CalRecycle

The California Department of
Resources Recycling and Recovery

Straw Wattle and Compost Filter Sock Performance Report for Retaining Stormwater Contaminants following the 2018 Camp Fire

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Executive Summary

This study compared the performance of two sediment retention products, straw wattles and compost filter socks, collectively referred to in this report as “products.” These products are used as best management practices (BMPs) for stormwater protection and erosion control following wildfires.

The goal of the study is to analyze and compare the performance of straw wattles and compost filter socks to retain contaminants from stormwater runoff after the 2018 Camp Fire. The straw wattle and compost filter sock samples studied in this research were deployed during the 2018 Camp Fire throughout the town of Paradise where they remained throughout the rainy season until collection in 2019. Sediment samples near these products were also collected and analyzed for contaminant concentrations to compare with the concentrations found in deployed products. This study characterized trace metals, per- and polyfluoroalkyl substances (PFAS), and polychlorinated dibenzo-p-dioxins and furans (PCDD/F, also known as dioxins and furans) in these straw wattle, compost filter sock, and sediment samples.

The findings demonstrated that organic and inorganic contaminants originating from wildland-urban interface (WUI) fires were mobilized into the stormwater collection system. Further, the study provided an indication of the concentrations and profiles of specific chemicals within the broad classes of contaminants (metals, PFAS, PCDD/F).

Key Findings

1. Compost filter socks retained more contaminants than straw wattles.

The results of this study clearly indicate that compost filter socks are more effective than straw wattles at retaining the contaminants studied. **Tables 7-9 and Figures 8-9** illustrate that compost filter socks retained higher concentrations of metals than straw wattles. For six of the seven measured metals, the differences were statistically significant.

Figure 15 shows the distribution of PFAS congeners by material, site, and abundance for all straw wattle, compost filter sock, and sediment samples that had detectable concentrations of PFAS. This figure illustrates that compost filter sock samples retained higher concentrations and more PFAS congeners than straw wattle samples. **Figure 16** shows PFAS controlled sorption experiments at low and high concentrations of PFAS. Low concentration tests show compost filter socks retained approximately 90% of the added PFAS while straw wattles retained little to no PFAS. Higher concentration tests were inconclusive.

For PCDD/F, straw wattle samples had no detectable concentrations, while all compost filter sock and sediment samples contained some detectable concentrations of PCDD/F (**Figures 11-12**).

2. Deployment and management of products are important to ensure BMPs are effective for the use of either erosion control and/or contaminant retention.

During field sampling, the team observed that some products were deployed and/or maintained in ways that would render them less effective, or ineffective for the use of either erosion control and/or contaminant retention (**Figures 17-20**). During sample collection, pictures were taken of other BMPs deployed. There were instances of straw wattles incorrectly placed on hard surfaces and pavement, or straw wattles not staked in place and overtopped by stormwater and debris. As another example, one of the compost filter socks was not staked in place nor strategically placed to intercept stormwater runoff in a trench.

“Hot spots” are locations with elevated chemical concentrations due to burned materials that inherently have high chemical concentrations, like household chemicals and electronics. Hot spots may require more active management of products when they are used for erosion control and contaminant retention. Post-cleanup removal of products used as BMPs should be consistently practiced.

The following are practice recommendations from this study. The final Stormwater Pollution Prevention Plan (SWPPP) should be made by a Qualified SWPPP Developer. Implementation or placement should be made by a Qualified SWPPP Practitioner, who should observe and document the products over the period of storms.

Practice Recommendations

- Use compost filter socks in urban areas that are typically hot spots for contamination after a wildfire (e.g., mobile home parks, commercial areas, etc.).
- Use compost filter socks near waterways to improve runoff quality.
- Use straw wattles and compost filter socks in areas needing soil stabilization and sediment retention.

Report

Introduction

The California Department of Resources Recycling and Recovery (CalRecycle) contracted a research study with Chico State Enterprises (CSE) and the Board of Regents Nevada System of Higher Education (NSHE). The goal of the study was to evaluate the efficacy of straw wattles (SW) and compost filter socks (CFS) to sorb contaminants from stormwater runoff after urban wildfires. SW and CFS were used during the 2018 Camp Fire in the town of Paradise, California, to control and mitigate erosion and contaminants in stormwater runoff. Project principal investigators collected samples of SW, CFS, retained sediments (SD), and ash from the Camp Fire burn area following the 2018-19 winter storm season in the months of late May to August before the cleanup. The cleanup began around January 2019 and lasted until October 2019 (Eto, 2024).

The impacts of wildfires were especially apparent at the wildland-urban interface (WUI) where human dwellings are located in or near wildland vegetation. The WUI is one of the fastest growing types of land use in the U.S., increasing the risk of wildfire ignitions and loss of human lives and homes (Radeloff et al., 2018). When a wildfire burns through urban landscapes, it results in a variety of waste materials, including construction and demolition debris, animal carcasses, vegetative matter, vehicles, vessels, electronic waste, household appliances, and medical waste. The large volume of uncommon, commingled waste is a challenge for the local public works and environmental departments to handle, especially when rain ensues (Maxwell and Matsler, 2023). Fire-damaged land is especially vulnerable to runoff and soil erosion with the loss of vegetation, loss of water collection and routing (e.g., house rain gutters), and increased bare soil (Jung and Hogue, 2008). Consequently, rain can move contaminants from the burned remains into the landscape, potentially polluting nearby water sources.

The results of this study will help inform CalRecycle's Disaster Debris Recovery Operations (DDRO) program and local agencies on appropriate uses of best management practices (BMPs) for mitigating contaminants from stormwater runoff after wildfires. SW and CFS are of particular interest because they are the most commonly used products. SW and CFS are both used in fire debris cleanup, although SW are preferred for economic reasons. Both products can reduce and intercept stormwater and sediment runoff, however, because of the different materials and structures, they can perform differently. With winter rains and the uncertain completion time of fire debris cleanups, there is an additional need for sediment retention products that can also filter stormwater runoff. To date, there are no studies that directly compare the efficacies of SW and CFS to capture and filter runoff for urban contaminants after a wildfire.

The primary objectives of the study were to:

- Evaluate the presence of trace metals, per- and polyfluoroalkyl substances (PFAS), and polychlorinated dibenzo-p-dioxins and furans (PCDD/F) in SW and CFS samples collected after the 2018 Camp Fire.
- Assess the sorption characteristics of SW and CFS samples to determine their efficacy as filter media to remove contaminants from water.

To achieve these objectives, metal analysis, total fluorine analysis, and commercial analyses of extractable organic fluorine, PFAS, and PCDD/F were performed.

Background of wildfire management in California

The state of California typically experiences wildfires during the summer season that have been increasing in intensity and damage in recent years. Most of the largest wildfires in California have occurred in the past 10 years (CAL FIRE, 2022). The risk of a similar event happening in the future is high.

In response to the destruction caused by wildfires, CalRecycle established and manages the Disaster Debris Recovery Operation (DDRO) under the leadership of the California Governor's Office of Emergency Services (Cal OES) and with the collaboration of other federal, state, and local agencies. This includes the California Department of Forestry and Fire Protection (CAL FIRE), Federal Emergency Management Agency (FEMA), California Department of Toxic Substances Control (DTSC), large landowners such as the U.S. Forest Service (USFS), county health officers, and local agencies, such as the county environmental health and public works departments.

There are two phases in the DDRO. During Phase 1, known as Household Hazardous Waste Removal, firefighting activities take place followed by DTSC work to clear properties of visible household hazardous waste and assess and remove asbestos materials. CAL FIRE responds to the fire and Cal OES can install sediment retention products. During Phase 2, known as Debris Removal, CalRecycle performs the following tasks:

- Assess and document the site
- Assess and remove asbestos
- Dispose or recycle debris
- Scrape the contaminated topsoil
- Confirm sampling of multiple soil samples from lots
- Remove hazardous trees

- Implement sediment retention products
- Perform a final inspection

After cleanup, CalRecycle removes the first set of sediment retention products deployed by discarding them in an approved, lined landfill. Placement of the second set of sediment retention products requires a Right of Entry (ROE) and is the last step before the homeowner receives back the property. Homeowners may keep, move, or discard erosion control measures (CalRecycle, 2024).

Description of sediment retention products as BMPs

SW are pre-manufactured rolls consisting of straw in a tubular net (**Figure 1B**). They are placed on slopes at intervals from the top to the toe of the slope to reduce stormwater runoff volume and velocity, intercept runoff, and spread out sediment and stormwater runoff. To install a SW, first dig a shallow trench and secure this product with wooden stakes every 4 feet until 6 inches from the end of the roll. Stakes must be less than 2 inches above the top of the SW. SW have to be maintained with the following actions to remain functional (CA DOT, 2018).

- Remove sediment and debris from behind
- Repair and adjust if concentrated runoff occurs beneath the SW
- Repair or replace if SW become split, unraveled, or torn
- Add stakes if SW sag or slump
- Replace broken wooden stakes

CFS are tubular nets filled with coarser composted woody material (**Figure 1B**). The main function of CFS is to allow water to filter through while retaining sediment. CFS can interrupt and divert runoff, trap pollutants, protect drain inlets, and provide perimeter control. On soil, CFS do not require trenching. However, it is recommended to trim or remove vegetation and debris to fasten them to the soil surface and stake them in place. Different kinds of stakes (e.g. wood, metal, steel, ropes of various shapes) have their own installation requirements for staking CFS to the ground. On hard urban landscapes, certain CFS are heavy enough to not require installation and would be lain around inlets, drainage systems, and receiving channels (Caltrans, 2018). CFS should be inspected regularly after significant rainfall events to remove accumulated sediment and debris and be reinstalled or replaced when damaged or rendered ineffective (Caltrans, 2018; **Figures 1A-1B**).

Figure 1A: Example placements of SW (left) and CFS (right) following the Camp Fire. Photo credits: Jackson Webster (left) and Sandrine Matiasek (right).

Figure 1B: The filling of a sample of SW (left) and CFS (right) collected after the Camp Fire. Photo credits: John Machado.

Urban contaminants

Following urban burning, countless known and unknown chemicals are likely to be present. Many chemicals associated with human activities or combustion products could be considered contaminants. The present study focused on three classes: trace metals, per- and polyfluoroalkyl (PFAS), and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/F). These compound classes are of particular interest because of their persistence in the environment and potential toxicity.

Trace metals can enter the environment from many sources that include:

- Natural processes (e.g., erosion, chemical, physical weathering)
- Altered landscapes (e.g., agriculture, mining, road building)
- Human-related activities (e.g. industrial discharge, consumer use).

Multiple studies have documented increased metals in stormwater runoff following wildfires (Hageman et al., 2008; Mendez, 2010; Burke et al., 2013; Burton et al., 2016; Pinedo-Gonzalez et al., 2017; Magliozzi et al., 2024). Fire increases the relative concentrations of metals in soil through enrichment as organic matter is combusted (Abraham et al., 2017; Bogorodskaya et al., 2011). Metals have negative effects on flora and fauna exposed to this post-fire runoff, resulting in losses in biodiversity in surface waters and soil health degradation in burned watersheds (Abraham et al., 2017). Potential human exposure can result in central nervous system disorders, cancer, and neuropathy (Jyothi, 2021). Understanding how to capture metals at their source can help mitigate the negative environmental and human health impacts of WUI fires.

PFAS are also found in urban environments after wildfire. PFAS are a class of organic contaminants that contain at least one fully fluorinated carbon atom. They include thousands of specific congeners (U.S. EPA, 2022). Because the carbon-fluorine bond is one of the strongest bonds in organic chemistry, PFAS are highly stable and breakdown slowly in the environment, giving them the name “forever chemicals” (Gagliano et al., 2020). On man-made products, PFAS resist heat, water, and oil. This makes PFAS useful in many applications, such as cookware coatings, firefighting foams, rugs, makeup, and many other industrial and consumer applications. In addition to their widespread use and persistence in the environment, PFAS are of growing concern due to their ubiquity and toxicity. PFAS can bioaccumulate in fish and wildlife and move through air, water, and soil systems.

PFAS can result in harmful human health effects, such as decreased fertility, developmental delays, suppression to vaccine response, increased risk of prostate, kidney, and testicular cancer, and other health consequences (Sunderland et al., 2019). The U.S. EPA set drinking water maximum contaminant levels (MCLs) for several PFAS. For the two of the most commonly produced PFAS, Perfluorooctanoic acid (PFOA) and Perfluorooctanesulfonic acid (PFOS), the MCL is set to 4.0 ppt individually. For GenX and perfluorohexanesulfonic acid (PFHxS), the MCLs are set to 10 ppt. PFAS can cause negative human health effects and are known or anticipated to occur in drinking water (U.S. EPA, 2024A).

PCDD/F are persistent organic pollutants that also take a long time to break down in the environment and are of particular concern for WUI fires. PCDD/F are not manufactured but formed through incomplete combustion (Hutzinger et al., 1985). In wildfires and uncontrolled burning, the variability in temperature and combustion completeness greatly increases the likelihood for PCDD/F formation (Ruokojarvi et al., 2000). PCDD/F are also created when plastics are burned (Organotini, et al., 2015).

Of the several hundred chemicals that exist, there are three major classes of PCDD/F: polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofuran (PCDFs), and polychlorinated biphenyls (PCBs) (EPA, 2023B). PCDD/F are extremely toxic chemical compounds that cause damage to the immune system, interference with hormones, developmental problems in children, and reproductive problems in adults. They are ubiquitous and accumulate in food chains, mainly in the fatty tissue of animals (U.S. EPA, 2023). While they cause the same toxic effects, different types of PCDD/F cause effects at various levels of exposure. To reflect the different toxicities, EPA uses Toxic Equivalency (TEQ) values to quantify the overall toxicity of a sample and to compare PCDD/F releases from different sources and/or different periods encompassing various types of PCDD/F (U.S. EPA, 2010; 2024B).

Background of Camp Fire

The Camp Fire started on November 8, 2018, and was fully contained on November 25, 2018. Due to ongoing dry weather, strong winds, and heavy fuel material, Camp Fire burned about 620 km^2 (153,336 acres) of WUI and forested land within the month. About 18,793 structures were destroyed: 13,972 single, multiple, and mixed commercial residences, 528 commercial buildings, and 4,293 other buildings (**Figure 2**). The towns of Paradise and Concow were almost completely destroyed, each losing about 95% of their structures.

The ash deposits from the fire contained high metal concentrations of arsenic (As), cadmium (Cd), cobalt (Co), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) (DTSC, 2018; TetraTech, 2019). Immediately following the fire and before it rained, SW and CFS were deployed throughout the town of Paradise.

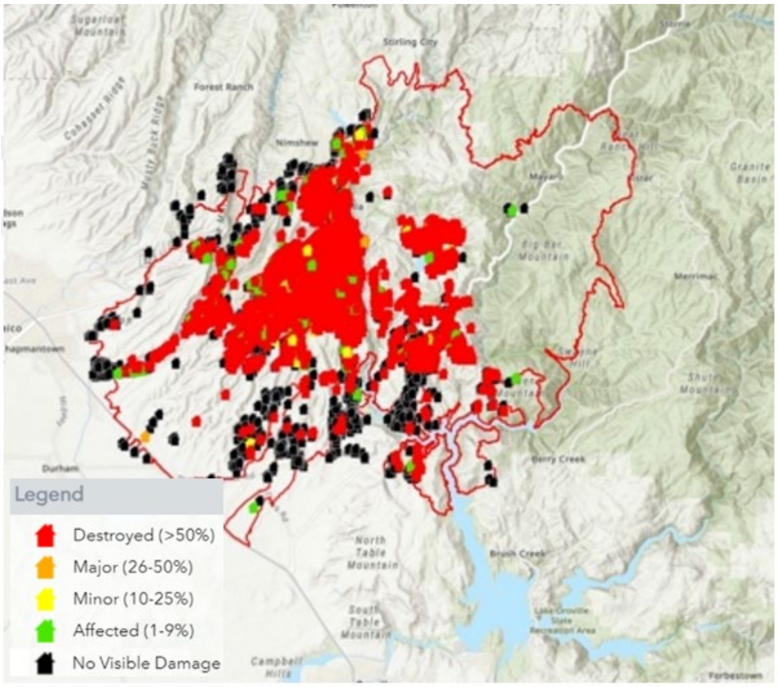

Figure 2: Camp Fire perimeter and structure status map (CAL FIRE, 2023).

Water quality after Camp Fire

Following the Camp Fire, the Watershed Emergency Response Team (WERT) investigated watershed conditions, potential risks to human safety, and the potential for post-fire debris flow (WERT, 2018). The team classified 63% of soils within the Camp Fire burn area as low burn severity, 16% as moderate burn severity, and 2% as high burn severity. Only 19% of the soil was classified as very low severity or unburned soil.

Post-fire peak storm flows corresponding to storm events with annual exceedance probability of 10% or greater were predicted to increase 10-70% over pre-fire storm flows. Erosion models predicted a 4-fold increase in sediment mobilization in burned areas compared to unburned areas. Sediment mobilization was predicted to occur at higher probability in water resources draining into Butte Valley from Paradise (WERT, 2018). The WERT evaluated 1,416 watershed basins for post-fire debris flow hazards. It was estimated that 420 of the 1,416 basins have a likelihood of 60% or greater probability of debris flows. Most of these basins are located along steep slopes that flank the North Fork Feather River and the West Branch of the Feather River upstream of Lake Oroville.

From December 2018 to May 2019, several teams monitored water quality in the region. On several monitoring occasions, concentrations of polycyclic aromatic hydrocarbons (PAHs) that exceeded the EPA human health threshold of 0.2 micrograms per liter (µg/L) were detected in eight of the 10 sampling sites (Parker & Wenzel, 2018). Several metals, including aluminum (Al), Cd, Cu, Pb and Zn, were found to be above the EPA habitat acute criteria in surface water (Magliozzi et al., 2024).

Methods

Study area

The town of Paradise is in the northwestern foothills of the Sierra Nevada, directly east of the California Central Valley and in the western area of the Camp Fire burn zone. The Camp Fire burn area contains multiple landforms, including basins and terraces in the east at approximately 1,500-meter elevation, floodplains, and alluvial fans in the west at 61-meter elevation, with creek systems throughout. This region experiences a typical Mediterranean climate characterized by hot, dry summers and cool, wet winters. Mean annual precipitation for Paradise is 140 cm as primarily rain, with few instances of snow (WERT, 2018). Fire is common in the Sierra foothills and in the area burned by the Camp Fire, 70% of which had burned between the 1960s and 2017. Notable recent fires in Paradise include the 2008 BTU Lightning Complex which burned 115 km^2 (28,465 acres) and the 2008 Humboldt Fire which burned 91 km^2 (22,583 acres) (WERT, 2018).

Prior to the Camp Fire, the ridges and higher elevation sections of the burn area were vegetated with ponderosa pine and montane conifers, fire adapted species (e.g. manzanita), and fire tolerant species (e.g. canyon live oaks). Lower elevations contained mixed shrub species dominated by oak woodlands, montane riparian regions, and valley foothill riparian areas (Burkett and Conlin, 2006; WERT, 2018). Soils consist of clay loams and gravelly clay loams ranging from 0.6 m to 2 m depth on steep slopes in the east to thicker (> 2 m) depths in the lower basin areas of the western half of the burn area (WERT, 2018). The eastern burn area contains sections of serpentine (ultramafic), metamorphic, and igneous rocks which have the potential to contain naturally elevated concentrations of asbestos, silver, mercury, manganese, molybdenum, cobalt, chromium, copper, nickel, lead, and zinc (WERT, 2018). This section of the burn area also contains multiple historic gold, arsenic, barium, copper, chromium, and lead mining sites along the upper reaches of the creeks running through Paradise. The western side of the burn area consists mainly of weathered Cenozoic-era volcanic and sedimentary rocks and does not contain the elevated concentrations of naturally occurring metals observed in the eastern burn zone (WERT, 2018).

Sample collection

22 samples of SW, CFS, and retained sediments (SD) were collected from the town of Paradise within the Camp Fire burn scar between May and August 2019 (**Figure 3; Tables 1-2**). Unused SW (n=1) and unused CFS (n=1) samples were collected in duplicates from nondeployed coils stored outdoors at the Paradise Town Hall to serve as experiment controls. These experiment controls were exposed to the same number of storm events as the deployed SW and CFS but did not receive any stormwater runoff. These samples from Paradise add up to a total of 24 samples. Samples were collected using plastic shovels, nitrile gloves, and plastic bags. Equipment was acid-washed with dilute nitric acid (HNO_3). SW (n=5) and CFS (n=8) samples were collected in ~30-cm segments throughout the town of Paradise at commercial, residential, and mobile home park sites. These products were deployed within days of the fire and prior to winter

storms to prevent sediment and ash from entering water resources via surface runoff. The mesh casing of each SW and CFS sample was opened using scissors to retrieve the inner straw or compost material. SD (n=9) was collected throughout the depth of retainment and across the length of each sampled section of SW or CFS using plastic shovels.

Table 1: Number of samples and controls collected in Paradise, CA

Sample Type	Number (n)
SW	5
CFS	8
SD	9
SW-CT	1
CFS-CT	1
Total	24

Table 2: Information about samples and analyses

Y: Yes, N: No; Sample IDs contain the sample type (SW: Straw wattle, CFS: Compost filter sock, SD: Sediment) followed by the urban area type (R: Residential, C: Commercial, MH: Mobile home, CT: Control). The number after the urban area indicates the specific site and the upper-case letter differentiates samples that were collected at the same location.

Sample ID	Sample Type	Urban area	Date Collected	GPS Coordinates (Decimal Degrees)	Metal analysis (CSUC)	EOF; PFAS & PCDD/F (Eurofins)
SW-C-1A	SW	Commercial	8/19/2019	39.761771, -121.603449	Y	Y
SW-C-1B	SW	Commercial	8/19/2019	39.761212, -121.603411	Y	Y
SW-MH-1	SW	Mobile Home	5/23/2019	39.777867, -121.576250	Y	Y
SW-MH-2	SW	Mobile Home	5/23/2019	39.774800, -121.598050	Y	N
SW-R-1	SW	Residential	6/13/2019	39.737950, -121.624050	Y	Y
CFS-C-1	CFS	Commercial	8/19/2019	39.761528, -121.604194	Y	Y
CFS-C-2	CFS	Commercial	8/19/2019	39.760694, -121.603611	Y	Y
CFS-MH-1	CFS	Mobile Home	5/23/2019	39.776967, -121.576117	Y	N
CFS-R-2A	CFS	Residential	5/23/2019	39.771639, -121.610278	Y	N
CFS-R-2B	CFS	Residential	5/23/2019	39.771556, -121.610389	Y	N
CFS-R-2C	CFS	Residential	5/23/2019	39.771502, -121.610425	Y	Y
CFS-R-3	CFS	Residential	5/29/2019	39.774880, -121.608170	Y	Y
CFS-R-4	CFS	Residential	6/21/2019	39.739889, -121.640500	Y	N
SD-C-1A	SD	Commercial	8/19/2019	39.761771, -121.603449	Y	N
SD-C-1C	SD	Commercial	8/19/2019	39.760695, -121.603621	Y	Y
SD-C-2	SD	Commercial	8/19/2019	39.761520, -121.604198	Y	Y
SD-MH-1	SD	Mobile Home	5/23/2019	39.777867, -121.576250	Y	Y

Table 2: Information about samples and analyses (continued)

Sample ID	Sample Type	Urban area	Date Collected	GPS Coordinates (Decimal Degrees)	Metal analysis (CSUC)	EOF; PFAS & PCDD/F (Eurofins)
SD-MH-2	SD	Mobile Home	5/23/2019	39.774800, -121.598050	Y	N
SD-MH-3	SD	Mobile Home	6/13/2019	39.740492, -121.679861	Y	Y
SD-R-1	SD	Residential	6/13/2019	39.737950, -121.624050	Y	Y
SD-R-2A	SD	Residential	5/23/2019	39.771423, -121.609919	Y	N
SD-R-2B	SD	Residential	5/23/2019	39.771544, -121.609672	Y	N
SW-CT	SW	Control	5/23/2019	39.749282, -121.634284	Y	Y
CFS-CT	CFS	Control	5/23/2019	39.749278, -121.634278	Y	Y

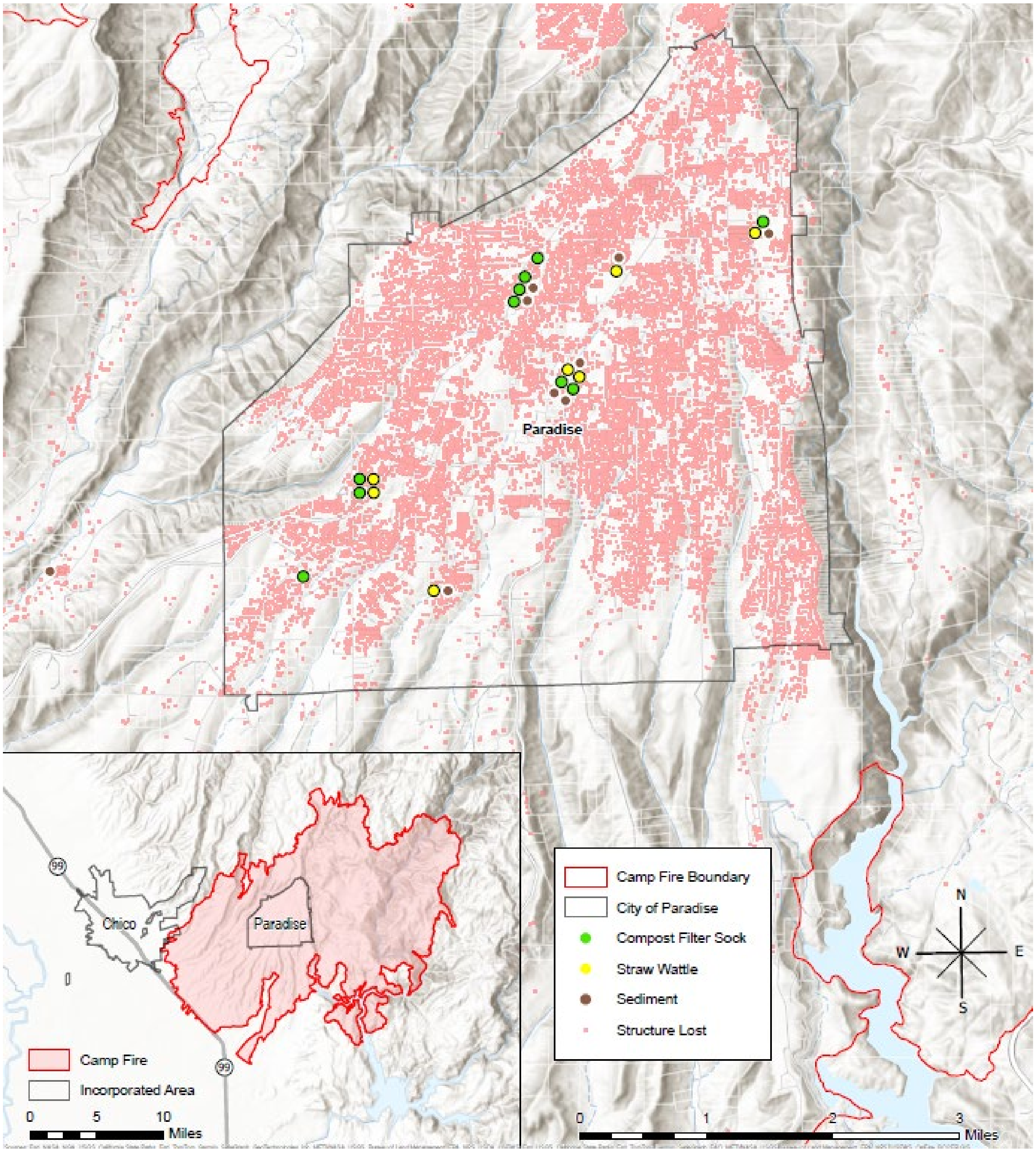

Figure 3: Map of sampling locations for CFS, SW, and SD deployed in the Town of Paradise after the 2018 Camp Fire. The fire boundary and structures destroyed by the fire are displayed. Map Created by Adil Syed with ArcGIS.

Sample storage and preparation

SW, CFS, and SD samples were stored in polyethylene plastic bags and chilled to 4°C in a cooler packed with ice packs during transport to the laboratory. They were stored frozen at -18°C until analysis in the spring of 2023.

Coming out of freezer storage, samples were thawed at room temperature, air-dried at 50°C for 72 hours, and homogenized. Samples sent to the commercial lab for processing were not homogenized, as the laboratory requested as little modification as possible. SD samples were ground using an agate mortar and pestle and passed through a 2-mm nylon sieve, both acid-washed with dilute HNO3 between samples. Straw wattle and compost materials were ground with a blade grinder and passed through a 2-mm nylon sieve, to increase surface area and ensure sample homogeneity. The grinder and sieve were washed between samples; first cleaned with a laboratory sponge and rinsed with Type 1 water (UV-oxidized, resistivity of 18.2 MΩ·cm at 25°C, EMD Millipore Synergy), then rinsed with trace metal grade HNO3. The equipment was then dried using Kimwipes.

Metal analysis

Seven trace metals, known to be elevated in stormwater runoff following this wildfire (Magliozzi et al., 2024), were selected for this study: arsenic (As), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn). Samples were digested in acid with a CEM MARS-6 microwave digestion system following modified versions of EPA Method 3051A for SW and CFS samples, and EPA Method 3050B for SD samples (U.S. EPA, 2007A; U.S. EPA, 1996).

For SW and CFS, 0.5 g of sample was placed in a digestion vessel with 10 mL of trace metal-grade HNO_3 and heated to 200°C for 15 minutes at 800 psi. For SD samples, 1.0 g of sample was placed in a digestion vessel with 10 mL of 50% (vol.) trace metal grade HNO_3 and heated to 95°C for 5 minutes at 800 psi. After allowing the vessel to cool, 5 mL of concentrated trace metal grade HNO_3 was added to the vessels and heated to 95°C for 5 minutes at 800 psi. Finally, 2 mL of Type 1 water and 3 mL of trace metal grade 30% hydrogen peroxide were added to the vessels and covered for 10 minutes.

Following digestion, all samples were diluted to 50 mL using Type 1 water, then underwent an additional 10-fold dilution to ensure metal concentrations were within a 1 – 10^4 µg/L standard curve. Digestion duplicates and blanks were generated for each digestion cycle and matrix-spiked samples were created by adding 100 µL of 10 mg/L SPEX ICP-MS Multi-Element Solution 2A (Claritas PPT, PN: CLMS-2AN) to samples. Reference materials also were included in each digestion cycle. Certified rye grass (European Reference Material, ERM-CD281) was digested with the SW and CFS samples, and certified metals in soil (Sigma-Aldrich, SQC001-30G) were digested with sediment samples.

Samples were analyzed for metal concentrations using a Perkin Elmer NexION 2000 Inductively Coupled Plasma Mass Spectrometer with a modified version of EPA Method 6020A (U.S. EPA, 1998). Check standards (50 µg/L) and instrument blanks were analyzed every 10-15 samples. Internal standard recoveries of scandium-45, rhodium-103, indium-115, and iridium-193 were constantly monitored and used to correct sample concentrations for instrument drift using SPEX Certiprep Internal Standard Mix 1 (Claritas PPT, CL-ISM1-500).

Seven elements were selected based on accuracy and reproducibility of QA/QC checks and their environmental relevance in stormwater (**Table 3**). Standard curve linearity for each element was ensured with correlation coefficients higher than 0.997. Detection limit was defined as three times the standard deviation of the instrument response for eight instrument blanks divided by the slope of the standard curve. Detection limit values ranged from 2.0 x 10^{-2} µg/L (Cr, Co, Ni) to 9.0 x 10^{-1} µg/L (Zn, **Table 3**). Mean digestion blanks concentrations were less than detection limits and were subtracted from sample concentrations. Digestion duplicate percent differences ranged from 0.4 to 5.2%. Matrix-spiked percent recoveries for each element ranged from 96 to 106%. Check standard percent differences ranged from 6.0 to 16%. Certified reference material rye wheat percent recovery ranged from 72 to 90%, with the arsenic concentration below the detection limit. The soil certified reference material percent recovery ranged between 72 to 127%.

Table 3: Results of QA/QC checks for trace metal analysis. Correlation coefficients (R^2) of standard curves, limits of detection (DL), check standard percent differences, and certified reference material percent recoveries.

Element	R^2	Limit of Detection (µg/L)	Check Standard Percent Difference (%)	Certified Reference Material Percent Recovery: Rye wheat, Soil (%)
As	0.9998	0.53	6.0	<DL, 110
Cr	0.9985	0.02	16	83, 105
Co	0.9987	0.02	14	N/A, 105
Cu	0.9986	0.08	9.7	89, 111
Ni	0.9987	0.02	12	84, 108
Pb	0.9988	0.13	8.9	89, 127
Zn	0.9970	0.90	6.6	72, 90

The retention of metals by sediment retention products was quantified by calculating control-corrected concentrations, obtained by subtracting the mean metal concentration measured in control duplicates (nondeployed from Paradise) from the metal concentration measured in each product. The Shapiro-Wilk test performed with RStudio software established that metal concentrations were not normally distributed. Consequently, the statistical significance of differences in metal concentrations was assessed with the Kruskal-Wallis test, a nonparametric alternative to the one-way ANOVA test, using RStudio.

Fluorine analyses

Total fluorine (TF) and extractable organic fluorine (EOF) contents in SW (n = 5), CFS (n = 8), SD (n = 9) samples, and aqueous- and solid-phase samples from sorption tests (n= 13) were analyzed at the University of Nevada, Reno.

TF and EOF were measured by combustion ion chromatography (CIC, **Figure 4**). Following freeze-drying, samples were combusted at ~1000°C to mineralize fluorine (inorganic and organic forms) to hydrofluoric acid which was then dissolved in water. This water was analyzed with CIC and the original fluorine content can be calculated from the resulting concentration of fluoride ions present in water. Because TF includes inorganic fluorinated compounds, such as sodium fluoride or calcium fluoride, in addition to fluorinated organic compounds, this analysis is the least selective and overestimates PFAS content.

The EOF analysis can be a better estimate for PFAS due to its higher specificity. In this method adapted from Higgins and Luthy (2006), organic fluorine was extracted from freeze-dried samples with methanol and ammonium hydroxide prior to CIC. The determined fluoride concentrations reflected the original EOF content in samples.

This method is not highly selective as it cannot be used to identify individual PFAS congeners. It can underestimate PFAS concentrations by generally accounting only for anionic PFAS. In samples containing high concentrations of fluoride, EOF analysis can overestimate organic fluorine and can yield highly variable results due to fluoride competing for binding sites on anion exchange cartridges during sample cleanup or on the ion chromatography column (DeNicola et al., 2023). Thus, results from environmental samples must be carefully screened for fluoride interference.

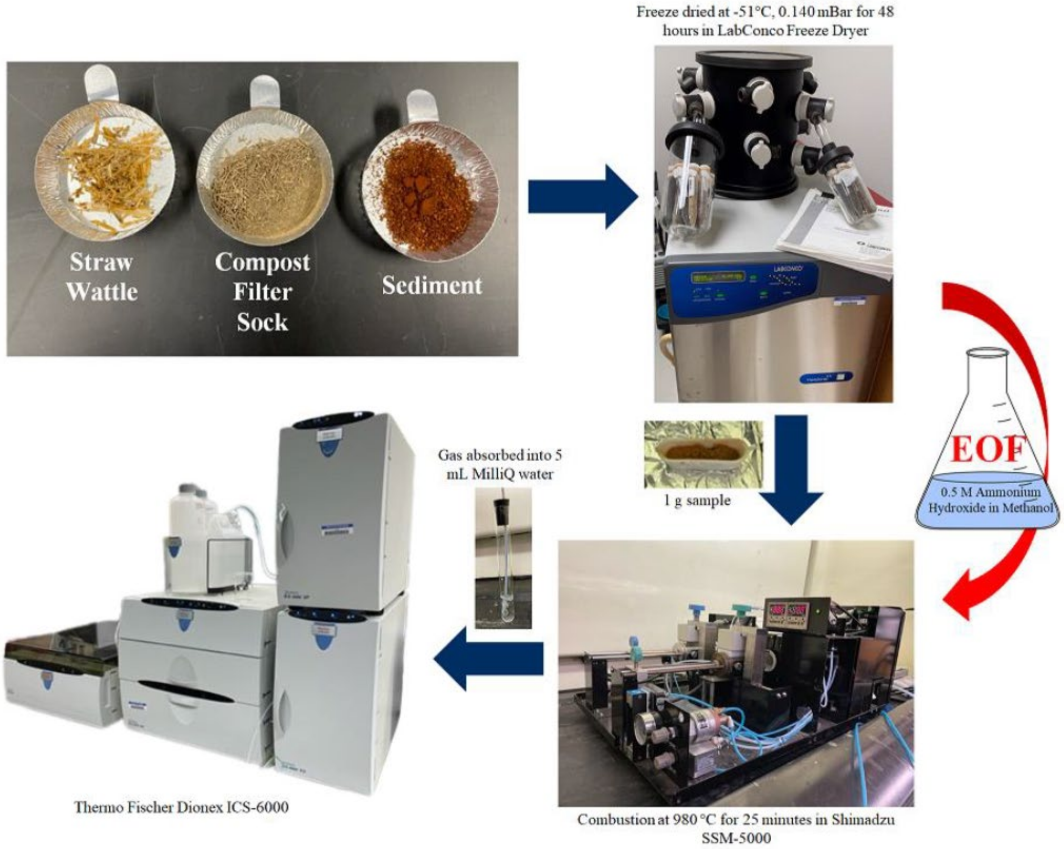

Figure 4: Visual schematic representations of the total fluorine (TF, blue) and extractable organic fluorine (EOF, blue and red) analyses.

Dried and homogenized SW, CFS, and SD samples along with aqueous samples from the sorption tests contained in polypropylene bottles were shipped overnight in ice-chilled coolers at ~4°C. Upon receipt in the laboratory, solid samples were kept at -20°C and aqueous samples were stored at 5°C until analysis. Solid samples were freeze-dried at -51°C, 0.140 mBar for a minimum of 48 hours. Approximately 0.3-1.0 g of each solid sample was prepared for analysis. For TF analysis, each solid sample was transferred into a ceramic boat and covered with ceramic fiber as necessary to prevent sample loss. Aqueous sample bottles were shaken before 1 mL was pipetted from the center of the bottle into a ceramic boat. Ceramic boats were then evaporated to dryness on a hot plate.

For EOF analysis, freeze-dried solid samples (0.3-1.0 g) were mixed with 3 mL of 0.5 M ammonium hydroxide in HPLC-grade methanol, vortexed on high for 30 seconds, sonicated for 30 minutes, and centrifuged at 4,650 rpm for 30 minutes. Extracts were evaporated twice under N_2 gas in a Biotage TurboVap until the remaining sample volume was approximately 1 mL before the transfer into ceramic boats and evaporation to dryness on a hot plate. No EOF analysis was conducted on aqueous samples.

A Shimadzu SSM-5000 combustion unit was set to 980°C with ultra-high purity oxygen gas at a flow rate of 0.25 L/min. Samples were combusted for 25 minutes and the effluent gas was collected into 5 mL Milli-Q water (UV-treated Type I water, resistivity > 18.2 MΩ•cm, Millipore Sigma). The effluent gas tube was rinsed between sample combustions. At the beginning of each analysis set, at least one blank was run, consisting of an empty ceramic boat.

A Thermo Fisher Dionex ICS-6000 ion chromatography unit was programmed to the following settings: pump at 0.25 mL/min, generator at 10 mM (gradient to 45 mM), suppressor at 28 mA (legacy, using hydroxide). The column temperature was set to 20°C and the cell heater to 30°C. When the monitored background level was below 1.0, the analysis could begin. Typical retention time for fluoride was 4.9 minutes or 5 minutes after a new AS20 column was installed. Each analysis consisted of Milli-Q water, a 9-point calibration curve, and the sorbent water samples generated from each sample combustion. Calibration curves and peaks were evaluated after each run, and fluoride concentrations were determined.

Commercial analyses of EOF, PFAS, and PCDD/F

The TF results guided the selection of 13 samples for commercial analysis. SWs (n=4), CFS (n=4), and SD (n=5) samples with the highest total fluorine concentrations, along with one nondeployed control of each material collected from Paradise, underwent comprehensive and EPA-compliant EOF, PFAS, and PCDD/F analyses at a commercial laboratory in Sacramento called Eurofins. The new controls from Salix Applied Earthcare were also analyzed for PFAS for comparison to the nondeployed controls sampled in Paradise. The following standard methods were used: Eurofins proprietary method ELLE by CIC for EOF, modified EPA Method 537 for 39 PFAS congeners (**Table 4**, U.S. EPA, 2020), and EPA Method 8290A for 17 PCDD/F (**Table 5**, U.S. EPA, 2007B).

For EOF, reporting limits ranged from 220 to 1,200 ng/g, the method blanks contained non-detect concentrations, and laboratory control sample (LCS) spike percent recoveries ranged from 103 to 106%. Duplicate LCS spike percent differences were 1% and 2%, respectively. The EOF results presented in the study are from Eurofins Scientific.

PFAS analysis was conducted using a modified version of EPA Method 537 by liquid chromatography with tandem mass spectrometry (U.S. EPA, 2020). Reporting limits were 0.20 µg/kg for all PFAS analyzed. The method blank contained non-detect concentrations for all analytes and isotope dilution percent recovery ranged from 65 to 96%, with the exception of perfluorobutanoic acid (PFBA) which was consistently below the acceptance limits. All analytes with recoveries below acceptance limits were still included in analyses. LCS spike percent recoveries ranged between 86 to 120%, with the exception of fluorotelomer carboxylic acid (FTCA, 46%). Two sample process control (SPC) spikes were analyzed and percent recoveries ranged from 69 to 126%, with the exception of perfluorobutanesulfonoic acid (PFBS, 427% and 772%). Isotope dilution percent recoveries for the SPC spikes ranged from 51 to 114%, with the exception of six analytes all below the acceptable limits; Perfluorotetradecanoic Acid (PFTeDA), N-Ethyl Perfluorooctanesulfonamidoacetic Acid (N-EtFOSA-M), N-Methyl Perfluorooctanesulfonamidoacetic Acid (N-MeFOSE-M), N-Ethyl, Perfluorooctanesulfonamidoacetic Acid (N-EtFOSE-M), 6:2 Fluorotelomer Carboxylic Acid (6:2 FTCA), and 8:2 Fluorotelomer Carboxylic Acid (8:2 FTCA).

Polychlorinated dibenzo-p-dioxin and furan (PCDD/F) analysis was also conducted by Eurofins Scientific using EPA Method 8290A using high-resolution gas chromatography/high resolution mass spectrometry (U.S. EPA, 2007). Isotope dilution recovery percent limits ranged from 40 to 135%, with all measurements qualifying as acceptable. Reporting limits for method blanks and LCS spikes ranged from 1 to 10 pg/g for the 17 analytes measured. All method blank had non-detect concentrations with isotope dilution recoveries ranging from 69 to 93%. LCS spike recoveries ranged from 86 to 107% with isotope dilution recoveries ranging from 67 to 102%. Duplicate LCS spike recoveries ranged from 88 to 107% with isotope dilution recoveries ranging 69 to 94%.

Table 4: Information about the PFAS congeners studied

Name	Abbreviation	Longest Carbon Chain	Total Carbons
Perfluoro(2-ethoxyethane)sulfonic acid	PFEESA	2	4
4,8-Dioxa-3H-perfluorononanoic acid	ADONA	3	7
Hexafluoropropylene oxide dimer acid	HFPO-DA	3	6
Perfluorobutanesulfonic acid	PFBS	4	4
Perfluorobutanoic acid	PFBA	4	4
Perfluoro-4-methoxybutanoic acid	PFMBA	4	4
Perfluoro-3-methoxypropanoic acid	PFMPA	4	4
Perfluoropentanoic acid	PFPeA	5	5
Perfluoropentanesulfonic acid	PFPeS	5	5
9-Chlorohexadecafluoro-3-oxanonane-1-sulfonic acid	9Cl-PF3ONS	6	8
Perfluorohexanoic acid	PFHxA	6	6
Perfluorohexanesulfonic acid	PFHxS	6	6
1H,1H, 2H, 2H-Perfluorohexane sulfonic acid	4:2FTS	6	6
Perfluoroheptanoic acid	PFHpA	7	7
Nonafluoro-3,6-dioxaheptanoic acid	NFDHA	7	7
Perfluoroheptanesulfonic acid	PFHpS	7	7
Perfluoroheptanesulfonic acid	PFHpS	7	7
11-Chloroeicosafluoro-3-oxaundecane-1-sulfonic acid	11Cl-PF3OUdS	8	10
Perfluorooctanoic acid	PFOA	8	8
Perfluorooctanesulfonic acid	PFOS	8	8
1H,1H, 2H, 2H-Perfluorooctane sulfonic acid	6:2FTS	8	8
N-ethyl perfluorooctanesulfonamidoacetic acid	NEtFOSAA	8	12
N-methyl perfluorooctanesulfonamidoacetic acid	NMeFOSAA	8	11
N-ethyl perfluorooctane sulfonamido ethanol	NEtFOSE	8	12
N-methyl perfluorooctane sulfonamido ethanol	NMeFOSE	8	11
Perfluorooctanesulfonamide	FOSA	8	8
N-ethyl perfluorooctane sulfonamide	NEtFOSA	8	12

Table 4: Information about the PFAS congeners studied (continued)

Name	Abbreviation	Longest Carbon Chain	Total Carbons
N-ethyl-perfluorooctane sulfonamido ethanol	NEtFOSE	8	8
Perfluorononanoic acid	PFNA	9	9
Perfluorononanesulfonic acid	PFNS	9	9
Perfluorodecanoic acid	PFDA	10	10
Perfluoroundecanoic acid	PFUnA	10	10
1H,1H, 2H, 2H-Perfluorodecane sulfonic acid	8:2FTS	10	10
Perfluorodecanesulfonic acid	PFDS	10	10
Perfluorododecanoic acid	PFDoA	12	12
Perfluorododecane sulfonate	PFDoS	12	12
Perfluorotridecanoic acid	PFTrDA	13	13
Perfluorotetradecanoic acid	PFTA	14	14
Perfluorotetradecanoic acid	PFTeA	14	14

Table 5: Information about the PCDD/F congeners studied

Congener Name	Congener Abbreviation	TEF (2005)
2,3,7,8-Tetrachlorodibenzo-p-dioxin	2,3,7,8-TCDD	1
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	1,2,3,7,8-PeCDD1	1
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	1,2,3,4,7,8-HxCDD	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	1,2,3,6,7,8-HxCDD	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	1,2,3,7,8,9-HxCDD	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	1,2,3,4,6,7,8-HpCDD	0.01
1,2,3,4,5,6,7,8-Octachlorodibenzo-p-dioxin	1,2,3,4,6,7,8,9-OCDD	0.0003
2,3,7,8-Tetrachlorodibenzofuran	2,3,7,8-TCDF	0.1
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,7,8-PeCDF	0.03
2,3,4,7,8-Pentachlorodibenzofuran	2,3,4,7,8-PeCDF	0.3
1,2,3,4,7,8-Hexachlorodibenzofuran	1,2,3,4,7,8-HxCDF	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	1,2,3,6,7,8-HxCDF	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran	1,2,3,7,8,9-HxCDF	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran	2,3,4,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1,2,3,4,6,7,8-HpCDF	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1,2,3,4,7,8,9-HpCDF	0.01
1,2,3,4,5,6,7,8-Octachlorodibenzofuran	1,2,3,4,6,7,8,9-OCDF	0.0003

Laboratory PFOA sorption experiments

To evaluate the PFAS sorption characteristics of CFS and SW samples, bench-scale sorption tests were conducted. Another set of unused SW and CFS, donated by a local BMP practitioner (Salix Applied Earthcare), were used in these experiments. Materials were rinsed to decrease the leachable fraction of organic matter, also known as dissolved organic carbon (DOC). Rinsing was conducted in 1 kg batches with deionized laboratory grade water until the DOC concentrations resulting from immersion were steady with each successive rinse. This material was then dried at 50°C for a period of 24 hours.

For low DOC sorption batch experiments, 1 g of rinsed material (compost or straw) was placed in a new (unused and uncleaned) 125 mL wide mouth HDPE vessel (Fisher Scientific, USA) and filled with 100 mL of UV-treated Type I water. For high DOC experiments, the rinse water from the material preparation was used to fill the vessels, resulting in DOC concentrations of ~700 mg/L. For medium DOC experiments the rinse water was diluted to DOC concentration of ~100 mg/L.

In some experimental vessels, parallel sorption tests were also conducted using 100 mg ash obtained from a woodstove that had received a mixture of pine, fir, and oak logs. One-liter solutions spiked with 50, 100, 500, and 1000 ppb PFOA were then added to the vessels. A PFOA stock solution was prepared by dissolving a vial of certified 100 mg PFOA into 5 mL HPLC grade methanol and adding this to 1 L of type 1 water. From this stock, serial dilutions were made into 1-L volumetric flasks precleaned with acid, methanol, and DI rinses. Spiked vessels were then shaken for 24 hours. After equilibration, the supernatant was carefully decanted from the vessels and all the remaining water was drained from the HDPE vessels. The supernatant was filtered using single-use HDPE syringes fitted with a 32-mm filter housing with a 0.45 µm Supor membrane. The straw and compost material were dried at 50°C for a period of 24 hours.

Both the aqueous and solid phase samples from the sorption experiments were analyzed at UNR using the total fluorine (TF) method described above which follows the Higgins method adapted from EPA Method 1633 (U.S. EPA, 2024C) In **Table 6** below, the first two columns display the concentrations of PFOA and TF in the aqueous solution (TF_{aq}). TF_{aq} is of particular interest because it is the concentration of fluorine in PFOA that was spiked into the vessel solution. In later sections, spikes of PFOA will be referred to as their equivalents of TF_{aq} concentrations. The concentration of TF that sorbed onto the solid material of SW and CFS, assuming that all the fluorine in PFOA has sorbed to the solid material, is displayed in the third column as $TF_{s.}$

Table 6: Concentrations of PFOA used in the sorption experiments with corresponding aqueous and solid concentrations of total fluorine (TF).

PFOA concentration (aqueous)	Total Fluorine concentration (TF_{aq}) *	Total Fluorine concentration (TF_{s})
50 ppb	3.4 ppb	3.4 ppm
100 ppb	6.9 ppb	6.9 ppm
500 ppb	34 ppb	34 ppm
1000 ppb	69 ppb	69 ppm

*There is 0.69 TF for every 1 PFOA.

Results

Metal results

Metal contents in SW, CFS, and SD samples

Trace metal (As, Co, Cr, Cu, Ni, Pb, Zn) concentrations in SW, CFS, and SD samples and controls varied by five orders of magnitude, ranging from 0.02 mg/kg (Co in SW) to 1,500 mg/kg (Zn in SD) (**Table 7**). SW samples contained the lowest metal concentrations overall ($p < 0.01$, Kruskal-Wallis test), while metal concentrations in CFS samples were similar to concentrations found in sediments retained by BMPs, except for Co, Cr and Ni concentrations, which were higher in sediment ($p < 0.05$).

Zn was the most abundant trace metal in all sample types, with median concentrations of 200 mg/kg in CFS, 140 mg/kg in SD, and 24 mg/kg in SW. Cu, Cr, and Ni were the next most abundant metals, followed by As, Co, and Pb. As was the metal present at the lowest median concentrations in CFS and SD samples (8.3 and 7.6 mg/kg, respectively), while Pb was the least abundant metal in SW samples (0.80 mg/kg; **Table 8**).

To give context to the metal concentrations found in SW, CFS, and SD samples, these concentrations were compared to the environmental cleanup goals for residential soils established by TetraTech after the Camp Fire for geologic unit topological vector profiles (tertiary volcanics, pyroclastics, andesite) where samples were collected (TetraTech, 2019, **Table 8**). The cleanup goal assigned to each metal was the more stringent of the following California and federal health-based and hazardous waste standards, unless the background concentration was higher: the California Department of Toxic Substances Control Human and Ecological Risk Office (DTSC HERO) levels, the California Hazardous Waste Total Threshold Limits Concentration, and the U.S. EPA Regional Soil Levels. While Zn was the most abundant metal in samples, none of the measured Zn concentrations exceeded the environmental cleanup goal (4999 mg/kg) for that metal. In contrast, the cleanup goal for As (7.7 mg/kg) was exceeded in four of nine SD samples and in five of eight CFS samples. Two SD samples also exceeded the cleanup goal for Pb (80 mg/kg).

Comparing metal concentrations in the nondeployed control products collected in Paradise to concentrations in the deployed SW and CFS samples can provide information on background metal contents in BMPs. There was no statistical difference in metal concentrations between nondeployed SW controls and SW samples (Kruskal-Wallis test, p: 0.053 – 0.44). In contrast, all metal concentrations in CFS samples were significantly different from concentrations in nondeployed CFS controls (Kruskal-Wallis test, p = 0.037), except for Cr (p = 0.19).

Table 7: Metal concentrations (mg/kg) in samples and controls. The control concentrations are the mean values obtained from duplicates.

SW: Straw wattle, CFS: Compost filter sock, SD: Sediment, R: Residential, C: Commercial, MH: Mobile home, CT: Control

Sample ID	As	Co	Cr	Cu	Ni	Pb	Zn
SW-C-1A	1.6	1.6	1.6	8.4	3.1	0.26	21
SW-C-1B	0.51	1.0	3.2	9.3	2.8	0.41	25
SW-MH-1	1.7	3.3	29	14	20	3.1	24
SW-MH-2	1.6	2.0	10	8.1	7.2	1.7	27
SW-R-1	1.1	0.34	0.90	4.3	1.1	0.10	25
CFS-C-1	7.7	19.9	28	101	40	40	859
CFS-C-2	8.9	6.9	43	72	37	48	797
CFS-MH-1	12	11.8	82	85	41	30	164
CFS-R-2A	26	11.7	93	225	41	33	307
CFS-R-2B	24	14.8	112	238	56	68	307
CFS-R-2C	5.0	10.7	77	44	52	15	109
CFS-R-3	2.3	6.9	36	85	29	27	188
CFS-R-4	7.4	27	117	122	66	46	215
SD-C-1A	18	19	103	127	68	82	758
SD-C-1C	8.0	22	127	62	73	34	98
SD-C-2	18	21	79	223	63	79	1486
SD-MH-1	7.6	13	97	136	52	88	1322
SD-MH-2	2.7	14	83	36	52	15	54
SD-MH-3	3.0	17	157	55	90	20	47
SD-R-1	4.6	23	128	44	70	24	81
SD-R-2A	9.9	22	141	66	82	24	199
SD-R-2B	5.1	8.5	49	76	34	34	141
SW-CT	b.d	0.02	0.42	2.8	0.32	0.24	19
CFS-CT	1.1	1.5	33	11	19	3.7	45

b.d.= below detection

Table 8: Summary of metal concentrations (mg/kg) measured in SW, CFS, and SD samples deployed in Paradise after the 2018 Camp Fire and in non-deployed controls compared to environmental cleanup goals for soils (TetraTech, 2019) with the number of samples that exceeded the cleanup goal, minimum, maximum, and median concentrations displayed for each material type.

	Cleanup Goal (mg/kg)	SW Min (mg/kg)	SW Max (mg/kg)	SW Median (mg/kg)	CFS Min (mg/kg)	CFS Max (mg/kg)	CFS Median (mg/kg)	SD Min (mg/kg)	SD Max (mg/kg)	SD Median (mg/kg)
As	7.7	0.5	1.7	1.4	1.1	26	8.3	2.7	18	7.6
Co	36	0	3.3	1	1.5	27	11	8.5	23	19
Cr	2499	0.4	29	1.6	28	120	72	49	160	100
Cu	2499	2.8	14	8.1	11	240	85	36	220	66
Ni	490	0.3	20	2.8	19	66	41	34	90	68
Pb	80	0.1	3.1	0.3	3.7	68	32	15	88	34
Zn	4999	19	27	24	45	860	200	47	1500	140

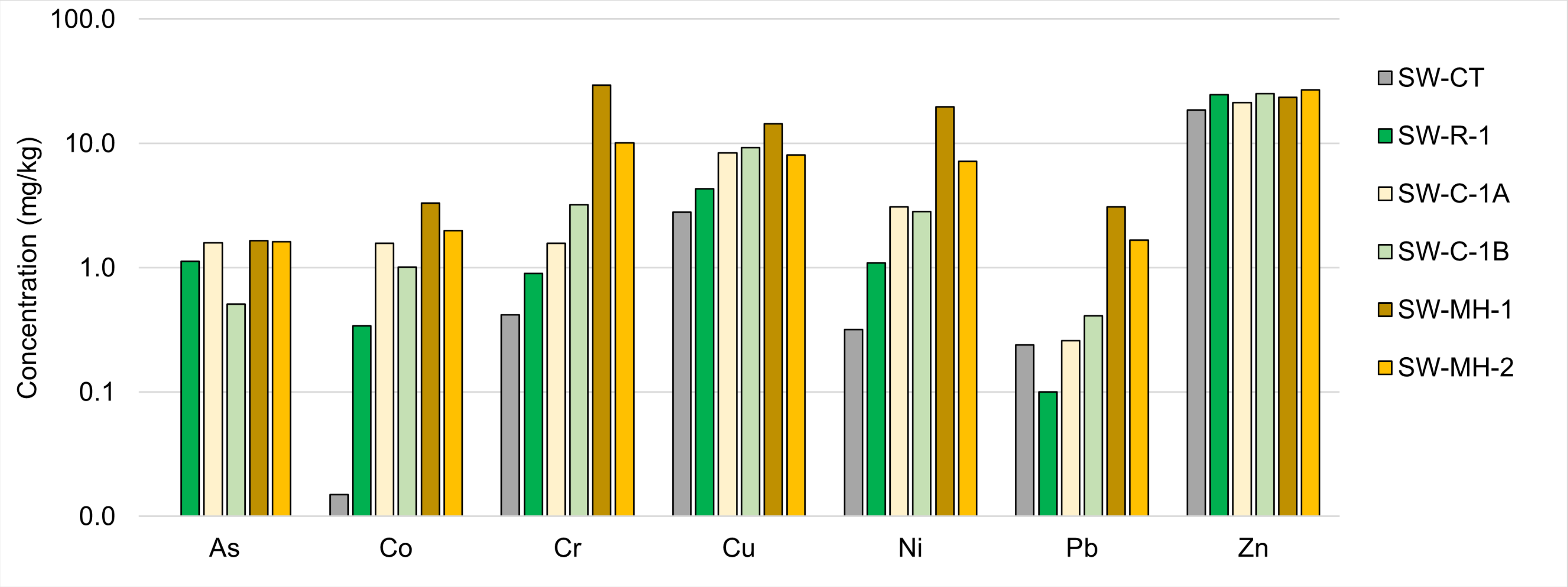

Figure 5: Concentrations of trace metals (As, Co, Cr, Cu, Ni, Pb, Zn) in SW.

SW samples (n=5) with trace metal control concentrations displayed in gray bars on the left of each dataset. The control concentration of As was below the detection limit, so it was assumed to be 0 mg/kg. Concentrations are displayed on a logarithmic scale. There are no SW samples that exceeded the environmental cleanup goals (Tetra Tech, 2019).

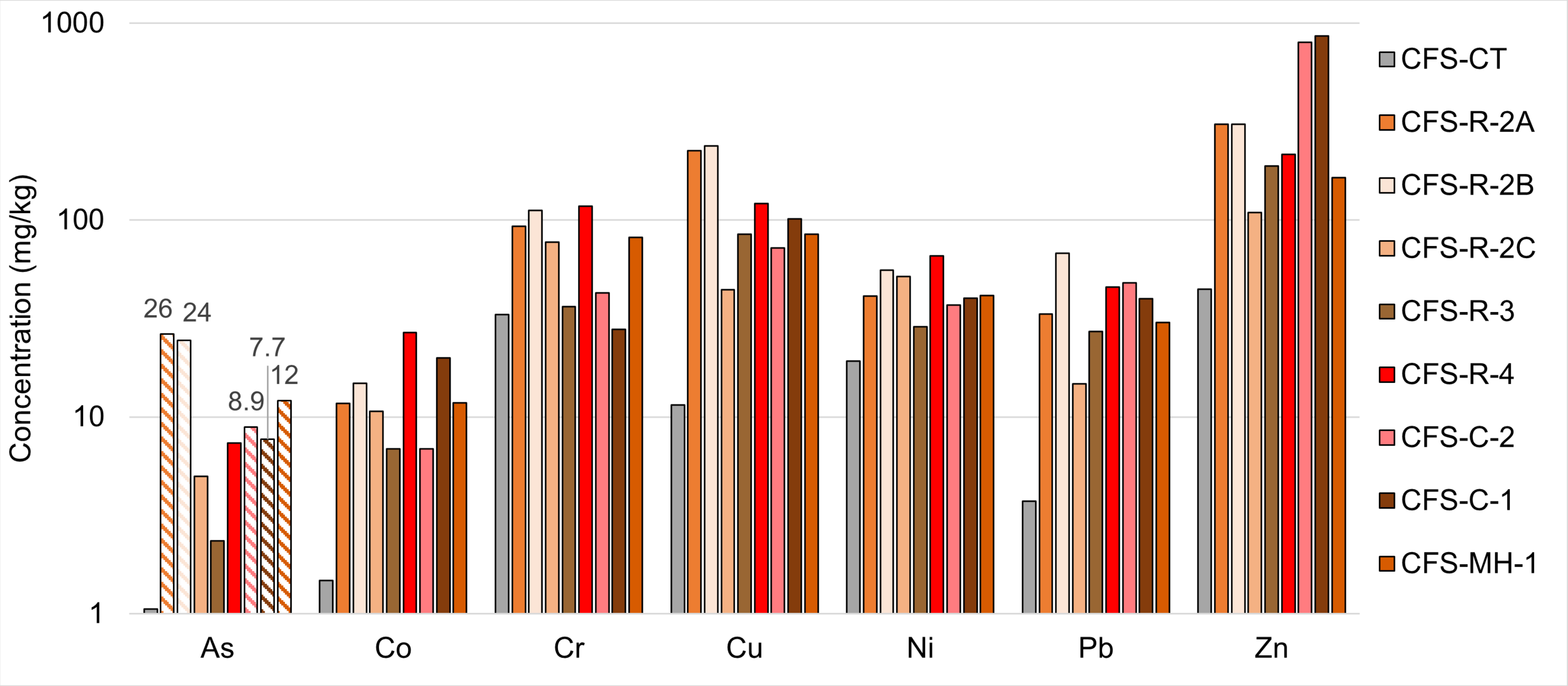

Figure 6. Concentrations of trace metals (As, Co, Cr, Cu, Ni, Pb, Zn) in CFS.

CFS samples (n = 8) with trace metal control concentrations are displayed in gray bars on the left of each dataset. Concentrations are displayed on a logarithmic scale. The striped bars denote samples that exceeded the environmental cleanup goal, with the concentration annotated above the striped bar. 5 samples of CFS exceeded the As environmental cleanup goal of 7.7 mg/kg (Tetra Tech, 2019).

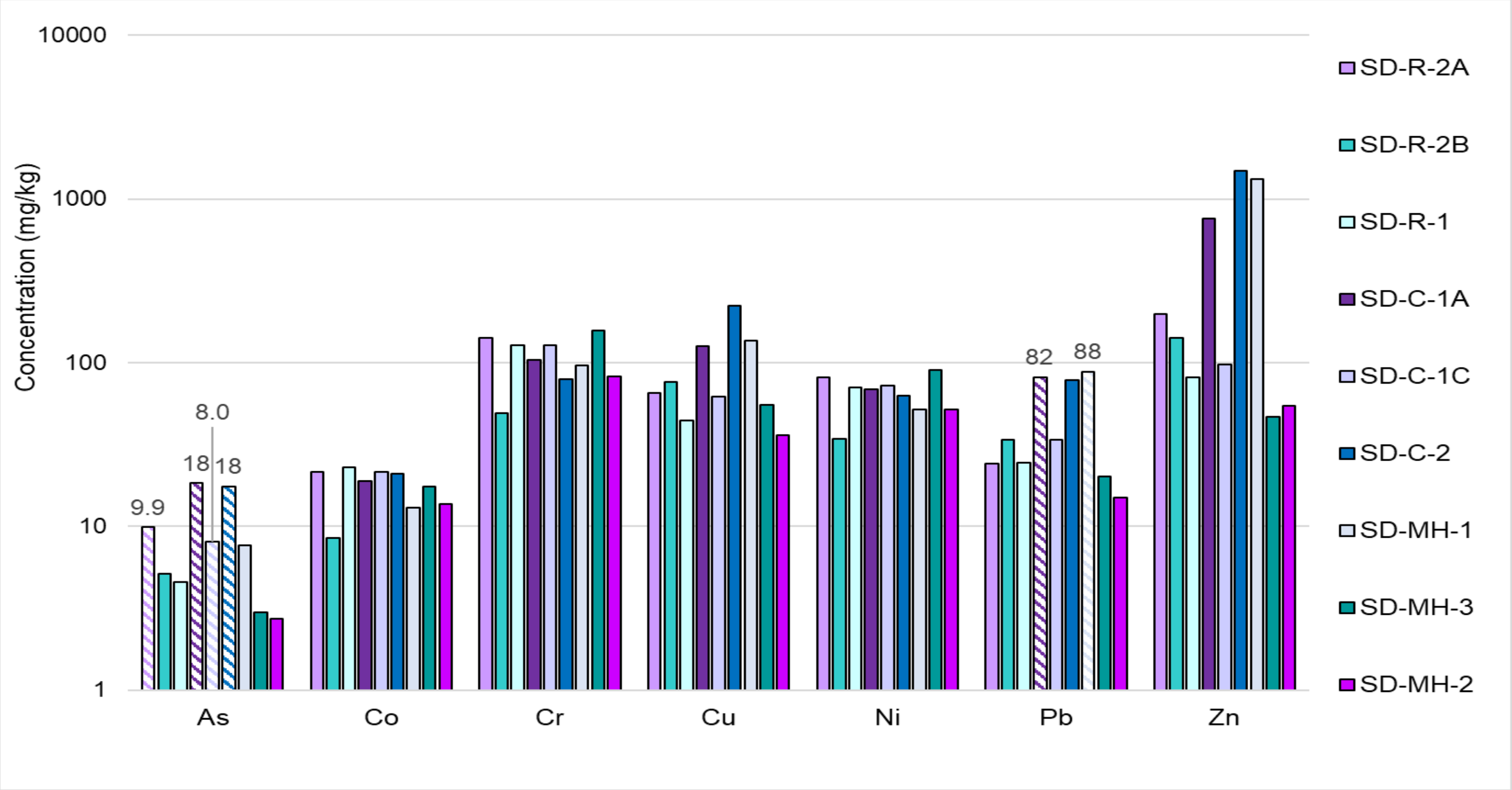

Figure 7. Concentrations of trace metals (As, Co, Cr, Cu, Ni, Pb, Zn) in SD.

SD (n = 9) trace metal concentrations retained are displayed on a logarithmic scale. The striped bars denote samples that exceeded the environmental cleanup goal. Four samples of SD exceeded the As environmental cleanup goal of 7.7 mg/kg and two samples of SD exceeded the Pb environmental cleanup goal of 80 mg/kg (Tetra Tech, 2019).

Metal retention in SW and CFS samples

To quantify the extent of metal retention in SW and CFS samples, the metal concentration measured in control duplicates of the same BMP material (non-deployed control from Paradise) was subtracted from the metal concentration measured in samples to obtain a control-corrected concentration representing metal retention (**Table 7**, **Figures 8-9**). SW-R-1 had a Pb concentration lower than the mean SW control concentration. Also, CFS-C-1 had a Cr concentration lower than the mean CFS control concentration (**Table 7**). Their control-corrected concentrations were therefore assumed to be zero.

The median control-corrected metal concentrations in SW and CFS samples varied by three orders of magnitude, ranging from 0.80 mg/kg for Pb in SW to 220 mg/kg for Zn in CFS (**Table 9, Figures 8-9**). Retained metal concentrations also displayed large variability between sample locations, as evidenced by standard deviation values (**Table 9**).

Control-corrected concentrations of trace metals in SW and CFS samples, which quantify the retention of metals in each BMP material, were compared using the Kruskal-Wallis statistical test (**Table 9**). Metal retention was significantly higher in CFS compared to SW for As, Co, Cu, Ni, Pb, and Zn, even with the large variability observed in each group of samples. In contrast, there was no statistical difference ($p > 0.05$) in metal retention between SW and CFS samples for Cr.

In **Table 9**, Zn had the highest retained control-corrected metal concentration in CFS samples by an order of magnitude (median: 220 mg/kg compared to Cu median: 81 mg/kg), while SW samples retained similar concentrations of Zn and Cu (medians: 6.1 and 5.6 mg/kg, respectively). Cr, Pb, and Ni had the next highest retained concentrations in CFS samples, followed by Co and As. In contrast, Cr, Ni, As, and Co were retained to similar extents in SW, while Pb was the least retained by SW samples (median: 0.80 mg/kg).

Table 9: Medians and standard deviations of control-corrected (retained) metal concentrations in SW and CFS samples, and p values for the Kruskal-Wallis test comparing control-corrected concentrations of metals in CFS and SW. Bolded p values ($p < 0.05$) represent statistical differences between the two materials.

Trace Metal	SW Median (mg/kg)	SW Standard Deviation (mg/kg)	CFS Median (mg/kg)	CFS Standard Deviation (mg/kg)	p-value Kruskal Wallis
As	1.7	0.5	7.3	8.9	**0.013**
Co	1.6	1.1	10	6.8	**0.003**
Cr	2.8	12	49	31	0.107
Cu	5.6	3.6	81	71	**0.003**
Ni	2.8	7.5	22	12	**0.008**
Pb	0.80	1.3	33	16	**0.003**
Zn	6.1	2.1	220	296	**0.003**

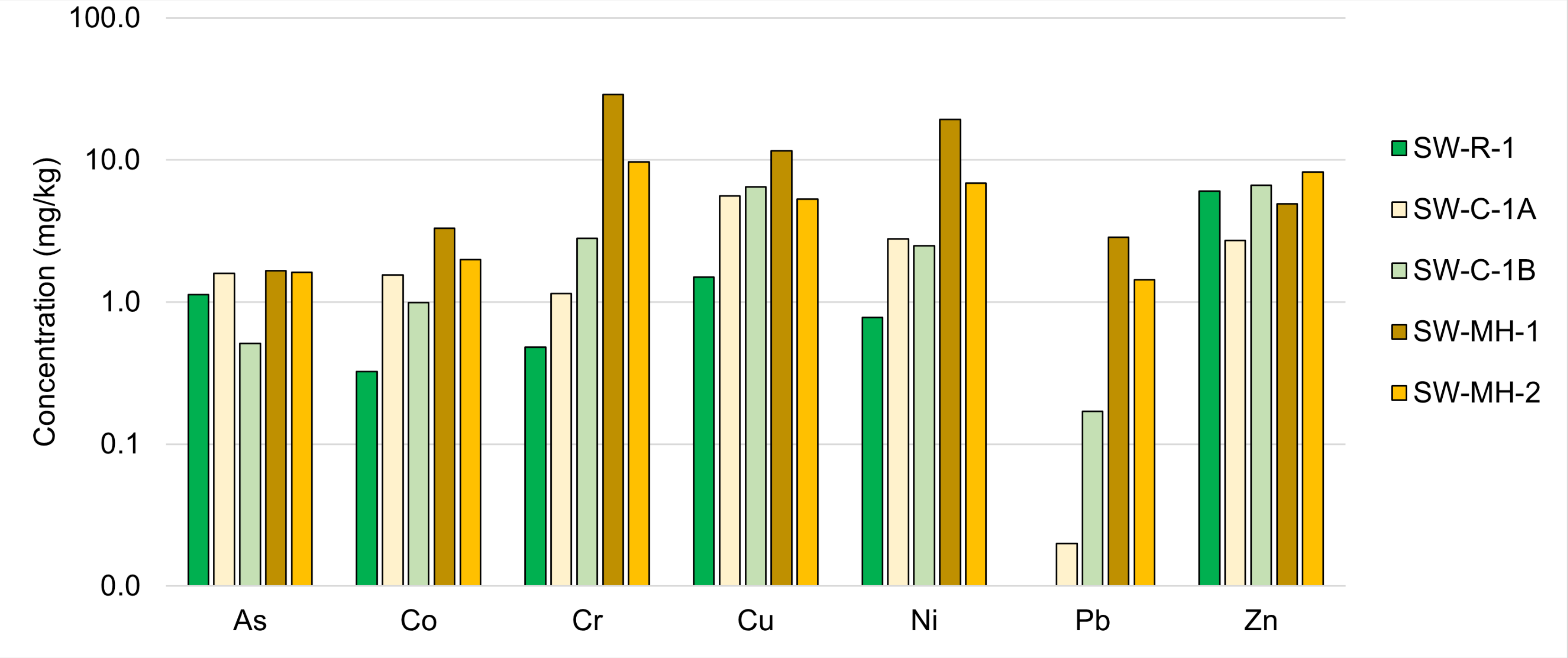

Figure 8: Control-corrected concentrations of trace metals (As, Co, Cr, Cu, Ni, Pb, Zn) in SW (n = 5).

For each trace metal, the mean concentration measured in SW controls was subtracted from the concentration measured in SW samples to obtain control-corrected concentrations. Concentrations are displayed on a logarithmic scale. There are no SW samples that exceeded the environmental cleanup goals (Tetra Tech, 2019).

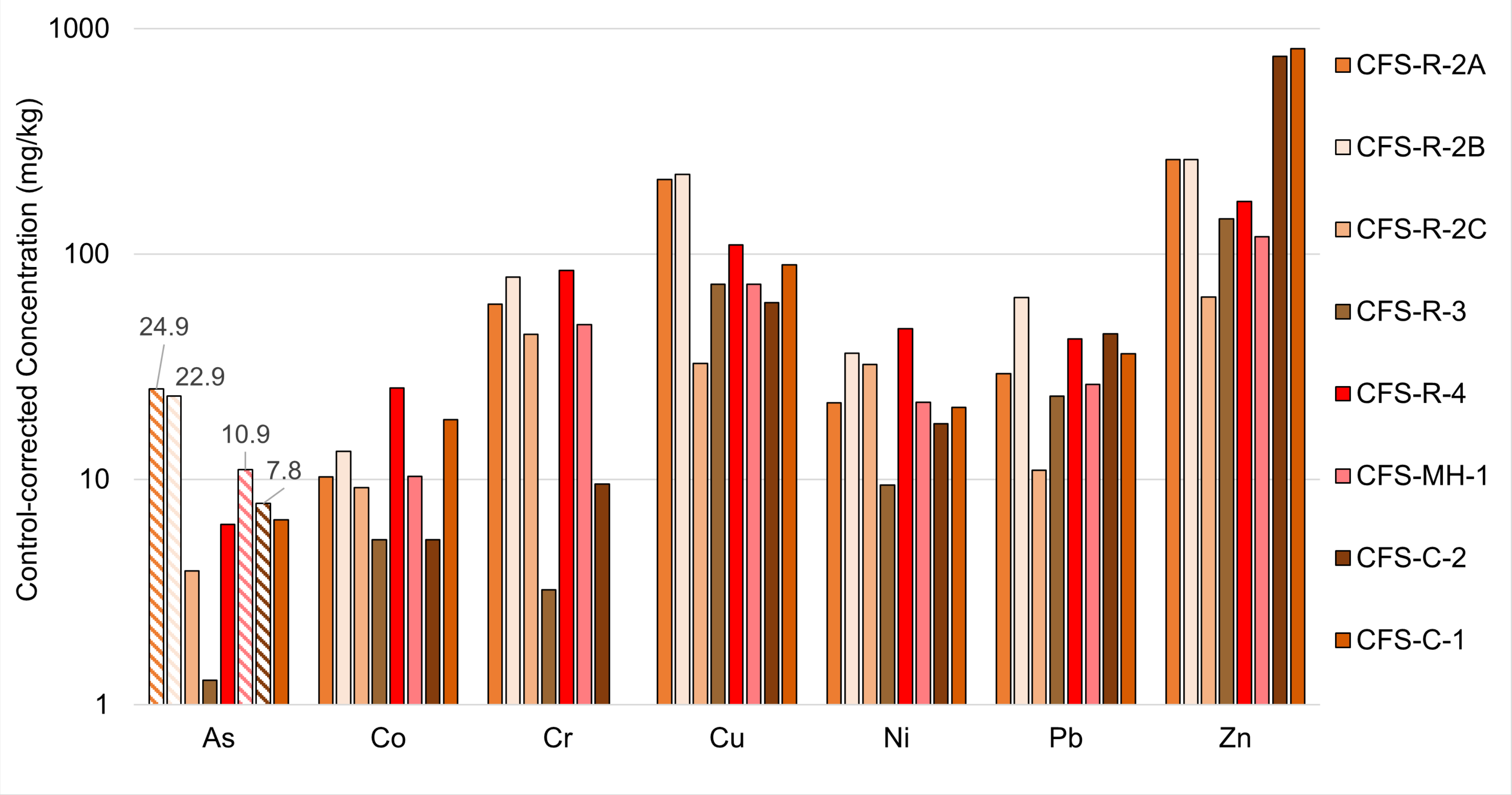

Figure 9: Control-corrected concentrations of trace metals (As, Co, Cr, Cu, Ni, Pb, Zn) in CFS (n = 8).

For each metal, the mean concentration measured in CFS controls was subtracted from the concentration measured in CFS samples to obtain control-corrected concentrations. Concentrations are displayed on a logarithmic scale. The striped bars denote samples that exceeded the environmental cleanup goal. Four CFS samples exceeded the As environmental cleanup goal of 7.7 mg/kg (Tetra Tech, 2019).

PCDD/F results

A total of 15 samples were analyzed for 17 PCDD/F congeners (**Table 2**). These included five SW (including the non-deployed control from Paradise), five CFS (including one control), and five SD samples. The highest concentration of octachlorodibenzodioxin (OCDD) was 15,000 ppt in sediment sample SD-MH-3 while all other samples contained less than 2,000 ppt OCDD (**Table 10**, **Figure 10**). The concentrations of most PCDD/F congeners were below 220 ppt. Toxic equivalency (TEQ) values were above 1 ppt in five samples. Of these samples, one sample SD-MH-3, had an order of magnitude of higher TEQ of 51 ppt that exceeds the soil remedial goal for contaminated soils in California (50 ppt; CA DTSC, 2021; **Figure 11**).

Out of the 15 total samples, PCDD/F were detected in 10 samples. SW samples had no detectable PCDD/F while all CFS and SD samples contained some detectable PCDD/F (**Figure 12**). A total of seven PCDD/F congeners were found, though only one sample (SD-MH-3) had all seven congeners. The OCDD congener was detected most frequently and at the highest concentration. OCDD was found in all 10 samples with detectable PCDD/F. 1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin (1,2,3,4,6,7,8-HpCDD) was detected in six of the 10 samples with PCDD/F, and 1,2,3,4,6,7,8-Heptachlorodibenzofuran (1,2,3,4,6,7,8-HpCDF) and Octachlorodibenzofuran (OCDF) were each detected in four of the samples with PCDD/F.

The detection of PCDD/F was closely linked to the type of material analyzed. No PCDD/F were found in the SW samples, four congeners were found in CFS samples, though only one sample (CFS-C1) had more than two. Sediments contained the largest number of congeners and the highest concentrations. Four of the five sediments analyzed had three or more congeners.

The co-located CFS and SD samples (CFS-C-1, SD-C-1C) had very similar profiles; behind the commercial site, PCDD/F congener distribution and concentrations between the two materials were nearly identical. The co-located parking lot samples (CFS-C-2, SD-C-2) were not as closely linked in concentration but contained a similar profile of PCDD/F congeners.

Table 10: PCDD/F detection by sample

Sample	1,2,3,6,7,8-HxCDD (ppt)	1,2,3,7,8,9-HxCDD (ppt)	1,2,3,4,6,7,8-HpCDD (ppt)	OCDD (ppt)	1,2,3,4,6,7,8-HpCDF (ppt)	OCDF (ppt)	2,3,7,8-TCDF (ppt)	Total	TEQ
SW-C-1A								0	
SW-C-1B								0	
SW-MH-1								0	
SW-R-1								0	
CFS-C-1			240	1900	54	110		2300	3.5
CFS-C-2			75	490				570	0.90
CFS-R-2C				410				410	0.12
CFS-R-3				390				390	0.12
SD-C-1C			230	1300	79	150	9.3	1800	4.5
SD-C-2			110	50	50			210	1.6
SD-MH-1			300	1500		150		2000	3.5
SD-MH-3	140	93	2000	15000	260	360	13	18000	51
SD-R-1				200				200	0.10
SW-CT								0	
CFS-CT				360				360	0.11

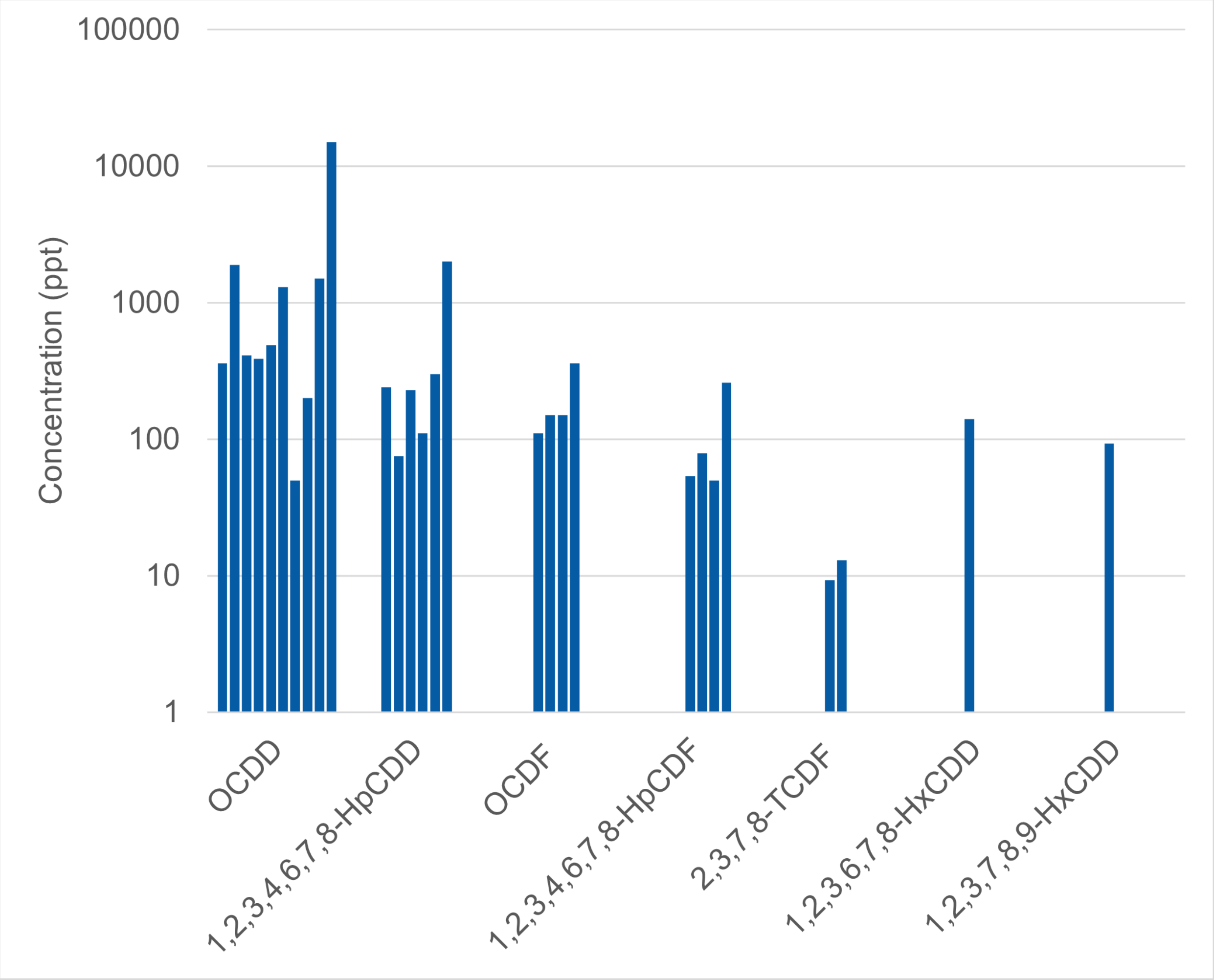

Figure 10: Distribution of congener concentrations across all samples

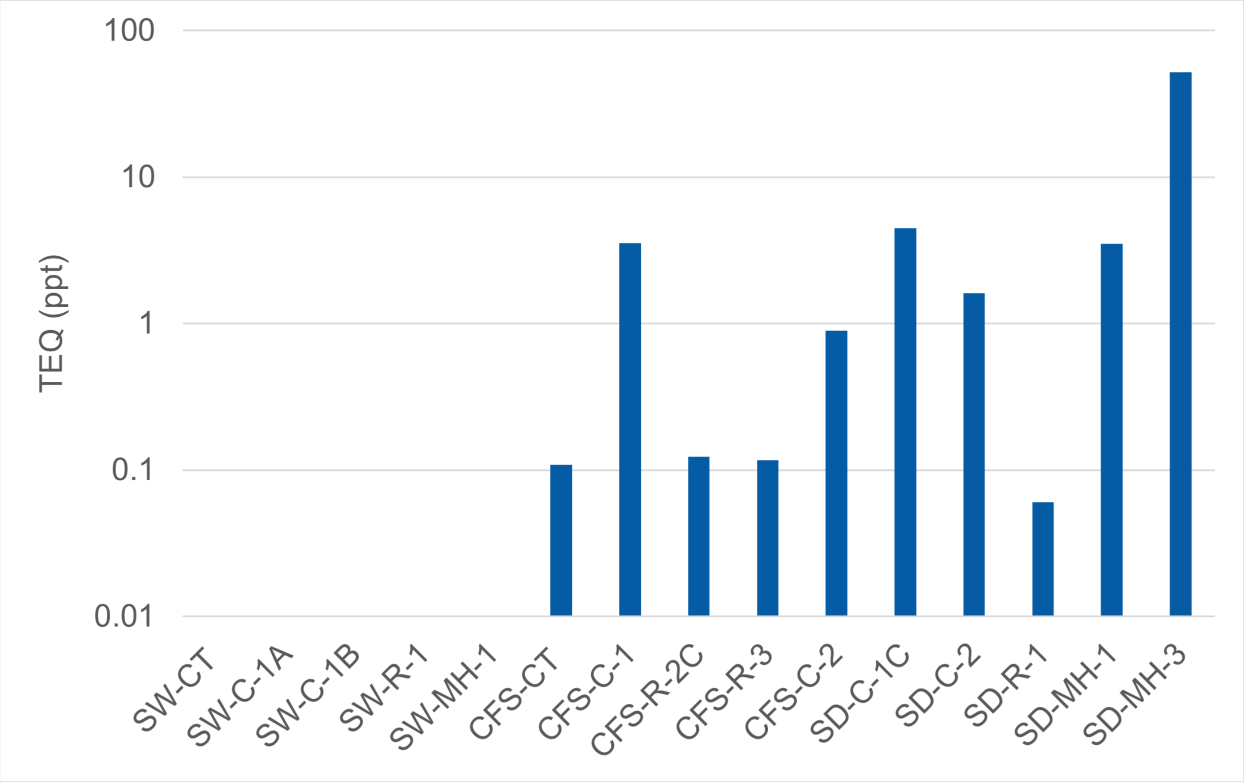

Figure 11: Toxic equivalency values (TEQ) for SW, CFS, and SD samples.

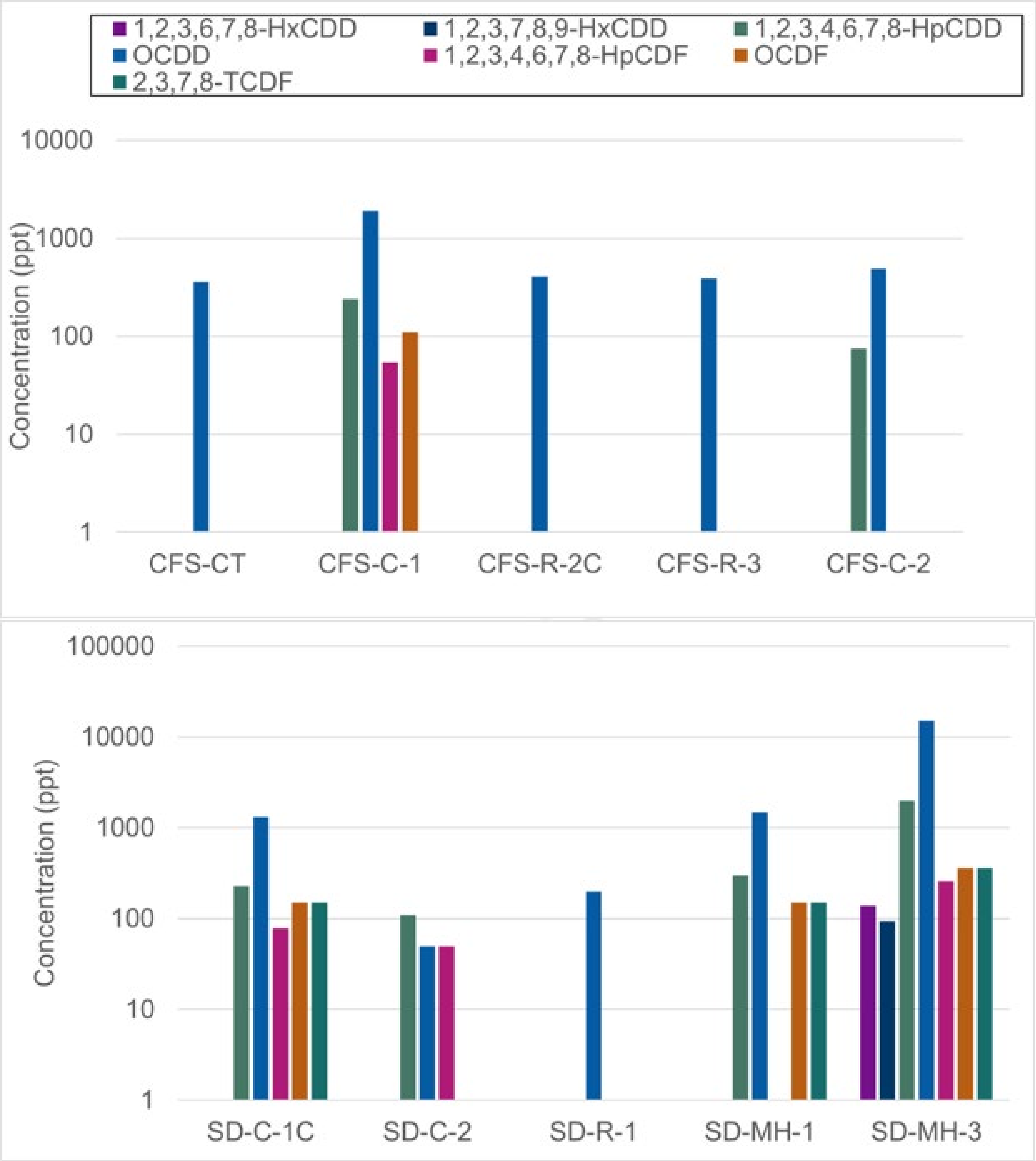

Figure 12: Concentrations of PCDD/F congeners in CFS samples (Top) and SD (Bottom). No PCDD/F congeners were found in SW samples.

PFAS results

A total of 15 field samples were analyzed for a suite of 39 PFAS congeners, including five SW samples (including the non-deployed control), five CFS samples (including the non-deployed control), and five SD samples (**Table 4**). 22 out of the 39 PFAS congeners were detected (**Table 11, Figures 13-14**).

Eight of the 15 samples had detectable PFAS. N-ethyl perfluorooctane sulfonamidoethanol (NEtFOSE) was the most frequently detected congener and was found in seven samples (**Figure 13**). The congeners PFOS, PFDoA, and PFDA were detected in four samples, and PFDS, PFBS, PFOA, and NEtFOSAA were each detected in three instances across all samples (**Figure 13**, **Table 11**).

The frequency and level of PFAS detection was related to the material analyzed. SD had the highest number of PFAS detections as well as the highest concentrations of PFAS (**Figure 15**). CFS samples had the same percentage of samples with detections, but the number of congeners and the total concentrations were generally not as high as SD. SW samples had two detections and only NetFOSE was detected. SD-C-1C, collected from behind a commercial retail grocery store, contained 21 of the 22 detected congeners, and also had the highest overall measurement of PFAS congeners at 51 ppb (**Figure 15**). The next highest concentrations of total PFAS were about one third of this concentration, detected in one SD and two CFS samples.

The detections of PFAS seemed to be location dependent as sediment and product accumulation were linked. Sediment at the commercial area contained high PFAS detects (21 of 22), and the compost material deployed (CFS-C-1) also contained high PFAS detects (9 of 22). Two SW samples had one congener (NEtFOSE) detection, the only detection in the field-deployed SWs. SD-C-2 from the parking lot of a commercial area had nine congener detections, and the co-located CFS sample (CFS-C-2) had three congener detections.

Total EOF was measured by two labs independently. Eurofins conducted EOF by CIC and reported a wide range of measurements. The highest total EOF was 65 ppb, and the lowest was 950 ppt. The EOF measurements had little correlation ($R^2 = 0.2$) with PFAS measurements. In the second lab analysis of EOF by UNR using the CIC, the concentrations ranged 22 to 250 ppt and had no correlation ($R^2 = 0.002$) with PFAS measurements.

Table 11: PFAS detections by samples.

Sample	PFBA (ppb)	PFPeA (ppb)	PFHxA (ppb)	PFHpA (ppb)	PFBS (ppb)	PFDS (ppb)	PFOA (ppb)	PFNA (ppb)	PFDA (ppb)	PFUnA (ppb)	PFDoA (ppb)	PFTrDA (ppb)	PFTeA (ppb)	PFOS (ppb)	FOSA (ppb)	NMeFOSAA (ppb)	NEtFOSAA (ppb)	NMeFOSA (ppb)	NMeFOSE (ppb)	NEtFOSE (ppb)	7:3 FTCA (ppb)	PFPeS (ppb)
SW-C-1A																						
SW-C-1B																				2.6		
SW-MH-1																				1.2		
SW-R-1																						
CFS-C-1							1.5		16		1.5		1.0	3.7		1.5	1.2		3.5	1.5		
CFS-C-2					10	1.1														2.5		
CFS-R-2C																						
CFS-R-3																				2.9		
SD-C-1C	0.78	0.64	1.6	0.57	0.27	1.6	3.1	1.2	4.3	1.5	3.6	0.59	2.5	7.7	2.3	3.9	3.8	0.33	5.7	3.9	0.58	
SD-C-2			0.54		7.6	2.1	0.32		0.42		0.36			1.1			1.4					0.21
SD-MH-1																						
SD-MH-3									0.37		0.26			2.1						0.22		
SD-R-1																						
SW-CT																						
CFS-CT																						

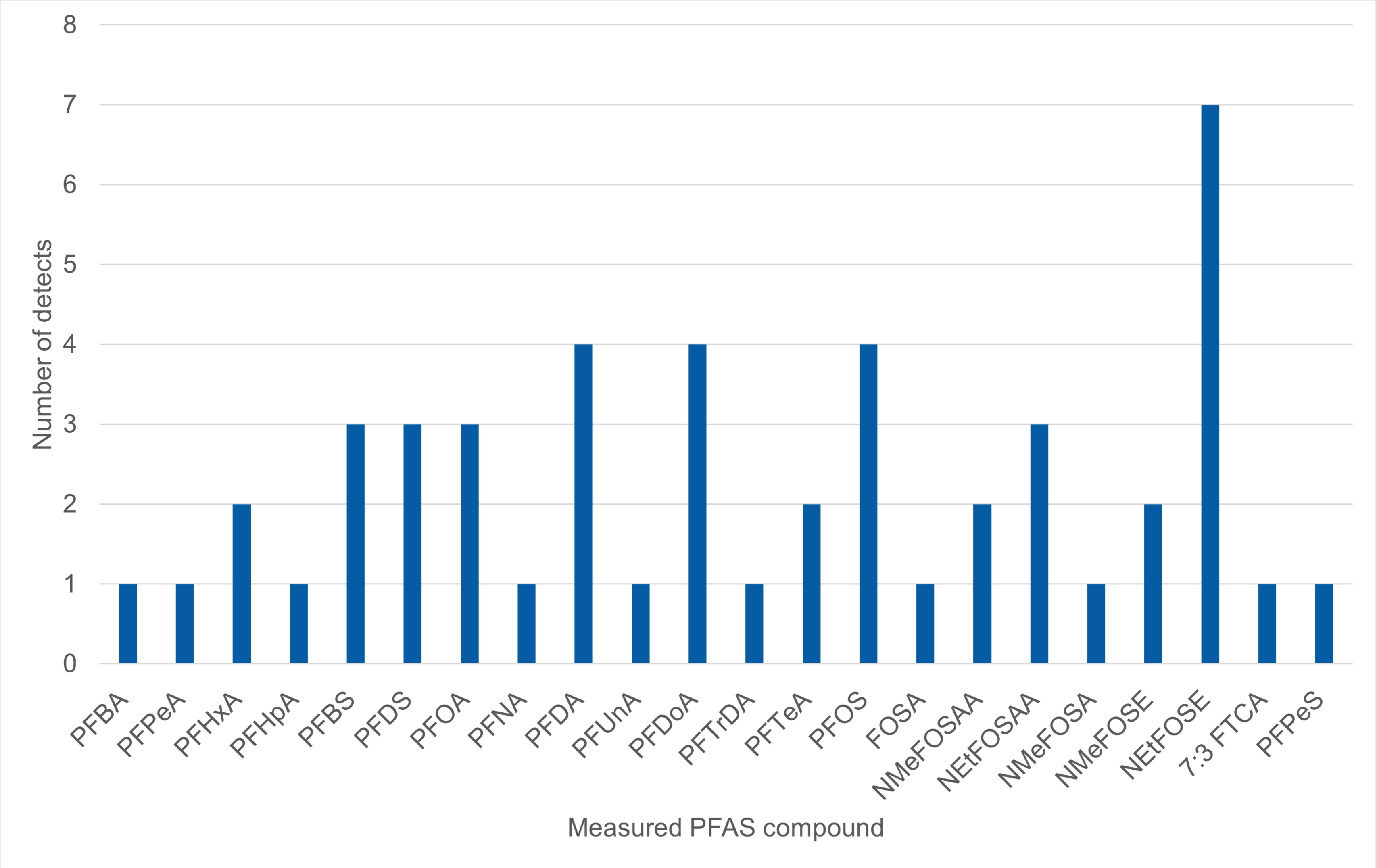

Figure 13: PFAS congener detections

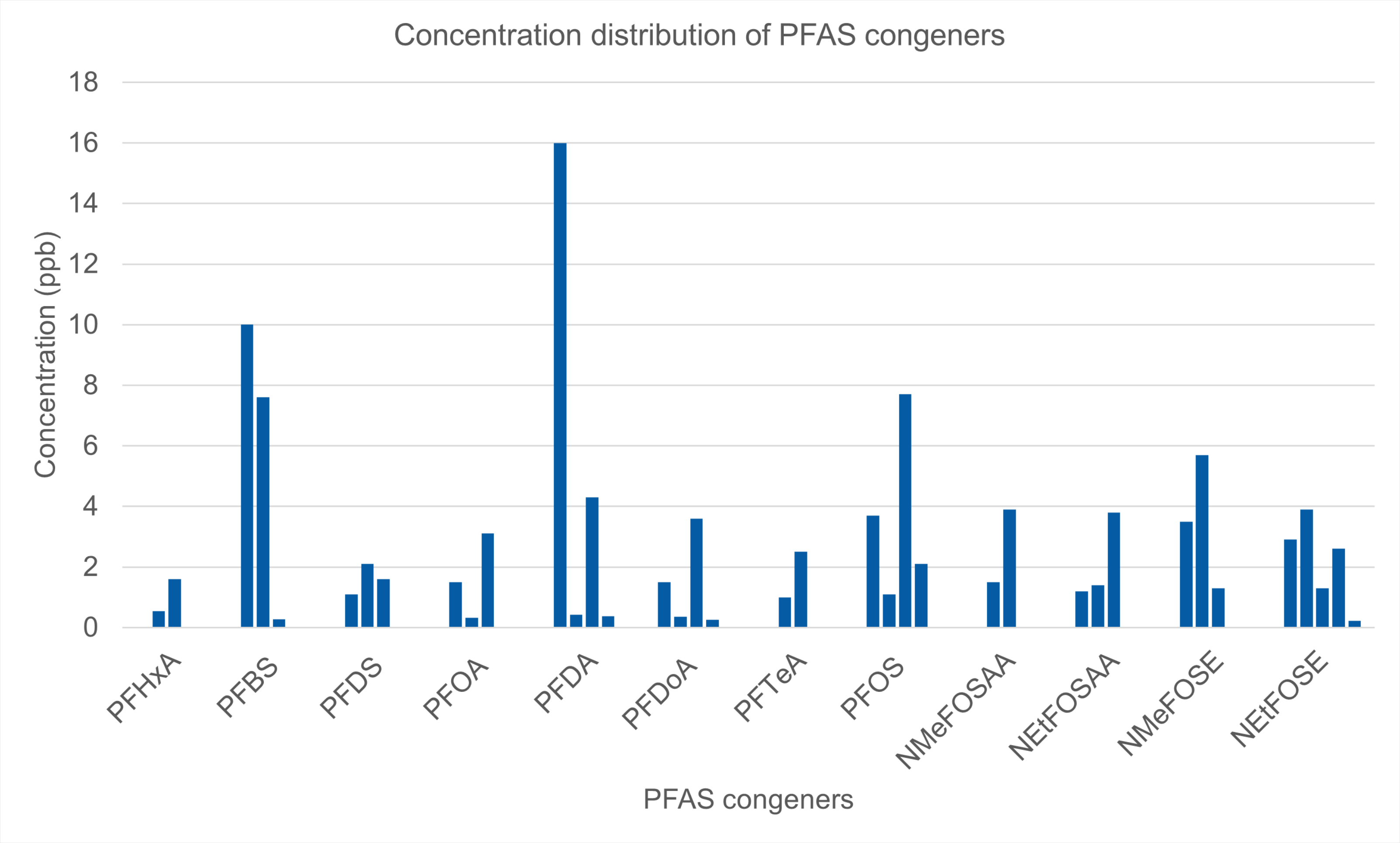

Figure 14: Concentration distribution of PFAS ordered by chain length of shortest to longest (left to right). Only the PFAS congeners that had more than one detect are displayed. Each bar represents an individual sample detection.

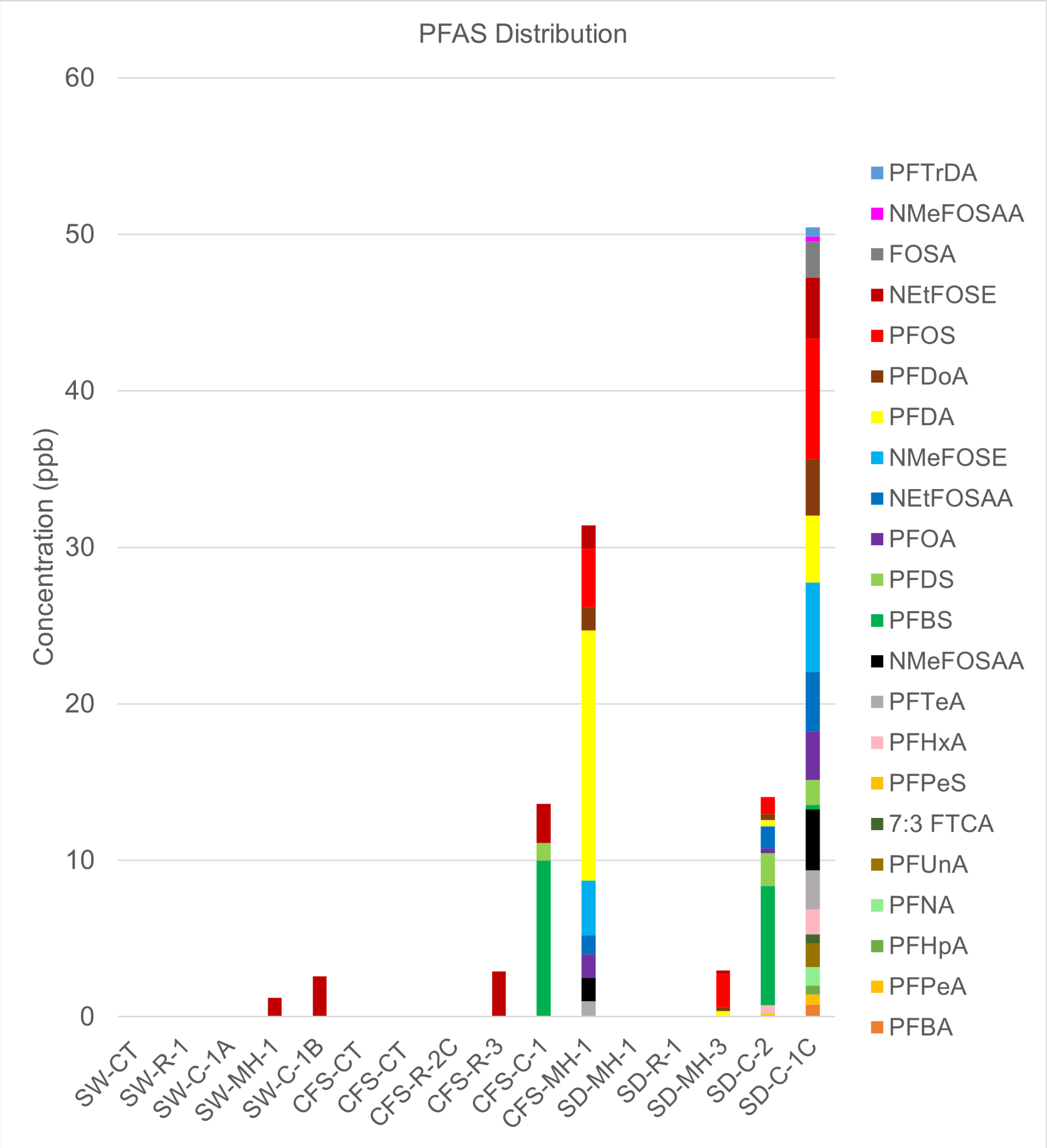

Figure 15: Distribution of PFAS congeners by material, site, and abundance.

Results of laboratory PFOA sorption experiments

Differences in sorption capacity were observed between SW and CFS samples across spiked DOC ranges (3.4 ppb, 6.9 ppb, 34 ppb, 69 ppb TF_{aq}). At spike concentrations of 3.4 ppb and 6.9 ppb TF_{aq}, the main trend of TF_s measurements was observed across the sorption tests conducted (**Table 12**, **Figure 16**). Specifically, CFS samples had higher TF_s concentrations than SW samples, and this was consistent even with varying levels of spiked DOC and ash.

In addition, the TF_s concentrations of SW and CFS samples increased with the amount of spiked DOC. The correlation coefficient (R^2) between DOC concentration and sorbed concentration was 0.50 for SW and 0.74 for CFS samples. This trend was observed at other spiked PFOA concentrations and suggested that the presence of DOC facilitated fluorine sorption to both materials.

Average sorbed concentrations were 0.0 ppm TF_s for SW and 3 ppm TF_s for CFS samples in the presence of low spiked DOC. Sorbed concentrations for SW were undetectable across all TF_{aq} concentrations. This indicated that PFOA sorption to SW samples spiked with low DOC did not occur regardless of spiked TF_{aq} concentration. SW samples spiked with low DOC had a negligible fluorine sorption capacity.

Table 12: Summary of results from PFOA sorption experiments.

Sample type	Spike concentration (ppm)	Average TF_s	Standard deviation	Percent recovery (%)
SW, Low DOC	3.4	0.00	1.32	0%
SW-d, Low DOC	3.4	0.00	1.31	0%
SW, High DOC	3.4	0.78	1.47	23%
CFS, Low DOC	3.4	2.99	0.45	88%
CFS-d, Low DOC	3.4	2.16	0.73	64%
CFS, High DOC	3.4	2.88	0.68	85%
SW, Ash	6.9	0.00	1.65	0%
SW, Low DOC	6.9	0.00	1.33	0%
SW, Medium DOC	6.9	1.05	1.40	15%
SW, High DOC	6.9	0.00	1.52	0%
SW-d, High DOC	6.9	2.89	2.83	42%
CFS, Ash	6.9	4.01	0.61	58%
CFS, Low DOC	6.9	3.54	0.46	51%
CFS, Medium DOC	6.9	3.90	0.96	57%
CFS, High DOC	6.9	4.45	0.98	65%
CFS-d, High DOC	6.9	5.93	0.76	86%

d = duplicate

Note: Percent recovery (%) refers to the amount recovered in solid phase.

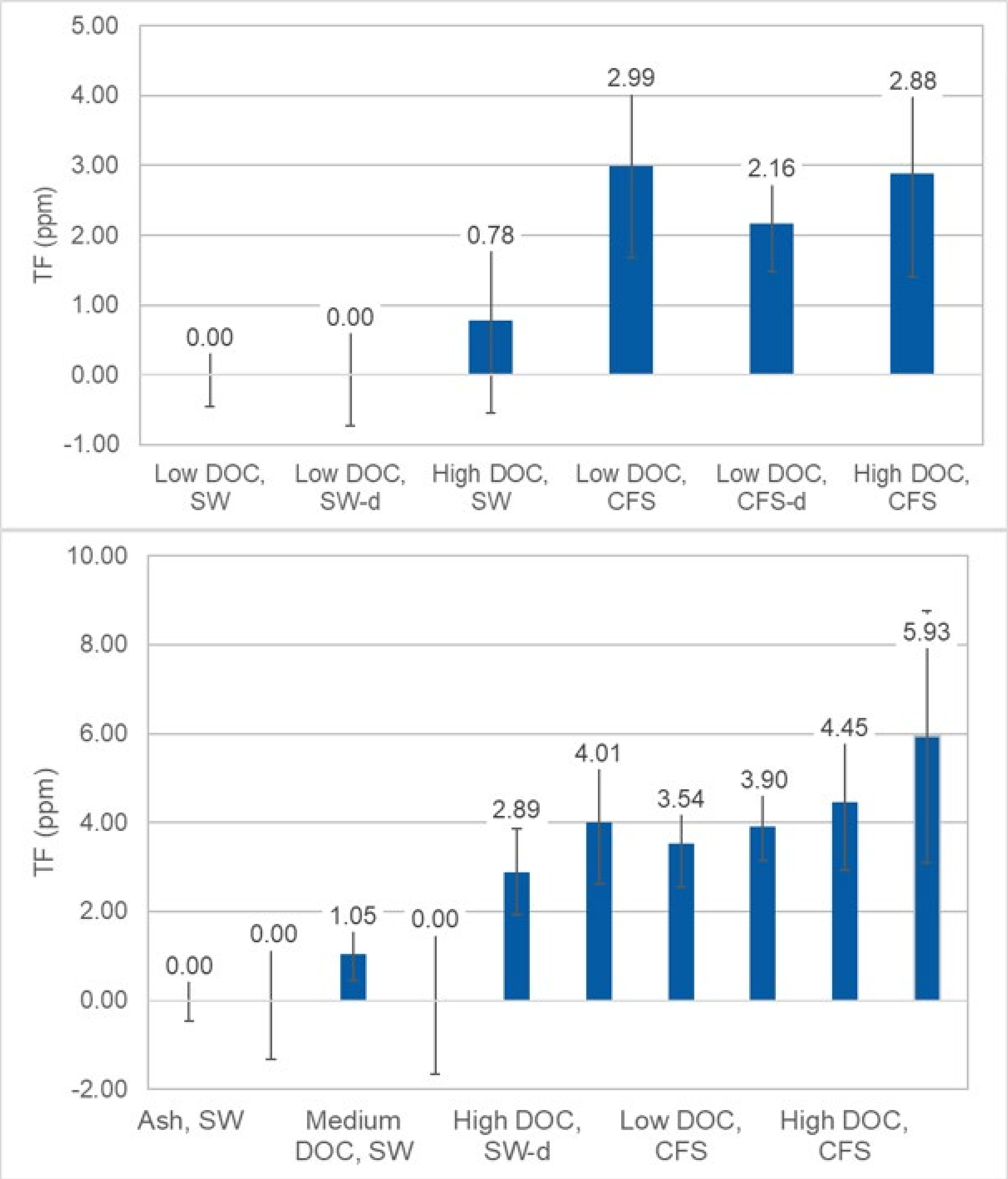

Figure 16: Average TF$_s$ concentrations in solid samples spiked with 3.4 ppm (top) and 6.9 ppm (bottom) TF as PFOA. d: duplicate.

Discussion

Metals, PFAS, and PCDD/F were retained by sediment retention products deployed after the 2018 Camp Fire, indicating that the chemicals were present in stormwater runoff that contacted the products during storm events following the fire. SD samples were found to have elevated metals, sometimes exceeding environmental cleanup goals. Detection of PFAS and PCDD/F varied, but these contaminants were consistently detected. Further, selected samples had PFAS and PCDD/F concentrations that were dramatically higher than other samples.

Without a better understanding of the environmental background levels of PFAS and PCDD/F in stormwater and sediment, as well as the flow volumes materials received, it is difficult to fully assess the efficiency of contaminant retention in this study. In general, the lack of comparable studies on post-fire occurrence of PFAS and PCDD/F make interpretation of data difficult. It is unknown why specific congeners were detected in specific locations and not others. Localized fuels are most likely the cause of congener detection differences.

This study was limited in the number of analyses conducted with only five samples per type (SW, CFS, SD), so even one high detection per type suggests a 20% "hit rate" for elevated chemical presence. These numbers would be improved with a greater number of samples and increased location specificity. It is likely that some results are highly influenced by localized fuels that release a specific contaminant profile and more extensive post-fire sampling would greatly improve the interpretation of these results.

While estimations of chemical sources and general trends contributing to the specific results are not possible with such limited data, the overall variability does specifically point to locations that could be "hot spots." This result is explained by the fact that the post-fire landscape is highly heterogenous. Chemical profiles are likely influenced by local chemical sources, such as types of structures, structure contents, automobiles, and other specific fuels.

Metals in environmental samples

There were several differences in the trace metal concentrations of SW, CFS, and SD samples. Five CFS and four SD samples exceeded the As environmental cleanup goal, two SD samples also exceeded the Pb environmental cleanup goal, while no SW samples exceeded environmental cleanup goals (**Tables 7-8**). These concentration exceedances emphasize the importance of proper management and disposal of BMP materials post-deployment.

SD sample concentrations were similar to CFS concentrations for As, Co, Cu, Pb, and Zn, or higher than CFS concentrations for Cr and Ni (**Table 8, Figures 5 and 7**). The sediment particles that sediment retention products intercepted were likely a main source of trace metals to the watershed following the Camp Fire. Ash collected at burned home sites after the Camp Fire contained trace metals in exceedance of environmental cleanup goals. Out of the 150 ash samples, there was one exceedance for Ni (mean concentration 58.9 mg/kg) to 53 exceedances for Pb (mean concentration 706 mg/kg) (TetraTech, 2019). This metal contamination was mobilized to nearby creeks, as water quality monitoring in Paradise showed that trace metal concentrations in the creeks of watersheds affected by the Camp Fire were higher than concentrations in an unburned control watershed (Magliozzi et al., 2024). Zn was found at the highest concentrations, followed by Cu, in creeks receiving runoff from the burned area. However, Cu was found in the highest concentrations in ash, followed by Zn (TetraTech, 2019). These differences reveal that metal content of ash sources on the landscape will influence metal profiles in the watershed and highlight the role of retention processes in sediment retention products.

CFS samples had higher control-corrected metal concentrations than SW samples (**Figures 8-9**), suggesting increased retention. Metal concentrations in deployed CFS samples were significantly different from concentrations in nondeployed CFS control from Paradise, except for Cr. This contrasts with the lack of significant differences in metal contents between deployed SW samples and nondeployed SW controls from Paradise, consistent with the low metal retention observed in SW. In addition to the CFS greater ability to trap particles and provide physical straining for particulate metals, the more heterogeneous organic matter composition of CFS compared to SW could provide stronger binding sites for metal ions, thereby enhancing the sorptive capacity of CFS (Huang et al., 2018). The differential retention observed between SW and CFS for specific metals could also be a result of variations in metal speciation, the pH and ionic strength of runoff, and the presence of other competing ions (Bodi et al., 2014). For instance, metals such as Cu and Pb have a strong affinity for organic matter due to their ability to form stable chelates, which might explain their higher concentrations in CFS (Fan et al., 2016).

The absence of significant differences in retained Cr concentrations between and among SW and CFS samples and their respective controls from Paradise suggests that the retention capacity of CFS for this particular metal may not be substantially different from that of SW. At high temperatures (>200°C), the most abundant form of Cr in soil, trivalent chromium (Cr III), can become oxidized into a much more carcinogenic form, hexavalent chromium (Cr VI), and released in ash plumes (Lopez et al., 2023). Even after months following a wildfire, ash-bound Cr VI can exhibit low leaching potential, which would result in Cr staying in sediment behind both BMPs, or within soils not carried by stormwater runoff.

Overall, CFS deployed after the Camp Fire demonstrated a greater ability to retain trace metals than SW. This clear trend is likely attributable to a combination of product design, with CFS designed to let storm runoff flow through, and of more heterogenous organic matter composition and particle sizes in CFS compared to SW.

PCDD/F in environmental samples

The results of the environmental sample analysis revealed the presence of PCDD/F in all the CFS and SD samples (**Table 10**). No PCDD/F was detected in SW samples. The most frequent congener detected in both CFS and SD samples was OCDD, which is also the least toxic of the PCDD/F suite (**Figure 10, Table 5**). Thus, although the concentrations of OCDD were relatively high in some samples, such as SD-MH-3 with an OCDD concentration of 15,000 ppt (**Figure 12**), the TEQ of this highest sample was 51 ppt, just above the soil remedial goal for contaminated soils in California of 50 ppt TEQ (CA DTSC, 2021).

The concentrations of most PCDD/F congeners were below 220 ppt (**Table 10**). In studies on combustion residues, reported concentration range depended on source fuels. For example, PVC and PVC clad wood has reportedly produced ash with 46 - 750 ppt TEQ (Carroll, 1996). In simulated house fires, polyhalogenated dibenzofurans (PCDF) have been reported to be as high as 5,300 ppt, with ash from electronic devices specifically containing 180 ppb PXDF (Organtini, et al., 2015). Ash from the Camp Fire was transported in the environment and mixed with sediments. Therefore, the total PCDD/F concentration in bulk sediment would likely be diluted (lower) compared to PCDD/F concentrations in a deposited combustion residue.

The total PCDD/F (TDF; Σ PCDD/F measured) sediment concentrations were in the range of 200 - 18,000 ppt (**Table 10**). For comparison, a study reported on base and storm flow TDF concentration in a San Franscico Bay tributary (Gilbreath and McKee, 2015). They found concentrations at low flow had an average of 0.23 ppt TDF and a high flow average of 3.7 ppt TDF while the low flow and high flow suspended sediment concentrations (SSC) averaged 68 ppm SSC and 290 ppm SSC, respectively. While this study is not a perfect comparison, a calculation of the sediment-normalized concentration for base flow in the SF tributary yields a range from 3,300 – 13,000 ppt TDF. The results of the present study are close to this range.

In the present study, CFS samples contained on the order of 360 – 2,300 ppt TDF. Interestingly, the CFS seemed to have a baseline of OCDD with 360 ppt detected in one of the non-deployed control materials from Paradise (only OCDD detected). It appears that SW did not accumulate additional PCDD/F.

At the commercial retail location C-1, where multiple sediments and products were collected, CFS samples were found to have multiple PCDD/F species (**Table 10**). The congener distribution followed the order OCDD > 1,2,3,4,6,7,8-HpCDD> OCDF = 1,2,3,4,6,7,8-HpCDF (**Figure 10**). This relationship is similar to other stormwater runoff results (OCDD >123478-HpCDD > OCDF> 2378-TCDF > 2378-TCDD; Suarez et al., 2006). OCDD was detected above 350 ppt in almost every sample and as high at 15,000 ppt in one sample (MH-3). OCDD has been previously observed to be the predominant congener in ash and fire residues as well as in soils in residential and urban areas (Ruokojarvi et al., 2000; Schumacher et al., 2005; Suarez et al., 2006).

PFAS in environmental samples

The number of PFAS congeners detected and the frequency of detection at various sites suggests that there was a broad presence of PFAS congeners throughout the stormwater systems following the 2018 Camp Fire (**Figures 13-14**). While this study did not address aqueous concentrations of PFAS, it can be assumed that storm runoff contained many PFAS congeners that were likely to be in both the dissolved phase and associated with suspended sediment.

The results identified SD samples as containing the greatest PFAS mass (**Figure 15**). Comparable studies with stormwater sediment PFAS concentrations are limited in availability. In sediments at the commercial area in Paradise and at a mobile home, the sediments were found to have total PFAS from around 3.0 ppb (SD-MH-3) up to around 51 ppb near the commercial retail store (SD-C-1C; **Figure 15**). For comparison, in a different study on street sweeping dust across the state of Florida (Ahmadireskety et al., 2021), swept material had a range of PFAS congeners, but median concentrations for most congeners were around 0.1 ppb. In the Ahmadireskety et al.(2021) study, the highest PFAS concentration found in urban areas was 22.7 ppb with another site having 9 and 8 ppb of total PFAS. The total number of detected PFAS congeners were between 6 and 25 for those sites. N-EtFOSAA had the highest total concentration, near an industrial site. The results of this current Camp Fire study indicate that PFAS in post-fire stormwater was much higher than those found in urban and even industrial street sediments.

The similarity in species distribution between SD and CFS samples in areas where the materials were collected from co-located sites suggests that the compost material was effective at retaining the PFAS that were transported in that area (**Table 11, Figure 15**). This similarity could be due to particulate retention in the BMP products or direct interaction due to hydrophobic surface reactions. As many of the congeners have moderately high hydrophobicity, it is likely they were associated with suspended sediments that were subsequently retained in the CFS. Notably, this was not the case for co-located SW samples. The difference in retention performance could be explained by the fact that SW does not have high flow-through rates compared to CFS. Because the stormwater is not passing through the material, the particulates do not enter the SW and become entrained, as they do in CFS.

The most frequently detected congener was NEtFOSE (**Figure 13**). The FOSA and FOSE congeners are products of perfluoroctanesulfonyl fluoride (POSF; 3M, 1999). As polymers, POSF congeners were widely used in the United States as surface treatments (carpets, fabric, apparel and leather) until 2002 when domestic production was dramatically reduced due to concerns from the U.S. EPA (U.S. EPA, 2000). POSF can either be methylated or ethylated to form the congeners N-methyl or N-ethylpeffluorooctanesulfonamide (FOSA), which can subsequently be reacted with ethylene carbonate to form either N-methyl or N-ethylperfluorooctanesulfonamidoethanol (FOSE). The FOSE congeners can be linked to a phosphate ester and used in paper products, particularly food packaging to avoid grease absorption. FOSE congeners degrade to PFOS. It is not entirely clear why this congener is detected with such frequency.

The congeners that were detected could be due to a combination of abundance in fuels prior to burning, and the chemical behavior in the environment once released. For instance, the presence of PFOS is reasonable as PFOS is the terminal breakdown product of the NEtFOSE congener. PFOS and the other common terminal product, perfluorooctanoic acid (PFOA), have moderately high hydrophobicity (log K_{ow}: 5.3 and 6.3, respectively), indicating that they have an elevated tendency to enter stormwater bound to particulate matter (Zhao et al. 2016). Similarly, other congeners such as PFDOA and PFDA are long chain (12 and 10 carbons, respectively), so they are more likely to be associated with sediments because the tendency for sediment sorption (K_d) goes up by 0.6 log units for each methylene (-CH2) unit (Higgins and Luthy, 2006). Based on the abundance of precursors and tendency for solid-phase adsorption, the presence of these congeners is unsurprising in the sediments collected.

The scarcity of studies following urban burning, and particularly focusing on stormwater sediment, makes contextualizing the species distribution and abundance difficult. In the only similar study following a series of wildfires in Northern California, Wang et al. (2022) reported measuring four PFAS in stormwater runoff: perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS), and perfluorooctane sulfonic acid (PFOS). These congeners were measured in the sub 10 ppt concentration range in water, with only PFOS detected above 5 ppt. The authors cautioned that sampling numbers were too low to assess the sources and what site characteristics may have driven these findings. The results of Wang et al. (2022) are comparable to other studies of PFAS in urban stormwater (not following WUI fire). Typical concentrations of PFAS congeners in urban runoff are typically reported higher than the Wang et al. (2022) results; in the sub-100 ppt range with PFOA and PFOS being the highest (Procopio et al, 2017; Houtz and Sedlak, 2012). The results of these comparable studies show that PFAS congeners are likely present in urban stormwater and post-fire stormwater, yet they do not seem to help explain the chemical species distributions. Further, this study has no comparable particle concentrations to assess relative PFAS content in suspended and deposited sediments analyzed.

EOF was found to have no correlation to the measured PFAS concentrations. The result is likely due to the large number of fluorinated congeners and organic fluorine-containing molecules that exist beyond what was measured. This study used the established EPA method 537 with 39 target analytes. There are over 14,000 PFAS chemicals, and many other chemical congeners with organofluorine (U.S. EPA, 2022). Therefore, the lack of correlation of target analytes with a total organic fluorine measurement is reasonable.

Laboratory PFOA sorption experiments

At spike concentrations of 3.4 ppb and 6.9 ppb TF_{aq}, trends of TF_s sorption were observed across the sorption tests conducted (**Figure 16**). Specifically, CFS samples had higher TF_s concentrations than SW samples, and this was consistent even with varying levels of spiked DOC and ash (**Figure 16**). It can be concluded that CFS materials have greater total fluorine (TF) sorption tendencies than SW materials.

The correlation coefficient (R^2) between DOC concentration and sorbed TF concentration was found to be 0.50 for SW and 0.74 for CFS samples. In lower concentration sorption experiments at 3.4 ppb TF_{aq}, 6.9 ppb TF_{aq}, SW spiked with high DOC contained mostly nonzero TF_s concentrations which suggested that the presence of DOC facilitated stronger sorption of TF_{aq} to SW. Compost also had a higher average sorbed concentration at high DOC than in low DOC, further suggesting that DOC was facilitating sorption. Regardless of DOC concentration, CFS had a higher fluorine sorption tendency than SW as observed in its higher average TF_s concentrations (**Figure 16**).

Many of the experimental results from the lab batch experiments were inconclusive due to problems with contamination and poor recovery. Many of the sorption isotherms tests showed little to no recovery, except for sorption tests that were conducted at 3.4 ppb TF_{aq} and 6.9 ppb TF_{aq} of PFOA (**Figure 16**). The results show an increase in sorption consistent with a linear response and good recovery at the lowest TF_{aq} concentrations. However, at higher concentrations of 34 and 69 ppb TF_{aq}, no additional TF_s was sorbed to the media. The increased TF_{aq} concentrations also seemed to decrease the aqueous recoveries. This could be due to several issues. It is possible that there were losses to the vessel walls, losses to filter membranes, or losses of TF_{aq} in the stock solution. Another possibility is that colloidal forms of PFAS could be occurring that form colloid surface coatings which remove dissolved PFAS and outcompete the media surface for sorption. This scenario may explain why higher concentrations of TF_{aq} lowered the concentration of TF_s recovered.

Notably, our preliminary experiments which were run to evaluate contamination issues, were not widely contaminated. However, as experiments were scaled up, there was TF contamination. A series of tests followed which led to the hypothesis that the building water may be contaminated, and the reverse osmosis systems may be unable to remove the fluorine in the feedwater. In any case, the presence of high levels of organic fluorine coupled with poor recoveries of some of the spiked fluorine, made assessment of the isotherm sorption experiment results unreliable at the higher concentrations of 34 ppb and 69 ppb TF_{aq}.

Key findings in detail

1. Compost filter socks retained more contaminants than straw wattles.

Retention of trace metals in products

- CFS samples retained higher concentrations of metals than SW samples. Of the seven trace metals detected in SW and CFS samples, six of the metals (As, Co, Cu, Ni, Pb, and Zn) were retained at higher concentrations in CFS compared to SW (**Tables 7-9, Figures 8-9**).
- Metal concentrations in deployed CFS samples were significantly different from concentrations in nondeployed CFS controls, except for Cr. There were no significant differences in metal concentrations between deployed SW samples and nondeployed SW controls, consistent with the low metal retention observed in SW.

Retention of PFAS in products

- Only one PFAS congener, NEtFOSE, was found in the SW samples analyzed, while 11 different congeners were found in CFS, and 22 different congeners were found in SD samples (**Figure 15**, **Table 11**).
- The relative potential for PFAS sorption was characterized through a set of controlled sorption experiments. In these experiments, SW and CFS materials were evaluated for relative PFOA sorption in the presence of low, medium, and high DOC concentrations, as well as with and without the presence of ash, all to mimic various conditions that might occur in field settings. Low concentrations (3.4 ppb and 6.9 ppb TF_{aq}) sorption tests showed CFS materials sorbed nearly all (around 90%) of the added PFOA while SW sorbed little to no PFOA (**Table 12**). However, at higher concentrations of TF_{aq}, tests were inconclusive due to poor recoveries.

Retention of PCDD/F in products

- No PCDD/F were found in SW, while all compost filter sock and sediment samples contained some detectable concentrations of PCDD/F (**Figures 11-12**). Four congeners were found in CFS, and seven in SD samples (**Table 10**).

2. Deployment and management of products are important to ensure BMPs are effective for the use of either erosion control and/or contaminant retention.

During field sampling, the team observed that some products were deployed and/or maintained in ways that would render them less effective, or ineffective in some cases for the use of either erosion control and/or contaminant retention (**Figures 17-20**).

During sample collection, pictures were taken of other BMPs deployed. There were instances of incorrectly deployed and/or maintained BMPs. For example, in **Figure 17**, a SW was incorrectly deployed on hard surfaces with part of the wattle on the pavement. The hard surface of the pavement prevented the required staking of SW into the ground. Rocks were instead used to hold the SW in place. The SW was insufficient in this instance because the large volume of stormwater runoff overtopped it and SWs should not be used on hard pavements. **Figure 18** shows similar instances of incorrect deployment as **Figure 17**.

In **Figure 19**, a CFS was not staked in place, The CFS may have been moved by weather elements, causing the CFS to not be strategically placed in the middle of the trench to intercept stormwater. In **Figure 20**, a very degraded SW behind the CFS was again not staked into the ground but only weighed down by rocks. This SW could have also been maintained or replaced because the end of its casing opened up, exposing the material to weather elements.

“Hot spots” are locations with elevated chemical concentrations due to burned materials that inherently have high chemical concentration. Hot spots may require more active management of products when they are used for erosion control and contaminant retention. Post-cleanup removal of products used as BMPs should be consistently practiced.

The following are practice recommendations from this study. The final Stormwater Pollution Prevention Plan (SWPPP) should be made by a Qualified SWPPP Developer (QSD). Implementation or placement should be made by a Qualified SWPPP Practitioner (QSP), who should observe and document the products over the period of storms.

Practice Recommendations

- Use CFS in urban areas that are typically hot spots for contamination after a wildfire (e.g., mobile home parks, commercial areas, etc.).
- Use CFS near waterways to improve runoff quality.
- Use SW and CFS in areas needing soil stabilization and sediment retention.

Conclusions

The present study has detailed the results of a field-sampling and laboratory-based assessment of contaminant retention in SW and CFS following the 2018 Camp Fire. The study has revealed that both inorganic and organic contaminants persist following wildfires and are present in stormwater systems.

To the authors’ knowledge, this is the first study to measure PFAS and PCDD/F in stormwater sediments following urban burning. While the total number of samples was small, the frequent detections coupled with some relatively high concentrations suggest that persistent organic chemicals should be considered for watershed management and protection when responding to urban disasters. The findings of PCDD/F in sediments contrast with a study of residential structure ash conducted following the Camp Fire (Tetratech, 2019) which found no presence of PCDD/F in ash. The discrepancy is likely due to the sources and temperature of burning between house fires and whatever the source of the sediment PCDD/F was. These discrepancies also show that post-fire environments are highly heterogenous, and contaminant profiles are likely to vary across current and historical land uses, burning characteristics, and the presence of specific infrastructure. Contaminants may be present at high concentrations in stormwater systems following fire. While this study is not comprehensive, it is clear that there are likely hot-spots of chemical contamination across the urban environment after WUI fires.

The study also suggests that installing sediment retention products quickly, correctly, and with sufficient capacity, is likely to be effective in reducing contaminant loading into local watersheds. The presence of chemicals in sediments behind products clearly shows that chemicals can be prevented from entering the environment. The team frequently observed failing sediment controls during sampling as many of the sediment retention products were not maintained, were overtopped, and insufficient quantities of products were in place by the end of the winter season.

The CFS were found to have contaminant profiles that closely resembled those profiles found in adjacent sediment, while SW did not have the same degree of contaminant uptake as CFS. These results indicate the added benefits of compost related to flow through, likely due to enhanced removal of suspended particles. It should also be noted that this study did not investigate the flow through, loading, or “filtering” efficiency of the sediment retention products. Loading, flow rate, sediment load, and flow effects, such as ponding, may influence the removals of contaminants by these materials.

The increased contaminant retention by CFS material in laboratory experiments clearly shows that there is a difference in surface reactivity between CFS and SW materials, thus contaminant uptake is not only related to particle capture. The side-by-side sorption tests showed increased PFOA sorption by CFS, while there was almost no PFOA sorption by SW. These experiments were not dependent on flow through as they were batch sorption tests conducted in controlled settings under identical conditions. Therefore, CFS materials do seem to be a better sorbent, at least for PFOA. The findings demonstrate that organic and inorganic contaminants originating from WUI fires are mobilized by stormwater runoff, and CFS are more effective than SW in retaining these contaminants.

Abbreviations and Acronyms

As- Arsenic

BMP- Best Management Practices

CAL FIRE- California Department of Forestry and Fire Protection

Cal OES- California Governor’s Office of Emergency Services

CalRecycle- California Department of Resources Recycling and Recovery

Cd- Cadmium

CFS- Compost Filter Sock

CIC- combustion ion chromatography

Co- Cobalt

Cr- Chromium

CSE- Chico State Enterprises

Cu- Copper

DDRO- Disaster Debris Recovery Operations

DOC- Dissolved organic carbon

DTSC- California Department of Toxic Substances Control

EOF- Extractable Organic Fluorine

FEMA- Federal Emergency Management Agency

HERO- Human and Ecological Risk Office

K_d- sediment sorption

Ni- Nickle

NSHE- Board of Regents Nevada System of Higher Education

Pb- Lead

PCDD/F- polychlorinated dibenzo-p-dioxins and furans

PFAS- Per- and polyfluoroalkyl substances

PFOA- Perfluorooctanoic acid

QSD- Qualified SWPPP Developer

QSP- Qualified SWPPP Practitioner

ROE- Right of Entry

SD- Sediment

SW- Straw Wattle

SWPPP- Stormwater Pollution Prevention Plan

TEQ- Toxic equivalency

TF- Total Fluorine

USFS- United States Forest Service

WERT- Watershed Emergency Response Team

WUI- Wildland-urban interface

Zn- Zinc

For PFAS acronyms, please refer to **Table 4**

For PCDD/F acronyms please refer to **Table 5**

Glossary of Terms

Best Management Practices — Proper methods for the control and dispersal of water.

Congener — Compounds with similar structures.

Erosion — Wearing away of the surface of the land, by action of water or wind due to timber harvesting operations.

Extractable organic fluorine — The isolation of organofluorine from the sample matrix.

Sediment retention products — Products to route field runoff through practices that filter, trap, or settle soil particles.

Sorption — a process in which one substance attaches to another substance, either physically or chemically.

Total Fluorine — The total amount of organic and inorganic fluorine measured directly in a sample.

Toxic equivalency — A weighted quantity measure based on the toxicity of each member of the PCDD/F congeners category relative to the most toxic members of the category.

Toxicity equivalent factors — Consensus estimates of congener-specific toxicity/potency relative to the toxicity/potency of an index chemical. TEFs are the result of expert scientific judgment using all of the available data and considering uncertainties in the available data.

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Appendix

Figure 17: Example of an improperly deployed and maintained straw wattle at a mobile home park burned by the Camp Fire. Photo Credit: Jackson Webster.

Figure 18: Another example of an improperly deployed and maintained straw wattle after the Camp Fire. Photo credit: Jackson Webster.

Figure 19: Example of an improperly deployed compost filter sock after the Camp Fire. Photo credit: Jackson Webster.

Figure 20: Example of improperly deployed straw wattle after the Camp Fire. Photo credit: Jackson Webster.