unlikely that these releases influenced PFAS concentrations at any of the locations sampled.

The Renton monitoring well network, although located within the urban part of the Cedar River subbasin, has been protected from hazardous substance releases for over 30 years. Our sampling of this well network has allowed us to assess PFAS impacts to a shallow, protected unconfined aquifer. Although there were some higher concentrations (> 30 ng/L) detected in the downtown monitoring wells, the median total PFAS concentration was low compared to samples collected at the groundwater-surface water interface.

The highest PFAS concentrations were observed in monitoring wells along the right (east) bank of the Cedar River (MW11, MW30S, and MW3). Wells MW11 and MW30S are located near a divide where groundwater flows either to the west-northwest or southwest (Hart Crowser 1989, PGG 2012). Changes to the pumping rate from the downtown production wells affects groundwater flow direction making the upgradient direction vary. Both wells are on the edge of the zone with the most restrictive use regulations.

It is difficult to determine potential sources due to the groundwater divide and variation in groundwater flow direction. The nearest potential sources of contamination in the approximate upgradient direction would be auto repair facilities, gas stations, and a bus depot. However, these facilities are located more than a 1,000 ft to the north and east. Total PFAS at the nearest upgradient well (MW10) is considerably lower. Overall, the use restrictions seem to have limited PFAS in the Cedar Valley aquifer. However, a more detailed review of aquifer characteristics, groundwater flow direction, current and previous land use, and additional sampling would be needed to better understand PFAS concentrations and potential sources.

A monitoring well located in Riverview Park (MW33) outside of the heavily urbanized area was similar in total PFAS concentration to the higher concentrations observed in the downtown area during the spring event. The only potential sources that we are aware of at Riverview Park are artificial fill, sampling equipment used during sampling, or spills. Artificial fill of unknown provenance underlies the segment of the railway that runs through the park to the south of the well. Operation of the railroad may have included historic spills containing PFAS chemicals. However, no spills were reported to Ecology near this location.

Due to sampling logistics related to well location, monitoring well MW33 was sampled using equipment that could not be confirmed was PFAS-free. Therefore, the pump and tubing may have contributed to the total PFAS concentration detected at this well.

Juanita Creek

Surface water total PFAS concentrations were similar during summer low flow and spring high flow, and upstream to downstream. In contrast, flows at the outlet were almost three times higher during spring high flow than during summer low flow. These findings suggest that inputs to the creek during spring high flow may be important. We might otherwise expect lower concentrations during spring high flow due to dilution. However, PFAS composition in Juanita Creek showed no differentiation by sampling event but instead showed increasing “mixture” along an upstream to downstream gradient. One explanation for the sustained total PFAS concentrations and increasing PFAS mixture from upstream to downstream could be contributions of PFAS from major tributary inflows to Juanita Creek.

High surface water total PFAS concentrations near the headwaters of Juanita Creek indicate upgradient sources. Relatively high concentrations of PFSAs (primarily PFBS, PFHxS, and PFOS) suggest non-atmospheric contributions of PFAS. Potential sources may be related to historical land uses; however, it remains unclear (further discussion below).

Groundwater total PFAS concentrations showed variability by location, decreasing upstream to downstream. Concentrations were higher during the spring near the headwaters and mouth of the creek but were higher during the summer at one upper reach site. This suggests a source is located upgradient of where the creek first daylights, whereas sources are not located near the downstream locations.

High total PFAS concentrations were detected in groundwater along the Lake Washington shoreline. These sample locations were within 700 ft of the low detections observed in groundwater at the mouth of the creek. Higher groundwater concentrations may be an indication of the proximity of the sample location to a point source. Therefore, the decrease in groundwater concentrations from upstream to downstream does not necessarily indicate PFAS inputs from groundwater only occur near the headwaters. This decrease in concentration may be more related to the limited number of groundwater sample sites.

The aquifer that is the source of Juanita Creek likely occurs in the coarse-grained glacial outwash deposits based on the topography and local geology (Brooks, 2017). Assuming the watershed divide coincides with the groundwater divide and topography approximates groundwater flow direction, potential upgradient point sources would have to be located somewhere within a very small area. The upgradient area consists of a school, a park, a church and residential housing.

Former use of a portion of the upgradient residential area includes a plant nursery that operated from approximately 1965 – 2005. The nursery was irrigated with stormwater runoff and well water mixed in a pond with pesticides and herbicides prior to application (City of Woodinville, 2014). Pesticides are known to contain PFAS, both indirectly, through leaching of PFAS from containers, and directly as active or inactive ingredients (Donley et al. 2024, Hall et al. 2024).

The former nursery is underlain by glacial till, which typically slows contaminant migration due to relatively low hydraulic conductivity (Vaccaro et al. 1998). Although infiltration of irrigation water was likely impeded by the till unit, it is plausible that four decades of irrigating with pesticides may have resulted in PFAS impacts to the underlying aquifer. However, without knowledge of the type or quantity of pesticides and herbicides used, confirmation of groundwater flow direction, and a more thorough assessment of historic land use, it cannot be determined whether the nursery was a PFAS source to groundwater and surface water observed at the headwaters of Juanita Creek. Other potential sources such as pesticide use for lawns and gardens, sewer or stormwater line leaks, or other sources not yet considered could result in contamination to the uppermost aquifer.

Thornton Creek

High concentrations and detections of several PFSAs, long-chain PFCAs, sulfonamide, and fluorotelomer compounds at the headwaters of the north branch of Thorton Creek (Ronald Bog) suggest upgradient sources, or sources associated with historic or current land use. PFAS levels in Ronald Bog could be associated with historic filling and dumping of construction-related debris to the bog for development (CDM Smith 2017), historic I-5 road construction, as well as stormwater discharges or runoff from surrounding lands. Wetland restoration activities involving large-scale excavation and removal of debris at Ronald Bog had also taken place in 2020, prior to sampling. Further study would be needed to assess whether these activities affected PFAS levels in the bog.

High total PFAS concentrations in groundwater south of Ronald Bog (at N 167th St) may be related to a historic landfill. This location was on the northern edge of the former Corliss landfill. Composition dominated by short-chain PFCAs and elevated concentrations of PFOS and PFOA are consistent with a previous study investigating leachate-impacted groundwater from 20 historic municipal landfills in Canada (Prop et al. 2021).

The Shoreline Recycling and Transfer Station is located on the northern portion of the former landfill. Due to the prevalence of PFAS in municipal waste a transfer facility could potentially be a source of PFAS contamination. A recent study found PFAS in residential and commercial waste collection vehicle leachate prior to landfilling, despite the short contact time (Liu et al. 2023). Additional information regarding management of waste and stormwater at the transfer station would be needed to assess whether operation of facilities such as this are resulting in releases of PFAS.

The highest surface water total PFAS concentration in Thornton Creek was observed in the upper reach of the south branch downstream of Northgate Mall, a large shopping mall complex (TC-S-5th). Total PFAS concentrations were also relatively high at the furthest upstream site along the south branch, presumably near the headwaters. PFAS analyte profiles at both sites suggest upgradient sources of PFAS, and different analyte profiles between the two sites suggest contributions from different sources. The high concentrations of N-MeFOSE (commonly associated with carpets and textiles), N-EtFOSE (commonly associated with paper and packaging), and the presence of 5:3 FTCA (commonly associated with landfill leachate) during summer low flow at the site downstream from the mall complex indicates localized inputs to this site. These analytes were not frequently detected in surface water samples in our study.

Sulfonamides were also detected in groundwater at the Northgate Mall location during the summer event suggesting groundwater input are likely contributing to sulfonamide concentrations observed in surface water. Detections of several long-chain PFAAs in groundwater, which are less mobile in groundwater (Weber et al. 2017), also indicate a localized source. The Northgate Mall area is mapped as modified land, including artificial fill with debris, underlain by coarse-grained outwash deposits in the vicinity of the creek (Booth et al. 2009). The area of fill extends across the Northgate Mall, I-5 and North Seattle College. Contaminated fill was also documented on nearby properties (Farallon Consulting 2023a, Sound Earth Strategies 2019). Uncontrolled fill from an unknown provenance could potentially be contaminated with PFAS containing material.

The highest total PFAS concentration in groundwater observed was below Meadow Brook Pond. Concentrations of N-MeFOSE and N-EtFOSE were detected at concentrations higher than typically observed in groundwater and the nearest upstream surface water sample along the south branch. Interestingly, we did not observe these analytes at the groundwater sample located between the two (approximately 500 feet east of the surface water location). The presence of sulfonamides in both surface water and groundwater at multiple locations at concentrations higher than typically observed suggests there may be a subbasin-wide source.

The nearest commercially zoned area to the samples above Meadowbrook Pond (south branch) is along Lake City Way NE (approximately 1300 ft west). Contaminated sites located along Lake City Way NE are underlain by coarse-grained glacial outwash deposits, leaving underlying aquifers more susceptible to contamination. Several contaminated sites, including former gas stations, a car wash, and auto repair facilities, are located at the headwaters of a small tributary to the south branch of Thornton Creek. Contaminated sites could potentially be a source of contamination to smaller tributaries of Thornton Creek or shallow groundwater.

In addition, several petroleum releases have also been documented on the hill to the southwest, in an area also underlain by coarse-grained deposits. Inferred groundwater flow direction based on topography would be toward the creek. It is unknown if subsurface releases such as these are associated with the release of PFAS. However, it is known that PFAS have a wide range of uses in the automotive and petroleum industries (Glüge et al 2020).

The geology underlying the south branch of Thornton Creek above Meadowbrook Pond at the surface water location is mapped as artificial fill containing debris, underlain by peat and alluvium (Booth et al. 2009). The fill is underlain by coarse-grained outwash deposits at the south branch groundwater location. As mentioned above, fill containing debris may be a source of PFAS. Differences in organic content and grain size could impact sorption to soil (Hubert et al. 2023) resulting in the differences observed in composition. In addition, grass and turf playfields are located near the sample locations. Areas not covered by impermeable surfaces (asphalt, concrete, buildings) would increase infiltration of rainwater, which would increase leaching if a subsurface source of PFAS was present.

A larger sample set focusing on artificial fill with waste material would be needed to understand if it is a source of PFAS to groundwater. Another possibility for widespread dispersal with potential impacts to groundwater could be lawn applications of fertilizers, pesticides, or compost, all of which are known to contain PFAS (Gaines 2023, Choi et al. 2019).

Meadowbrook Pond, which is a stormwater detention facility, may also play a role in the fate and transport of PFAS. Previous studies have detected PFAS in urban stormwater pond sediments (Flanagan et al. 2021, Olmstead et al. 2021). Although accumulated sediments are monitored and removed periodically, the sediments could act as a source if seasonal changes in biogeochemical conditions cause desorption and remobilization of PFAS.

Meadowbrook Pond is located at the site of the former Lake City Sewage Treatment Plant, which operated from 1953 to 1967 (Thornton Creek Watershed Management Committee 2000, Gibbs 1958). The Lake City system was known to have an issue with excessive infiltration from groundwater through joints and other openings in pipes (Gibbs 1958). Groundwater infiltration is still a major issue for sewer pipes in King County (King County 2024). King County estimates that inflow and infiltration from stormwater, surface water, and groundwater account for 75% of peak flows in their sewer system during the winter months. If infiltration of groundwater is occurring, sewage is also likely seeping out through holes, cracks, joints, and leaky connections. Infiltration of PFOS contaminated groundwater into sewer lines from contaminated sites has been an issue documented at multiple sites in Michigan (EGLE 2020).

In addition to infiltration issues, multiple studies have observed high fecal bacteria throughout the Thornton Creek watershed (Thornton Creek Watershed Management Committee 2000, Frodge 2013, Ecology's 2022 Draft Water Quality Assessment[1]), which indicates sewage leaks may be a major issue impacting the creek. Sewer main lines generally run adjacent to the north and south branches of Thornton Creek, with numerous side sewer connections. Municipal wastewater is known to contain PFAS (Bothfeld and Mathieu 2022). Releases from sewage pipes or connections could contribute to PFAS concentrations observed in surface and groundwater in the Thornton Creek subbasin.

Fairweather Creek

Fairweather Creek, one of the smaller subbasins in our study area, had the highest total PFAS concentrations in surface water and had one of the highest surface water concentrations of PFOS and PFOA in our study. Fairweather Creek also had the second highest concentration of PFDA (9.4 ng/L), after a residential stormwater sample (BEL-SHORE; 9.8 ng/L). At all other tributary sites, PFDA concentrations were less than 2 ng/L. PFDA has been associated with grease-proof coatings on food packaging, and stain-resistant treatments on furniture, upholstery, carpets, and other consumer products (EPA 2023).

The relatively high concentrations and detections of long-chain PFCAs during both summer low and spring high flow events, and during both sampling years suggest that PFAS in the Fairweather Creek may be associated with stormwater runoff, sediments, and/or continuous localized sources. The high total PFAS concentration relative to the percentage of imperviousness within the subbasin suggest that factors other than runoff from impervious surfaces may be important, and/or that there may be sources of relatively high contamination affecting PFAS levels in runoff or other pathways to the creek.

Kelsey Creek

The relatively low total PFAS concentration upstream from tributary inflows to Kelsey Creek, and variability in concentrations upstream to downstream along the mainstem indicate likely contributions of PFAS from tributary inflows to the creek. It is unclear whether the relatively high concentration near the headwaters of Kelsey Creek (outlet of Larsen Lake) is associated

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[1] https://apps.ecology.wa.gov/ApprovedWQA/CandidatePages/CandidateSearch.aspx

with inputs to Larsen Lake, or if it reflects the conditions of the media from which the sample was collected. Field notes indicated wetland-like conditions at this site, with surrounding wetland vegetation and tannin-color water (an indication of high organic carbon content).

At the outlet of Valley Creek to Kelsey Creek, both the high total PFAS concentration and relatively high concentrations of long-chain PFAAs suggest that PFAS inputs could be associated with stormwater, sediments, localized inputs, and/or previous construction activities related to in-stream restoration just upstream of the site. Valley Creek runs through a former gas station and automotive repair facility that operated from 1971 to 2008 (Ecology 2013). Multiple releases to soil and groundwater are known to have occurred. In addition, inflows from Sears Creek, which empties into Valley Creek just upstream of its outlet to Kelsey Creek, could have affected PFAS levels at this site. No major PFAS inputs to Valley Creek were indicated upstream near the headwaters.

West Tributary, which flows through an industrial area of the Kelsey Creek subbasin, had the highest concentrations of short-chain PFCAs among tributary surface water samples. West Tributary also had the second highest concentrations of long-chain PFCAs, after Valley Creek, along with low levels of 6:2 FTS and 5:3 FTCA. Composition of the West Tributary groundwater was very similar to surface water. Concentrations of short-chain PFCAs at the southern location were among the highest observed in groundwater. Long-chain PFCAs and small detections of 6:2 FTS, 8:2 FTS, and 5:3 FTCA were also observed in groundwater. Based on the high concentrations and detections of fluorotelomers, the use of AFFF at the former fire training area may be contributing to PFAS observed in the surface water and groundwater.

In addition to the fire training facility, other industrial facilities include a former paper facility that operated from 1966-2011 (CDM Smith 2023), a bus base including maintenance, fueling, and washing facilities (Parametrix 2021), light rail operations and maintenance facility, and a grocery warehouse and food processing facility including truck maintenance and truck wash (EPI 2001). The majority of confirmed releases and subsequent remediation and cleanup efforts are petroleum related. These facilities and releases may also be sources to Kelsey Creek and groundwater entering the creek. Additional investigation would be needed to determine if these facilities are contributing to the PFAS concentrations at this location.

McAleer Creek

The highest total PFAS concentration in the McAleer Creek subbasin was found at the upstream-most site in Hall Creek near the outlet of Hall Lake, while concentrations declined towards the outlet of McAleer Creek to Lake Washington. This finding suggests sources to Hall Lake, potentially related to stormwater runoff into the lake (Gray & Osborne, Inc. 2002).

Ravenna Creek

Total PFAS concentrations in the upper reach of Ravenna Creek, within Ravenna Park, were lower compared to downstream. The creek enters an underground pipe at the south end of Ravenna Park and resurfaces in Union Bay Natural Area. Low concentrations within

Ravenna Park combined with the impervious piping indicate that the main PFAS inputs to the creek lie downstream.

However, cracks or damage to the pipe and a shallow water table and upward gradient toward the historic creek bed could result in groundwater infiltration into the pipe. In addition, underground pipes and utilities can create conduits for contaminants. Disturbed soil surrounding pipe is often more permeable than the native material and can act as a preferential pathway. Water entering a preferential pathway can travel faster and further than typical for groundwater transporting contamination than it would otherwise travel. Faster flow could also reduce sorption to soil (Piwoni and Keeley 1990).

All sample locations in the Ravenna Creek subbasin south of Montlake Boulevard NE/NE 45th Street were within the ~200-acre footprint of the Montlake Dump. Within the Union Bay Natural Area, high total PFAS concentrations and the presence of 5:3 FTCA, a common indicator of landfill leachate, indicates that the historic dump likely contributed PFAS to the surrounding environment.

High PFAS concentrations were also observed in groundwater at the two monitoring wells, which are also within the footprint of the dump (UW-1 and UW-2). Municipal waste landfill leachate tends to be dominated by short-chain PFCAs and PFOA (Hamid et al. 2018), which is what was observed in the monitoring wells. A study investigating PFAS concentrations in 20 historic landfills with closure dates 30 or more years ago (Prop et al. 2021) frequently found elevated concentrations of PFBA, PFPeA, PFHxA, PFOA, and PFOS in leachate impacted groundwater. In the present study, PFOS was the dominant compound at the groundwater location nearest Montlake Boulevard NE and short-chain PFCAs became more dominant to the south. This suggests either an upgradient PFOS source outside of the landfill footprint or variation in composition related to waste type. As noted above, the landfill was used by both the City of Seattle and industries/private use. A designated area for industry/private use could result in unique composition profiles for different areas of the landfill.

The high total PFAS and PFOS groundwater concentrations observed south of Montlake Boulevard where the creek daylights (RC-MONTLAKE-GW) could also be associated with nearby contaminated sites. This location is located at the northern edge of the dump and may contain PFAS contributions from outside the landfill footprint. Upgradient of this location, releases to the subsurface were identified at a former dairy processing facility, gas stations, automotive repair facilities, car wash facilities, a television repair shop, and a dry cleaner (Farallon Consulting 2023b, Environmental Associates, Inc. 2001). Additionally fill that contains debris (wood, concrete, metal, plastic) was encountered during subsurface investigations, north of NE 45th Street (Farallon Consulting 2023b).

Florida completed a statewide PFAS pilot study at dry cleaning sites and found a wide range of PFAS concentrations in groundwater were generally co-located with chlorinated solvent plumes (Barnes et al. 2021). Composition was generally dominated by PFOA and PFOS with maximum concentrations in groundwater of 2,640 ng/L and 3,480 ng/L, respectively. We are not aware of any studies investigating PFAS at petroleum contaminated sites such as gas stations and auto repair facilities. Considering their use in the petroleum and automotive industry (Gaines et al. 2022), it is possible PFAS are also associated with these types of contaminated sites.

The two monitoring wells sampled were located on a small urban organic farm and educational facility. Farming practices, such as the use of biosolids, compost, or recycled water for irrigation, have the potential to impact PFAS concentrations. For example, previous studies have found PFAAs were the dominant PFAS in yard waste compost and short-chain PFAAs and PFCAs, particularly PFHxA, dominated municipal waste compost (Saha et al. 2024; Choi et al. 2019). Moreover, higher concentrations were found in compost that contained food packaging (Choi et al. 2019). Although soil amendments are potential sources of PFAS, current and historic practices need to be reviewed to determine the likelihood that compost and other biowastes contributed to groundwater PFAS concentrations in the present study.

A previous study at these monitoring wells also observed relatively high concentrations of PFPeA at UW-MW1 compared to UW-MW2 but did not observe it in surface water samples collected from Ravenna Creek (Prasad, 2021). Importantly, low-density polyethylene (LDPE) tubing was left in the wells from a 2016 investigation and was not removed until approximately 24-hours before Prasad's 2020 sampling event. The tubing was suspected to have contributed to the high concentration of PFPeA. LDPE tubing has been noted as a potential source of PFAS, and so current PFAS sampling practices discourage its use in favor of HDPE (MPART 2018). Leaching of C4 through C11 PFCAs, particularly PFBA and PFPeA, from one brand of LDPE tubing has been documented (Denly et al. 2019).

Tubing was removed from both monitoring wells in March of 2022 prior to sampling for the present study. The wells were purged dry in an attempt to collect samples in August 2022. During the spring 2023 event, less than one liter of water was purged from each well before they went dry and were left to recharge overnight. The heterogeneous fill and discontinuous aquifer at the landfill site were likely responsible for the slow recharge encountered at these wells. If leaching from the tubing did occur, sorption of PFAS to aquifer material or filter pack surrounding the screen would be slow to flush out of the system. Additionally, it is unknown what volume of water would be required to sufficiently purge a well impacted by PFAS leaching from tubing. Despite the potential contribution of PFPeA from the LDPE tubing, results from this study suggests the landfill leachate likely contributed to the PFPeA observed in the monitoring wells based on concentrations observed at other locations within the landfill footprint.

Madrona Beach

The highest total PFAS concentration of Phase 1 groundwater samples (105 ng/L) was observed along the shore of Lake Washington at Madrona Beach. However, Phase 2 summer and spring groundwater sampling results at this location were among the lowest concentrations detected in groundwater (< 3 ng/L).

This may be a result of the PushPoint sampling method used at this site, which required removal of the device after each event. An attempt was made to resample the same location; however, we were unable to insert the PushPoint sampling deep enough to collect groundwater samples during Phase 2 sampling. Thus, the Phase 2 samples were collected further from the shoreline, but were within 10 ft of the Phase 1 location. It is possible that variability in flow patterns may have resulted in lower concentrations observed during Phase 2. This area is mapped as landslide debris with indistinct morphology (Booth et al. 2009). Springs discharging along the upland hillside into the landslide debris may cause variable transport pathways to the lake which could alter discharge points along the shoreline.

Alternatively, a sewage release to Lake Washington may have impacted Phase 1 results. Releases of untreated sewage to Lake Washington occurred in January 2021 and August 2022 (King County 2022). The releases, which occurred approximately 0.5 miles north, increased bacteria levels at Madrona Beach. The 2021 release was much larger (2.2 million gal) than the 2022 release (15,000 gal) and contained a mix of sewage and stormwater. Untreated sewage and stormwater are known to contain PFAS. For example, one study found an increase in detected PFAS following a release of stormwater and untreated sewage in a marine environment (Ford and Ginley 2024). It is possible the larger release impacted PFAS concentrations, and the smaller release did not. PFAS may have been retained in the sediments and then remobilized through desorption processes when there is a reversal in concentration gradient, as observed by Reif et al. (2022), due to dilution of sewage/stormwater with lake water.

Based on high PFAS concentrations detected in the springs discharging from the hillside above the Madrona shoreline location (MADRONA-SP), we suspect an upland source of PFAS. Similarities in composition between shoreline groundwater and spring samples also suggest an upland source contributed to the high Phase 1 total PFAS concentration along the shoreline. Like other study locations, the upland area was primarily residential with a mix of commercial businesses (laundromat, gas station, and autobody shop) and petroleum releases.

Wetmore Slough

Follow-up sampling in the Wetmore slough area was conducted downgradient from the former Genesee landfill, near the Lake Washington shoreline. High PFOS concentrations were detected during follow-up sampling and were more the twice the Phase 1 concentration (summer 2023 event). High PFOS concentrations have also been observed by other studies at historic landfills (Hepburn et al. 2019). However, a high proportion of PFOA to total PFAA concentration in groundwater samples is considered an indicator of municipal landfill derived PFAA (Hepburn et al. 2019) and was not observed in the present study. The Wetmore slough sampling location is also next to a recreational rowing and sailing center. Previous research has detected PFAS in maritime textiles including sail covers, seat covers, console housing and bimini tops, all of which may play a role in total PFAS concentrations at this sampling location (Janousek et al. 2019). We also detected high total PFAS and PFOS concentrations at a shoreline groundwater location, Mt Baker Beach, approximately one mile north. Similarities in composition and long chain PFCAs detected at both Wetmore and Mt Baker Beach suggest a nearshore localized source.

Conclusions

Results of this 2020 – 2023 field study support the following conclusions:

- Except for the upper Cedar River subbasin, PFAS detections were widespread within the Greater Lake Washington watershed, and PFAS were detected in all sampled pathway types (tributary surface water, stormwater, groundwater, and bulk atmospheric deposition) indicating ongoing diffuse sources.
- The Sammamish and Cedar Rivers represented the largest instantaneous load contributions of total PFAS to Lake Washington, although they represented among the lowest concentrations. However, the small tributaries still represented about 37% of the total PFAS instantaneous load and represented the highest surface water concentrations in the watershed. Thus, both the large rivers and small tributaries were important contributors of PFAS to the lake, and more generally to PFAS contamination in the watershed.
- TOP assay results showed that there could be a large presence of unknown PFAS precursor compounds in Lake Washington and tributary sediments.
- The sources of PFOS (the main type of PFAS previously found in Lake Washington fish) were largely diffuse, with higher concentrations from some localized inputs.
- Bulk atmospheric deposition was not found to be a pathway of PFOS to the lake. However, for PFCAs, bulk atmospheric deposition is a pathway to the lake — with more analyte detections and higher concentrations in spring and fall, and fewer detections and lower concentrations in winter. Results also showed that deposition from atmospheric sources were greater in the more urbanized area (Seattle) than the less urbanized area (North Bend).
- Comparisons of total PFAS and analytes in the surface water during the dry/summer/low flow and wet/spring/high flow seasons suggest that both runoff and non-runoff related pathways may be important contributors of PFAS to the receiving water bodies. Regarding runoff — PFAS concentrations in the tributary subbasins were correlated with the amount of impervious surfaces in the subbasin.
- This study suggests that in addition to AFFF, landfills, and industrial facilities, unidentified point sources in commercial and residential areas are a source of groundwater contamination. These sources are contributing to the load of PFAS entering surface water in the Greater Lake Washington.

Recommendations

Results of this 2020 – 2023 field study support the following recommendations.

- **Model PFAS pathways**. In our study, we were able to compare PFAS load contributions among tributary surface water locations. To compare relative contributions of PFAS among the different pathway types and localized source areas was challenging. Creating a model could further assess the relative importance among PFAS pathways to the lake, including surface water, groundwater, stormwater, and bulk atmospheric deposition.
- **Conduct Sammamish River subbasin assessment**. On average, the Sammamish River represented about 45% of the surface water total PFAS instantaneous load contribution to Lake Washington. Further assessment is needed to:
    - better understand PFAS sources and pathways within the Sammamish subbasin,
        - and whether those differ from other subbasins within the Greater Lake Washington watershed.
- **Delineate PFAS sources within urban residential and commercial land uses.** Assessing current and historical land uses was important in understanding PFAS levels in the watershed. Current land use surrounding the lake is predominantly urban residential and commercial. Further delineation within these categories, and further assessment of historical land uses will provide better understanding about sources and pathways in the watershed.
- **Reduce nonpoint pollution**. Addressing PFAS in fish from Lake Washington will in part involve continued strategies to address nonpoint pollution in the watershed, including runoff from impervious surfaces, and further assessments of more localized potential sources areas.
- **Conduct assessment near Cedar River mouth**. Results indicated PFAS sources near the Cedar River mouth could be AFFF-related, and/or related to industrial activities. Further assessment of this area could include:
    - identifying current and/or historical land use practices which may be contributing PFAS to the river,
    - and delineating the extent of the groundwater plume along the river — confirming whether AFFF-related activities have recently or historically occurred (use or spill).
- **Analyze PFAS in Lake Washington sediment cores.** Analyses of PFAS in sediment core samples from Lake Washington would provide greater understanding of historical and future changes in PFAS levels over time — as PFAS regulations have shifted along with their use and manufacture. For example, we may gain understanding about how shifts away from legacy PFAS such as PFOS to shorter-chain PFAS and PFAS precursors are reflected in the local environment.
- **Gather further information about current and historical AFFF uses and practices.** AFFF use has been the most identified source of PFAS impacting groundwater in Washington State. Over 60% of Washington residents get their drinking water from groundwater.

Ecology's AFFF collection and disposal program[1] has been working with fire departments and training facilities to collect and dispose of stockpiles of AFFF. These efforts focused on current stockpiles.

  - Additional information from municipalities, fire departments, fire training facilities, airports and other facilities regarding AFFF equipment maintenance practices and current and historic use of AFFF would help identify potential source areas.
  - Department of Energy's Guide for Investigating Historic and Current Uses[2] could be used as a starting point to design a questionnaire for facilities with likely or known AFFF use/storage throughout Washington state.

- **Investigate PFAS use, storage, and potential for release by facility-type**. PFAS are used in thousands of processes and products but very little is known about the relationship between facilities that use PFAS-containing substances and subsurface releases that have the potential to impact groundwater. Future studies investigating PFAS releases by facility-type and potential for migration of PFAS to groundwater could help identify and prioritize releases with the potential to impact drinking water sources.
- **Assess PFAS levels at contaminated sites in non-industrial urban areas**. Dry cleaning facilities and automotive facilities (gas station, auto repair facilities, body shops) are the most common type of contaminated sites in Washington in non-industrial urban areas. A pilot-study in Florida demonstrated that chlorinated solvent plumes associated with dry cleaning facilities often have a co-located PFAS plume (Barnes et al. 2021). Further assessment of the prevalence of PFAS associated with these types of releases would further our understanding of urban sources, particularly in residential areas.
- **Quantify PFAS mass discharge between groundwater and surface water**. A follow-up study quantifying PFAS mass discharge between groundwater to surface water in the Greater Lake Washington watershed would further our understanding of how the groundwater pathway contributes to PFAS load in Lake Washington and its tributaries.
- **Conduct PFAS bioaccumulation/biomagnification study in Lake Washington**. Since this study focused on assessing sources and pathways of PFAS to Lake Washington. A study focused on dietary transfer and food web dynamics within the lake would provide useful information on how PFAS accumulate and biomagnify in Lake Washington fish, which would be valuable for assessing risks to ecological and human health.
- **Monitor PFAS levels in Lake Washington and its watershed over time**. Our study provided a thorough dataset of PFAS concentrations in the Greater Lake Washington watershed and in various sample media. Future monitoring of PFAS levels in Lake Washington, its fish, and surrounding watershed will help to assess changes over time. The resulting dataset may provide useful information for other studies of PFAS contamination in urban watersheds.

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[1] https://ecology.wa.gov/waste-toxics/reducing-toxic-chemicals/product-replacement-program/afff-disposal#:~:text=AFFF%20is%20used%20to%20fight,contamination%20in%20our%20drinking%20water

[2] https://www.energy.gov/sites/default/files/2023-02/Final%20Appendix%20A%20Survey%20Questions-Historical_.pdf

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Glossary, Acronyms, and Abbreviations

Glossary

Alluvium: loose clay, silt, sand, or gravel that has been deposited by running water in a stream bed

Analyte: A chemical substance that is the subject of chemical analysis

Aquitard: a geologic formation or stratum that lies adjacent to an aquifer and that allows only a small amount of liquid to pass.

Bulk Atmospheric Deposition: sum of wet and dry fractions of atmospheric deposition

Clean Water Act: A federal act passed in 1972 that contains provisions to restore and maintain the quality of the nation's waters. Section 303(d) of the Clean Water Act establishes the TMDL program.

Conductivity: A measure of water's ability to conduct an electrical current. Conductivity is related to the concentration and charge of dissolved ions in water.

Dissolved oxygen (DO): A measure of the amount of oxygen dissolved in water.

Effluent: An outflowing of water from a natural body of water or from a man-made structure. For example, the treated outflow from a wastewater treatment plant.

Eutrophication: excessive richness of nutrients in a lake or other body of water, frequently due to runoff from the land, which causes a dense growth of plant life and death of animal life from lack of oxygen

Glaciomarine: describing an environment containing both glacial ice and marine water

Hydraulic head: a measure of the potential energy stored in the fluid

Lacustrine: relating to or associated with lakes

Nonpoint source: Pollution that enters any waters of the state from any dispersed land-based or water-based activities, including but not limited to atmospheric deposition, surface-water runoff from agricultural lands, urban areas, or forest lands, subsurface or underground sources, or discharges from boats or marine vessels not otherwise regulated under the NPDES program. Generally, any unconfined and diffuse source of contamination. Legally, any source of water pollution that does not meet the legal definition of "point source" in section 502(14) of the Clean Water Act.

Parameter: Water quality constituent being measured (analyte). A physical, chemical, or biological property whose values determine environmental characteristics or behavior.

pH: A measure of the acidity or alkalinity of water. A low pH value (0 to 7) indicates that an acidic condition is present, while a high pH (7 to 14) indicates a basic or alkaline condition. A pH of 7 is considered neutral. Since the pH scale is logarithmic, a water sample with a pH of 8 is ten times more basic than one with a pH of 7.

Point source: Sources of pollution that discharge at a specific location from pipes, outfalls, and conveyance channels to surface water. Examples of point source discharges include municipal wastewater treatment plants, municipal stormwater systems, industrial wastewater treatment facilities, and construction sites where one or more acres of land are disturbed.

Pollution: Contamination or other alteration of the physical, chemical, or biological properties of any waters of the state. This includes change in temperature, taste, color, turbidity, or odor of the waters. It also includes discharge of any liquid, gaseous, solid, radioactive, or other substance into any waters of the state. This definition assumes that these changes will, or are likely to, create a nuisance or render such waters harmful, detrimental, or injurious to (1) public health, safety, or welfare; (2) domestic, commercial, industrial, agricultural, recreational, or other legitimate beneficial uses; or (3) livestock, wild animals, birds, fish, or other aquatic life.

PushPoint Sampler: a device used to collect sediment pore water samples in the field, typically consisting of a pointed tubular stainless-steel tube with a screened zone at one end and a sampling port at the other

Stormwater: The portion of precipitation that does not naturally percolate into the ground or evaporate but instead runs off roads, pavement, and roofs during rainfall or snow melt. Stormwater can also come from hard or saturated grass surfaces such as lawns, pastures, playfields, and from gravel roads and parking lots.

TOP assay: a method used to convert oxidizable PFAS precursors into PFAAs, which are measured using a targeted PFAS analytical method

Watershed: A drainage area or basin in which all land and water areas drain or flow toward a central collector, such as a stream, river, or lake at a lower elevation.

303(d) list: Section 303(d) of the federal Clean Water Act requires Washington State to periodically prepare a list of all surface waters in the state for which beneficial uses of the water – such as for drinking, recreation, aquatic habitat, and industrial use – are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that fall short of state surface water quality standards and are not expected to improve within the next two years.

Acronyms and Abbreviations

AFFF	aqueous film-forming foam
AQP	Air Quality Program
ARFF	aircraft rescue and firefighting
BGS	below ground surface
DOC	dissolved organic carbon
EAP	Environmental Assessment Program
Ecology	Washington State Department of Ecology
EIM	Environmental Information Management database
EIS	Extracted Internal Standards
EPA	U.S. Environmental Protection Agency
HDPE	high-density polyethylene
HWTR	Hazardous Waste and Toxics Reduction program
IT	Information Technology
ITRC	Interstate Technology & Regulatory Council
MEL	Manchester Environmental Laboratory
MQO	measurement quality objective
MS/MSD	matrix spike/matrix spike duplicate

n	number of samples
NA	not applicable
ND	non-detect
ORP	oxidation reduction potential
PBDE	polybrominated diphenyl ethers
PCA	principal component analysis
PFAA	perfluoroalkyl acid
PFAS	per- and polyfluoroalkyl substances
PFCA	perfluoroalkyl carboxylic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonic acid
PFSA	perfluoroalkane sulfonic acid
pH	power of hydrogen
QAPP	quality assurance project plan
RPD	relative percent difference
SOP	standard operating procedures
TOC	total organic carbon
TOP assay	total oxidizable precursor assay
TSS	total suspended solids
USGS	U.S. Geological Survey
WSDOT	Washington State Department of Transportation
WQP	Water Quality Program
WRIA	Water Resource Inventory Area
YSI	Yellow Springs Instruments

Units of Measurement

°C	degrees centigrade
cfs	cubic feet per second
cm	centimeter
FNU	Formazin nephelometric units
ft	feet
g	gram, a unit of mass
km	kilometer, a unit of length equal to 1,000 meters
m	meter
mg	milligram
mg/L	milligrams per liter (parts per million)
mL	milliliters
mm	millimeters
mV	millivolts
ng/g	nanograms per gram (parts per billion)
ng/L	nanograms per liter (parts per trillion)
NTU	nephelometric turbidity units
µS/cm	microsiemens per centimeter, a unit of conductivity

Appendices

Appendices A – G

Appendix A. Target PFAS Analytes

Table A1. List of 40 target PFAS analytes.

Analyte Group	Analyte Name	Abbreviation	Carbon Chain Length	CAS Number
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorobutanoic acid	PFBA	C4	375-22-4
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluoropentanoic acid	PFPeA	C5	2706-90-3
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorohexanoic acid	PFHxA	C6	307-24-4
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluoroheptanoic acid	PFHpA	C7	375-85-9
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorooctanoic acid	PFOA	C8	335-67-1
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorononanoic acid	PFNA	C9	375-95-1
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorodecanoic acid	PFDA	C10	335-76-2
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluoroundecanoic acid	PFUnA	C11	2058-94-8
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorododecanoic acid	PFDoA	C12	307-55-1
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorotridecanoic acid	PFTrDA	C13	72629-94-8
Perfluoroalkyl carboxylic acids (PFCAs)	Perfluorotetradecanoic acid	PFTeDA	C14	376-06-7
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorobutanesulfonic acid	PFBS	C4	375-73-5
Perfluoroalkane sulfonic acids (PFSAs)	Perfluoropentanesulfonic acid	PFPeS	C5	2706-91-4
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorohexanesulfonic acid	PFHxS	C6	355-46-4
Perfluoroalkane sulfonic acids (PFSAs)	Perfluoroheptanesulfonic acid	PFHpS	C7	375-92-8
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorooctanesulfonic acid	PFOS	C8	1763-23-1
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorononanesulfonic acid	PFNS	C9	68259-12-1
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorodecanesulfonic acid	PFDS	C10	335-77-3
Perfluoroalkane sulfonic acids (PFSAs)	Perfluorododecanesulfonic acid	PFDoS	C12	79780-39-5
Fluorotelomer sulfonic acids	1H,1H, 2H, 2H-Perfluorohexane sulfonic acid	4:2 FTS	-	757124-72-4
Fluorotelomer sulfonic acids	1H,1H, 2H, 2H-Perfluorooctane sulfonic acid	6:2 FTS	-	27619-97-2
Fluorotelomer sulfonic acids	1H,1H, 2H, 2H-Perfluorodecane sulfonic acid	8:2 FTS	-	39108-34-4
Fluorotelomer carboxylic acids	3-Perfluoropropyl propanoic acid	3:3 FTCA	-	356-02-5

Analyte Group	Analyte Name	Abbreviation	Carbon Chain Length	CAS Number
Fluorotelomer carboxylic acids	2H,2H,3H,3H-Perfluorooctanoic acid	5:3 FTCA	-	914637-49-3
Fluorotelomer carboxylic acids	3-Perfluoroheptyl propanoic acid	7:3 FTCA	-	812-70-4
Perfluoroalkane sulfonamides	Perfluorooctanesulfonamide	PFOSA	-	754-91-6
Perfluoroalkane sulfonamides	N-methylperfluorooctanesulfonamide	N-MeFOSA	-	31506-32-8
Perfluoroalkane sulfonamides	N-ethylperfluorooctanesulfonamide	N-EtFOSA	-	4151-50-2
Perfluoroalkane sulfonamido acetic acids	N-methyl perfluorooctanesulfonamidoacetic acid	N-MeFOSAA	-	2355-31-9
Perfluoroalkane sulfonamido acetic acids	N-ethyl perfluorooctanesulfonamidoacetic acid	N-EtFOSAA	-	2991-50-6
Perfluorooctane sulfonamide ethanols	N-methylperfluorooctanesulfonamido ethanol	N-MeFOSE	-	24448-09-7
Perfluorooctane sulfonamide ethanols	N-ethylperfluorooctanesulfonamido ethanol	N-EtFOSE	-	1691-99-2
Ether acids	Perfluoro-3-methoxypropanoic acid	PFMPA	-	377-73-1
Ether acids	Perfluoro-4-methoxybutanoic acid	PFMBA	-	863090-89-5
Ether acids	Nonafluoro-3,6-dioxaheptanoic acid	NFDHA	-	151772-58-6
Ether acids	Hexafluoropropylene oxide dimer acid (GenX)	HFPO-DA	-	13252-13-6
Ether acids	4,8-Dioxa-3H-perfluorononanoic acid	ADONA	-	919005-14-4
Ether acids	Perfluoro(2-ethoxyethane) sulfonic acid	PFEESA	-	113507-82-7
Ether acids	9-Chlorohexadecafluoro-3-oxanonane1- sulfonic acid (F-53B Major)	9Cl-PF3ONS	-	756426-58-1
Ether acids	11-Chloroeicosafluoro-3-oxaundecane1- sulfonic acid (F-53B Minor)	11Cl-PF3OUdS	-	763051-92-9

PFAS = per- and polyfluoroalkyl substances. Chemical Abstracts Service Registry Number.

Appendix B. Phase 1 Sampling Sites

Table B1. Phase 1 lake, tributary, groundwater, and bulk atmospheric deposition sampling sites.

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)
Lake	Lake Washington	LAKE-MID	47.6365, -122.268611	Water (2), Sediment (1)
Lake	Lake Washington	LAKE-NORTH	47.68549, -122.23666	Water (2), Sediment (1)
Lake	Lake Washington	LAKE-SOUTH	47.57536, -122.26777	Water (2), Sediment (1)
Lake	Lake Washington	MC-OUTLET	47.64746, -122.3017637	Water (2)
Lake	Lake Washington	NS-ANDREWS	47.5581, -122.25964	Water (2), Sediment (1)
Lake	Lake Washington	NS-BAKER	47.58435, -122.28537	Water (2), Sediment (1)
Lake	Lake Washington	NS-CEDAR	47.50658, -122.21394	Water (2), Sediment (1)
Lake	Lake Washington	NS-COAL	47.5769, -122.19495	Water (2), Sediment (1)
Lake	Lake Washington	NS-JUANITA	47.69811, -122.220321	Water (2), Sediment (1)
Lake	Lake Washington	NS-KENMORE	47.753409, -122.262669	Water (2), Sediment (1)
Lake	Lake Washington	NS-LM	47.748433, -122.275166	Water (2), Sediment (1)
Lake	Lake Washington	NS-MADRONA	47.611176, -122.28084	Water (2), Sediment (1)
Lake	Lake Washington	NS-MAY	47.52723, -122.20898	Water (2), Sediment (1)
Lake	Lake Washington	NS-MEYDENBAUER	47.60962, -122.21257	Water (2), Sediment (1)
Lake	Lake Washington	NS-MOSS	47.67235, -122.2116	Water (2), Sediment (1)
Lake	Lake Washington	NS-PONTIAC	47.69492, -122.2683	Water (2), Sediment (1)
Lake	Lake Washington	NS-RAINIER	47.52211, -122.25922	Water (2), Sediment (1)
Lake	Lake Washington	NS-SEWARD	47.54593, -122.25438	Water (2), Sediment (1)
Lake	Lake Washington	NS-SHERIDAN	47.7375, -122.284621	Water (2), Sediment (1)
Lake	Lake Washington	NS-TAYLOR	47.51308, -122.245592	Water (2), Sediment (1)
Lake	Lake Washington	NS-UNION	47.65159, -122.29081	Water (2)
Lake	Lake Washington	NS-WOLF	47.66324, -122.26458	Water (2), Sediment (1)
Lake	Lake Washington	NS-YARROW	47.652474, -122.209491	Water (2), Sediment (1)
Tributary	Cedar River	CR-CGNA	47.462542, -122.0890528	Water (2), Biofilm (1)
Tributary	Cedar River	CR-LANDSBURG	47.374945, -121.971843	Water (2), Biofilm (1)

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)
Tributary	Cedar River	CR-MOUTH	47.49513, -122.213614	Water (2)
Tributary	Cedar River	CR-USGS-GAGE	47.482306, -122.202778	Water (7), Biofilm (1)
Tributary	Coal Creek	CC-MOUTH	47.566686, -122.180519	Water (2), Sediment (1)
Tributary	Coal Creek	CC-NEWCASTLE	47.537141, -122.131959	Water (2), Sediment (1)
Tributary	Denny Creek	DC-MOUTH	47.708849, -122.250522	Water (2), Sediment (1)
Tributary	Fairweather Creek	FWC-MOUTH	47.636766, -122.230779	Water (2)
Tributary	Forbes Creek	FC-108TH	47.695922, -122.19755	Water (2)
Tributary	Juanita Creek	JC-MOUTH	47.705022, -122.216747	Water (7), Sediment (1), Biofilm (1)
Tributary	Juanita Creek	JC-WINDSOR	47.730768, -122.194051	Water (2), Sediment (1)
Tributary	Kelsey Creek	KC-405	47.6015134, -122.184517	Water (2)
Tributary	Kelsey Creek	KC-KCP	47.6060245, -122.1623475	Water (2), Sediment (1)
Tributary	Kelsey Creek	M-SLOUGH	47.5821, -122.186264	Water (2)
Tributary	Lyons Creek	LC-MOUTH	47.752901, -122.277207	Water (2)
Tributary	Lyons Creek	LC-TERRACE	47.789913, -122.298133	Water (2), Sediment (1)
Tributary	Mapes Creek	MAPES-MOUTH	47.523462, -122.264219	Water (2), Sediment (1)
Tributary	May Creek	MAY-MOUTH	47.53002, -122.20081	Water (7), Sediment (1)
Tributary	May Creek	MAY-NILE	47.51529, -122.14326	Water (2)
Tributary	McAleer Creek	MC-196TH	47.7705851, -122.3114713	Water (2), Sediment (1)
Tributary	McAleer Creek	MC-MOUTH	47.752079, -122.281876	Water (2), Sediment (1)
Tributary	Ravenna Creek	RC-WAHKIAKUM	47.658005, -122.296513	Water (2), Sediment (1)
Tributary	Sammamish River	SR-145TH	47.732761, -122.145717	Water (2), Sediment (1), Biofilm (1)
Tributary	Sammamish River	SR-96TH	47.748167, -122.213333	Water (7)
Tributary	Sammamish River	SR-MARYMOOR	47.662275, -122.124367	Water (2), Biofilm (1)
Tributary	Sammamish River	SR-MOUTH	47.7543874, -122.2495176	Water (2), Sediment (1)
Tributary	Taylor Creek	TC-LAKERIDGE	47.50821, -122.24792	Water (2)
Tributary	Thornton Creek	TC-MOUTH	47.695957, -122.275806	Water (7)
Tributary	Thornton Creek	TC-N-10TH	47.725131, -122.319932	Water (2), Sediment (1)
Tributary	Thornton Creek	TC-S-5TH	47.704162, -122.3223024	Water (2)

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)
Tributary	Yarrow Creek	YC-101ST	47.643769, -122.205626	Water (2)
Tributary	Yarrow Creek	YC-34TH	47.6401347, -122.1857002	Water (2), Sediment (1)
Groundwater	Cedar River	CR-1	47.495456, -122.213806	Water (1)
Groundwater	Cedar River	CR-2	47.497414, -122.214377	Water (1)
Groundwater	Cedar River	CR-3	47.492896, -122.212856	Water (1)
Groundwater	Lake Washington	GC2	47.504585, -122.203108	Water (1)
Groundwater	Lake Washington	CHISM	47.599751, -122.210887	Water (1)
Groundwater	Lake Washington	HBP	47.661043, -122.207281	Water (1)
Groundwater	Lake Washington	JBP	47.703911, -122.215164	Water (1)
Groundwater	Lake Washington	KENNYDALE	47.523048, -122.207994	Water (1)
Groundwater	Lake Washington	LB	47.757573, -122.265619	Water (1)
Groundwater	Lake Washington	LCWP	47.752776, -122.274394	Water (1)
Groundwater	Lake Washington	LFPWD	47.763084, -122.279484	Water (1)
Groundwater	Lake Washington	MADRONA	47.609257, -122.282376	Water (1)
Groundwater	Lake Washington	MB	47.694947, -122.271522	Water (1)
Groundwater	Lake Washington	MBAY	47.61126, -122.211857	Water (1)
Groundwater	Lake Washington	NEWCASTLE	47.566617, -122.191589	Water (1)
Groundwater	Lake Washington	WETMORE	47.570296, -122.278652	Water (1)
Groundwater	Lake Washington	SE1	47.729146, -122.263078	Water (1)
Groundwater	Lake Washington	SE2	47.731724, -122.263688	Water (1)
Groundwater	Lake Washington	SE3	47.733634, -122.26491	Water (1)
Bulk Atmospheric Deposition	Beacon Hill Air Quality Station	BH-ATM	47.568228, -122.308639	Water (4)

Appendix C. Phase 2 Sampling Sites

Table C1. Phase 2 lake, tributary, groundwater, and bulk atmospheric deposition sampling sites.

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)	Phase 1 Samling Site
Lake	Lake Washington	LAKE-MID	47.6365, -122.268611	Water (1)	X
Lake	Lake Washington	LAKE-NORTH	47.68549, -122.23666	Water (1)	X
Lake	Lake Washington	LAKE-SOUTH	47.57536, -122.26777	Water (1)	X
Lake	Lake Washington	MADRONA-SW	47.609294, -122.282355	Water (1)	
Lake	Lake Washington	WETMORE-SW	47.570336, -122.279044	Water (1)	
Tributary	Cedar River	CR-CGNA	47.462542, -122.089053	Water (2), Biofilm (1), Macroinvertebrate (1)	x
Tributary	Cedar River	CR-CML	47.36958, -121.62443	Water (1)	
Tributary	Cedar River	CR-CRP	47.465538, -122.125549	Water (2)	
Tributary	Cedar River	CR-FVHRP	47.405995, -122.039295	Water (2), Biofilm (1), Macroinvertebrate (1)	
Tributary	Cedar River	CR-LANDSBURG	47.374945, -121.971843	Water (2), Biofilm (1), Macroinvertebrate (1)	X
Tributary	Cedar River	CR-LPNA	47.4395, -122.06641	Water (2)	
Tributary	Cedar River	CR-NF2252	47.414966, -121.791567	Water (1)	
Tributary	Cedar River	CR-RRP	47.469633, -122.1504774	Water (2), Biofilm (1), Macroinvertebrate (1)	
Tributary	Cedar River	CR-RVP	47.476822, -122.17981	Water (2), Biofilm (1), Macroinvertebrate (1), Suspended Sediment (2)	
Tributary	Cedar River	CR-USGS GAGE	47.482306, -122.202778	Water (5), Biofilm (1), Macroinvertebrate (1)	X
Tributary	Juanita Creek	JC-NE129TH	47.71621, -122.20696	Water (2), Suspended Sediment (2)	

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)	Phase 1 Samling Site
Tributary	Juanita Creek	JC-MOUTH	47.705022, -122.216747	Water (5)	X
Tributary	Juanita Creek	JC-NE132ND	47.720763, -122.201284	Water (2)	
Tributary	Juanita Creek	JC-SOURCE	47.73546, -122.17933	Water (2)	
Tributary	Juanita Creek	JC-WINDSOR	47.730768, -122.194051	Water (2)	X
Tributary	Thornton Creek	TC-BELOW-MBP	47.704678, -122.285424	Water (2), Suspended Sediment (2)	
Tributary	Thornton Creek	TC-MOUTH	47.695957, -122.275806	Water (5)	X
Tributary	Thornton Creek	TC-N-10TH	47.725131, -122.319932	Water (2)	X
Tributary	Thornton Creek	TC-N-AB-MBP	47.708299, -122.289946	Water (2)	
Tributary	Thornton Creek	TC-N-RONALD	47.754911, -122.332348	Water (2)	
Tributary	Thornton Creek	TC-N-TWIN	47.7385022, -122.3294508	Water (2)	
Tributary	Thornton Creek	TC-S-5TH	47.704162, -122.322302	Water (2)	X
Tributary	Thornton Creek	TC-S-AB-MBP	47.70689, -122.29608	Water (2)	
Tributary	Thornton Creek	TC-S-BARTON	47.7026325, -122.3343991	Water (2)	
Tributary	Thornton Creek	TC-S-KNA	47.703255, -122.309576	Water (2)	
Tributary	Fairweather Creek	FWC-MOUTH	47.636766, -122.230779	Water (1)	X
Tributary	Fairweather Creek	FWC-NE24TH	47.632825, -122.230376	Water (1)	
Tributary	Hall Creek	HC-BALL-IN	47.7891542, -122.3301083	Water (1)	
Tributary	Hall Creek	HC-HALL-LK-OUT	47.807799, -122.311713	Water (1)	
Tributary	Kelsey Creek	KC-405	47.601513, -122.184517	Water (1)	X
Tributary	Kelsey Creek	KC-ABOVE-KCP	47.6195237, -122.1616649	Water (1)	
Tributary	Kelsey Creek	KC-ABOVE-VC	47.624178, -122.152918	Water (1)	
Tributary	Kelsey Creek	KC-KCP	47.606024, -122.162347	Water (1)	X
Tributary	Kelsey Creek	KC-LARSEN	47.60603, -122.140508	Water (1)	
Tributary	Kelsey Creek	KC-PHANTOM	47.5926116, -122.1290016	Water (1)	
Tributary	Kelsey Creek	M-SLOUGH	47.57964, -122.18757	Water (1)	X
Tributary	McAleer Creek	MC-196TH	47.770585, -122.311471	Water (1)	X

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)	Phase 1 Samling Site
Tributary	McAleer Creek	MC-BALL-OUT	47.7756374, -122.3159586	Water (1)	
Tributary	McAleer Creek	MC-MOUTH	47.752079, -122.281876	Water (1)	X
Tributary	Ravenna Creek	RC-CARP	47.654697, -122.295149	Water (1)	
Tributary	Ravenna Creek	RC-CENTRAL	47.654835, -122.293093	Water (1)	
Tributary	Ravenna Creek	RC-MONTLAKE	47.660877, -122.297846	Water (1)	
Tributary	Ravenna Creek	RCr-MOUTH	47.65602, -122.29699	Water (1)	X
Tributary	Ravenna Creek	RC-RP	47.671979, -122.306503	Water (1)	
Tributary	Ravenna Creek	RC-WAHKIAKUM	47.658005, -122.296513	Water (1)	X
Tributary	Valley Creek	VC-ABOVE-BGC	47.6557439, -122.1534627	Water (1)	
Tributary	Valley Creek	VC-BELOW-BGC	47.652561, -122.150239	Water (1)	
Tributary	Valley Creek	VC-MOUTH	47.62438, -122.1534	Water (1)	
Groundwater	Lake Washington	BAKER	47.5839264, -122.2876144	Water (2)	
Groundwater	Lake Washington	HARRISON	47.622526, -122.282338	Water (2)	
Groundwater	Lake Washington	JBP	47.703911, -122.215164	Water (1)	X
Groundwater	Lake Washington	JBP2	47.703847, -122.214796	Water (1)	
Groundwater	Lake Washington	KING-ST	47.5979534, -122.2863373	Water (2)	
Groundwater	Lake Washington	MADRONA	47.609294, -122.282355	Water (2)	X
Groundwater	Lake Washington	MB	47.694947, -122.271522	Water (2)	X
Groundwater	Lake Washington	WETMORE-PH2-N	47.570326, -122.279053	Water (1)	
Groundwater	Lake Washington	WETMORE-PH2-S	47.5702905, -122.2790133	Water (1)	
Groundwater	Cedar River	CR-1-PH2	47.4953268, -122.213751	Water (2)	
Groundwater	Cedar River	CR-4-PH2N	47.49171, -122.212422	Water (1)	
Groundwater	Cedar River	CR-4-PH2S	47.4916177, -122.2123861	Water (1)	
Groundwater	Cedar River	CR-5	47.491063, -122.2122178	Water (2)	
Groundwater	Cedar River	CR-6	47.490697, -122.212077	Water (1)	
Groundwater	Cedar River	CR-7	47.490194, -122.211889	Water (1)	
Groundwater	Cedar River	CR-8	47.489755, -122.211702	Water (1)	

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)	Phase 1 Samling Site
Groundwater	Cedar River	CR-BBNA-DD	47.384922, -121.991278	Water (1)	
Groundwater	Cedar River	CR-FVHRP-GW	47.4046255, -122.037829	Water (2)	
Groundwater	Cedar River	CR-LANDSBURG-GW	47.375029, -121.97198	Water (2)	
Groundwater	Cedar River	CR-LPNA-GW	47.445478, -122.06903	Water (2)	
Groundwater	Cedar River	CR-LPNA-SP	47.442515, -122.066966	Water (2)	
Groundwater	Cedar River	LRNA-SP	47.37535, -121.97624	Water (2)	
Groundwater	Juanita Creek	JC-MOUTH-GW	47.705048, -122.216755	Water (2)	
Groundwater	Juanita Creek	JC-SOURCE-GW	47.735488, -122.179416	Water (2)	
Groundwater	Juanita Creek	JC-WINDSOR-GW	47.7305774, -122.1940854	Water (2)	
Groundwater	Juanita Creek	JC-WINDSOR-GW2	47.730651, -122.193968	Water (1)	
Groundwater	Thornton Creek	TC-BELOW-MBP-GW	47.704834, -122.285543	Water (2)	
Groundwater	Thornton Creek	TC-MOUTH-GW	47.695957, -122.275806	Water (1)	
Groundwater	Thornton Creek	TC-N-10TH-GW	47.725012, -122.320056	Water (2)	
Groundwater	Thornton Creek	TC-N-167THST-GW	47.750313, -122.3329015	Water (2)	
Groundwater	Thornton Creek	TC-N-AB-MBP-GW	47.708557, -122.2899295	Water (2)	
Groundwater	Thornton Creek	TC-N-TWIN-GW-N	47.7385495, -122.3294874	Water (1)	
Groundwater	Thornton Creek	TC-N-TWIN-GW-S	47.738477, -122.329487	Water (1)	
Groundwater	Thornton Creek	TC-S-5TH-GW	47.704162, -122.322302	Water (2)	
Groundwater	Thornton Creek	TC-S-AB-MBP-GW	47.706892, -122.293274	Water (1)	
Groundwater	Thornton Creek	TC-S-KNA-SP	47.703203, -122.309414	Water (2)	
Groundwater	Ravenna Creek	RC-MONTLAKE-GW	47.660906, -122.29764	Water (2)	
Groundwater	Ravenna Creek	RC-MOUTH-GW	47.656112, -122.2969921	Water (2)	
Groundwater	Ravenna Creek	RC-RP-GW	47.6716117, -122.303999	Water (2)	
Groundwater	Ravenna Creek	RC-WAHKIAKUM-GW	47.658002, -122.296567	Water (2)	
Groundwater	Monitoring Well	RENTON-MW10	47.4871264, -122.2004993	Water (2)	
Groundwater	Monitoring Well	RENTON-MW11	47.4857696, -122.2043975	Water (2)	
Groundwater	Monitoring Well	RENTON-MW25S	47.4821768, -122.2054857	Water (2)	

Location Type	General Location	Location ID	Latitude, Longitude (WGS84)	Matrices Collected (Number of Events)	Phase 1 Samling Site
Groundwater	Monitoring Well	RENTON-MW26R	47.4783596, -122.2071561	Water (2)	
Groundwater	Monitoring Well	RENTON-MW3	47.4839321, -122.2037724	Water (2)	
Groundwater	Monitoring Well	RENTON-MW30S	47.485064, -122.20615	Water (2)	
Groundwater	Monitoring Well	RENTON-MW31S	47.4807941, -122.1952848	Water (2)	
Groundwater	Monitoring Well	RENTON-MW33	47.476543, -122.181488	Water (2)	
Groundwater	Monitoring Well	RENTON-MW36S	47.4738061, -122.1693381	Water (2)	
Groundwater	Monitoring Well	RENTON-MW4	47.4849903, -122.2001456	Water (2)	
Groundwater	Monitoring Well	RENTON-MW42	47.4809364, -122.1961173	Water (2)	
Groundwater	Monitoring Well	RENTON-MW5	47.4857958, -122.1970079	Water (2)	
Groundwater	Monitoring Well	NSEATTLE-MW	47.70088, -122.331797	Water (1)	
Groundwater	Monitoring Well	UW-MW1	47.658442, -122.292447	Water (1)	
Groundwater	Monitoring Well	UW-MW2	47.658126, -122.292754	Water (1)	
Groundwater	Spring	FRINK-SP	47.599686, -122.290343	Water (2)	
Groundwater	Spring	LICTON-SP	47.699093, -122.339202	Water (2)	
Groundwater	Spring	MADRONA-SP	47.611364, -122.28551	Water (2)	
Bulk Atmospheric Deposition	Beacon Hill Air Monitoring Station	BH-ATM	47.568278, -122.308889	Water (5)	X
Bulk Atmospheric Deposition	North Bend Ecology Air Monitoring Station	NB-ATM	47.4900742, -121.7729967	Water (5)	

Appendix D. Maps of Phase 2 Surface Water and Groundwater Sampling Sites

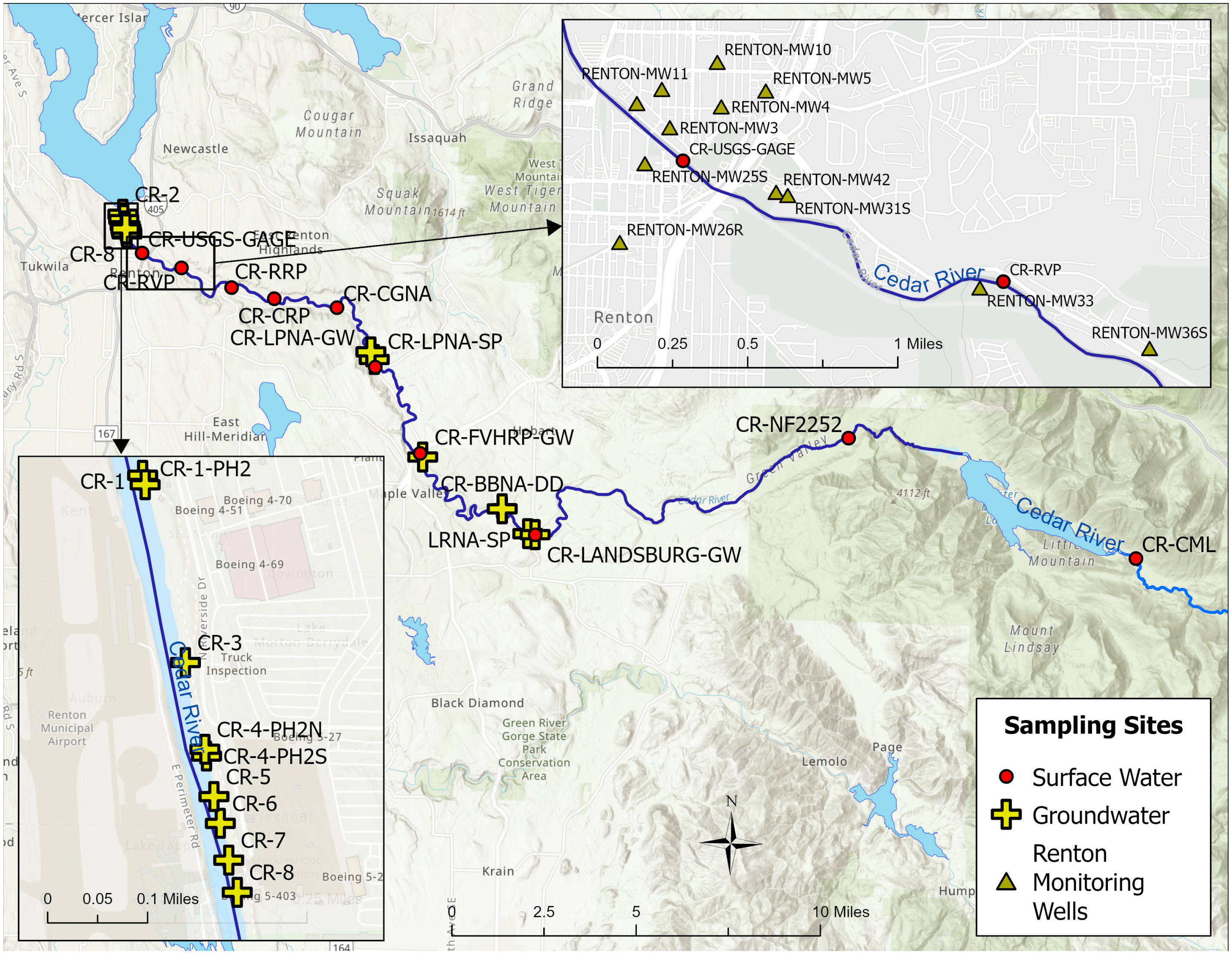

Figure D1. Map of Phase 2 surface water and groundwater sampling sites in the Cedar River subbasin.

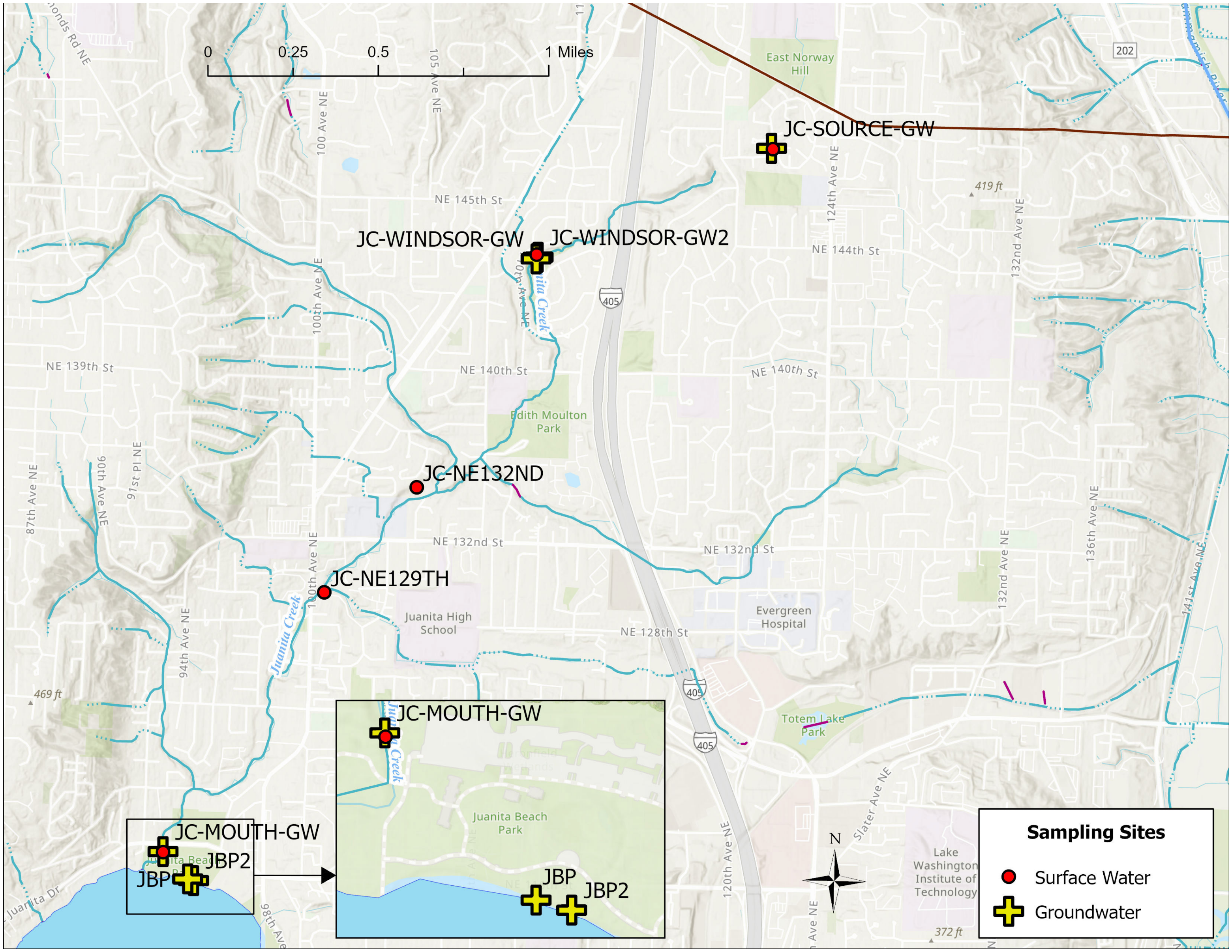

Figure D2. Map of Phase 2 surface water and groundwater sampling sites in the Juanita Creek subbasin.

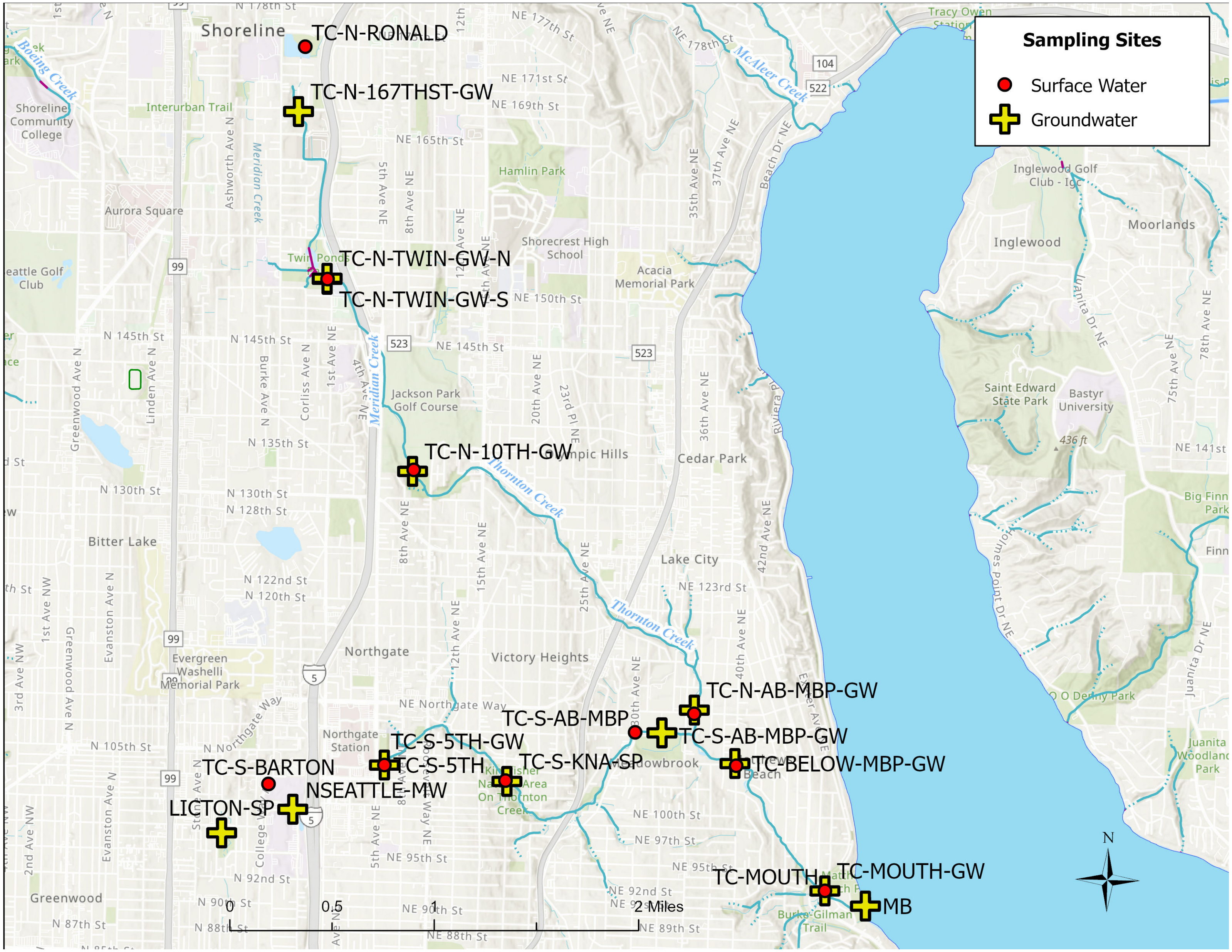

Figure D3. Map of Phase 2 surface water and groundwater sampling sites in the Thornton Creek subbasin.

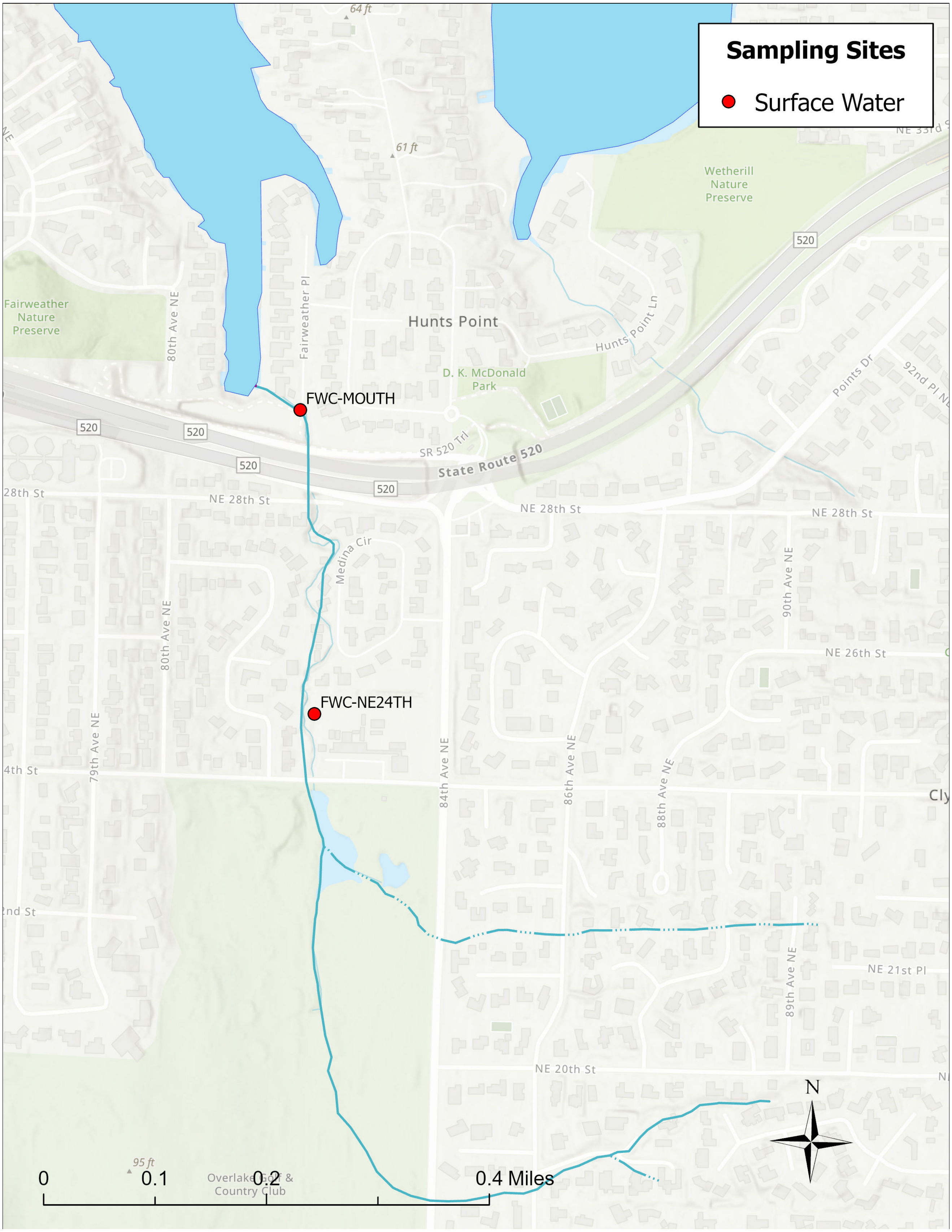

Figure D4. Map of Phase 2 surface water sampling sites in Fairweather Creek.

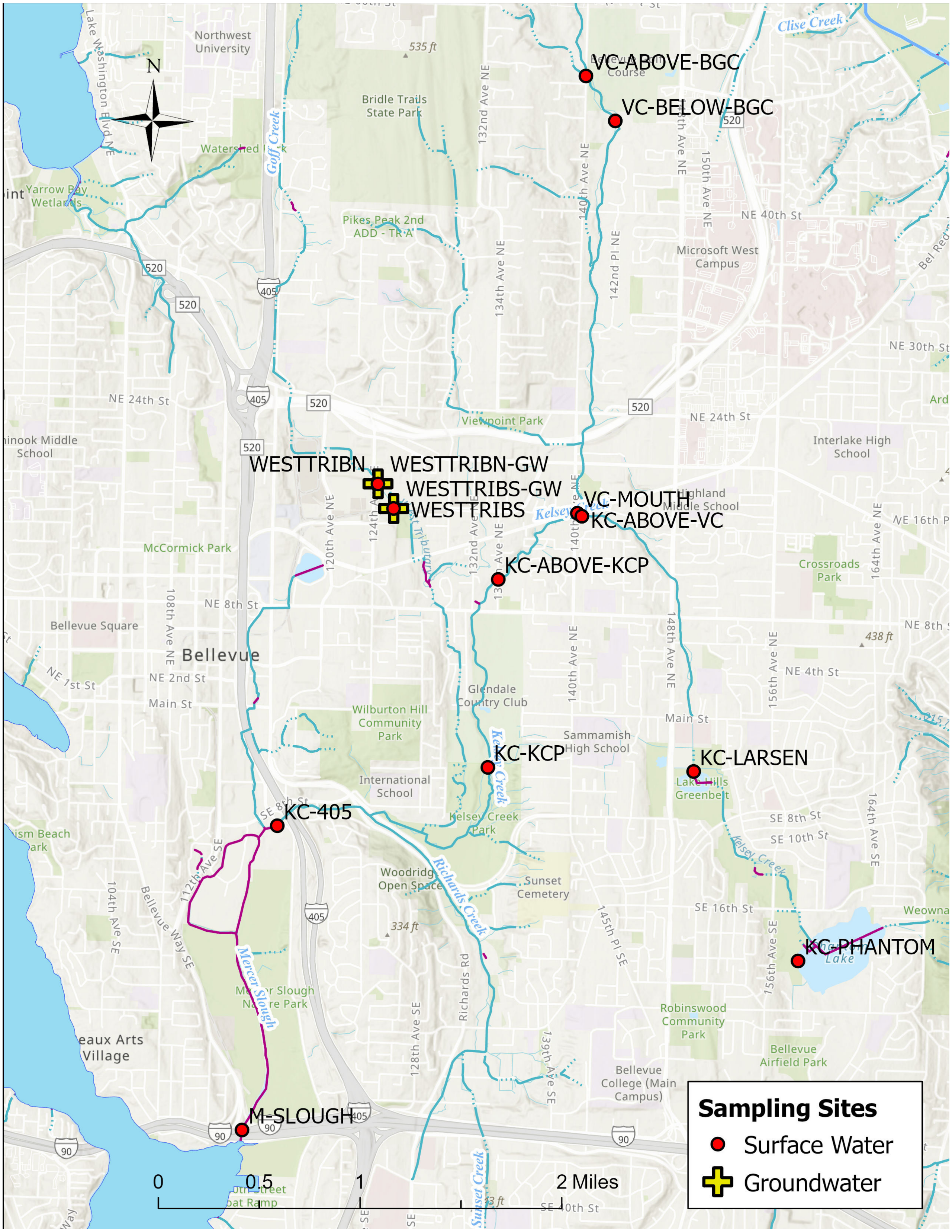

Figure D5. Map of Phase 2 surface water and groundwater sampling sites in the Kelsey Creek subbasin.

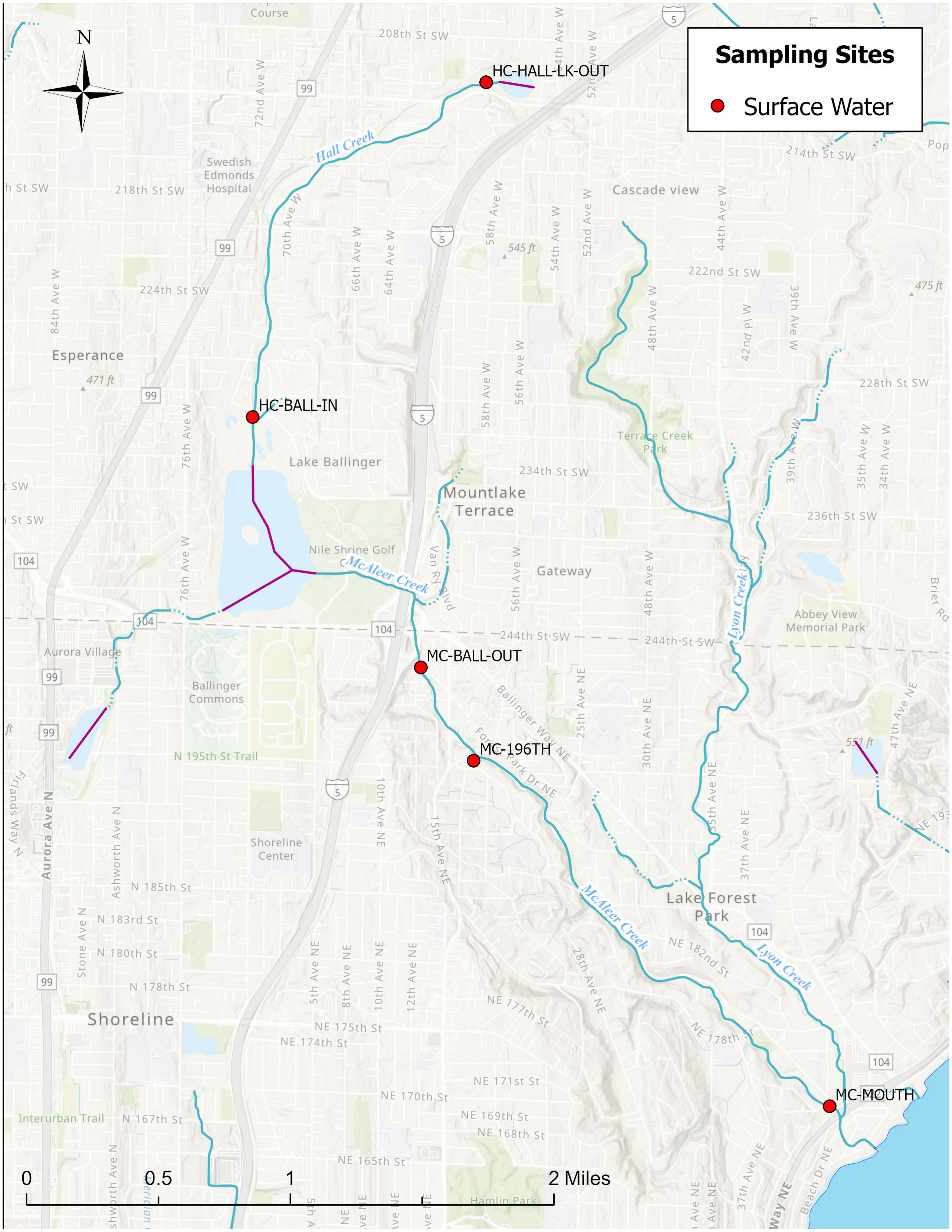

Figure D6. Map of Phase 2 surface water sampling sites in the McAleer Creek subbasin.

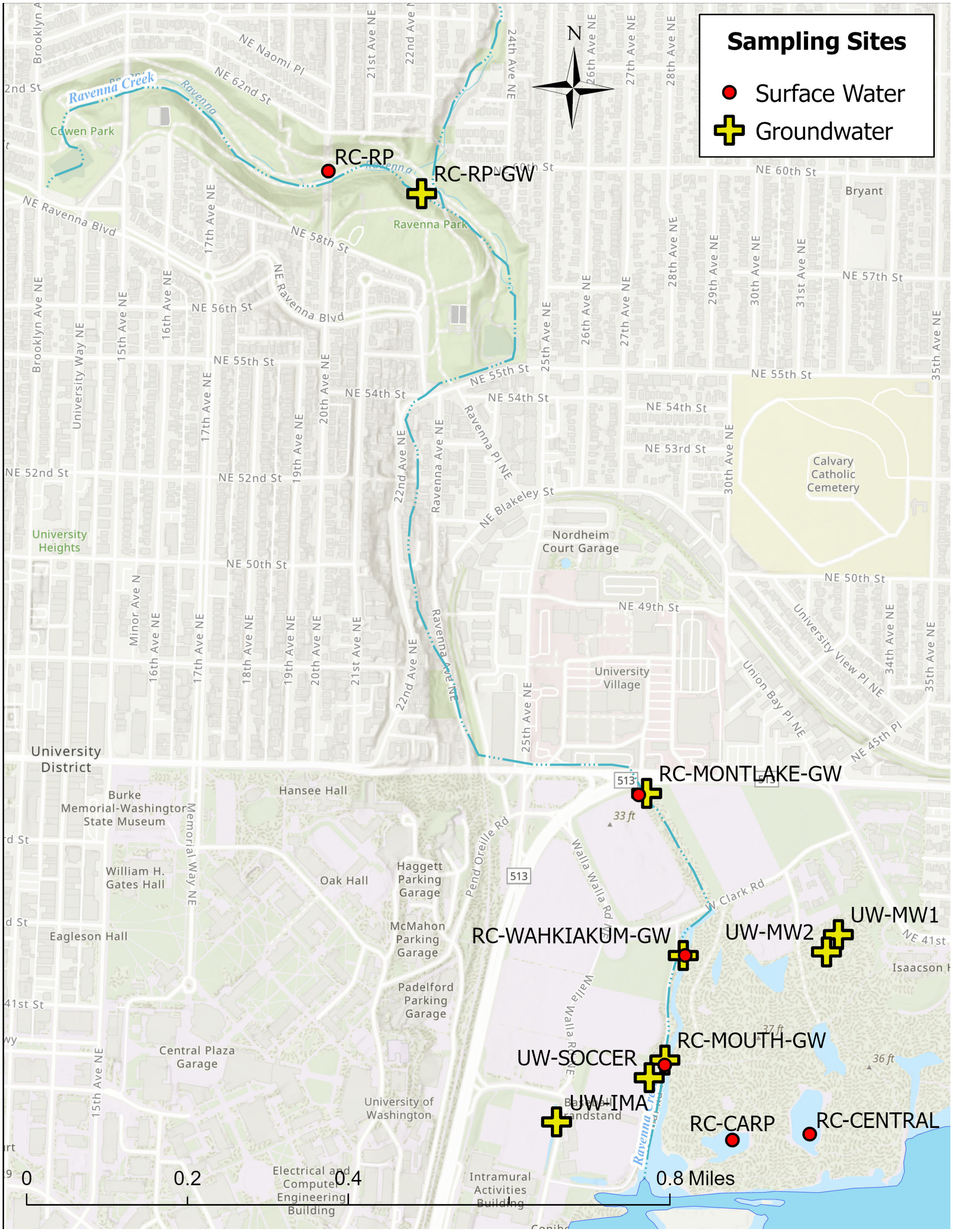

Figure D7. Map of Phase 2 surface water and groundwater sampling sites in the Ravenna Creek subbasin.

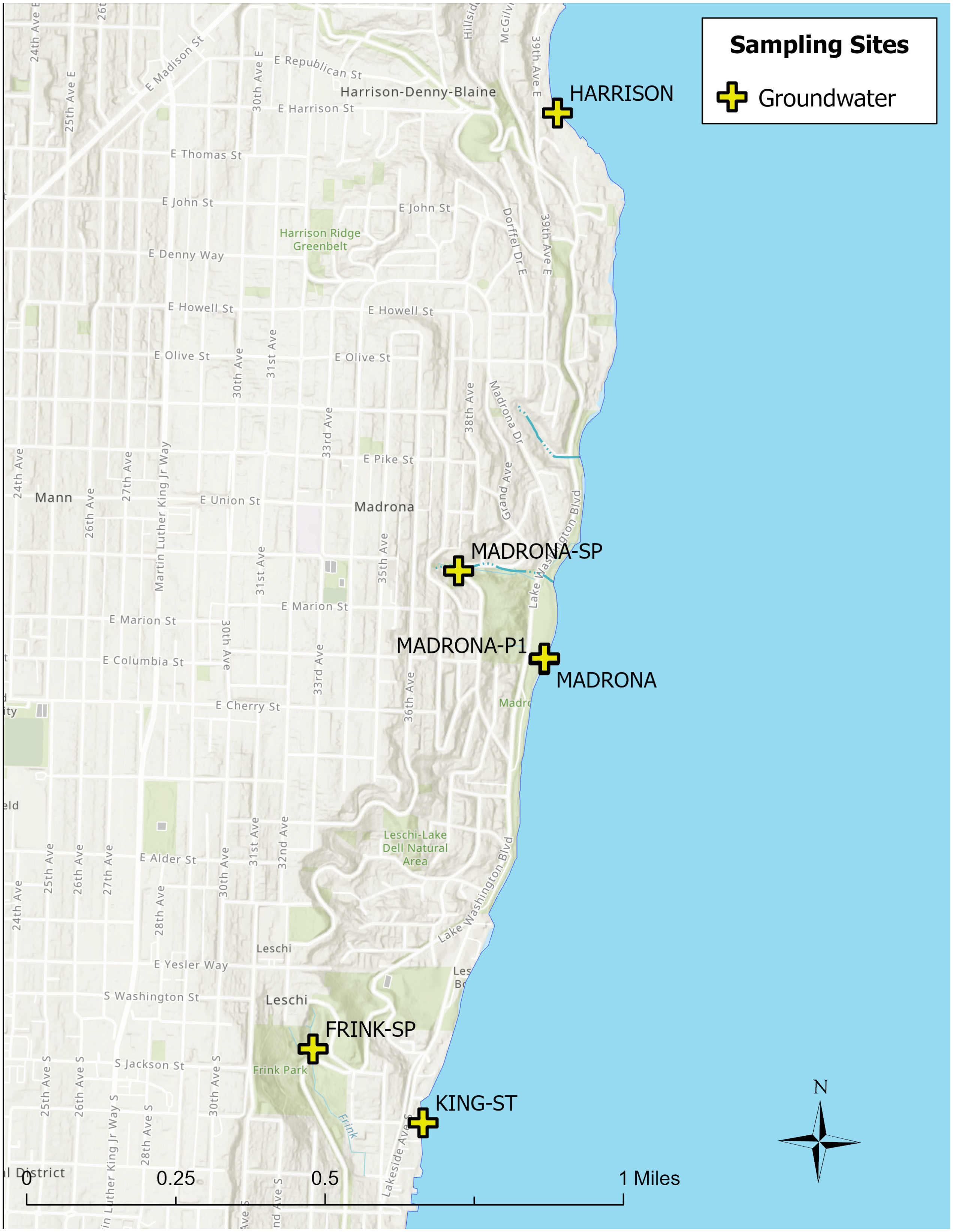

Figure D8. Map of Phase 2 groundwater sampling sites in the Madrona Beach area along the Lake Washington shoreline.

Appendix E. Correlation Between Sediment Grain Size and Total Organic Carbon Content

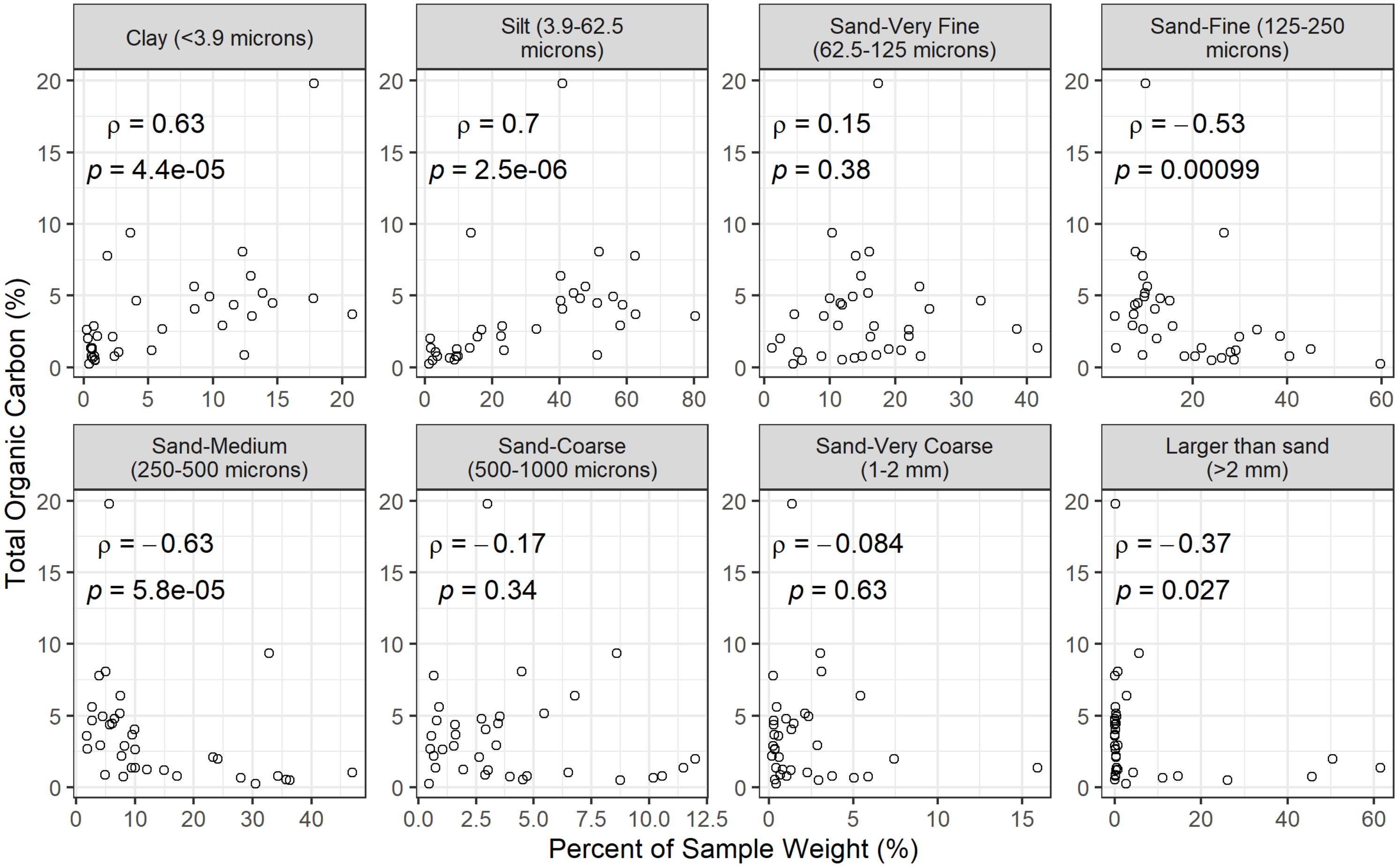

Figure E1. Correlations between total organic carbon content (%) and the percentage of each grain size class in lake and tributary sediment samples collected in the Greater Lake Washington watershed during Phase 1.

Spearman correlation (ρ) and significance values (p) are shown. Grain size classes are arranged small to large from top left to bottom right.

Appendix F. Sediment Grain Size Distribution for Phase 1 Samples

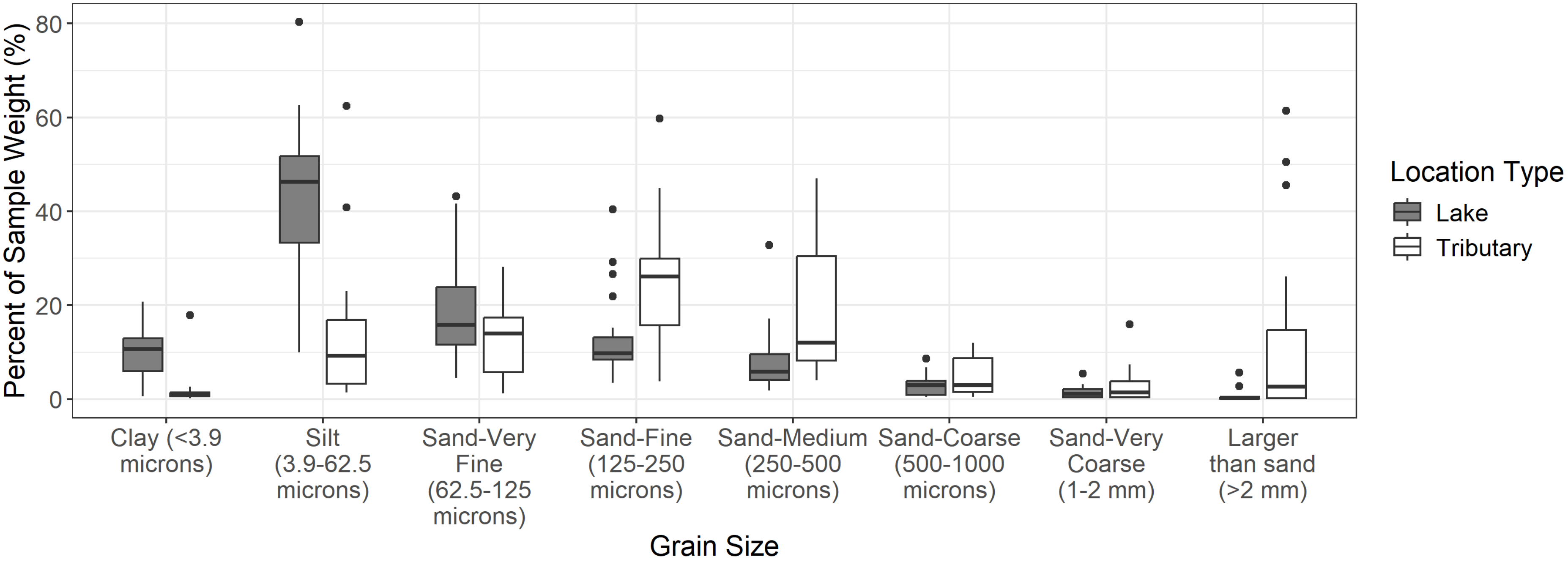

Figure F1. Grain size distribution of lake and tributary sediment samples (N=37) collected during Phase 1 in the Greater Lake Washington watershed

Appendix G. Sediment Grain Size Distribution for Phase 2 Samples

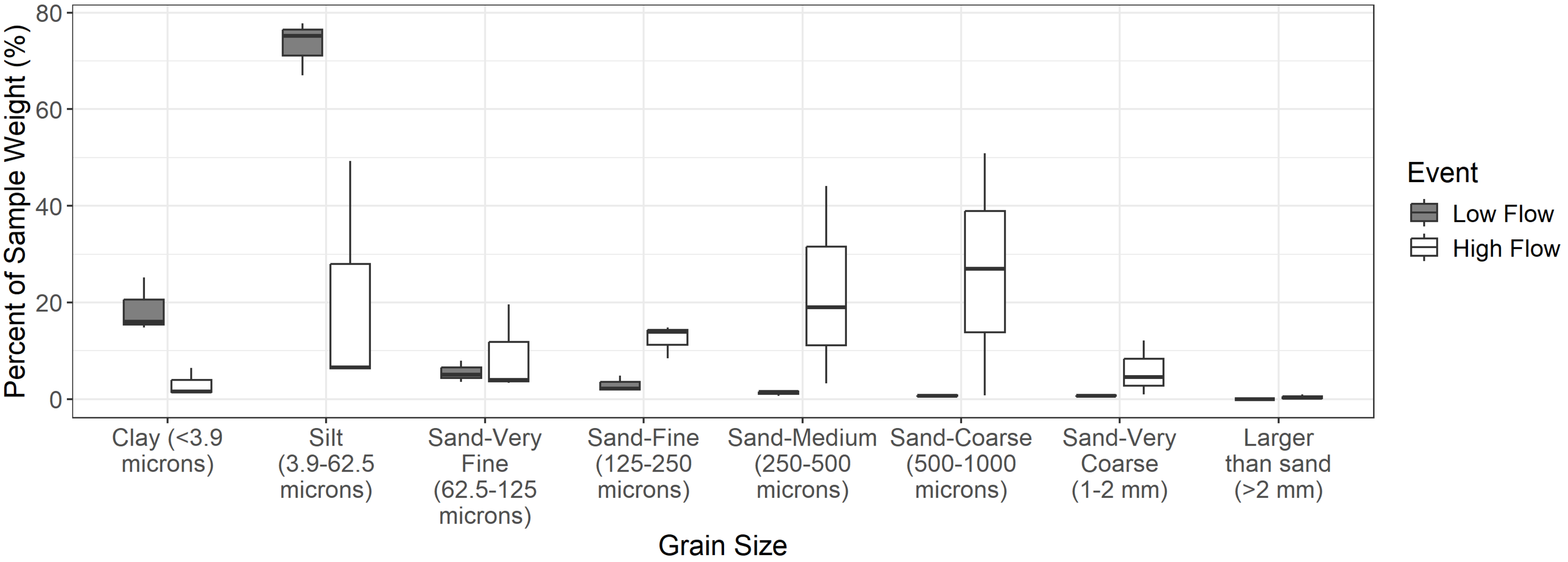

Figure G1. Grain size distribution of Phase 2 suspended sediment samples (N=6) collected from the Cedar River, Juanita Creek, and Thornton Creek.