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1.5.1	Leaching Methods
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Information in these Priority Topics was prepared to supplement the information published in Section 11 and the PFAS Analytical Methods Excel file. Information about Strategic Environmental Research and Development Program (SERDP) and Environmental Security Technology Certification Program (ESTCP) projects for Sampling and Analysis can be found at https://serdp-estcp.mil/.

1.5.1 Leaching Methods

Approaches, methods, and tools available to evaluate PFAS leachability were previously discussed in Section 10.4.1. Information presented here provides an update on the current state of methods and studies since the last update of the Guidance Document in 2023.

The US Environmental Protection Agency (USEPA) has published two hazardous waste methods (SW-846) relating to leaching. One, SW-846 Method 1311 (USEPA 1992), the Toxicity Characteristic Leaching Procedure (TCLP), simulates landfill leachate conditions to evaluate the mobility of organic and inorganic analytes present in liquid, solid, and multiphase wastes. The other, SW-846 Method 1312 (USEPA 1994), the Synthetic Precipitation Leaching Procedure (SPLP), simulates weathering in situ by acid rain to model the leaching of organic and inorganic analytes present in liquids, solids, and multiphase wastes under normal weather conditions.

In addition to these two methods, the Leaching Environmental Assessment Framework (LEAF) is the result of long-standing collaboration between experts and scientists in association with the Vanderbilt University School of Engineering, the Energy Centre of the Netherlands, Arcadis-US Inc., and the USEPA Offices of Research and Development and Solid Waste and Emergency Response. In 2011, USEPA released a report providing detailed background information for the LEAF Methods (USEPA 2011). The LEAF documentation provides for standardizations of multiple leach testing methods and pairs premade laboratory data collection templates with both laboratory data management and data analysis tools.

LEAF testing is currently represented by four defined methods:

- Method 1313—Liquid-Solid Partitioning as a Function of Extract pH Using a Parallel Batch Extraction Procedure (USEPA 2017)
- Method 1314—Liquid-Solid Partitioning as a Function of Liquid-Solid Ratio for Constituents in Solid Materials Using an Up-Flow Percolation Column Procedure (USEPA 2017)
- Method 1315—Mass Transfer Rates of Constituents in Monolithic or Compacted Granular Materials Using a Semi-Dynamic Tank Leaching Procedure (USEPA 2017)
- Method 1316—Liquid-Solid Partitioning as a Function of Liquid-Solid Ratio Using a Parallel Batch Extraction Procedure (USEPA 2017)

These four leaching test procedures follow a tiered approach presented by Kosson et al. (2002) (Vanderbilt University 2025) and can provide insights into the rate and extent of leaching processes through evaluation of pH, liquid-to-solid ratio, and leaching time representing the tested conditions.

Use of such innovative testing, data management, modeling, and presentation tools is of immense benefit to environmental practitioners related to data evaluation and sharing and reflects the increasing availability of powerful data processing technologies.

The framework is supported by the following downloadable tools (Vanderbilt University 2025)

- Preliminary versions of USEPA leaching test methods
- Data templates for collecting, archiving, and uploading laboratory leaching test data
- Latest “for distribution” version of LeachXS Lite
- Various publications (or links to sources) that support and demonstrate LEAF for several materials

The downloads are free but require application submission to obtain a free user license. The USEPA has developed a LEAF How-To Guide and currently maintains a website regarding LEAF Methods and Guidance (USEPA 2019). The relevant LEAF supporting downloads are only available through Vanderbilt University. These links are provided above.

Both the LEAF methods and guidance and the two SW-846 methods have not, however, been validated specifically for use in evaluating per- and polyfluoroalkyl substances (PFAS) mobility (Strock 2024). SERDP has funded one project (ER23-3761) focused on the optimization of LEAF for the evaluation of PFAS mobility. Vanderbilt University and the USEPA are evaluating these optimized protocols through a multilaboratory validation study. Release of these optimized protocols is pending the outcome of this validation effort. Release of these optimized protocols is pending the outcome of this validation effort.

Outside of the US, entities have published their own leaching procedures. For example, Australia published the Australian Standard Leaching Procedures tests, AS4439.2:2019: *Wastes, sediments, and contaminated soils. Part 2: Preparation of leachates—Zero headspace procedure* and AS4439.3:2019: *Wastes, sediments, and contaminated soils, Part 3: Preparation of leachates—Bottle leaching procedure*. However, these tests have also not been optimized for the evaluation of PFAS leachability. See Appendix H of HEPA (Heads of EPA Australia and New Zealand) (2025) for more information. Kleja et al. (2025) evaluated the ISO leaching tests and Kalbe, Pieochotta, and Bandow (2024) evaluated European leaching tests for use on soils containing PFAS.

New Jersey Department of Environmental Protection (NJDEP) has a developed guidance and a spreadsheet calculator for site-specific soil-to-groundwater screening levels and soil-water partition coefficients (Kd) for specific PFAS using Synthetic Precipitation Leaching Procedure (SPLP) results. Within the “Soil and Soil Leachate Migration to Ground Water Exposure Pathway Calculator,” SPLP analytical results are adjusted to field conditions through NJDEP’s calculator and are intended to be compared to existing soil leachate remediation standards (NJDEP 2025).

A number of publications have evaluated the application of existing leaching methods to PFAS (for example, Navarro et al. 2024; Kalbe et al. 2024; Rayner et al. 2022), all of which support the conclusion that optimization for the evaluation of PFAS is needed. Others evaluated PFAS leaching in situ using high-resolution characterization techniques (for example, Bigler et al. 2024). PFAS sorption and desorption processes with a focus on understanding leaching is another active research area. For example, Bierbaum et al. (2023) examined rate-limited leaching in column and lysimeter experiments, the relevance of sorption to the air-water interface, and colloid-facilitated transport. USEPA (2025) provides information about using SPLP with modifications for PFAS.

1.5.2 Fish Sampling and Laboratory Homogenization

The goal of this section is to summarize current knowledge on fish sample collection and preparation. The sample collection discussion presents information about containers, shipping, and general best practices, as available in the literature. Seasonal variability is discussed. Avoiding cross contamination with processing steps such as homogenization is discussed, including parameters to be considered (for example, whole body fish, fillets, skin on/skin off). Sampling procedures for fish are discussed in Section 11.1.7.11. Information presented here provides an update on the current state of methods and studies since the last update of the Guidance Document in 2023.

1.5.2.1 Sampling Design Considerations

Multiple studies in the literature have measured PFAS in fish, but few standardized approaches inform field collection and laboratory preparation of fish. Most recently, USEPA released Method 1633A, a standardized method including general guidance for sample collection, preparation, and analysis of 40 PFAS in biological tissues (USEPA 2024). FDA C-010.03 provides laboratory preparation and analysis guidance for 30 analytes in food matrixes, including fish and shellfish, with validation in salmon and clams (USFDA 2024). USDA CLG-PFAS 2.04 provides laboratory preparation and analysis of 16 PFAS in catfish muscle, as well as some mammalian and poultry tissues (USDA 2023) .

Beyond USEPA Method 1633A, more detailed protocols for fish sampling activities remain limited. However, some existing guidance designed for other contaminants may provide general guidance for fish sampling efforts for PFAS (USEPA2000). Additionally, general techniques and best practices that delineate sampling in other matrixes may be applicable to fish and other biota. For example, field and laboratory activities should avoid PFAS-containing equipment and supplies to minimize

the risk of cross-contamination, as recommended for collection of water, sediment, and other ecological matrixes (MI DEQ 2019; USEPA 2024).

PFAS levels in fish vary based on organism phenology or seasonal environmental fluctuations impacting ambient PFAS levels (Petali et al. 2024; Taylor 2019). For example, fish have been demonstrated to transfer PFAS to their eggs depurate PFAS during egg-laying (Conard et al. 2022). Seasonal migration, dietary changes, or behavioral changes can also impact organismal interactions with the environment and resulting PFAS accumulation (Petali et al. 2024). Seasonal hydrodynamics may cause changes to ambient PFAS levels in water and may impact observed body burdens in resident fish, though some data suggest migratory patterns may primarily drive PFAS body burdens as compared to ambient water levels (Pignotti et al. 2017; Tokranov et al. 2021; Taylor 2020). Stocked fish may rapidly accumulate PFAS in situ depending on ambient water concentrations, therefore stocking date should be considered if sampling stocked fish (Danielson 2023). Additionally, operational and safety constraints may limit site accessibility to certain seasons or periods. The timing of collection activities should consider relevant seasonality or periodicity impacting target species; those studies seeking interannual comparisons should collect fish within a similar time frame each year.

Varied diet, habitat, and migratory choices result in inherent variability in PFAS concentrations found in fish, with implications for sample size goals (Petali et al. 2024). Guidance offered by a state fish collection standard operating procedure suggests at least ten fish should be collected, including both sexes (MI DEQ 2019). A modeling effort examining the impact of sample size on derived mean and median concentrations suggested collection of at least 20 individuals per location, with a suggested sample size of 30–50 individuals, to derive robust concentration medians for use in human health risk assessment (Taylor 2019). In practice, budget constraints and laboratory bandwidth are also key factors dictating sample size.

The size of fish included in PFAS sampling may vary, as the relationship between fish length and PFAS body burden is unclear. This uncertainty has notable implications for study design. Some studies suggest PFAS levels correlate to animal size—for example, mass or length—while others find no such relationship or nuanced size–PFAS relationships in different species or habitats (Taylor 2019; Hedgespeth et al. 2023; Fair et al. 2019). In practice, a range of fish lengths may be sampled to achieve population summary data, while only fish of a specific size class, as an indicator of age class, may be targeted for derivation of more accurate bioaccumulation or biomagnification characteristics.

Given the unique factors influencing PFAS body burdens in fish, a work plan delineating study design, field sampling plans, and associated quality assurance measures should be constructed prior to initiating field efforts, specifically considering how sampling elements facilitate PFAS study goals. For example, if spatial comparison is an objective, wide-ranging, common species that occur across the region of interest should be targeted. If human health risk assessment is a project goal, fish species and sizes typically consumed by anglers should be sampled. If ecological risk assessment is the primary goal, fish species and sizes regularly consumed by birds, mammals, or other focal predatory species should be collected.

1.5.2.2 Sampling, Equipment, and Supplies

Multiple mechanisms exist to collect fish, including seine trawls, electrofishing, rod and line, or fish traps; shellfish may be collected by hand or bottom trawl. Each approach is inherently opportunistic, as sampling efforts are constrained to those individuals randomly inhabiting a given location and capable of interacting with the collection gear. No data exist documenting the impact of collection mechanism on PFAS body burdens in fish, though some guidance suggests the risk of contamination during collection is minimal given the presence of PFAS within internal tissue reservoirs and the order of magnitude difference of biota concentrations (typically ppb) compared to water (typically ppt).

Sample container material recommendations depend on the size and number of fish or shellfish to be collected. USEPA Method 1633A recommends wrapping collected fish in aluminum foil or storing in polyethylene tubing, whereas MI DEQ (2019) guidance suggests rinsing fish with ambient water before storing individually in polyethylene bags. Literature studies used polyethylene bags, high-density polyethylene (HDPE) or polypropylene vessels, or glass vessels to store fish samples (Blazer et al. 2024; Fair et al. 2019; Soudani et al. 2024). USEPA Method 1633A recommends storing collected fish or fish tissues at or below 6°C from the time of collection until receipt by the processing laboratory for a maximum of 24 hours. If a longer transport time is necessary, 1633A recommends freezing the sample upon collection before storage and shipment on dry ice or otherwise chilling to maintain sample temperature below 6°C.

Once received by the laboratory, USEPA Method 1633A recommends sample storage in a manner protected from light at or below -20°C until prepared for laboratory activities. USEPA Method 1633A also recommends that sample storage should be no more than 90 days from collection to sample extraction. Preliminary data from a 2023 USEPA retrospective study of

archived tissues suggest fish fillet tissue samples may be frozen for up to 5 years without significant impacts on PFAS measurements, but further study may be required to validate holding times for specific species and tissue types to be used for regulatory purposes (McCarty et al. 2024).

1.5.2.3 Sample Dissection, Homogenization, and Lyophilization

Materials that directly contact fish tissue during dissection and sample processing should be PFAS-free. Recommended decontamination approaches include rinsing equipment and consumables with water and available/appropriate solvents prior to use and between samples (MI DEQ 2019; USEPA 2024). Existing guidance suggests general dissection procedures used to obtain tissue samples for other contaminant monitoring efforts may be used to obtain fish tissue samples for PFAS analysis, accounting for PFAS-specific contamination risk (MI DEQ 2019; NY DEC 2023; USEPA 2000). Fish plasma, muscle fillets with and without skin, muscle plugs, liver, and internal organs have been used for PFAS measurements in addition to whole fish carcass (whole body) samples (Nilsen et al. 2024; Blazer et al. 2024; Levanduski et al. 2024; Miranda et al. 2023). Although skin-off fillets have been found to contain lower PFAS concentrations compared to skin-on fillets, the decision to retain skin should consider routine consumption trends and prior sampling practices to ensure data comparability to any archived samples (Figueroa-Munoz et al. 2025). Muscle plugs have been used for PFAS analysis, though no published information currently compares concentrations derived via plugs to whole muscle samples, as has been demonstrated for heavy metals (Miranda et al. 2023; Stahl et al. 2021). Additionally, no published information documents the impacts of homogenization on subsequent PFAS measurement in tissue, though a limited comparative effort suggests variable homogenization efficiency between different tools (Shaw et al. 2024). Lyophilization may be used during tissue preparation to address variable water content in different tissue matrixes and increase extraction efficacy of organic solvents. Literature suggests partial loss of some volatile compounds during lyophilization of tissue with good retention of long-chain PFAAs (Balgooyen et al. 2025; Carrizo et al. 2023). USEPA Method 1633A does not include lyophilization, and multiple datasets have derived PFAS measurements directly from wet tissue (Blazer et al. 2024; USEPA 2022).

1.5.2.4 Analysis

Analytical strategies validated for analysis of fish tissue are detailed by recently published methods such as USEPA Method1633A (USEPA 2024), FDA C-010.03 (USFDA 2024), and USDA CLG-PFAS 2.04 (USDA 2023) , as well as a substantial body of peer-reviewed literature (Valsecchi et al. 2013; Nakayama et al. 2019). These methods provide separation and detection conditions for specific anionic analytes in fish tissue and other complex matrixes. Regardless of applied method, the complexity of endogenous chemicals (for example, lipids, fatty acids, and proteins) inherently present in fish tissues varies by species and life stage, which may present unique analytical challenges. Some endogenous chemicals may interfere with analytical measurements of PFAS conducted using tandem mass spectrometry platforms. Variable preparation strategies, separation techniques, or high-resolution mass spectrometry (HRMS) can be leveraged to confirm or refute potential interferences (Bangma et al. 2024;, MI EGLE 2024). Endogenous molecules may also impact ionization efficiency; some peer-reviewed studies include the use of matrix-matched curves to better account for matrix effects during analysis and quantification (Carrizo et al. 2023; Malinsky et al. 2011). Continued efforts seek to refine analytical strategies for fish tissues and other complex matrixes, leveraging techniques such as HRMS and non-targeted analysis, ion mobility, total oxidizable precursor (TOP) assay, and extractable organofluorine analysis to expand analytical coverage of PFAS in fish tissues (Boatman et al. 2024; Pickard et al. 2022; Pickard et al. 2024). These methods have not been validated or standardized across the range of fish species and tissue types; therefore, data derived using these methods should be used with caution.

1.5.3 Ultrashort-chain PFAS Analysis

This section provides information regarding the analysis of ultrashort-chain PFAS (ultrashort PFAS), including the feasibility and challenges of using existing PFAS analytical methods. This topic supplements Section 11.2, which addresses analytical methods and techniques. For additional information about ultrashort PFAS, see Section 1.1.3.1. For additional examples and occurrences and health effects associated with ultrashort PFAS, see Section 1.4.4. In addition, see the updated Table 4-1 for physical and chemical properties of some ultrashort PFAS.

1.5.3.1 Overview

The measurement of ultrashort PFAS was not seen to be as important as the measurement of short-chain and long-chain PFAS in the early days of PFAS testing. However, the implementation of “replacement” PFAS, the higher concentrations of ultrashort PFAS in the environment, and the presence of ultrashort PFAS almost everywhere increased the importance of ultrashort PFAS measurement over time. Much of the development for the measurement of ultrashort PFAS has occurred in recent years as the need to evaluate them has increased (Ateia et al. 2019). USEPA added two ultrashort PFAS to the

Regional Screening Levels table: perfluoropropanoic acid (PFPrA) and 1,1,1-trifluoro-N-[(trifluoromethyl)sulfonyl]methanesulfonamide (TFSi).

1.5.3.2 Ultrashort-chain PFAS

Table 1-11 lists ultrashort PFAS that currently have commercially available analytical reference materials. As of September 2025, no consensus methods have been published for this class of compounds. Most ultrashort PFAS analyses will not include all these compounds, but any of them may be expected to be reported from an ultrashort PFAS analysis. Note that the salt forms associated with these acids will have different CAS Registry numbers. See "Anions Versus Acids" in Section 2.2.3.1 for an explanation.

Table 1-11. Commonly reported ultrashort-chain PFAS

Analyte Name	Acronym	CAS Number
2,2,3,3-Tetrafluoropropanoic acid	2,2,3,3-TFPA	756-09-2
2,3,3,3-Tetrafluoropropanoic acid	2,3,3,3-TFPA	359-49-9
Perfluoropropanoic acid	PFPrA	422-64-0
1,1,1-Trifluoro-N-[(trifluoromethyl)sulfonyl]methanesulfonamide	TFSi	82213-65-3
Trifluoroacetic acid	TFA	76-05-1
Trifluoromethanesulfonic acid	TFMS	1493-13-6
Perfluoropropane-sulfonic acid	PFPrS	423-41-6
Perfluoroethanesulfonic acid	PFEtS	354-88-1
Difluoroacetic acid	DFA	381-73-7
Perfluoro-2-methoxyacetic acid	PFMOAA	674-13-5

Numerous methods have been developed and published for the analysis of ultrashort PFAS (for example, Wang et al.2024; Zou et al. 2024; Liang, Steimling, and Chang 2023; Jacob and Helbling 2022). A significant challenge in measuring ultrashort PFAS is their incompatibility with reversed-phase liquid chromatography, which is the preferred technique for analysis of PFAS having four or more carbon atoms.

This challenge is commonly overcome using a different stationary phase. Various choices are available, including hydrophobic-interaction liquid chromatography (HILIC), ion exchange chromatography, and various hybrids of those reversed-phase, HILIC, and ion exchange stationary phases (Bjornsdotter et al. 2020). The ideal column, mobile phases, and gradient will depend on the ultrashort PFAS being analyzed.

Aside from issues with retention on the cartridge or column, another major challenge with measuring ultrashort PFAS is blank contamination, particularly from TFA. This can impact the interpretation and quantitation of analytical results. Prior work has shown that laboratory consumables, particularly those containing fluoropolymers, can leach ultrashort-chain PFAS such as TFA (Joudan et al. 2024).

1.5.3.3 Simultaneous Analysis of Ultrashort-chain and Other PFAS

For the most part, the use of these alternative liquid chromatography (LC) stationary phases prevents the inclusion of larger PFAS in the same analysis, although 4-, 5-, and even 6-carbon PFAS may be analyzed along with the ultrashort PFAS. Larger PFAS typically have less retention on these columns as the number of carbons increases, although exceptions may exist. More complicated chromatographic systems involving combinations of columns can be attempted to address this issue, but the desire to have a single analysis must be balanced against the increased complexity and vulnerability to problems that increase with such an approach.

Recently, LC stationary phases that mix either HILIC or ion exchange or both with reversed-phase LC stationary phases provide some hope that ultrashort PFAS may be analyzed in the same analysis as larger PFAS (Waters Corporation 2023; Restek Corporation 2024). This aspect of ultrashort PFAS analysis is very much in its early phase, and one can reasonably expect that things will change. Until new approaches to simultaneously analyze a wide range of PFAS with varying carbon chain lengths emerge, analysis of ultrashort PFAS will often be a separate analysis from the short- and long-chain PFAS.

1.5.3.4 Sample Preparation

Sample preparation before analysis is another area of active development. Water samples of various sorts are typically handled by weak-anion exchange (WAX)/reverse-phase hybrid SPE sorbents. Solid samples likewise follow other methods in that an initial extraction of the solid in basic methanol is followed by dilution in water and SPE in the same manner as water samples. The QuECHeRs (Quick, Easy, Cheap, Rugged, and Safe) method has been explored for food samples, but it is not clear whether stable-isotope-labeled extraction standard recoveries were comparable to those obtained with WAX SPE (Santa-Mayor et al. 2023).

Other options for sample preparation for the analysis of ultrashort PFAS include preparation for direct injection analysis and on-line SPE (Jacob and Helbling. 2022). Direct injection analysis is particularly appropriate for many ultrashort PFAS due to the near-ubiquitous presence of ultrashort PFAS in all parts of the environment and the relatively high concentrations in the environment of many of them. One study from the State of California Division of Drinking Water showed elevated detection limits when performing direct injection analysis compared to WAX SPE (CA Water Boards 2024).

On-line SPE combines the process of extraction and analysis of a sample into a single process by placing SPE columns in-line with the analytical LC column. This technique simplifies and reduces the cost of SPE when it works well, but the relatively small bed volume of SPE columns and reusable nature of them can lead to degradation in SPE performance, contamination, and building of back-pressure that can cause analysis to halt.

1.5.3.5 TOP Assay

Many implementations of the TOP assay have not included any measurement of ultrashort PFAS (Neuwald et al. 2022). Conversion of precursors containing four or more carbons to end products having three or fewer carbons is therefore not assessed. Likewise, conversion of molecules with two or three carbons is not assessed when ultrashort PFAS are not measured.

TOP assay typically produces a higher concentration of perfluorocarboxylic acids (PFCAs) with decreasing chain length thereof. Thus, in a typical measurement of PFCAs having between 4 and 14 carbons, perfluorobutanoic acid (PFBA) is the most increased in abundance, followed by perfluoropentanoic acid (PFPeA), then perfluorohexanoic acid (PFHxA). That pattern suggests that one should reasonably expect that PFPrA would be more abundant than PFBA, and TFA more abundant than PFPrA. Although the precise proportion of conversion products is not known, it is a reasonable assumption that a significant proportion of conversion products is not captured when ultrashort PFAS are not measured. Note that other possible distributions of products could occur, depending on the precursors present. More information about TOP assay is included in Section 11.2.2.2.

One example of a study in which ultrashort PFAS were analyzed after oxidation can be found in Tsou et al. (2023). This study showed, as expected, that both PFPrA and TFA were found in samples that had been oxidized. The relative proportions of the ultrashort PFAS to one another and short- and long-chain PFAS varies from sample to sample, but the ultrashort PFAS content is, as one would anticipate, significant relative to the other PFAS. Another example can be found in Janda et al. (2019), which provides information on an improved protocol for the TOP assay when targeting 2- to 14-carbon chains.

1.5.3.6 High-Resolution Mass Spectrometry of Ultrashort-chain PFAS

Studies of ultrashort PFAS using HRMS for non-targeted analysis (NTA) are relatively few, but ultrashort PFAS should be detectable in NTA, provided they are high enough in concentration to be detected. Those ultrashort PFAS with reference standards can be unambiguously identified. Other compounds may be more difficult to identify.

As with any HRMS NTA, the precise concentration of a compound cannot be known without a quantitative standard. Both ionization and fragmentation can be (and usually are) affected by changes in molecular composition or structure. Therefore, one cannot determine a detection limit without a quantitative standard. These factors result in some uncertainty in the ability to detect compounds in an NTA.

As with targeted analysis, high polarity of ultrashort PFAS introduces another difficulty in that a column other than the usual reversed-phase column must be used to separate the ultrashort PFAS from other components to avoid distortion of the mass spectrum from interfering compounds.

1.5.4 Consumer Product Testing

This section describes the current testing methods, types of analyses, data uses, and regulations related to the testing of

PFAS in consumer products such as textiles, food packaging, cosmetics, nonstick cookware, and juvenile products. This topic supplements Section 11.1.7, which discusses PFAS sampling in various media but does not discuss consumer products.

1.5.4.1 Current Testing Methods

A variety of analytical methods is available for PFAS testing; however, only a few of them have been validated for consumer products and other chemical products with complicated matrixes. Testing of PFAS in consumer products can be addressed through multiple analyses and may use a workflow combining fluorine content analysis, target analysis, and non-targeted or suspect screening analysis methods. See ASTM F3700-25 Standard Guide for Selecting and Applying Analytical Methods to Evaluate PFAS in Consumer and Related Products (ASTM 2025).

Consumer products testing presents unique challenges due to the sample matrix, differentiating them from environmental media testing. The sample matrix for products is typically a complex composition from diverse materials (paper, fabric, natural materials, plastics). PFAS could be included in consumer products at percent levels to provide a certain function or could be residues or impurities from the manufacturing process. Other chemicals in the matrix can interfere with PFAS analytical processes from sample preparation to instrument analysis. Extra techniques are required to reduce matrix effects but issues such as co-elution in chromatography, ion suppression in mass spectrometry, or signal masking, may still be present. Sample preparation can be difficult for consumer products made with polymeric PFAS.

1.5.4.2 Targeted Analysis

In targeted analysis, testing methods generally involve LC or gas chromatography (GC), and mass spectrometry (MS). LC/MS is capable of monitoring short-chain PFAS and ionic PFAS, while GC/MS will measure the more neutral and volatile PFAS (TemaNord 2024). In commercial labs, LC/MS is more available for PFAS analyses.

Validated methods for PFAS testing in consumer products are very limited. USEPA has a single laboratory validated method from 2024 for testing 31 PFAS in plastic container walls such as HDPE by LC/MS/MS, and it may be used with modifications for solid samples such as fabric or packaging paper (USEPA, 2024). ISO published an LC/MS/MS method to determine extractable nonvolatile PFAS in leather and coated leather (ISO 23702-1:2023 | IULTCS/IUC 39-1) (ISO 2023). This method is specifically designed for long-chain PFAS (C7-C14). Europe combined LC and GC in method EN 17681-1:2025 (European Committee for Standardization (CEN) 2025) and EN 17681-2: 2022 (European Committee for Standardization (CEN) 2022) to test textiles and textile products for selected extractable PFAS (PFOA, PFOS).

In instances when a validated method does not exist for a specific consumer product, methods validated for soil, sludge, and solids, such as USEPA Method 1633A (USEPA 2024), DIN 38414-14 (DIN 2014, ASTM D7979 (ASTM 2020, and ISO 25101(ISO 2009), are often modified or used as the basis for testing. All targeted methods have a list of predefined PFAS. A primary limitation of these methods comes from the availability of analytical reference standards for individual PFAS.

Other targeted analytical techniques are still in the research phase and lack established or standard methods. For example, pyrolysis-gas chromatography-mass spectrometry (pyr/GC/MS) can be used for the bulk characterization of fluorinated polymers (polymeric PFAS) in consumer products. Pyr/GC/MS methods involve pyrolyzing a sample directly (without extraction) at temperatures between 250°C and 1,000°C, followed by online analysis of the degradation products. Skedung et al. (2024) used a pyrolysis method, applying a maximum pyrolysis temperature of 700°C to 45 consumer products and identified polytetrafluoroethylene (PTFE, most used fluorinated polymer) in cookware, dental products, and electronics at concentrations as low as 0.1–0.2 wt%. It was also possible to use pyr/GC/MS to distinguish between three different side-chain fluorinated polymers in textiles (Skedung et al. 2024). Recent work has established solid-phase microextraction protocols for both aqueous- and gas-phase PFAS analysis (Martínez-Pérez-Cejuela et al. 2025). Such work may allow for outgassing or emissions testing of consumer products; ASTM (2025) acknowledges the application of “off-gassing techniques to determine the potential release of PFAS from products over the product life cycle” (F3700-25, 1.4). Earlier emissions work employed thermal desorption focusing on fluorotelomer alcohols (FTOHs) (Robbins et al. 2023).

In addition to these targeted methods, some laboratories have applied TOP assay to measure PFAAs derived from precursors that undergo oxidation by hydroxyl radicals. To measure precursor concentrations, the most common target PFAS are measured before and after the oxidation using conventional targeted PFAS analysis such as LC/MS (Ateia et al. 2023). However, some PFAS precursors may not be extracted, and incomplete oxidation can occur (for example, PFAS with ether functional groups that are resistant to oxidation), or oxidation products could be missed (for example, ultrashort PFAS) in traditional target analysis. The limitation of TOP assay on ultrashort PFAS is not limited to consumer product testing and is also discussed in Section 1.5.3.5.

Targeted analysis is the most sensitive and confirmative method to measure PFAS in consumer products. But major challenges still exist, including the lack of standard methods, the limited number of individual PFAS that can be measured, and the need for optimalization of extraction procedures (TemaNord 2022).

1.5.4.3 Fluorine Screening

Although they are not yet validated, this section discusses several useful methods for detecting fluorine instead of individual PFAS. For fluorine screening, it is important to note that while all PFAS contain fluorine, not all fluorine-containing compounds are considered PFAS. NTAs discussed in this section include methods that look for totals, such as extractable organic fluorine or total fluorine, and do not reference the NTA looking at suspect screening or unknown targets present in a sample.

Polymeric vs. Nonpolymeric PFAS

Many state-level legislative actions for consumer product bans use total organic fluorine (TOF) as the determining factor for presence or absence of PFAS. If a legislative action does not specifically exclude polymeric PFAS in the TOF content, then it should be assumed that polymeric PFAS is included as part of the TOF. For example, New Mexico's legislation on PFAS (HB 212), signed on April 8, 2025, specifically excludes fluoropolymers from the legislative action. Inclusion or exclusion of polymeric PFAS is an important factor in determining which analytical tests are the best fit for determining organic fluorine content. For more information about analytical tests for determining organic fluorine, see Section 1.4.5.

Types of Analysis

TOF refers to *all* organic fluorine present in a given sample. Although there is currently no test to directly measure TOF, there are methods to approximate a TOF concentration. The current conventional tests for fluorine include:

- Total fluorine (TF), which includes all fluorine present in a sample, inorganic and organic, including polymeric and nonpolymeric forms if present. This can be used for both aqueous and solid samples.
- Inorganic fluoride (IF), which includes soluble inorganic fluoride. This can be used for both aqueous and solid samples.
- Extractable organic fluorine (EOF), which includes organofluorine compounds, encompassing both PFAS and non-PFAS. This can be used for aqueous and solid samples.
- Adsorbable organic fluorine (AOF), which includes organofluorine compounds that will adsorb to carbon, encompassing both PFAS and non-PFAS. This applies only to aqueous samples.

Each of these have some limitations. Where TF includes all fluorine present in a sample, this could overestimate the concentration of organofluorine compounds in the sample. EOF will include only the organofluorine compounds amenable to the extraction process. For example, aqueous samples will depend on the type of solid-phase extraction (SPE) cartridge used. Weak-anion exchange (WAX) cartridges will target anionic PFAS and mostly exclude cationic and zwitterionic PFAS. A hydrophilic-lipophilic balance cartridge may include a larger range of PFAS (Idowu et al. 2025). For solid samples, EOF efficiency will be based on the extraction solvent used and the affinity for each class of PFAS to that extraction solvent (example methodology can be found in Spaan et al. 2020; Spaan et al. 2021). EOF may contain some polymeric PFAS but will not include all polymeric PFAS present in a sample. AOF is limited to aqueous samples. The affinity for shorter chain PFAS (< C4 chain length) to carbon may present a negative bias for shorter chain PFAS. AOF may include some but will not include all polymeric PFAS if present.

To put this in perspective, the potential concentrations for each group would be TF > TOF > EOF > AOF (Ateia et al. 2023). As there is currently no method to directly measure TOF, it is typically reported mathematically by subtracting the IF concentration from the TF concentration, but this method has limitations. These limitations include differences in reporting limits, with limits for inorganic fluoride in the mg/kg (ppm) range and limits for TF and EOF in the ng/g (ppb) range. Use of this calculation may also be confounded by insoluble inorganic fluoride, including naturally occurring minerals such as fluorite, also known as fluorspar (CaF2). There are currently no published methods for the quantification of insoluble inorganic fluoride. Both EOF and AOF may be reported as TOF, however, the limitations discussed above should be considered when using this convention. It is important to consider the type of consumer product being tested and the potential types of IF, as well as polymeric PFAS, that could be present in a sample. For example, testing TF may be sufficient for determining presence or absence of PFAS in a sample when IF is not present. Conversely, a consumer product high in insoluble inorganic fluoride may need a different approach. It is equally important to consider the end use of the data because TOF may detect organofluorine chemicals that are not PFAS.

These analyses are not specific to consumer product testing but are very relevant tools when TOF or TF are part of a regulation for PFAS in consumer products.

Combustion Ion Chromatography

Combustion ion chromatography (CIC) is used to measure TF, IF, AOF, and EOF, see Section 11.2.2.4. If a sample is combusted by CIC prior to adsorption or extraction, then the resulting fluoride concentration corresponds to TF, which refers to both the organically and inorganically bound fluoride present in the sample. TF may not be as reliable of a proxy for PFAS if, for example, fluorine-containing minerals or IF are present at high concentrations in the sample.. USEPA Method 1621 quantifies the concentration of AOF in aqueous samples by adsorbing the organofluorine compounds to a granular activated carbon cartridge that then is combusted (USEPA 2024). EOF can measure organic fluorine in both aqueous and solid samples by extracting the organofluorine compounds from the sample prior to combustion. The methods used for EOF extraction are similar to those used for targeted PFAS extraction with the addition of a step that removes the inorganic fluorine from the sample (Ruyle et al. 2023). There is no standardized EOF method, but it has been validated through interlaboratory comparisons (Kärrman et al. 2021). Ruyle et al. (2023) provides a table of recommended best practices for EOF analysis that can be referenced because there is no standardized method yet.

Other Total Fluorine and Organofluorine Methods

- Particle-induced gamma-ray emission (PIGE), see Section 11.2.2.3. This is a nondestructive rapid screening tool that measures TF. Like TF above, PIGE does not distinguish between types of fluorine and would include inorganic fluorine, polymeric PFAS, nonpolymeric PFAS, and other organic fluorine.
- X-ray photoelectron spectroscopy (XPS). This analytical technique is used to measure fluorine on the surface of a product, looking at the top few nanometers of a sample (Tokranov et al., 2018). This process is highly sensitive to fluorine. XPS can distinguish between different types of chemical bonds, differentiating classes of PFAS (but not identifying individual compounds). There are sample preparation challenges, because XPS requires samples to be analyzed under high vacuum, which can cause issues for certain matrixes.
- Soluble inorganic fluoride in consumer products can be measured by extracting with water or weak acid followed by analysis by a published method. Methods that include soluble inorganic fluoride include USEPA Method 300.1 (USEPA 1997); ASTM D1179 (ASTM 2021); and ASTM D4327 (ASTM 2025).
- TOP assay, see Section 11.2.2.2. The TOP assay is a proxy method to determine additional PFAS present in a sample that cannot be measured by current targeted methods. This method uses the current targeted analytical methods, with a pre- and post-oxidation sample analysis. The difference in PFCAs between these two samples is attributed to the conversion of oxidizable precursors to PFCAs.

Fluorine Screening Methods

There are currently no validated methods that are specific to TOF in consumer products. In European standards, there is no method specified for the determination of organic fluorine in consumer products, only a reference to EN 14582:2016 Characterization of Waste (European Committee for Standardization 2016), which specifies a combustion method for the determination of halogen and sulfur contents. Oxygen combustion in closed systems and determination methods is frequently referenced for consumer product testing. A few research papers compared the target analysis with TF or TOF analysis and revealed the huge discrepancy between fluorine and speciated PFAS. It was consistently found in consumer product testing that the total amount of fluorine cannot be correlated with the sum of speciated PFAS. For example, Rodgers et al. (2002) tested 54 samples with total fluorine over 10 ppm for 36 targeted PFAS via a methanol extraction followed by LC/MS/MS and found that the total amount of fluorine attributable to methanol-extractable target PFAS analytes and to PFAAs generated from oxidizable precursors in samples never exceeded 0.1% of the total F measured in these products.

1.5.4.4 Non-targeted Analysis

Suspect screening and NTA are designed to detect and identify a wide array of known and unknown PFAS without prior knowledge of the chemicals' exact structures or identities, see Section 11.4. Though NTA, such as HRMS, has been successfully applied to a wider range of PFAS in contaminated matrixes, these analytical approaches are not yet common in consumer product analysis (Bugsel, Zweigle, and Zwiener 2023). Most method developments are in the research phase.

A notable study by Stroski et al. (2024) developed NTA workflows to investigate unknown PFAS in various food packaging materials, including paper-based and foil wrappers. The research identified both legacy and emerging PFAS, such as 6:2 fluorotelomer phosphate diester (6:2 diPAP) and its transformation product 2H-perfluoro-2-octenoic acid, also known as 6:2 fluorotelomer unsaturated carboxylic acid (6:2 FTUCA), in these materials. Beyond food packaging, NTA has revealed the

presence of PFAS in several other consumer products, including textiles, cosmetics, and household items. A recent study by the National Institute of Standards and Technology (NIST) employed liquid chromatography-HRMS combined with suspect screening to analyze over 4,000 PFAS in firefighter turnout gear textiles (NIST 2025). One major challenge of using HRMS for PFAS detection in consumer products is the lack of analytical standards and reference spectra for the thousands of known and emerging PFAS. In addition, complex product matrixes (for example, cosmetics, textiles) can interfere with PFAS signal detection and introduce false positives or reduce confidence in compound annotations (Bugsel, Zweigle, and Zwiener 2023). Recent work has applied machine learning and two-dimensional GC analysis for consumer product emissions testing, which may establish a new workflow for PFAS emissions testing (Watson et al. 2025).

1.5.4.5 Regulations

The Regulatory Programs Table includes information about regulations related to the ban of PFAS in certain categories of consumer products.

Manufacturers can use PFAS testing data to show compliance with regulations, ensure consumer safety, maintain brand reputation, or identify potential contamination sources and take appropriate measures to ensure the integrity of their products. On the other hand, regulatory agencies use PFAS testing data to bring the regulated industries into compliance, guide remediation or prevention efforts, and inform rule-making decisions.

Compliance and enforcement usually require testing to follow strict quality assurance/quality control (QA/QC) procedures, use accredited laboratories, and use standardized analytical methods that are rigorously validated for various PFAS and matrixes.

1.5.5 ISM for PFAS Sampling

Incremental sampling methodology (ISM) is a method for collecting and compositing soil samples to achieve a result that is more representative of a given decision unit. Practitioners have found that the sampling density afforded by collecting many increments, together with the disciplined processing and subsampling of the combined increments, in most cases yields more consistent and reproducible results than those obtained by discrete sampling approaches (ITRC 2020). Given the unique properties and potentially low action levels associated with PFAS, there is substantial interest in determining whether ISM is an effective approach for these compounds and whether there are concerns or limitations to its use. This topic supplements Section 11.1, which discusses sampling equipment and procedures for PFAS in soil but does not address ISM. Section 1.7.2.4 discusses the application of ISM-type sampling for biosolids land application sites.

Although peer-reviewed research on ISM for PFAS sampling is, to date, very limited, the Department of Defense (DoD) Environmental Data Quality Workgroup (EDQW) has evaluated the use of ISM for PFAS and identified several factors to consider that were enumerated in a white paper released on 31 January 2025 (DOD EDQW 2025). In short, the current DoD position is that ISM sampling and analysis protocols should not be adopted for PFAS without first performing a validation study of the protocol to determine and remedy sources of positive and negative bias inputs followed by publication of the validated method by the USEPA. The issues identified by the EDQW include some field sampling but are mainly laboratory processes.

Following is a general outline of the concerns highlighted in the DoD report (DOD EDQW, 2025): The grinding step cannot be performed on the ISM soils due to the decontamination procedure that would be required after each sample would cause increased surface rusting of the equipment. Another potential concern related to grinding is the level of volatilization of the more volatile PFAS chemicals that may result from the heat generated during the grinding process. The drying and grinding steps could also potentially promote volatilization and loss of the ultrashort- and short-chain PFAS analytes. Extracted internal standard (EIS), ongoing precision and recovery (OPR), and low-level ongoing precision and recovery (LLOPR) spikes require reference materials that are not commercially available, and even if they were, would likely be cost-prohibitive and taxing on manufacturers of such standards. Finally, when collecting soil samples in the field, typical ISM sampling is performed within the top 18 inches of the soil surface. However, the report cited concerns that PFAS would likely be deeper in the soil column.

Although DoD has published their reservations with several aspects of ISM, at least three states (Hawai‘i, Maine, and Michigan) are actively using modified versions of ISM for investigations involving PFAS impacts (HI DOH 2025; ME DEP 2023; ME DEP 2023; MI EGLE 2024). Specific Maine Department of Environmental Protection (DEP) methods are not outlined in reports but are referred to in their published homeowner guidance and legislative report. Hawai‘i employs the Decision Unit and Multi Increment methodologies, which are similar to the ISM-type investigation methods (HI DOH, 2025). Michigan has used the full ISM to investigate areas that land-applied biosolids (MI EGLE 2024). Generally, states seem to use the planning

and data quality objectives from ISM and find them applicable to sampling for PFAS in some scenarios, but there are still some debates on the 8330b laboratory processing methods. For example, Maine DEP has used a modified ISM procedure that excludes drying and sieving in the laboratory as an effort to alleviate the issues outlined by the DoD (see Section 1.7.2.4).

1.5.6 Concrete and Asphalt Sampling and Analysis

Frequent, historical use of aqueous film-forming foam (AFFF) at various firefighting training grounds (for example, military bases, airports, and oil refineries) has led to significant contamination of PFAS in many industrial areas to the infrastructure materials present. Two such materials of concern are the contamination present in concrete and asphalt, which are common matrixes at industrial sites. There are currently no standardized methods for sampling and analysis of PFAS in these matrixes, although many principles applied to sampling and analysis of other solid matrixes may apply.

For the purpose of this section, concrete matrixes are inorganic matrixes formed from aggregate, sand, and a complex cement (for example, Portland cement), which can vary significantly regarding ratios and input materials. Other materials, such as reinforcing steel bars and plastic fibers, have also commonly been used to enhance the engineered functions. Concrete properties can vary over time as minerals age and fracture, with rehealing also possible. Douglas et al. (2023) reviewed literature on PFAS in concrete and asphalt and found relatively little information available at present, though information on the distribution and leaching of PFAS is emerging. Although essentially a monolithic material, porosity is important for concrete sorption properties, including anticipated transport, ingress, and egress of water and PFAS (Douglas et al. 2023). Asphalt is a mixture dominated by aggregate in an organic bitumen matrix. Partitioning of PFAS in asphalt may therefore be more influenced by hydrophobic interactions than electrostatic interactions that may play a more prominent role for concrete (Douglas et al. 2023). Asphalt may also change significantly in properties over time as lower molecular weight organic solvents are lost to volatilization and other components are subject to oxidative change. Douglas et al. (2023) also concluded that use and contamination with other products, such as hydraulic fluids, oils, and degreasing solvents, can also change PFAS interactions with both concrete and asphalt.

PFAS sampling in concrete and asphalt is an important tool in managing these materials post-PFAS contamination (for example, reuse, remediate, or dispose), understanding if the contaminated matrix could pose as a secondary source of PFAS runoff during precipitation events, and determining the fate and transport of PFAS through the matrix. To date, most studies have focused on concrete with regards to understanding PFAS impacts. Best practices regarding sample collection, extraction, and analysis of concrete matrixes are discussed in the following sections. Additionally, the fate and transport of PFAS within the matrix is also discussed to highlight considerations when trying to collect a representative sample.

Although there is currently no federal or state guidance available, the purpose of this section is to promote consistency of approaches for sampling and analysis of concrete and asphalt until more formal guidance is available. This topic supplements Section 11.1.7, which discusses PFAS sampling procedures for various media but does not cover concrete or asphalt.

1.5.6.1 Sample Collection and Extraction

Due to the complexities of concrete and asphalt matrixes, and the potential for changes over time, sampling is not straightforward. Work at a concrete firefighting training pad from Baduel et al. (2015) showed that PFAS concentrated in the top few centimeters, with penetration to depth largely correlating with chain length or molecular weight. In contrast, Williams et al. (2023) showed that for a different firefighting training pad, there was considerably more variability in concentrations and relationships between different PFAS with depth, with chain length a poor predictor of penetration distance. These differences could be due to the chemistry of the AFFF released, environmental conditions (for example, precipitation), or the exact composition of the concrete, as this is not uniform. Any sampling plan for PFAS analysis in these matrixes must consider potential for spatial heterogeneity and site-specific conditions.

Asphalt and concrete samples are collected as cores for transport to the laboratory. Standard coring techniques can be used. Air- and water-cooled corers are available. If water-cooled corers are used, the water used for lubrication/cooling should be verified as PFAS-free to an appropriate limit of reporting prior to use. Cores can be subsampled by dry cutting with a diamond saw into segments for monolith leach testing and for subsampling monoliths (Vo et al. 2023; Williams et al. 2023; Srivastava et al. 2024). Potential for PFAS transformation due to heat when cutting or coring does not appear to have been studied.

Cores can be collected using drilling or coring techniques that use PFAS-compatible materials (for example, stainless steel). Reused parts contacting the core should be decontaminated between each core collected. Once cores have been collected,

they should be transported to the laboratory in a PFAS-compatible container (for example, an HDPE bag) at 4°C.

Although cores are valuable, obtaining sufficient cores to understand horizontal spatial variability will rarely be practical. Researchers to date have reported collecting surface samples by creating dust with a handheld electric drill (Williams et al. 2023; Vo et al. 2023; Thai et al. 2025; Thai et al. 2025). This method has also been used for sample preparation for laboratory analysis (see Section 1.5.6.2).

Shipping, storage, and holding times should follow guidance for other solid matrixes, for example, USEPA Method 1633A, in the absence of guidance specific to concrete and asphalt.

Samples need to undergo a methanol solvent extraction for analysis, which requires the asphalt/concrete to be finely ground to ensure sufficient extraction from the solid phase. Methods such as drilling or crushing can be used to create asphalt/concrete powder that is then subjected to the methanol extraction process. To date there has not been a standardization on the drilling/crushing of asphalt/concrete. Variables such as particle size, drill hole depth, drill bit diameter, and drill speed could alter concentrations of PFAS that are obtained in the dust sample. Though no standardized method has been developed for sample collection, any drill bit or saw blade, for example, used to process the sample for extraction should be cleaned with methanol prior to sample collection.

Methanol extraction can be enhanced by the addition of an acid or base to improve extraction efficiencies of charged PFAS. Enhanced methanol extraction in addition to sequential extraction could benefit the analysis of more complex matrixes, such as asphalt.

Information about SERDP and ESTCP projects designed to establish standardized methods for sampling asphalt and concrete contaminated with PFAS can be found on their website (https://serdp-estcp.mil).

1.5.6.2 Sample Analysis

LC/MS/MS is the most common analytical technique used for the measurement of PFAS in environmental samples. Paired with solid-phase extraction (to remove matrix bias and improve detection limits) or centrifugation, this method has been used for analyzing PFAS in both asphalt and concrete samples (Baduel et al. 2015; Li et al. 2024; Liu et al. 2024; Srivastava et al. 2022). There are no specific instrumental analytical considerations for concrete and asphalt extracts, but proper sample preparation is crucial to obtain more accurate results.

1.5.6.3 Fate and Transport of PFAS in Concrete

The fate and transport of PFAS within concrete is important in understanding the extent of contamination that occurs within the matrix. Horizontal transport of PFAS from a source typically behaves similarly to the natural flow path of the concrete pad. Williams et al. (2023) studied a concrete pad from a fire training area where AFFF has been used. They found that concentrations of PFAS varied considerably within relatively short distances. Understanding the contaminant flow path may help in collecting spatially representative samples. Additionally, Williams et al. (2023) identified that PFAS was found to migrate within the vertical length of the core. Depth of PFAS impacts varied depending on the concrete material (which may affect diffusion/back diffusion), age of the material, bulk flow over the material (for example, stormwater), and the material's adsorption/desorption capacity. Additionally, transport of PFAS tended to be species dependent, for both horizontal and vertical movement (Williams et al. 2023).

The transport of PFAS from a concrete slab during precipitation events can indicate whether the matrixes serve as a secondary source of contamination. Thai et al. (2022) conducted a study of concrete from a fire training area where AFFF has been used to understand the effect of rainfall to release PFAS from the concrete. During successive rainfall simulations the similar concentrations of PFAS in the simulated runoff suggested the water evaporation between simulations may generate a potential wicking effect in the concrete bringing PFAS back to the surface (Thai et al. 2022). The PFAS can be released with the next stormwater runoff event, supporting the theory that concrete could serve as a secondary source of PFAS contamination. Ponding was another scenario that Thai et al. (2022) investigated, with regards to surface water impacts from PFAS-impacted concrete. Results indicated that an initial rapid release of PFAS concentrations occurred, while subsequent ponding events noted decreased concentrations (though similar concentrations between subsequent events). This would also support the potential for concrete to serve as a secondary source of contamination (Thai et al. 2022).

1.5.6.4 Leaching Testing of Concrete

To date, information regarding leachability of PFAS from concrete is limited, one study (Srivastava et al. 2024) is discussed below, and further research is required to develop and validate leaching tests under field conditions to fully assess leaching

behavior in concrete. However, it is a topic of interest because understanding leaching behavior can determine disposal needs, potential remediation methods, and reuse options.

Based on a 2024 study (Srivastava et al. 2024) some leaching characteristics and considerations were preliminarily determined. Using a modified LEAF 1315 method (USEPA 2017), three long-chain (PFOS, PFOA, and PFHxS) and one short-chain (PFHxA) compound were analyzed for their leaching behaviors. The study determined that, of the four constituents, PFOA was retained in the concrete, with relatively low leachability, while PFHxS readily leached out of the material. This highlights the variability in behavior based on species present.

Unlike samples analyzed in a powder form, leaching tests in this study (Srivastava et al. 2024) were conducted with quartered pucks extracted from a core collected in the field, resulting in a more complex surface (rather than a homogenized mixture). As such, leaching behavior was assumed to be driven by characteristics of the concrete being tested. For example, the age of the concrete was an important factor. As concrete ages, it experiences a net loss of alkalinity and cations via surface wicking (rainfall or other fluid distribution over the surface). This can influence adsorption and desorption within concrete pores. Development of secondary minerals over time may also decrease accessibility to pore space and adsorption/desorption potential of PFAS.

Similarly, van der Waals interactions may influence sorption/desorption of charged species to the surface of the concrete. Hydrophobic properties can also develop within the matrix and influence the ability for PFAS to bind to the matrix (Srivastava et al. 2024). The formation of micelles (colloidal particles with a hydrophobic core and a hydrophilic exterior), which has previously been observed in sediment matrixes, may also be a factor in leaching behavior for concrete given its porous nature (Srivastava et al. 2024). Further research is needed to fully understand and define these interactions, and resulting concentrations of PFAS in leachate produced from concrete. Better understanding of matrix interactions and characteristics that affect observed PFAS concentrations can help determine best practices for representative sample collection for this analysis method in the future.

1.5.7 Field Screening Tools

The goal of this section is to briefly summarize the development and availability of options for field screening for PFAS. Field screening tools are of interest because of the potential for portability (on-site use), ease of use, rapid results, and cost-effectiveness, all of which can improve the effectiveness and efficiency of PFAS analysis, and, ultimately, site characterization and remediation. However, there are currently no validated analytical methods that apply to field screening tools. This topic supplements Section 11.2, which addresses laboratory analytical methods/techniques.

Although not yet widely accepted by regulators, several PFAS screening approaches and tools are in various stages of development and validation (Erickson 2023; Menger et al. 2021), and some have recently been commercialized. SERDP released a Statement of Need in November 2024 titled *Real-Time Sensors for Detection and Quantitation of PFAS in Soil and Groundwater at AFFF-Impacted Sites* (SERDP 2024). More information about SERDP projects can be found on their website (https://serdp-estcp.mil).

Sensing technologies that have been evaluated in academic laboratories include those based on electrochemistry, optics, and nonconventional approaches, such as hydrogel swelling and thermal detection (Thompson, et al. 2024). Nanoparticle-based sensors, which may be made of metal-based nanoparticles, quantum dots, or carbon dots, can measure PFAS by detecting the optical or electrochemical changes that occur because of interaction with PFAS (Sendão et al. 2023). Different technology approaches provide different results, with some tools yielding total PFAS or TOF results and others reporting individual compounds, such as PFOS or PFOA, down to single-digit nanogram per liter levels for comparison to drinking water standards. As an alternative approach, fixed commercial laboratories are offering modified HPLC-MS/MS analyses, generally based upon ASTM D8241-25 (ASTM 2025), USEPA Method 537 (USEPA 2020), or USEPA Method 8327 (USEPA 2021), but perhaps omitting some sample preparation or quality assurance/quality control steps to achieve faster turnaround times, on the order of 1–10 days. Additionally, these streamlined analyses can be performed for a lower cost, as much as 50% reduced from full, National Environmental Laboratory Accreditation Program (NELAP)-certified analyses. It should be noted that screening tools, even when fully developed, properly validated, and commercialized, may not be approved by regulatory agencies for decision-making regarding site risks, comparison to standards, or assessment of remedial actions. It is likely that results will have to be confirmed with an accredited laboratory using a validated and published analytical method.

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