

University of Nevada, Reno

**Per- and Polyfluoroalkyl Substances and Organofluorine in
Lakes and Waterways of the Northwestern Great Basin**

A thesis submitted in partial fulfillment of the
requirements for the degree of Master of Science in
Hydrology

by

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Abstract

Per- and polyfluoroalkyl substances (PFAS) are a class of thousands of anthropogenic chemicals that are ubiquitous in the environment and have been linked to numerous adverse health effects in humans and aquatic organisms. Known anthropogenic sources include wastewater treatment plant effluents, aqueous film forming foam (AFFF) fire retardants, urban runoff, and atmospheric deposition. Although numerous environmental monitoring studies have been conducted, only one has evaluated surface water contamination in the Northwestern Great Basin. I sought to close that knowledge gap by evaluating the occurrence of 19 PFAS using liquid chromatography tandem mass spectroscopy (LC-MS/MS) and extractable organofluorine (EOF) in grab samples from 15 lakes and 10 rivers throughout the region, including headwater lakes, rivers, and endorheic lakes that are the terminus of their watersheds. In this analysis I found that the highest levels of contamination were in sites near aqueous film forming foam (AFFF) training areas, the next highest near wastewater treatment plants, and slightly elevated concentrations near urban areas. I found evidence of PFAS accumulation in endorheic lakes by comparing inflow and lake body concentrations. I also showed that fluorotelomer sulfonate (FTS) species can be used as a tracer for AFFF impacts during source and transport pathway investigations. The results of my EOF analysis indicate that total PFAS loads are likely much higher than what is indicated by LC-MS/MS analysis, however it is difficult to state how much higher due to reliability issues with EOF analysis.

Dedication

I dedicate this thesis to the friends, family, loved ones, and colleagues who have supported me throughout my career and through this program. I couldn't have made it here or done this without you.

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Introduction

Per- and polyfluoroalkyl substances (PFAS) are a class of over 9000 synthetic chemicals that have been in production and use since the 1950s (ITRC, 2020). They are used in a wide range of industries such as hydrophobic and oleophobic treatments for textiles, as surfactants, non-stick coatings, and temperature resistant materials (Glüge, et al., 2020). PFAS are characterized by a completely (per-) or partially (poly-) fluorinated alkyl carbon chain (ITRC, 2020). Longer fluorinated chains cause PFAS to be more hydrophobic and shorter chains cause them to be less hydrophobic, while their functional groups can make them neutrally charged, anionic, cationic, or zwitterionic, in addition to having a wide range of molecular weights and structures (ITRC, 2020). The range of physiochemical properties of PFAS allow for several routes of environmental transport and accumulation in most environmental compartments (Ahrens & Bundschuh, 2014). The carbon-fluorine bonds in PFAS make them extremely recalcitrant, earning them the moniker “forever chemicals” (Hamid & Li, 2016; Ahrens & Bundschuh, 2014).

Globally persistent and present, PFAS are present in surface waters, groundwater, and the atmosphere (ITRC, 2020; Ahrens & Bundschuh, 2014; Tokranov, et al., 2021; Petre, et al., 2021). Extremely remote regions including snow and surface waters north of the Arctic Circle, the Himalayas, and the deep ocean contain PFAS (Miner, et al., 2020; Kwok, et al., 2013; Miranda, 2021; Yamashita, 2008). Furthermore, measured concentrations often exceed US EPA lifetime drinking water advisory levels (Podder, Sadmani, Reinhart, Chang, & Goel, 2021; EPA, 2022). Despite numerous studies documenting the widespread occurrence of PFAS in surface waters around the world I

only identified one peer-reviewed publication studying surface water contamination in the Great Basin, their associated endorheic (terminal) lakes which accumulate salt and may therefore accumulate recalcitrant compounds, and no publications for the Sierra Nevada (Bai & Son, 2021; Sharma & Hanigan, 2021). This region is home to over five million people, endemic and endangered wildlife species, agricultural operations, and a rapidly growing outdoor recreation sector, all of which would be negatively impacted by degraded water quality from PFAS contamination (Torregrosa & Devoe, 2008; Nevada Division of Natural Heritage, 2021; Chambers, Devoe, & Evenden, 2008).

PFAS have been shown to accumulate and cause health effects in humans and aquatic wildlife (Ahrens & Bundschuh, 2014; Ankley, et al., 2021). In humans, exposure to PFAS can cause a wide range of adverse health effects at low levels such as cancer, autoimmune disease and high cholesterol, among others (Barry, Winqvist, & Steenland, 2013; NTP, 2016; Steenland, Zhao, Winqvist, & Parks, 2013; Darrow LA, 2013; Lopez-Espinosa, Mondal, Armstrong, Eskenazi, & Fletcher, 2016). Limited evidence has shown that previously unstudied PFAS also have significant health effects associated with them (Sunderland, et al., 2019). Furthermore, PFAS can be acutely toxic and cause chronic health effects such as reproductive, growth and developmental issues in invertebrates, amphibians and fish with invertebrates being the most vulnerable (Ankley, et al., 2021). In both humans and aquatic wildlife the bioaccumulation and health effects associated with species are typically greater for PFAS with longer fluorinated chain lengths, additionally the nature of the functional group and co-occurrence of different PFAS can increase their toxicity (Ankley, et al., 2021; Ahrens & Bundschuh, 2014; ITRC, 2020)

One of the most significant point sources of PFAS contamination is aqueous film forming foams (AFFF) used to combat petrochemical fires, typically at airports and military bases (Ahrens & Bundschuh, 2014). AFFF training and firefighting events release large quantities of PFAS into the environment leading to high concentrations of PFAS in the soil, concrete, and groundwater below the site and high levels in surrounding surface water bodies (Ahrens & Bundschuh, 2014; Thai, et al., 2022). Because of PFAS ubiquity in wastewater streams, another common point source of PFAS contamination is wastewater treatment plants (WWTPs) effluent when it is released into aquifers, surface waters or used for irrigation (Ahrens & Bundschuh, 2014; Hamid & Li, 2016; Hubbard, et al., 2022; Szabo, Coggan, Robson, Currell, & Clarke, 2018). Additionally, WWTP biosolids have been shown to be a source of soil, surface water, and groundwater PFAS contamination when applied as a soil amendment in agriculture (Hamid & Li, 2016). Due to PFAS uses in domestic products such as paints, lubricants, electronics, and water repellents another environmental source can be urban runoff and landfill leachates (Glüge, et al., 2020; Zhao, 2013; Ahrens & Bundschuh, 2014; Helmer, Reeves, & Cassidy, 2022) Lastly, degradation of volatile PFAS species in the atmosphere and subsequent deposition as less volatile species is often responsible for PFAS occurrence in very remote regions of the earth (Ahrens & Bundschuh, 2014; Stock, Furdui, Muir, & Mabury, 2007).

Due to the number of PFAS species in existence, targeted analysis methods such as liquid chromatography tandem mass spectroscopy (LC-MS/MS) are limited by the availability of analytical standards. For this reason, non-target screening methods are required for accurate assessment of total PFAS contamination. One such non-target method is

extractable organofluorine (EOF) analysis using weak anion exchange (WAX) solid phase extraction (SPE) (McDonough, Guelfo, & Higgins, 2019). This method is well suited for the analysis of anionic and neutral PFAS species. Drawbacks include poor retention of cationic PFAS species and lack of differentiation between organofluorine (OF) and fluoride (McDonough, Guelfo, & Higgins, 2019). Other research teams have demonstrated that the majority of OF in environmental samples is not identified in an LC-MS/MS assay (Yeung, et al., 2013; Miyake, et al., 2007). This unidentified OF may be representative of precursors to targeted species, intermediate degradation products, and other PFAS not accounted for by a targeted species analysis, in addition to residual fluoride that is retained and eluted during extraction (breakthrough).

I sought to understand the occurrence of PFAS and organofluorine in surface waters of the Northwestern Great Basin, Central Sierra Nevada, and Upper Sacramento River watershed. Collectively, these water ways represent major sources of water to human urban and rural populations in California and the Northwestern Great Basin, in addition to providing habitat for numerous endemic and threatened wildlife species (Nevada Division of Natural Heritage, 2021). To do this I measured 19 PFAS species via LC-MS/MS and EOF in 10 waterways and 15 lakes.

Materials and Methods

Study Area

I defined the Northwestern Great Basin as the area North to South from the Black Rock Desert of Nevada to Mono Lake California and West to East from the California Border to the city of Fallon Nevada. It is a basin and range environment, consisting of numerous

north-south running mountain ranges and valleys receiving water primarily from mountain precipitation through numerous rivers and streams as well as groundwater infiltration. The unique geologic setting makes it prone to the development of endorheic (terminal) basins and lakes that have no hydrologic outflows, aside from evaporation and infiltration (Wurtsbaugh, et al., 2017). Small rural agricultural communities dominate human development, in addition to a few heavily developed cities and military bases (Torregrosa & Devoe, 2008). To compensate for the lack of precipitation several water-engineering projects such as groundwater extraction, surface water diversions, and reclaimed water use are implemented to allow human development to occur (Chambers, Water Resources and the Great Basin, 2008).

Alpine lakes in the Tahoe Basin, Upper American River Watershed, and Upper Sacramento River Watershed (Mt. Shasta) regions were also sampled as low human activity, non-terminal lakes for comparison to the study region. The Tahoe Basin and Upper American River Watershed are large hydrologic basins in the Sierra Nevada Mountains of California and Nevada that receive their water from local snowmelt through surface runoff and numerous streams, some of which pass through urban areas. The Upper Sacramento River Watershed around Mt. Shasta is an expansive mountainous zone receiving water from local snowmelt with very little human activity and no significant human development. Briefly, the sites sampled, and potential contaminant sources are described here, in Figure 1, and in appendices A (sampling site metadata), B (source locations), and C (Watershed Landcover).

Carson River Watershed

In the Carson River watershed four sites were sampled along the Carson River (August 2022), two in Lahontan Reservoir (January 2022) and five in the Carson Sink (January 2022) at the terminus of the Carson River. Waste water treatment plants in this watershed include the Carson City Water Reclamation Facility (CCWRF), the Rolling-A Waste Water Treatment Plant (RAWWTP) in Dayton NV, and the Fallon Waste Water Treatment Plant (FWWTP). A known AFFF fire training area (FTA) exists at the Fallon Naval Air Station (FNAS) in Fallon, with a documented PFAS plume in the groundwater below it (EWG, 2022). Additionally, this watershed is hydrologically connected to the Truckee River watershed via the Truckee Canal.

Truckee River Watershed

In the Truckee River Watershed 5 sites were collected offshore around the southern and eastern banks of Lake Tahoe and 1 in the Tahoe Keys Marina (September 2021), 14 sites were sampled across 2 sampling events (October 2021, June 2022) in the Truckee River using timed sampling to sample the same plug of water with each collection (Bohman, 2000), one was taken of the Truckee Meadows Waste Water Treatment Plants (TMWRF) effluent (October 2021), and 2 were sampled along the Truckee Canal (June 2021). In the city of Reno NV five sites were sampled targeting surface waters around Reno-Tahoe International Airport (RNO) (June 2021). The Tahoe-Truckee Sanitation Agency (TTSA) discharges wastewater effluent to the groundwater in the town of Truckee and TMWRF, a surface water effluent discharging WWTP, is located just above the confluence of the Truckee River and Steamboat Creek on Steamboat Creek (EPA, 2012). RNO is an AFFF FTA with a documented PFAS plume in the groundwater below it (EWG, 2022).

Walker Basin

The Walker River was sampled at 3 points (June 2022). Walker Lake was sampled at 3m at 3 locations and on the surface at 1 (November 2021). No WWTPs or AFFF FTAs were identified by this research team in this watershed.

Mono Basin

In the Mono Basin Mono Lake was sampled at the surface at 2 points and a sample was collected in Lee Vining Creek below the town of Lee Vining CA (January 2021). No WWTPs or AFFF FTAs were identified by this research team in this watershed.

Swan Lake

Swan Lake was sampled at the surface on its east shore and in its inflow from an unnamed drainage canal that receives effluent from the Reno Stead Wastewater Treatment plant (RSWRF) (August 2021). This site may receive PFAS contamination from a former FTA, the Dodd/Deal Fire Training Academy (DDFTA), that has been remediated for soil and groundwater hydrocarbon contamination but may have been used for AFFF training resulting in a contaminated groundwater plume (McGinley and Associates, Inc., 2002). Similarly, it may receive PFAS contamination from incidental AFFF use to extinguish fires at the Reno Stead Airport (RSA).

Anonymized Terminal Lakes

A handful of terminal lakes have been anonymized here at the landowner's request. Terminal Lake 1 (TL1) was sampled at 4 locations on the lake and in its inflow (March 2022). It is downstream from two WWTPs, and one known AFFF FTA. Terminal Lake 2

(TL2) was sampled at 2 points in addition to one of its source creeks (October 2021), No WWTPs or AFFF FTAs were identified by this research team in this watershed.

Other Alpine Lakes

In the Upper American River Watershed two sites were sampled at Serene Lake near Truckee CA (October 2021). In the Upper Sacramento Watershed Cliff Lake was sampled at two points, and Castle Lake was sampled at two points (September 2021). No WWTPs or AFFF FTAs were identified by this research team in this watershed.

Sampling Methods

At each sampling site, I collected triplicate samples for water quality parameters, EOF, and LC-MS/MS. Surface water grab samples were collected directly in sample bottles, samples taken at depth were collected using a van-dorn sampler that was rinsed with sample water on-site prior to sample collection. All sample containers were borosilicate glass or high-density polyethylene (HDPE) and were baked at 500°C for a minimum of 3 hours or triple rinsed with methanol, respectively. Water quality and EOF bottles were triple rinsed with sample water prior to filling, LC-MS/MS bottles were not rinsed to preserve ascorbic acid preservation agent in them. All field samples were stored on ice as soon as possible after collection and were transferred to a refrigerator at 5°C.

Sample Processing

Water quality and EOF samples were filtered as soon as possible after collection, typically within one to seven days, see appendix A for dates of collection and filtration. Water quality and EOF samples were filtered separately using stacked 0.5µm (Advantec GC-50) and 2.7µm (Whatman GF/D) glass fiber filters that had been baked at 500 °C for

a minimum of 3 hours. EOF sample filtration involved additional steps including the rinsing of the hopper, filter and flask of the vacuum filtration apparatus using ~10mL of MeOH that was then recombined with the filtered sample, to capture any PFAS that adhered to the walls of the apparatus. LC-MS/MS samples were mailed, unprocessed, to SNWA for filtration, extraction and analysis within a month of collection. Upon receipt LC-MS/MS samples were filtered using prewashed glass fiber filters, as described in Gonzalez, Thompson, Quinones, Dickenson, & Bott, 2021.

Sample Extraction and Analysis

Liquid Chromatography Tandem Mass Spectroscopy

After filtration all LC-MS/MS samples were extracted using methods described elsewhere (Gonzalez, Thompson, Quinones, Dickenson, & Bott, 2021). Briefly, Samples were adjusted to pH <2 and spiked with isotopically marked standards. Pre-packed 200mg, 6cc hydrophilic-lipophilic balance (HLB) cartridges (waters corporation) were used in a Dionex Autotrace 280 for solid phase extraction, cartridges were sequentially conditioned using MTBE, methanol, and reagent water prior to sample loading, following sample loading cartridges were rinsed with reagent water and dried for 30 minutes using N₂ gas, they were then eluted using methanol and concentrated to 500µL using N₂ gas.

Analytes were separated using a 50x4.6mm Kinetex C18 column with a pore size of 2.6 µm and a binary gradient of 5.0 mM ammonium acetate (v/v) in water (A) and 100% methanol (B) at a flow rate of 500 uL/min. An Agilent G1312A binary pump and HTC-PAL autosampler (CTC Analytics) with an injection volume of 2 µL was used for all

analytes. Tandem mass spectrometry was performed using an API 4000 triple-quadrupole mass spectrometer (Applied Biosystems) using ESI negative ionization.

The following analytes were assessed: perfluorobutanoic acid (PFBA), perfluoropentanoic acid (PFPeA), perfluorohexanoic acid (PFHxA), perfluoroheptanoic acid (PFHpA), perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorodecanoic acid (PFDA), 6:2 fluorotelomer unsaturated carboxylic acid (6:2 FTUCA), 8:2 fluorotelomer unsaturated carboxylic acid (8:2 FTUCA), 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propanoic acid (Gen-X or HFPO-DA), perfluorobutane sulfonate (PFBS), perfluorohexane sulfonate (PFHxS), perfluorooctane sulfonate (PFOS), 4:2 fluorotelomer sulfonate (4:2 FTS), 6:2 fluorotelomer sulfonate (6:2 FTS), 8:2 fluorotelomer sulfonate (8:2 FTS), perfluorooctane sulfonamide (PFOSA), N-ethyl-N-[(heptadecafluorooctyl)sulphonyl]glycine (N-EtFOSAA), N-methyl perfluorooctane sulfonamidoacetic acid (N-MeFOSAA).

Extractable Organofluorine

SPE for EOF was conducted using the method described by Miyake, et al., 2007. modified for use with a Dionex Autotrace 280 solid-phase extraction instrument according to Thermo-Scientific (Zhang, Qiu, Ullah, & Liu, 2020). Briefly, 250mL of sample was adjusted to pH ~4 using glacial acetic acid. 200 mg, 6cc Oasis WAX cartridges were sequentially conditioned with 5mL 0.1% NH₄OH/MeOH, 5mL milli-Q water prior to sample loading. Following sample loading cartridges were sequentially rinsed with 20mL of 0.01% NH₄OH in water, 10 mL of milli-Q water, 10 mL of ammonium acetate buffer (pH 4) and 10mL of 20% MeOH in water, cartridges were then

dried with N₂ gas for 20 minutes. Finally, samples were eluted with 10 mL of 0.1% NH₄OH/MeOH and concentrated to 1mL under N₂ gas.

CIC analysis was conducted using the method described by Miyake, et al., 2007. Briefly, a combustion ion chromatogram (CIC) (Metrohm, Herisau) which was operated at 1050°C under 300mL/min O₂ combustion gas flow. 100mL/min Argon carrier gas was used to blow combustion end products into deionized absorber water. A combustion time of 20 minutes was applied. Samples pass through a 1mL loop and were analyzed by a 930 Compact IC Flex equipped with an A Supp5 analytical column.

Extractable Organofluorine Quality Assurance Experiment

To test the occurrence of F⁻ breakthrough in SPE for EOF I added a series of NaF and PFOA spikes to milli-Q water. These samples were extracted and analyzed for EOF in the same manner as my field samples described above. NaF spikes were added such that the solutions had 0, 0.05*10⁶, 0.5*10⁶, and 10⁶ ppt of F⁻ and PFOA spikes were added such that the solutions had 0, 10, 100, 1000 ppt of F bound to PFOA (IE organofluorine). This resulted in an assay containing 16 solutions. Each solution was made in bulk and separated into three triplicates that were extracted and analyzed together.

Water Quality Parameters

After filtration, sample conductivity was analyzed using a TitraLab AT-1000 series workstation. Conductivities among field triplicates were averaged and reported in appendix A. Fluorine concentrations were determined using a Dionex ICS-6000 ion chromatogram. I averaged the results of my water quality parameter field triplicate values for reporting and analysis.

Data Analysis

Liquid Chromatography Tandem Mass Spectroscopy

I reported LC-MS/MS results in two ways, as the sum of the 19 targeted species and as the profile (relative abundance) of species found. I summed the average of triplicate values for each species to calculate the sum of targeted species. For each species I calculated a standard deviation among the field triplicates, I then used these standard deviations to calculate a propagated error of the sum of detected species ($z = x + y$, $\sigma_z = ((\sigma_x)^2 + (\sigma_y)^2)^{1/2}$). I reported this propagated error as error bars on sums of species in Figure 2. I made the profiles of species found using the average of individual species concentrations among field triplicates.

Extractable Organofluorine

I reported EOF as the average of field triplicates, when EOF values were below detection limit EPA guidelines for chemical concentrations near detection limit were followed, reporting one half of the detection limit when contamination was suspected (EPA, 2022). I also calculated standard deviations of these triplicates and reported them as error bars on my EOF values in Figure 3. To determine the quantity of OF identified via LC-MS/MS I converted my LC-MS/MS values to fluorine equivalents. To do this I multiplied the species concentration found by the ratio of fluorine in that species to its total molecular mass. I then used the fluorine equivalent values to find the sum of species and propagated error of the sum as described above for LC-MS/MS sum of species. To find the percent of LC-MS/MS identified OF I divided the sum of fluorine equivalent values by the EOF value at the corresponding sampling site and multiplied the result by

100. I was then able to use the percent of identified organofluorine to infer how much more PFAS was likely to be in the sample than what is suggested by the LC-MS/MS assay (EG 10% identified = ~10x more present than reported in LC-MS/MS assay). I calculated an error range for the percent of identified OF as propagated error using the propagated error of my summed species converted to fluorine equivalents and the standard deviations calculated for my EOF values ($z = x/y$, $\sigma_z = z * ((\sigma_x/x)^2 + (\sigma_y/y)^2)^{1/2}$). Both the median percent identified OF and median error in percent identified organofluorine are presented in the text below. To show the correlation between fluoride concentration and EOF a regression analysis was used and the R^2 value is reported in Figure 5.

Results and Discussion

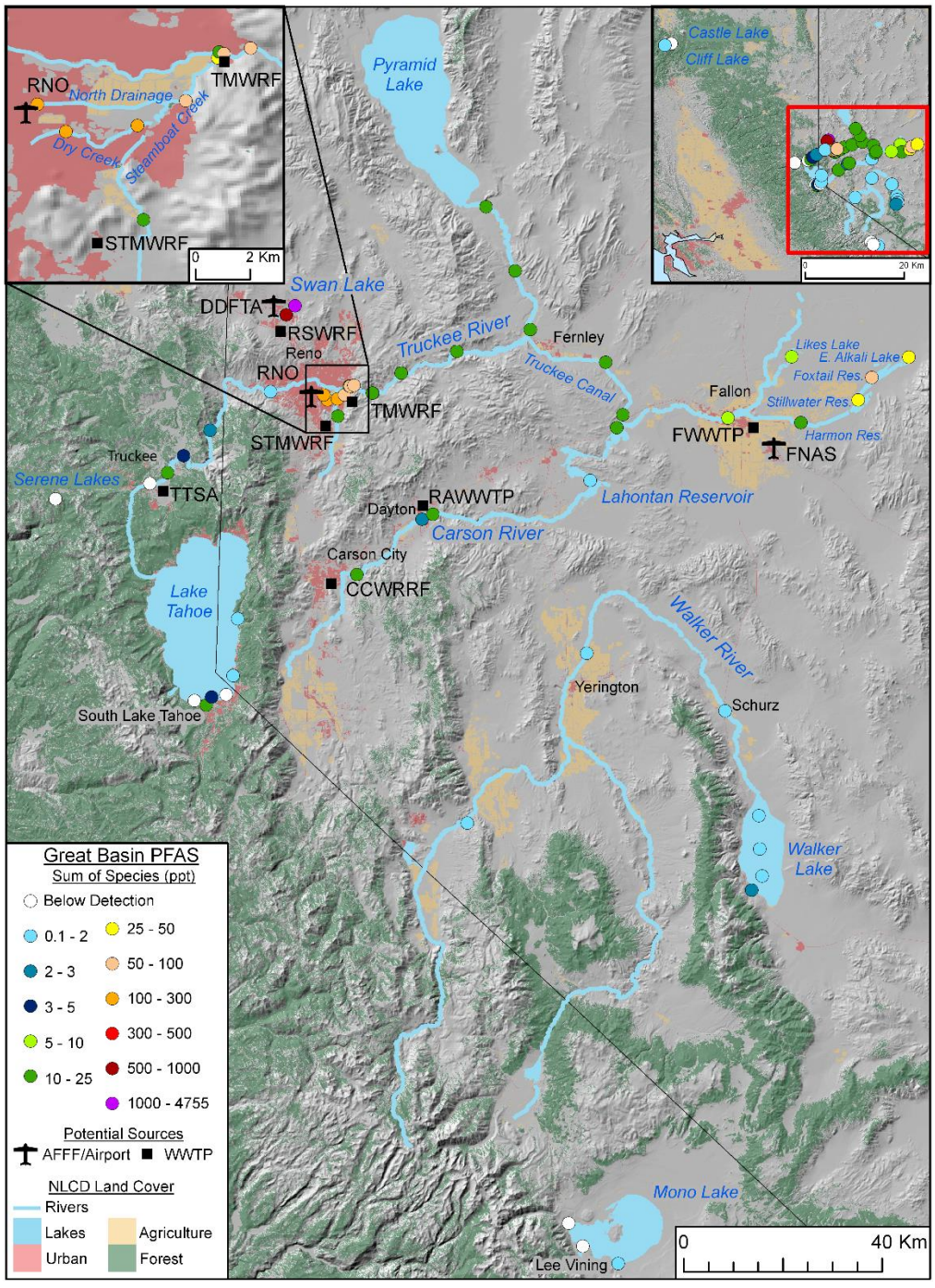
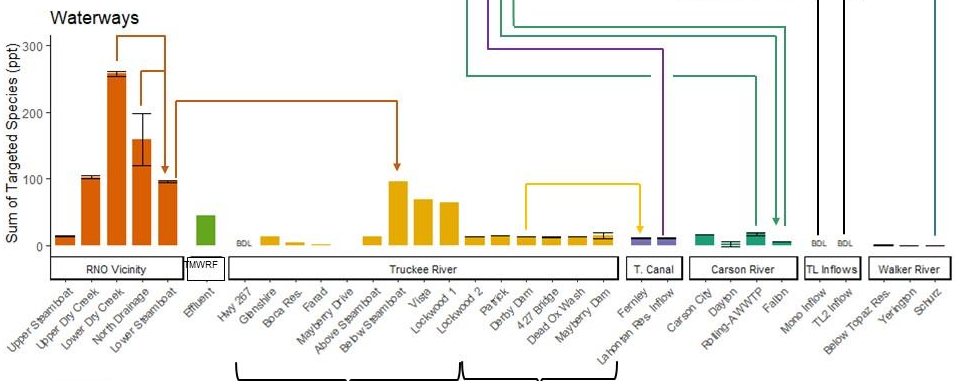
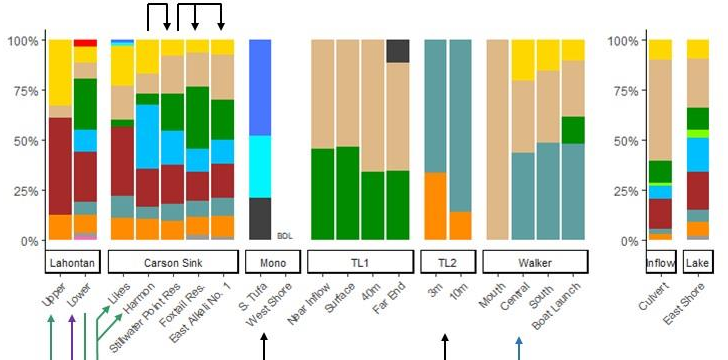
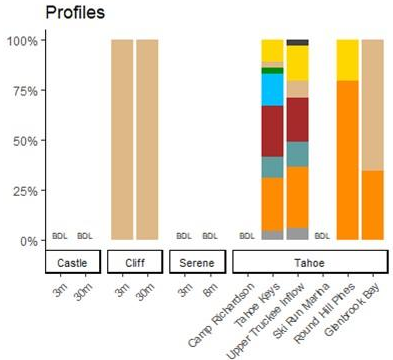
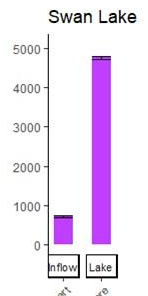
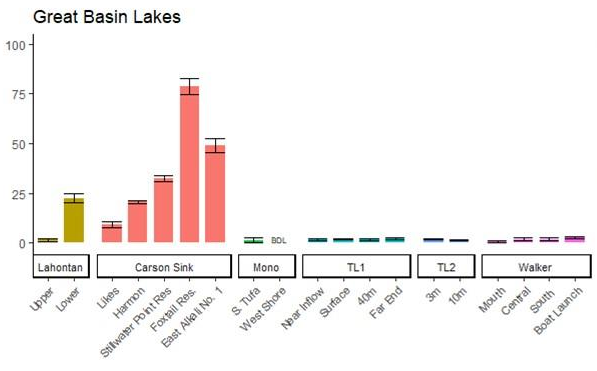
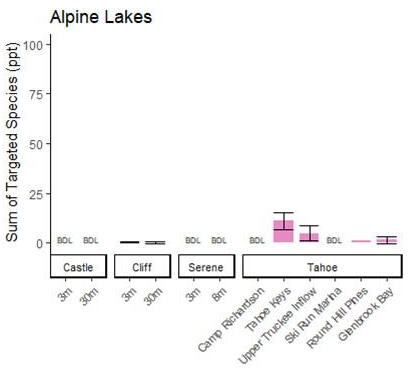


Figure 1: Map of study area and sites sampled. The concentrations shown are the sum of the average of field triplicates for targeted species.



Select Hydrologic Connections

- 4:2 FTS
- 6:2 FTS
- 8:2 FTS
- N-EiFOSAA
- N-MeFOSAA
- PFOSA
- PFBS
- PFHxS
- PFOS
- PFBA
- PFFeA
- PFHxA
- PFHpA
- PFOA
- PFNA
- PFDA

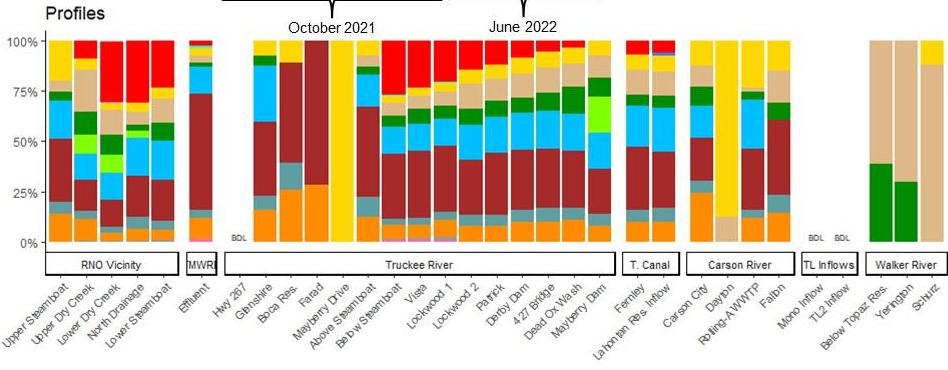


Figure 2. Summed target species from LC-MS/MS analysis and species profiles. Error bars are propagated error of standard deviations among field triplicates for detected target species, sites with no error bars were not sampled in triplicate. Arrows indicate hydrologic connections and flow paths between select field sites and watersheds.

Targeted Species Analysis Summary

Sub-Class	Species	% of Sites Detected at	Max (ppt)	Average (ppt)	Median (ppt)
Perfluoro-carboxylic acid (PFCA)	PFOA	60.0	390.00	15.3	2.0
	PFHpA	58.5	310.00	11.6	1.2
	PFHxA	53.8	990.00	43.8	4.5
	PFPeA	44.6	930.00	42.1	4.1
	PFNA	21.5	90.00	8.9	0.8
	PFDA	12.3	19.00	3.8	0.7
	PFBA	9.2	230.00	52.5	13.0
Perfluoro-sulfonic Acid (PFSA)	PFHxS	72.3	1400.00	41.9	1.5
	PFBS	63.1	500.00	18.3	1.8
	PFOS	58.5	570.00	22.2	1.1
Precursors	6:2 FTS	26.2	80.00	14.6	1.9
	N-MeFOSAA	9.2	1.20	0.6	0.5
	N-EtFOSAA	7.7	2.90	1.2	1.0
	PFOSA	6.2	0.84	0.6	0.6
	4:2 FTS	1.5	0.92	0.8	0.8
	8:2 FTS	1.5	0.88	0.7	0.6
	6:2 FTUCA	0.0	0.00	0.0	0.0
	8:2 FTUCA	0.0	0.00	0.0	0.0
	Gen X	0.0	0.00	0.0	0.0

Table 1: LC-MS/MS target species detection frequency and ranges within study area. All target species had minimum concentrations below detection limit.

Throughout the study area the most commonly detected targeted species were terminal degradation products (i.e. PFCAs and PFSA), and precursor species were found somewhat less often. This may be attributable to degradation of precursors and legacy PFAS in the environment and WWTPs into terminal degradation products (Ahrens &

Bundschuh, 2014; Lenka, Kah, & Padhye, 2021). Of the precursors found 97.5% by mass were FTS species (8,6,4:2 FTS), 2.3% were N-MeFOSAA and N-EtFOSAA, and 0.3% were PFOSA. These non-FTS precursors have been shown to undergo long range transport and degradation in the atmosphere and are likely representative of diffuse contamination from atmospheric deposition (Stock, Furdui, Muir, & Mabury, 2007).

The highest concentrations of PFAS were found in sites near AFFF FTAs, lower concentrations were associated with urban areas and wastewater treatment plants, and the lowest concentrations found were in areas with minimal or no human development, as shown in Figure 1. This is similar to the findings of Bai & Son, 2021 for the Truckee river, in which the highest concentrations were found in the city of Reno near the AFFF FTA. Interestingly, my analysis of Lake Tahoe and the Truckee River had generally lower concentrations than theirs indicating that temporal variance in PFAS concentration is likely to occur in this region.

Comparison of Alpine and Terminal Lakes

When no known PFAS sources were present, alpine lakes had concentrations at or near the detection limit (<1 ppt), and Great Basin lakes had concentrations of ~2 ppt. This is shown in Figures 1 and 2. Great Basin Lakes generally had higher concentrations than alpine lakes, as shown in Figure 2. This is largely explained by the absence of significant sources around our alpine lake sites and demonstrates that headwaters are likely to have very low levels of PFAS contamination. A notable exception to this was the Tahoe Keys Marina which had the highest concentration of PFAS of all the alpine lake sites sampled. This is likely because of the large quantity of urban runoff received from the City of

South Lake Tahoe and limited mixing with the rest of the lake, limiting dilution. The next highest concentration alpine lake site was adjacent to the Tahoe Keys offshore of the inflow of the Upper Truckee River, here the elevated concentration is likely attributable to PFAS transported into the lake from the City of South Lake Tahoe by the river that has not been diluted into the lake yet. Other adjacent sites were below detection limit likely due to sufficient dilution to bring their PFAS load below detection limit.

Accumulation in Endorheic Lakes

By comparing PFAS concentrations in terminal lakes to those of their inflows it was found that in most cases lake concentrations are 2-5 times higher than those of their primary inflow, as shown in Figure 2. This indicates that PFAS are accumulating in the waters of these lakes. Examples of this include the lakes in the Carson Sink, Mono Lake, Walker Lake, TL2, and Swan Lake. The notable exception to this was TL1 which had ~10x higher concentration in its inflow than in the body of the lake.

Another line of evidence that PFAS will accumulate in lake waters is the significantly higher concentrations of PFAS in Lower Foxtail Reservoir (LFR) and East Alkali Lake No. 1 (EA1) than in Stillwater Point Reservoir (SPR), the lake they each receive water from. These lakes are managed as intermittent wetlands and have naturally intermittent outflows and human controls on their outflows making it reasonable to say that these lakes can behave like endorheic lakes for several seasons or years, accumulating salts and recalcitrant organic compounds (C. Lunderstadt, personal communications, October 25, 2022). This notion is supported by their higher conductivities than their primary source water, the Carson River, and increases in these qualities from SPR to EA1 and LFR (See

Appendix A). This notion of concentration of PFAS is further supported by the similarity in PFAS profiles observed within these 3 lakes (see Figure 2), having no large additions of any species, indicating no additional contamination sources were present in EA1 or LFR.

Long Range Transport in Waterways

By considering the summed PFAS species concentrations and profiles along the rivers sampled I found that the degree of long-range transport of target species varied between different waterways. Direct comparisons are difficult to make due to temporal, spatial, and strategic variance in sampling. However, some observed patterns are summarized here.

PFAS concentrations and profiles were remarkably similar in the Walker River (Average = 0.89, Coefficient of variance = 21.4%), Truckee Canal (Average = 11.40, Coefficient of variance = 1.9%), and in the Truckee River reach from Lockwood to Marble Bluff dam (Average = 14.19, Coefficient of variance = 6.8%). This similarity can be attributed to the lack of diluting inflows and notable PFAS sources in these reaches. These sites demonstrate a setting in which PFAS can undergo significant long-range transport in surface waters.

At other sites, such as the sampling event conducted on the Truckee to Reno reach of the Truckee River, and in the Carson River, notable variance in the concentrations along the reach were observed. Any increases in concentration represent the presence of a source or confluence with a higher concentration waterway. Less obvious are the mechanisms leading to decreases in concentration. In montane sites such as the Truckee River from

Truckee to Reno, I expect that the decreases in PFAS concentration can be explained by water diversions and dilution with less contaminated water from the numerous creeks and reservoirs that feed into the river along this reach. This does not explain the decreases seen from Carson City to Dayton NV along the Carson River, as this reach lacks inflows from creeks and reservoirs. It is possible that the decrease observed along this reach can be explained by the diversion of in-stream flows for irrigation in the agricultural operations around Dayton, binding of PFAS in irrigation water to the soil causing a reduction in PFAS concentration in return flows, ultimately leading to a dilution effect in the river. This theory, or a similar mechanism such as direct dilution from an unidentified spring, is supported by the modest reduction in conductivity observed at the Dayton site relative to the chemistry of the Carson City site, which indicates the input of some more dilute water source.

Another noteworthy abnormality in the data presented here is the notable increase in the concentration of PFAS from Upper Dry Creek to Lower Dry Creek. Given that Upper Dry creek is in closer proximity to RNO it was expected to have a greater than or equal concentration to Lower Dry Creek. This may be explained by an increasing ratio of contaminated groundwater derived flows to surface derived flows along the length of this waterway. Thorough investigation and explanation of this phenomena is beyond the scope of this study.

Fluorotelomer Sulfonate as an Indicator of Aqueous Film Forming Foam Impacts

FTS species formed a large fraction (>10%) of the PFAS load in sites associated with the AFFF FTA at RNO. Because 6:2 fluorotelomer species are commonly used in AFFF

chemistries and are readily degraded to 6:2 FTS, it can be used here as an indicator of AFFF impacts (Ruyle, et al., 2021; Choi, Helbling, Liu, Olivares, & Higgins, 2022). This association has been shown in engineering and environmental contexts (Gonzalez, Thompson, Quinones, Dickenson, & Bott, 2021; Karrman, Elgh-Dalgren, Lafossas, & Moskeland, 2011). Significant aerobic degradation of the FTS species to PFPeA and PFHxA that occurs in the sediments must also be considered when evaluating species profiles for source identification (Zhang, Lu, Wang, & Buck, 2016). This degradation can reduce 6:2 FTS concentrations to levels near or below detection limit, particularly in legacy contamination sites with ample time for degradation to occur. For this reason, the presence of PFPeA and PFHxA may be considered evidence of AFFF impacts if they are suspected, and viable conditions for degradation are present. Furthermore, the presence of FTS species will be a function of the degradation of fluorotelomer AFFF chemistries into FTS and fluorotelomer alcohols (FTOH), and the rate of this primary degradation will determine the amount of FTS available for detection (Choi, Helbling, Liu, Olivares, & Higgins, 2022). For these reasons an expanded and AFFF degradation pathway targeted species assay that includes primary AFFF species, FTS and other intermediate species, and terminal degradation products would prove useful for future AFFF source tracking projects. This would reduce the number of stages of the degradation pathway in which intermediate species can be missed in analysis.

In this study we were able to use 6:2 FTS as a tracer to explain the increase in PFAS contamination from upper to lower Lahontan Reservoir. In the Truckee River Watershed 6:2 FTS can be tracked from its origin in the surface waters surrounding the Reno Tahoe International airport, dilution in Steamboat creek, further dilution in the Truckee River,

stable transport along the Truckee into the Truckee Canal, and finally observation in the lower Lahontan Reservoir (See Figure 2). Because this species was not observed in the Carson River watershed above this point it can be determined that it, and the other PFAS contributing to the increased concentration lower Lahontan Reservoir are derived from the Truckee River watershed. This demonstrates an instance in which diversion of surface waters for supplemental flows to neighboring watersheds directly caused a significant reduction in water quality for downstream users. Furthermore, through repeat sampling of the Lockwood site across two seasons I showed three times more PFAS in the Fall than in the following Summer. Indicating that contamination of the lower Lahontan reservoir may vary in severity temporally.

We also demonstrated that the decommissioned FTA near Swan Lake likely has legacy contamination in the groundwater below it that impacts the lake, using 6:2 FTS as a tracer. Swan lake and its inflow had the highest concentrations of PFAS observed in this study, including a small fraction of FTS precursor species. Although total precursors accounted for <0.1% of the PFAS mass detected, FTS presence indicates that fluorotelomer AFFF were likely used at the FTA, possibly explaining the abnormally high concentration of PFAS at this site. The low concentration of FTS present may be explained by degradation over time given that the FTA was decommissioned roughly 30 years ago, this is supported by the presence of the terminal degradation products PFHxA and PFPeA.

It is interesting to note that none of the sites in Stillwater National Wildlife Reserve (SNWR) had notable precursor contamination, despite their proximity to a documented

AFFF FTA. This may be due to degradation of FTS to PFHxA and PFPeA. This is supported by the fact that PFPeA only occurs downstream of FNAS in the Carson Sink (See Figures 1 and 2). Alternatively, this may also indicate that they are not receiving contaminated water from the FTA, but rather another source such as reclaimed water from the WWTP in Fallon. In each of these cases the presence/absence of 6:2 FTS may provide a strong clue as to the source of PFAS contamination in the surface water in question.

Unidentified Organofluorine Analysis

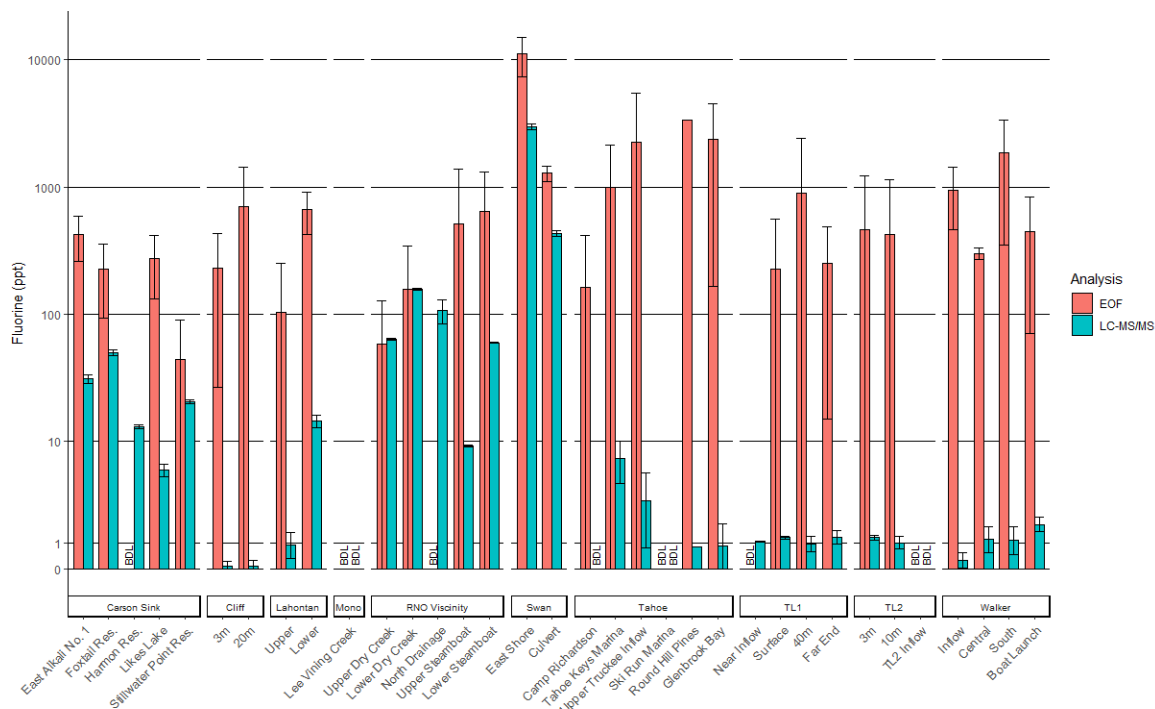


Figure 3: Comparison of EOF and summed targeted species. Error bars are one standard deviation among field triplicates, sites with no error bars were not sampled in triplicate.

By comparison of the LC-MS/MS results, converted to F equivalents, and our EOF results the percentage of organofluorine identified by our LC-MS/MS assay was shown to

be highly variable among field sites, ranging from ~0.1 – ~600% with a median of 0.6% \pm 0.9%. This implies that the total PFAS load is much greater than what my LC-MS/MS findings suggest at many sites. Sites with a high percentage of identified OF were all associated with AFFF FTAs and other point sources, and those with the lowest percent of identified OF were remote reference sites. This is similar to the findings of other research papers (Koch, et al., 2019; Miyake, et al., 2007).

It is important to consider the limitations of EOF analysis. These include high variance in concentration among triplicate field samples, in some cases resulting in standard deviations greater than the average (Figure 3). This occurred in roughly half of our field sites when some replicates were below detection limit and others somewhat above it. Identifying the cause of these issues is beyond the scope of this study and warrants its own publication, though interference from environmental fluoride, poor PFAS recovery, and issues with detection of very low concentrations of fluoride during ion chromatography may be at play. There was a high degree of uncertainty around the true organofluorine load in our field samples. These EOF values should be considered a rough estimate of total PFAS and demonstrating that the total PFAS load is likely to be somewhat higher than what is shown by the LC-MS/MS assay.

Extractable Organofluorine Quality Assurance Experiment

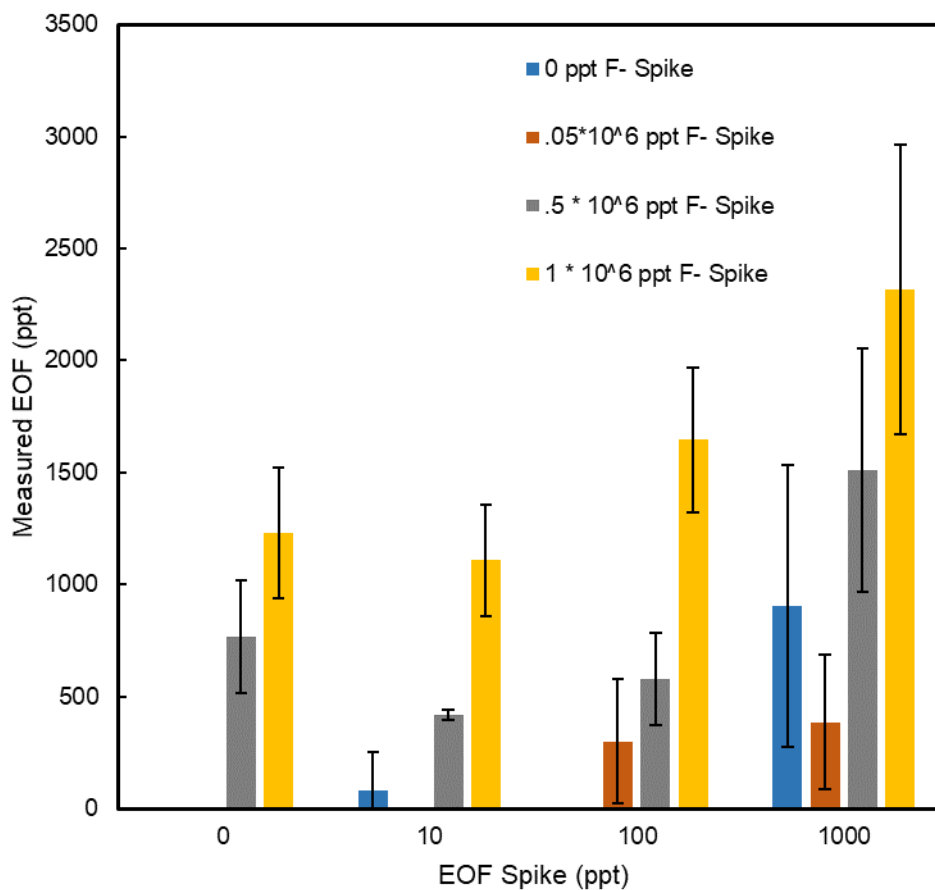


Figure 4: Results of EOF quality assurance experiment. Error bars are standard deviations among laboratory triplicates.

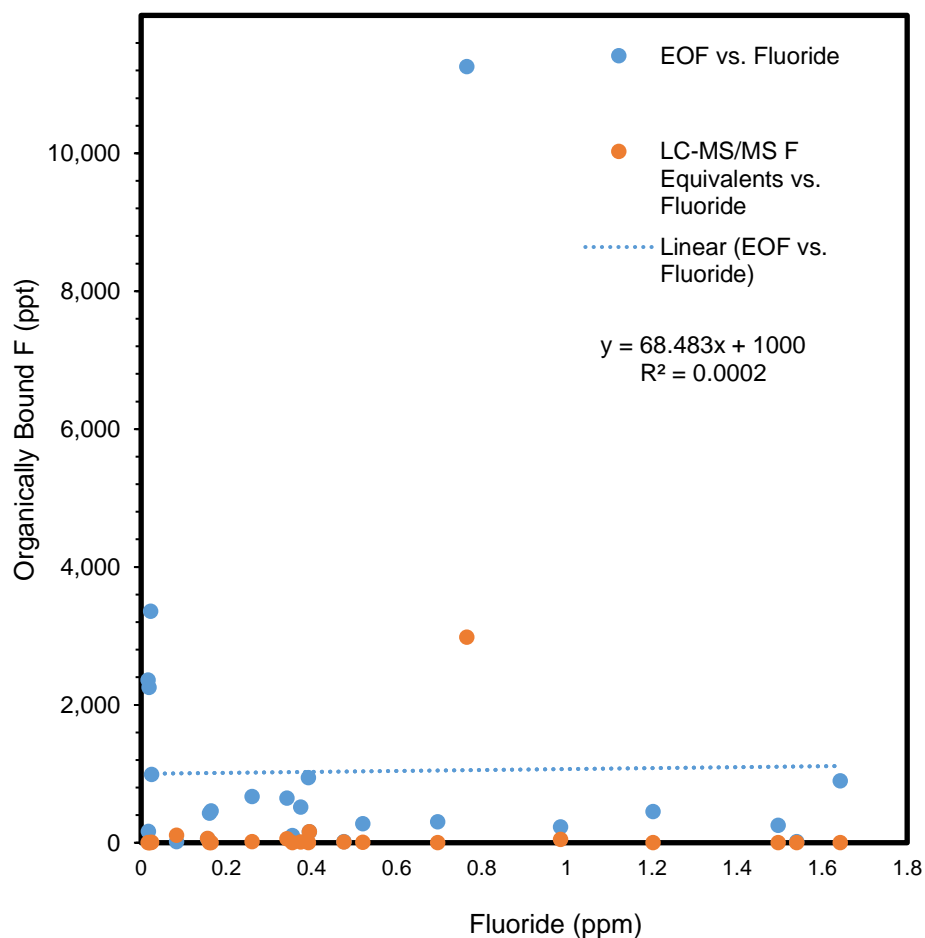


Figure 5: Correlation between EOF results and fluoride. LC-MS/MS results converted to fluoride equivalents shown for reference to known OF concentration.

During EOF quality assurance experiments I found evidence of inorganic fluorine breakthrough and poor analyte recovery in spiked laboratory samples using the EOF extraction and analysis method applied to my field samples, results are shown in Figure 4. I observed EOF values 2-10 times greater than known OF spike concentrations, with the increase from OF spike concentration roughly correlating to the NaF spike concentration used. When the observed breakthrough rate ($\sim 0.13\%$) was used as a corrective factor for field samples ($[\text{True EOF}] = [\text{measured EOF}] - ([\text{F}^- \text{ in field sample}]$

* (breakthrough rate))) roughly half of my samples had negative values for “true EOF”. This indicates that the breakthrough rate found in our laboratory samples is not reflective of the breakthrough rate in environmental samples. This is corroborated by the low R^2 value (0.0002) found during regression analysis of environmental fluoride concentrations and observed EOF values, as shown in Figure 5. I suspect that this is due to binding site competition on the WAX media in which F^- is preferentially desorbed in favor of stronger acids such as Cl^- and SO_4^{2-} . Many of the field samples with the highest F^- concentrations had the highest conductivities and overall load of competing anions, this may explain why many of the samples with the highest concentrations of F^- gave negative values when the correction factor was applied and why no correlation was observed between F^- content and EOF. Further investigation of this phenomena was beyond the scope of this study.

The inconsistency and error seen in these laboratory experiments even when little or no F^- was added show that the method is not very reliable at OF concentrations found via LC-MS/MS in the study area. This is because the OF spike concentrations used here were reflective of the sum of targeted species values found in the study area. In Figure 4 I show that these are often below detection limit and only come above detection limit when known OF concentration is like that of the highest concentrations found in the region (~1000 ppt OF, similar to Swan Lake). Furthermore, in Figure 4 there is a visible trend in which the more F^- that was added the higher the EOF value is in most cases, regardless of OF spike concentration, indicating that breakthrough can and does occur in low ionic strength samples. However, the production of negative EOF values by implementation of the observed breakthrough rate as a corrective factor in environmental samples and lack

of correlation between fluoride and EOF demonstrates that breakthrough may be somewhat mitigated in high ionic strength systems. These seemingly contradicting factors demonstrate how nebulous EOF analysis can be and indicate that reported EOF values and analysis done using them must be considered with abundant caution both here and in other publications.

Conclusions

Here I showed that PFAS are ubiquitous throughout the surface waters of the Great Basin, and that their concentrations are below or near detection limit in most locations in the Tahoe Basin and in the headwaters of the Sacramento and American Rivers.

Additionally, my findings suggest that PFAS are accumulating in terminal lakes, indicating the load of PFAS in these lakes is likely to increase with time. This is likely to be exacerbated with lake level declines associated with increased evapotranspiration and reduced inflow volumes in the same way that salinities are increased by these forcings (Wurtsbaugh, et al., 2017).

Without temporal sampling, seasonal lake volume, PFAS concentration, and inflow rate data the dynamics of PFAS accumulation in endorheic systems remains obscured. I suggest this is an area for further research given the potential impacts of PFAS accumulation in these lakes on both the endemic fish and migratory waterfowl that inhabit them and the humans who hunt and consume these animals for sport and recreate in their waters. Understanding the rate at which PFAS are accumulating in these systems and what the drivers of this accumulation are is key to understanding the threat that they pose to humans and wildlife alike.

In waterways, PFAS concentrations varied widely but were almost always above detection limit and were often among the highest concentration sites in the study. In some waterways PFAS showed remarkable long range transport capability and in others its concentration varied widely along the reach, likely due to dilution, diversions, and additional contamination sources. Illuminating the phenomena that dictate PFAS long range transport in waterways is an issue that warrants its own publication, requiring a thorough understanding of the hydrology and water management of the reach and appropriate mass balance of PFAS in water and sediment, which is beyond the scope of this paper.

I also demonstrated that FTS species can be used as an indicator of AFFF impacts in forensic analysis of PFAS sources and transport pathway identification in surface waters. To improve this methodology, expanded PFAS assays targeting fluorotelomer AFFF species, FTS species, intermediate degradation products, and terminal degradation products are recommended for forensic analysis of AFFF impacts.

Because a median of only $0.6\% \pm 0.9\%$ of OF was identified by the LC-MS/MS assay it can be inferred that the true PFAS load in many of my samples is much higher than what is shown by the LC-MS/MS assay. However, in many cases the results of my EOF analysis were difficult to interpret due to high standard deviations among triplicates.

Additionally, I present laboratory data demonstrating some of the issues associated with F breakthrough and OF recovery in EOF analysis.

In this study I provide a snapshot of PFAS contamination in surface waters throughout the Northwestern Great Basin from 2021-2022. Giving researchers, regulators, land

managers, and utility providers with a starting point for assessment of PFAS occurrence and hazards in their respective management areas. In addition to this I have shown how a particular indicator species may be used to implicate a contamination source far from the impacted area.

Appendix A: Table of Sampling Site Locations

Watershed	Water Body	Location	Depth (m)	Latitude	Longitude	Collected	Filtered	Conductivity (uS/cm)	
Carson Basin	Carson River	Below Carson City	0	39.17987	-119.67316	8/2/2022	8/4/2022	715.00	
		Dayton	0	39.28326	-119.52513	8/2/2022	8/4/2022	628.33	
		Below Rolling-A WWTP	0	39.29312	-119.49963	8/2/2022	8/4/2022	703.00	
	Lahontan Reservoir	Floodplain	0	39.36161	-119.13425	1/12/2022	1/20/2022	366.00	
		Shoreline	0	39.46111	-119.07155	1/12/2022	1/20/2022	425.00	
	Carson River	Fallon	0	39.48025	-118.81499	8/2/2022	8/3/2022	332.33	
	Likes Lake	Shoreline	0	39.59382	-118.66843	1/12/2022	1/20/2022	666.33	
	Harmon Reservoir	Shoreline	0	39.47447	-118.64399	1/6/2022	1/18/2022	1246.00	
	Stillwater Point Reservoir	Shoreline	0	39.51760	-118.51047	1/6/2022	1/18/2022	2553.33	
	Foxtail Reservoir	Shoreline	0	39.55852	-118.47940	1/6/2022	1/11/2022	4033.33	
East Alkali Lake No. 1	Shoreline	0	39.59673	-118.39354	1/6/2022	1/10/2022	4730.00		
Lemmon Valley	Swan Lake Inflow	Culvert	0.1	39.64670	-119.85716	8/23/2021	8/26/2021	804.33	
	Swan Lake	East Shore	0.1	39.66327	-119.83700	8/23/2021	8/31/2021	9455.00	
Mono Lake	Lee Vining Creek	Lee Vining Creek	0	37.97347	-119.11039	1/23/2022	1/25/2022	52.33	
	Mono Lake	South Tufa	0	37.94338	-119.02849	1/23/2022	1/25/2022	65533.33	
		West Shore	0	38.01436	-119.14393	1/23/2022	1/25/2022	49933.33	
Terminal Lake #1	Terminal Lake #1	Shore - Near Inflow	0	39.90748	-119.52928	3/16/2022	3/17/2022	8000.00	
		Open Water - Surface	0	38.01435	-119.14394	3/16/2022	3/17/2022	7986.67	
		Open Water - 40m	40	38.01435	-119.14394	3/16/2022	3/17/2022	7943.33	
		Far End	0	40.19100	-119.62865	3/16/2022	3/17/2022	8053.33	
Terminal Lake #2	Terminal Lake #2 Inflow	Inflow	0	41.52587	-119.05941	10/15/2021	10/26/2021	120.50	
	Terminal Lake #2	Open Water - 3m	3	41.50269	-119.06351	10/15/2021	10/26/2021	542.33	
		Open Water - 10m	8	41.50269	-119.06351	10/15/2021	10/26/2021	530.67	
Truckee River	Lake Tahoe	Camp Richardson	0.1	38.94220	-120.04178	9/17/2021	9/23/2021	97.60	
		Tahoe Keys Marina	0.1	38.93505	-120.01395	9/17/2021	9/23/2021	131.53	
		Upper Truckee Litoral Shelf	0.1	38.94970	-120.00230	9/17/2021	9/27/2021	95.73	
		Outside Ski Run Marina	0.1	38.95477	-119.96890	9/17/2021	9/23/2021	96.30	
		Round Hill Pines	0.1	38.98988	-119.95435	9/18/2021	9/23/2021	97.10	
		Glen Brook Bay	3	39.09310	-119.94789	9/11/2021	9/14/2021	97.73	
	Truckee River	Truckee River	Hwy 267 Overpass	0	39.332690	-120.162970	10/9/2021	10/11/2021	N/A
			Glenshire	0	39.353280	-120.122750	10/9/2021	10/11/2021	N/A
			Boca Res. Confluence	0	39.385280	-120.086670	10/9/2021	10/11/2021	N/A
			Farad	0	39.433060	-120.026110	10/10/2021	10/11/2021	N/A
			Marble Bluff Dam	0	39.506670	-119.888060	10/11/2021	10/11/2021	N/A
			Above Steamboat Crk. Confluence	0	39.521030	-119.703670	10/12/2021	10/14/2021	N/A
			Mixing with Steamboat Crk.	0	39.520450	-119.702130	10/12/2021	10/14/2021	N/A
			Below Steamboat Crk. Confluence	0	39.522330	-119.69331	10/12/2021	10/14/2021	N/A
			Lockwood	0	39.509560	-119.648860	10/13/2021	10/14/2021	N/A
			Lockwood	0	39.50965	-119.64938	6/15/2022	6/20/2022	203.33
			Patrick	0	39.54728	-119.58305	6/15/2022	6/20/2022	210.67
			Derby Dam	0	39.58942	-119.45398	6/15/2022	6/21/2022	201.67
			Hwy 427 Overpass	0	39.63235	-119.28316	6/16/2022	6/21/2022	200.13
			Dead Ox Wash	0	39.73829	-119.32242	6/16/2022	6/17/2022	221.00
			Mayberry Dam	0	39.85309	-119.39485	6/17/2022	6/17/2022	264.67
	Truckee Canal	Fernley	0	39.5768	-119.10473	6/16/2022	6/21/2022	123.07	
		Lahontan Res. Spillway	0	39.46279	-119.06882	6/16/2022	6/21/2022	176.10	
	Steamboat Creek	Upper (Above RNO)	0	39.46576	-119.72867	6/9/2022	6/10/2022	727.67	
	Dry Creek	Upper	0	39.49478	-119.75424	6/9/2022	6/10/2022	258.67	
		Lower	0	39.49677	-119.73063	6/9/2022	6/10/2022	398.67	
	North Drainage	Upper	0	39.50380	-119.76363	6/9/2022	6/10/2022	568.33	
	Steamboat Creek	Lower (Below RNO)	0	39.50494	-119.71469	6/9/2022	6/10/2022	621.50	
	Upper American River	Serene Lake	Open Water - 3m	3	39.29853	-120.38246	10/15/2021	10/18/2021	31.40
			Open Water - 8m	8	39.29853	-120.38246	10/15/2021	10/18/2021	33.53
	Upper Sacramento River	Castle Lake	Open Water - 3m	3	41.22760	-122.38262	9/11/2021	9/15/2021	41.50
			Open Water - 30m	30	41.22760	-122.38262	9/11/2021	9/15/2021	51.00
		Cliff Lake	Open Water - 3m	3	41.20008	-122.49015	9/11/2021	9/14/2021	16.19
		Open Water - 20m	20	41.19940	-122.48965	9/11/2021	9/14/2021	26.34	
Walker Basin	Walker River	Below Topaz Res.	0	38.73532	-119.40076	6/1/2022	6/2/2022	163.50	
		Yerington	0	39.04841	-119.13313	6/1/2022	6/2/2022	221.67	
		Schurz	0	38.94912	-118.80806	6/1/2022	6/2/2022	508.33	
	Walker Lake	Near Mouth	3	38.76070	-118.72518	10/28/2021	11/2/2021	31700.00	
		Center	3	38.70007	-118.72152	10/28/2021	11/2/2021	31733.33	
		South End	3	38.65093	-118.71477	10/28/2021	11/2/2021	31966.67	
	Boat Launch	0	38.75163	-118.72489	10/28/2021	11/2/2021	31600.00		

Appendix B: Table of Major Source Locations

Aqueous Film Forming Foam Sites

Site ID	Name	Details	Lat	Long
DD FTA	Dodd/Deal Fire Academy	Decommissioned, possible legacy AFFF site	39.65446	-119.87075
RSA	Reno Stead Airport	Probable AFFF use in emergencies	39.662	-119.8682
RNO	Reno-Tahoe International Airport	Active AFFF training area	39.50099	-119.76746
FNAS	Fallon Naval Air Station	Active AFFF training area	39.42718	-118.70625

Waste Water Treatment Plants

Site ID	Name	Effluent Discharge Method	Lat	Long
RSWRF	Reno Stead Water Reclamation Facility	Irrigation Reuse Outfall to Surface Waters	39.64453	-119.86942
TTSA	Tahoe-Truckee Sanitation Agency	Discharge to Groundwater	39.3411	-120.12627
FWWTP	Fallon Wastewater Treatment Plant	Outfall to Surface Waters	39.4633	-118.75538
TMWRF	Truckee Meadows Water Reclamation Facility	Outfall to Surface Waters	39.5178	-119.70195
STMWRF	South Truckee Meadows Water Reclamation Facility	Irrigation Reuse	39.45797	-119.74381
CCWRRF	Carson City Resource Recovery Facility	Irrigation Reuse	39.16221	-119.73227
RAWWTP	Rolling-A Wastewater Treatment Plant	Irrigation Reuse Discharge to Groundwater	39.29248	-119.50653

Appendix C: Table of Landcover by Watershed

Large Watersheds											
Watershed	Subbasin Unit (HUC-8)	Sampled Water Bodies	AFFF Sites	WWTPs	Open Water	Perennial Ice/Snow	Developed	Undeveloped	Agriculture	Wetlands	Total
Carson Basin	Carson Desert	Carson River Likes Lake Harmon Reservoir Stillwater Point Reservoir Foxtail Reservoir East Alkali Lake No. 1	Fallon Naval Air Station (FNAS)	Fallon Waste Water Treatment Plant (FWWTP),	62.25	0	84.37	5,044.90	234.5	190.19	5,616.23
Carson Basin	Middle Carson	Lahontan Reservoir Carson River	None Identified	Carson City Water Reclamation Facility (CCWRRF), Rolling-A Wastewater Treatment Plant (RAWTP)	35.32	0	72.56	1992.62	12.29	29.87	2,142.67
Carson Basin	Upper Carson	None	None Identified	None Identified	5.63	0.01	127.02	2170.26	82.28	90.37	2,475.54
Carson Basin	Total	N/A	1 Site	3 Sites	103.2	0.01	283.95	9207.78	329.07	310.43	10234.44
Walker Basin	Walker Lake	Walker Lake	None Identified	None Identified	119.95	0	14.22	1947.26	3.03	3.07	2,087.54
Walker Basin	Walker	Walker River	None Identified	None Identified	7.35	0	30.21	2051.39	148.7	71.46	2,309.11
Walker Basin	West Walker	Walker River	None Identified	None Identified	10.25	0.46	36.94	2447.09	139.03	56.64	2,690.41
Walker Basin	East Walker	None	None Identified	None Identified	13.36	0.98	9.71	2963.52	28.98	85.12	3,101.66
Walker Basin	Total	N/A	None Identified	None Identified	150.91	1.44	91.08	9409.26	319.74	216.29	10188.72
Truckee Basin	Pyramid-Winnemucca Lakes	None	None Identified	None Identified	466.75	0	13.69	3075.11	8.35	12.91	3,576.81
Truckee Basin	Truckee	Truckee River	Reno-Tahoe International Airport (RNO),	Tahoe-Truckee Sanitation Agency (TTSA), Truckee Meadows Water Reclamation Facility (TMMWRF), South Truckee Meadows Water Reclamation Facility (STMWRF)	51.45	0	394	2645.57	9.35	43.04	3,143.39
Truckee Basin	Lake Tahoe	Lake Tahoe	None Identified	None Identified	508.21	0.01	80.56	703.92	0	16.19	1,308.90
Truckee Basin	Total	N/A	1 Site	3 Sites	1026.41	0.01	488.25	6424.6	17.7	72.14	8029.1
Mono Basin	Mono Lake	Mono Lake Lee Vining Creek	None Identified	None Identified	195.51	3.94	21.35	2563.03	0.01	22.27	2,806.09

Medium Watersheds

Watershed	Subwatershed Unit (HUC-12)	Sampled Water Bodies	AFFF Sites	WWTPs	Open Water	Perennial Ice	Developed	Undeveloped	Agriculture	Wetlands	Total
Lemmon Valley	Lemmon Valley	Swan Lake Swan Lake Inflow (Culvert)	Dodd/Deal Fire Training Academy (DDFTA) (Decommissioned), Reno Stead Airport	Reno Stead Water Reclamation Facility (RSWRF)	6.63	0	39.24	203.42	0.02	0.25	249.57
Terminal Lake #2	Redacted	Terminal Lake #2 Terminal Lake #2 Inflow	None Identified	None Identified	2.12	0	0.01	141.97	0	4.1	148.18
Terminal Lake #1	Redacted (multiple)	Terminal Lake #1	1 Anonymized Site	3 Anonymized Sites	1026.41	0.01	488.25	6424.6	17.7	72.14	8029.1

Small Watersheds

Watershed	Delineation Point	Sampled Water Bodies	AFFF Sites	WWTPs	Open Water	Perennial Ice	Developed	Undeveloped	Agriculture	Wetlands	Total
Castle Lake	Outflow	Castle Lake	None Identified	None Identified	0.19	0.00	0.00	1.03	0.00	0.00	1.22
Cliff Lake	Outflow	Cliff Lake	None Identified	None Identified	0.06	0.00	0.00	1.38	0.00	0.00	1.44
Serene Lake	Outflow	Serene Lake	None Identified	None Identified	0.29	0	1.1	3.72	0	0.02	5.13

Note on Appendix C: I pulled data in these tables from the USGS 2019 National Landcover Database using ModelMyWatershed.org. Categories are presented in units of Km² and have been condensed in the following way: Developed = developed, open + developed, low intensity + developed, medium intensity + developed, high intensity; Undeveloped = deciduous forest + evergreen forest + mixed forest + shrub/scrub + grassland/herbaceous; Agriculture = pasture/hay + cultivated crops; Wetlands = woody wetlands + emergent herbaceous wetlands.

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