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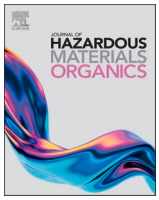

Occurrence of ultrashort-chain per- and polyfluoroalkyl substances in water samples from Ohio, Indiana, and Illinois

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ABSTRACT

Ultrashort-chain per- and polyfluoroalkyl substances (PFAS) are the most mobile and potentially the most persistent PFAS analogues. Although well known for their use in different industries, their environmental occurrence and impact still constitute knowledge gaps. This study analyzed sixteen PFAS, with emphasis on five ultrashort-chain PFAS, in samples collected from Ohio (rainwater, surface water, drinking water and wastewater effluent), Indiana (surface water), and Illinois (surface water). Trifluoroacetic acid (TFA) was found in all the samples: 91 ng L^{-1} in rainwater, 51–123 ng L^{-1} in drinking water, 41–315 ng L^{-1} in surface water, and 89–387 ng L^{-1} in wastewater effluent. Trifluoromethanesulfonic acid (TFMS) was quantified in three samples; the estimated concentration in wastewater effluent was 179 ng L^{-1}. Perfluoropropanoic acid (PFPrA) was detected in all samples (estimated concentration range 5–144 ng L^{-1}), except for rainwater. Overall, these three quantified ultrashort-chain PFAS are more prevalent and occurred at higher concentrations than longer-chain homologues (< 70 ng L^{-1}). Perfluoroethanesulfonic acid (PFEtS) and perfluoropropanesulfonic acid (PFPrS) were below the limit of quantification in all samples. This study contributed to filling the knowledge gap of the environmental occurrence of ultrashort-chain PFAS, for which available data are few, particularly in the United States. The results set the foundation for expanded analysis of ultrashort-chain PFAS across more extensive geographic regions and implied the necessity for environmental and health impact studies.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a class of more than 14,000 synthetic compounds widely applied in goods, industry, and aqueous film-forming foam (AFFF) [1]. PFAS are persistent and toxic chemicals, and their presence in the environment can lead to exposure for humans, animals, and plants, either directly or indirectly, through pathways such as drinking water, surface water, air, and soil [2–5]. The environmental and human health concerns of perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) drove major manufacturers to phase out these two PFAS of use/production in the early 2000's [6,7]. As a result, the short-chain homologues ($4 < \#C < 8$), which are more mobile in the environment than the long-chain species ($\#C \geq 8$) [3], have been used as replacements. Regulatory scrutiny is shifting to include shorter-chain species: recently, three short-chain PFAS, perfluorohexanesulfonic acid (PFHxS), perfluorobutanesulfonic acid (PFBS) and hexafluoropropylene oxide dimer acid (HFPO-DA) were included in the United States Environmental Protection Agency's National Primary Drinking Water Regulation [8]. Following this trend, questions regarding the ultrashort-chain PFAS ($\#C < 4$) emerge [9,10]; their occurrence, environmental chemistry, health impact, and treatability are poorly understood.

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Ultrashort-chain PFAS are potentially very persistent and very mobile (vP/vM) chemicals, with well-known utilizations. They have been used in laboratories (e.g., trifluoroacetic acid (TFA) as mobile phase modifier of liquid chromatography and for peptide synthesis), and in electronic and optical equipment manufacturing [11–13]; moreover, they have been reported as the transformation products of longer-chain PFAS and other fluorinated structures (e.g., hydrofluorocarbon refrigerants and CF_3-containing pesticides) that cannot further be easily degraded [13–15]. The few studies investigating the occurrence of ultrashort-chain PFAS reported consistently higher concentrations in comparison to short- and long-chain PFAS in water. A study conducted in Germany identified three ultrashort-chain PFAS concentrations accounting for 98 % of all quantified PFAS in drinking water sources [16]. The highest detected TFA concentration was 38 μg L^{-1} in rainwater from a study based in Munich, Germany [17]. Perfluoropropanesulfonic acid (PFPrS) was detected at concentrations as high as 8 μg L^{-1} in groundwater in Australia [18]. The flux of trifluoromethanesulfonic acid (TFMS) per snow precipitation event in the remote Artic locations in Norway was in the range of 1.5–57 ng m^{-2} with no seasonal variations [19]. Results pertaining to the occurrence of ultrashort-chain PFAS in the United States (U.S.) are emerging, but few. Perfluoropropanoic acid (PFPrA) was found at 0.45–6.52 ng L^{-1} in bottled water in the U.S. [20]. PFPrS was detected as high as 63 μg L^{-1} in groundwater close to military bases where AFFF was used during trainings [21]. Their occurrence in environmental water samples, drinking water, and wastewater remains a knowledge gap in the U.S. As an essential basis for future environmental studies on the ultrashort-chain PFAS, environmental occurrence data is urgently needed.

This study aims to provide one of the first datasets for the never-before-described occurrence of ultrashort-chain PFAS in water samples in the U.S. states of Ohio, Indiana and Illinois. Targeted analytes included five ultrashort-chain PFAS (TFA, PFPrA, TFMS, perfluoroethanesulfonic acid (PFEtS), and PFPrS), six short- and long-chain perfluorocarboxylic acids ($C_{4–9}$) and five short- and long-chain perfluorosulfonic acids ($C_{4–8}$). Samples included drinking water, river water, lake water, runoff water, rainwater and wastewater treatment plant (WWTP) effluent.

2. Experimental

2.1. Chemicals

Analytical standards and reagents used in this study are described in Table S1 and S2 of the Support Information (SI).

2.2. Sample collection

Samples from 16 sampling sites including drinking water (grab samples), river water, lake water, rainwater, runoff water and WWTP effluent (grab samples) were collected directly into 1 L high-density polyethylene (HDPE) bottles from June 2024 to December 2024 in the U.S. states of Ohio, Indiana and Illinois (Table 1 and Fig. 1). For certain locations (OH4–6 and OH7–8), samples were collected on different days at the same site to assess potential variations in PFAS concentrations over time. OH6 and OH8 were collected during the same rainfall event. A low-density polyethylene funnel was placed on top of an uncapped 1 L HDPE sampling bottle; they were used to collect rainwater on a balcony of a residential area.

2.3. Sample preparation

Each sample was filtered using 0.45 μm polyethersulfone (PES) membranes (28147–640; Pall Corporation, Port Washington, NY, U.S.) before apportioning two 250 mL aliquots for duplicate solid phase extraction (SPE) on a 20-position SPE manifold (WAT200607; Waters Corporation, Milford, U.S.). Prior to SPE, each portion was spiked with surrogates $^{13}C_3$-PFBA and $^{13}C_4$-PFOS, both at 200 ng L^{-1}. Oasis WAX cartridges (186009347, Waters Corporation, Milford, U.S.) were used for the SPE of target analytes and ^{13}C-labeled standards. Detailed SPE procedures are available in the SI (Text S1). After the final eluates were obtained and transferred into polypropylene autosampler vials, the samples were dried under nitrogen flow and shipped to the Arizona Laboratory of Emerging Contaminants (ALEC) for analysis. Immediately before instrumental analysis, samples were reconstituted with a methanolic solution containing the injection internal standards (Table S3).

Table 1
Acronyms, water types, weather conditions, and locations of samples analyzed.

Sample ID	Type of sample	Weather during sampling	Location
DW1	Drinking water	Dry	Cincinnati, OH
DW2	Drinking water	Dry	Cincinnati, OH
RW	Rainwater	Wet	Cincinnati, OH
RF	Runoff water	Wet	Cincinnati, OH
LM1	Lake water	Dry	Lake Michigan, Chicago, IL
LM2	Lake water	Dry	Lake Michigan, Gary, IN
LE	Lake water	Dry	Lake Erie, Cleveland, OH
IR	River water	Dry	Illinois River, Utica, IL
WbR	River water	Dry	Wabash River, West Lafayette, IN
WR	River water	Dry	White River, Indianapolis, IN
CR	River water	Dry	Cuyahoga River, Cleveland, OH
OH1	River water	Dry	Ohio River, Marietta, OH
OH2	River water	Dry	Ohio River, Washington, WV
OH3	River water	Dry	Ohio River, Cincinnati, OH
OH4	River water	Dry	Ohio River, Cincinnati, OH
OH5	River water	Dry	Ohio River, Cincinnati, OH
OH6	River water	Wet	Ohio River, Cincinnati, OH
OH7	River water	Dry	Ohio River, Cincinnati, OH
OH8	River water	Wet	Ohio River, Cincinnati, OH
OH9	River water	Dry	Ohio River, Rabbit Hash, KY
WW1	Wastewater treatment plant effluent	Dry	Cincinnati, OH
WW2	Wastewater treatment plant effluent	Dry	Cincinnati, OH

2.4. Sample analysis

An UltiMate 3000 Rapid Separation LC high pressure liquid chromatograph (Dionex, Sunnyvale, CA, U.S.) coupled to a Q-Exactive Focus Hybrid Quadrupole-Orbitrap mass spectrometer (Thermo Scientific, San Jose, CA, U.S.) was used for the analysis; a mixed-mode guard column and chromatographic column Obelisc N (SiELC Technologies, Wheeling, IL, U.S.) were employed to achieve separation of analytes. The analytical methods were previously developed by the authors and details are available elsewhere [22,23]. Skyline software 24.1.0.414 (Seattle, WA, U.S.) was used to process the data. Only peaks with average m/z error < 10 ppm were quantified. The water quality parameters of selected samples were analyzed and described in Text S2.

2.5. Method performance

Method performance validation procedures were available in Text S3; the results are listed in Tables S3, S4, and Fig. S1.

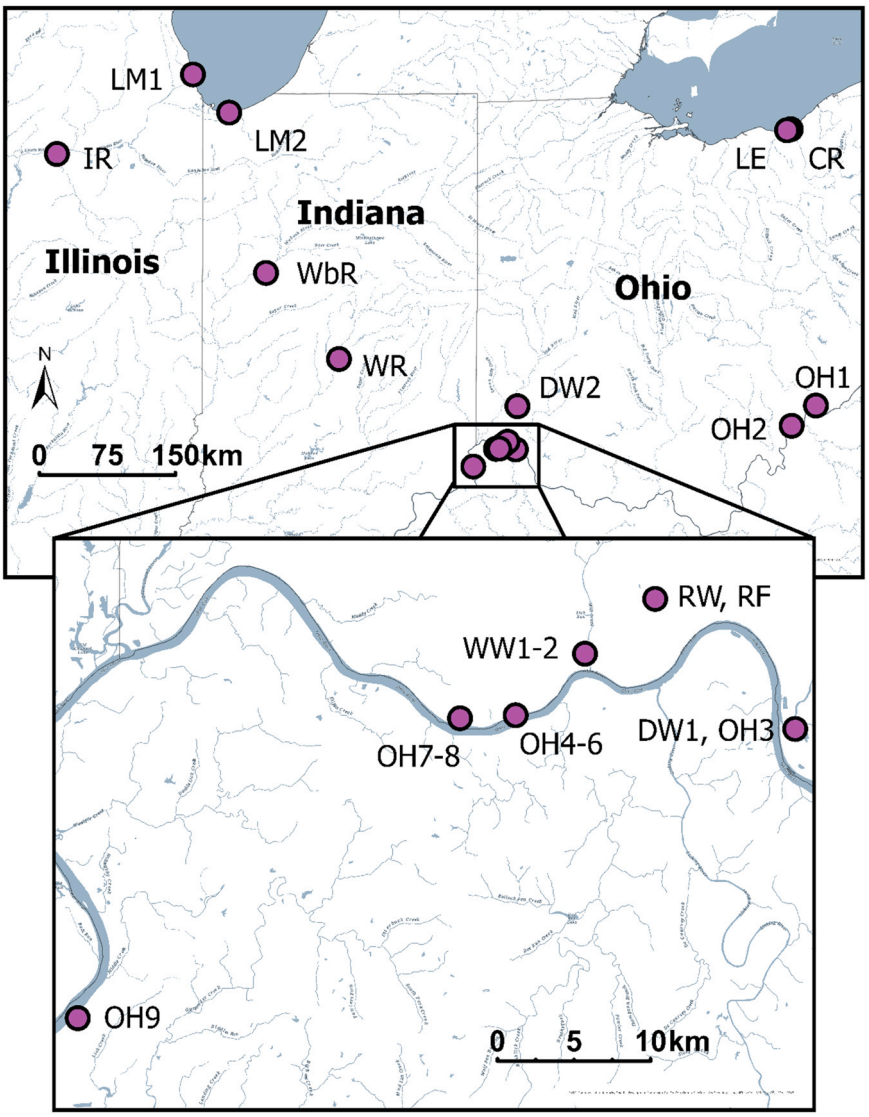

Fig. 1. Locations of water samples collected in this study (figure created with assistance of ArcGIS Pro 3.3.0).

2.6. Quality assurance and quality control (QA/QC)

To avoid cross contamination, sampling bottles were not reused. The selection of membranes to filter samples before SPE, syringe filter for the SPE extract, and autosampler vial was based on our previous study to guarantee accuracy of analysis [23]. Instrumental blanks (methanol) and quality control solutions (standard solutions in methanol) were injected every 10 sample injections to monitor possible carry-over effects and the stability of the instrumental system. The surrogates were used to evaluate the method recoveries for each sample. Laboratory Reagent Blank (LRB – ultrapure water) and Laboratory Fortified Blank (LFB – spiked ultrapure water) were processed with each batch of SPE. A Field Reagent Blank (FRB) was added to the rainwater sampling procedure to ensure no contamination was coming from the funnel used during sampling.

3. Results and discussion

3.1. Concentrations of ultrashort-chain PFAS in water samples

Among the five targeted ultrashort-chain PFAS, TFA and PFPrA were detected in all the samples above the concentrations found in LRB and FRB (except for the rainwater in which PFPrA concentration was below the FRB) (Figs. 2 and S2). With yet-unknown sources, previous studies have also quantified TFA and PFPrA in procedural blanks [4,24,25]. TFA presented the highest concentrations in the majority of the samples, varying from 41 ng L^{-1} in Cuyahoga River (CR) to 387 ng L^{-1} in the WWTP effluent (WW2). The concentrations of these ultrashort-chain PFAS were significantly higher than the short- and long-chain analogues (Figs. 2, 3, S3). The drinking water sample (DW1; 51 ng L^{-1}) was obtained immediately after the last step of treatment, i.e., finished water; the treatment plant employed filtration steps (granulated activated carbon, and rapid sand filtration) and UV disinfection. The value is similar to the average concentration (79 ng L^{-1}) previously reported for drinking water from Indiana, U.S. [25]. In comparison, Scheurer *et al.* detected TFA along drinking water treatment trains in southwest Germany at concentrations higher than 1 μg L^{-1} [26]. The high levels of TFA in finished water indicate the challenges in the complete removal of these smaller molecules in the current drinking water treatment plants. The tap water (DW2) had a concentration (123 ng L^{-1}) that is more than double the DW1 concentration, possibly attributed to the difference in water sources, difference in water treatment process and/or the date of sampling.

TFA was detected in the rainwater sample (RW; 91 ng L^{-1}), which could be ascribed to wet deposition of TFA formed after the degradation of hydrofluorocarbons in the atmosphere [17]. This concentration is

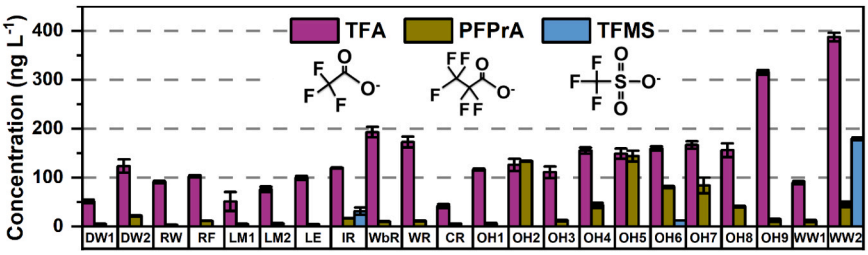

Fig. 2. Concentrations of ultrashort-chain PFAS in water samples from Ohio, Indiana and Illinois. PFEtS and PFPrS were lower than the limit of quantification in all the samples analyzed and therefore are not shown in the graph.

within the range of values (4–1200 ng L^{-1}) obtained for TFA in rainwater from another study conducted in Ohio and Indiana [27]. Freeling *et al.* reported even higher TFA concentration in meteoric waters in Germany (maximum concentration of 38 μg L^{-1}) [17]. Non-PFAS water quality parameters for runoff water (RF) collected in an urban/-residential area close to the rainwater sampling site during the same rainfall event were significantly different from RW, indicating the input of different components in the matrix due to the process of runoff (Table S5), but TFA concentrations in RW and RF were comparable, suggesting insignificant TFA contributions from roads, houses and automotive traffic. The Cuyahoga River sample (CR; 41 ng L^{-1}) collected in downtown Cleveland, i.e., before it flows into Lake Erie, showed lower TFA concentration than in Lake Erie (LE; 99 ng L^{-1}). A similar trend was reported for some lakes in the Western U.S.; a probable explanation is that TFA, existing in its deprotonated form, is stable and lacks volatility, which results in accumulation in lakes with no outflow [28]. Lake Michigan (LM1, LM2) (51 and 76 ng L^{-1}) showed lower concentrations than Lake Erie. Illinois River (IR; 119 ng L^{-1}) had similar concentrations to upstream Ohio River samples. Wabash River (WbR) and White River (*WR*) in Indiana showed relatively high concentrations (193 and 173 ng L^{-1}). Concentrations along the Ohio River were relatively constant (111–167 ng L^{-1}), except for OH9 (315 ng L^{-1}). Although changes in other water quality parameters (Table S5) between OH5 and OH6 and between OH7 and OH8 might indicate the influx of rainwater and runoff water to the Ohio River during a rainfall event, the concentrations of TFA remained at similar levels (Table S6). This is reasonable because TFA concentrations in rainwater, runoff, and river water were comparable (Fig. 2). Therefore, the influx of rainwater and runoff is not expected to cause substantial changes in TFA levels within the river. Notably, TFA concentrations in surface waters of this study were much lower than concentrations documented in Germany (300–12,400 ng L^{-1}) [16]. In WWTP effluent (WW1 and WW2), TFA concentrations showed discrepancies, varying from 89 to 387 ng L^{-1}. Since the WWTP effluents were grab samples that represented instantaneous concentrations, this variation may result from the temporal variation of influent sources to the WWTP, including possible pulsed input of non-domestic wastewater [29]. The presence of PFAS in quantifiable levels in drinking water and WWTP effluents agrees with studies that demonstrated the challenge of WWTP and potable water treatment plant removal of PFAS, especially the shorter and more hydrophilic homologues [30].

PFPrA presented relatively lower concentrations among all samples (5–16 ng L^{-1}), except in Ohio River samples (6–144 ng L^{-1}) and in a WWTP effluent sample (WW2; 44 ng L^{-1}). Comparable levels of PFPrA were found in German drinking water sources (13–179 ng L^{-1}), and in WWTP samples in the U.S. (6–11 ng L^{-1}) [16,31]. Unlike TFA that presented at relatively constant concentrations along the Ohio River, PFPrA levels fluctuated both across different sampling dates at the same site and among samples collected from different river locations: 43–144 ng L^{-1} for the same sampling site collected on three different days (OH4–6), 133 ng L^{-1} in a sample collected in Washington, WV (OH2), and the lowest concentration of 6 ng L^{-1} in an upstream location (OH1).

TFMS was less prevalent than TFA and PFPrA among samples analyzed in this study, it was only quantified in Illinois River (IR; 31 ng L^{-1}), Ohio River (OH6; 12 ng L^{-1}) and WWTP effluent (WW2; 179 ng L^{-1}). Without commercially available mass-labeled TFMS standard, $^{13}C_3$-PFBS was used as its internal standard, so TFMS quantification can be considered an estimate. Like the case for TFA, differences in TFMS concentrations in WWTP effluent samples can be explained by variation of influent received in the treatment plant. Jacob & Helbling also described TFMS in surface water (5 ng L^{-1}) and WWTP effluents (12–175 ng L^{-1}) [32]. In AFFF-impacted samples, TFMS was found in even higher concentrations at up to 3200 ng L^{-1} [33].

PFEtS and PFPrS results were below the LOQ in all samples of this study. These two sulfonic acids are associated with the use of AFFF and some specific industries [21,32]. They were reported at high concentrations in AFFF samples (PFEtS: 7–13 mg L^{-1}; PFPrS: 120–270 mg L^{-1}), in groundwater of U.S. military bases (PFEtS: 11–7500 ng L^{-1}; PFPrS: 19–63,000 ng L^{-1}), U.S. Department of Defense sites (PFEtS up to 25 μg L^{-1}; PFPrS up to 420 μg L^{-1}), and in industrial wastewater (PFEtS: 0.4–33 ng L^{-1}; PFPrS: 9 ng L^{-1}) [21,32,33]. To the best of our knowledge, no samples in this study had known impacts from AFFF or relevant industries.

The short- and long-chain PFAS concentrations were consistently lower than TFA, PFPrA and TFMS (Figs. 2, 3, and S3). Our results also agree with occurrence studies from other countries that identified concentrations of ultrashort-chain PFAS being significantly higher than the short- and long-chain PFAS [16]. PFBS, one of the replacements for long-chain PFAS, was prevalent, being quantified in 64 % of the samples. The highest PFBS concentration was found in WW1 at 70 ng L^{-1}, which was six times lower than the TFA concentration in the same sample (Fig. S3). Although the use of PFOA and PFOS was phased out, rooted in their persistent nature, they were still prevalently detected. PFOA was quantified in all the samples, except in WbR, W*R* and CR in which concentrations were below the respective LRB.

3.2. QA/QC evaluations

The recoveries for the quality control (QC) injections demonstrated the stability of the instrumental system throughout the analysis of all the samples (Fig. S4). Instrumental blank signals were all below the LOQ (except for PFPrS, PFHxS and PFOS), and LRB signal were all below the LOQ (except for TFA, PFPrA, and PFOA). TFA, PFPrA, PFOA and PFHxS had quantifiable concentrations in LRB and/or FRB (Fig. S2), consistent with previous studies also describing the challenges in obtaining contamination-free solvents and materials [4,24,25]. The LRB and FRB concentrations were not subtracted from the concentrations obtained for the samples. Recoveries of surrogates in SPE revealed good method accuracy (Fig. S5). A more realistic evaluation considering LRB and Laboratory Fortified Matrix (LFM) showed SPE recoveries in ultrapure water in the range of 61–164 % and in river water in the range of 10–96 % (Fig. S6), implying that true concentrations in real water matrix reported in this study could be considerably higher. Note that the trace-level analysis of ultrashort-chain PFAS has been challenging due to the lack of commercially available mass-labeled standards; as a result,

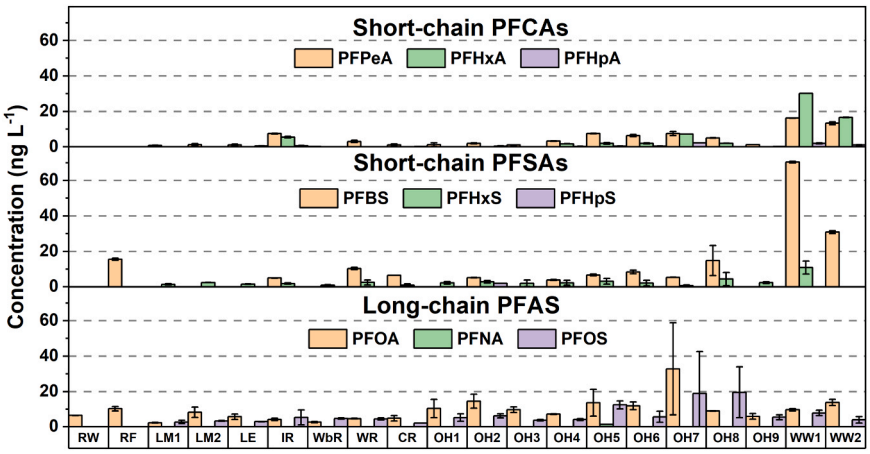

Fig. 3. Concentrations of short- and long-chain PFAS in water samples from Ohio, Indiana and Illinois. PFPeS and PFBA are not shown because they were lower than the limit of quantification in all the samples.

recent publications proceeded without the use of the respective surrogates and injection internal standards [16,18,24]. In this study, commercial $^{13}C_2$-TFA was only available after the SPE was performed. Despite the recognition that it would have been better used as a surrogate, its inclusion as an injection internal standard still represented a methodological improvement. As mass-labeled standards become available in the future, trace-level analysis of ultrashort-chain PFAS will be further improved. Although there might be concerns of possible loss of TFA owing to the use of heated nitrogen evaporator, other studies have already described TFA recoveries varying from 68 % to 95 % applying this technique [25,34]. Our LFB also showed excellent TFA recovery, suggesting little loss was incurred during the drying process.

4. Implications

As an emerging class of PFAS, it is essential to thoroughly characterize the environmental occurrence and distribution of ultrashort-chain PFAS prior to advancing research into their fate and transport, toxicity, and management. This study provides one of the first comprehensive datasets on the environmental occurrences of ultrashort-chain PFAS for above-ground water in the U.S. The high concentrations and widespread occurrence reported in this study are consistent with their high aqueous mobility and persistence. We suspect these PFAS species may occur ubiquitously across a broader range of water matrices and geographic regions, as reported in studies carried out in other countries [16,18,19]. Given the exceptional mobility and persistence of these PFAS species [35], if found to be environmentally hazardous, they may be the most challenging PFAS to manage. Their potential occurrence in source drinking water is especially concerning as existing treatment trains may struggle to remove them. Our results thus urge for investigation into expanded analysis of ultrashort-chain PFAS across a broader range of samples, followed by research into their fate and transport, toxicity, and treatability.

CRediT authorship contribution statement

Xi-Zhi Niu: Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Karen Noda Morishita:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation. **Alexander Pimenidis:** Investigation, Formal analysis. **Leif Abrell:** Writing – review & editing, Validation, Software, Methodology, Investigation, Formal analysis, Data curation. **Jon Chorover:** Writing – review & editing, Resources, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.hazmo.2025.100006.

Data availability

Data will be made available on request.

References

[1] United States Environmental Protection Agency, 2022. CompTox Chemicals Dashboard v2.5.3, PFAS|EPA: PFAS structures in DSSTox (update August 2022).
[2] R.A. Dickman, D.S. Aga, A review of recent studies on toxicity, sequestration, and degradation of per- and polyfluoroalkyl substances (PFAS), J. Hazard Mater. 436 (2022) 129120, https://doi.org/10.1016/j.jhazmat.2022.129120.
[3] Y. Wang, U. Munir, Q. Huang, Occurrence of per- and polyfluoroalkyl substances (PFAS) in soil: sources, fate, and remediation, Soil Environ. Health 1 (2023) 100004, https://doi.org/10.1016/j.seh.2023.100004.
[4] M. Sadia, I. Nollen, R. Helmus, T.L. Ter Laak, F. Béen, A. Praetorius, A.P. Van Wezel, Occurrence, fate, and related health risks of PFAS in raw and produced drinking water, Environ. Sci. Technol. 57 (2023) 3062–3074, https://doi.org/10.1021/acs.est.2c06015.

[5] M.E. Morales-McDevitt, J. Becanova, A. Blum, T.A. Bruton, S. Vojta, M. Woodward, R. Lohmann, The air that we breathe: neutral and volatile PFAS in indoor air, Environ. Sci. Technol. Lett. 8 (2021) 897–902, https://doi.org/10.1021/acs.estlett.1c00481.
[6] United States Environmental Protection Agency. Fact Sheet: 2010/2015 PFOA Stewardship Program. Assess. Manag. Chem. TSCA. ⟨https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/fact-sheet-20102015-pfoa-stewardship-program⟩. Accessed 15 Apr 2025.
[7] M. Land, C.A. De Wit, A. Bignert, I.T. Cousins, D. Herzke, J.H. Johansson, J. W. Martin, What is the effect of phasing out long-chain per- and polyfluoroalkyl substances on the concentrations of perfluoroalkyl acids and their precursors in the environment? A systematic review, Environ. Evid. 7 (2018) 4, https://doi.org/10.1186/s13750-017-0114-y.
[8] Environmental Protection Agency, 2024. PFAS National Primary Drinking Water Regulation 40 CFR Parts 141 and 142 [EPA-HQ-OW-2022-0114; FRL 8543-02-OW] RIN 2040-AG18.
[9] H.P.H. Arp, A. Gredelj, J. Glüge, M. Scheringer, I.T. Cousins, The global threat from the irreversible accumulation of trifluoroacetic acid (TFA), Environ. Sci. Technol. 58 (2024) 19925–19935, https://doi.org/10.1021/acs.est.4c06189.
[10] X. Lim, There's a new acid in our rain — should we be worried? Nature 643 (2025) 894–897, https://doi.org/10.1038/d41586-025-02259-6.
[11] European Chemicals Agency. Substance Inforcard - Trifluoromethanesulphonic acid.
[12] European Chemicals Agency. Substance Inforcard - Trifluoroacetic acid.
[13] Y. Zhi, X. Lu, G. Munoz, L.W.Y. Yeung, A.O. De Silva, S. Hao, H. He, Y. Jia, C. P. Higgins, C. Zhang, Environmental occurrence and biotic concentrations of Ultrashort-Chain perfluoroalkyl acids: overlooked global organofluorine contaminants, Environ. Sci. Technol. 58 (2024) 21393–21410, https://doi.org/10.1021/acs.est.4c04453.
[14] H. Joerss, F. Freeling, S. Van Leeuwen, J. Hollender, X. Liu, K. Nödler, Z. Wang, B. Yu, D. Zahn, G. Sigmund, Pesticides can be a substantial source of trifluoroacetate (TFA) to water resources, Environ. Int 193 (2024) 109061, https://doi.org/10.1016/j.envint.2024.109061.
[15] Z. Fan, M.N. Nadagouda, X.-Z. Niu, M.J. Bentel, Micelle-Mediated hydrated electron capture enables efficient C–F bond activation and PFAS mineralization, Environ. Sci. Technol. 59 (2025) 16764–16774, https://doi.org/10.1021/acs.est.5c04621.
[16] I.J. Neuwald, D. Hübner, H.L. Wiegand, V. Valkov, U. Borchers, K. Nödler, M. Scheurer, S.E. Hale, H.P.H. Arp, D. Zahn, Ultra-Short-Chain PFASs in the sources of German drinking water: prevalent, overlooked, difficult to remove, and unregulated, Environ. Sci. Technol. 56 (2022) 6380–6390, https://doi.org/10.1021/acs.est.1c07949.
[17] F. Freeling, D. Behringer, F. Heydel, M. Scheurer, T.A. Ternes, K. Nödler, Trifluoroacetate in precipitation: deriving a benchmark data set, Environ. Sci. Technol. 54 (2020) 11210–11219, https://doi.org/10.1021/acs.est.0c02910.
[18] S. Ghorbani Gorji, R. Mackie, P. Prasad, E.R. Knight, X. Qu, S. Vardy, K. Bowles, C. P. Higgins, K.V. Thomas, S.L. Kaserzon, Occurrence of Ultrashort-Chain PFASs in Australian environmental water samples, Environ. Sci. Technol. Lett. 11 (2024) 1362–1369, https://doi.org/10.1021/acs.estlett.4c00750.
[19] M.K. Björnsdotter, W.F. Hartz, R. Kallenborn, I. Ericson Jogsten, J.D. Humby, A. Kärrman, L.W.Y. Yeung, Levels and seasonal trends of C_1 –C_4 perfluoroalkyl acids and the discovery of trifluoromethane sulfonic acid in surface snow in the Arctic, Environ. Sci. Technol. 55 (2021) 15853–15861, https://doi.org/10.1021/acs.est.1c04776.
[20] S.J. Chow, N. Ojeda, J.G. Jacangelo, K.J. Schwab, Detection of ultrashort-chain and other per- and polyfluoroalkyl substances (PFAS) in U.S. Bottled water, Water Res. 201 (2021) 117292, https://doi.org/10.1016/j.watres.2021.117292.
[21] K.A. Barzen-Hanson, J.A. Field, Discovery and implications of C_2 and C_3 perfluoroalkyl sulfonates in aqueous Film-Forming foams and groundwater, Environ. Sci. Technol. Lett. 2 (2015) 95–99, https://doi.org/10.1021/acs.estlett.5b00049.
[22] X.-Z. Niu, L. Abrell, R. Sierra-Alvarez, J.A. Field, J. Chorover, Analysis of hydrophilic per- and polyfluorinated sulfonates including trifluoromethanesulfonate using solid phase extraction and mixed-mode liquid chromatography-tandem mass spectrometry, J. Chromatogr. A 1664 (2022) 462817, https://doi.org/10.1016/j.chroma.2022.462817.
[23] K. Noda Morishita, H. Lee, C. Han, X.-Z. Niu, Syringe filters and autosampler vials impact on analysis of Long-, Short-, and Ultrashort-Chain Per- and polyfluoroalkyl substances, ACS EST Water 5 (2025) 1344–1352, https://doi.org/10.1021/acsestwater.4c01090.
[24] S.-H. Liang, J.A. Steimling, M. Chang, Analysis of ultrashort-chain and short-chain (C1 to C4) per- and polyfluorinated substances in potable and non-potable waters, J. Chromatogr. Open 4 (2023) 100098, https://doi.org/10.1016/j.jcoa.2023.100098.
[25] G. Zheng, S.M. Eick, A. Salamova, Elevated levels of Ultrashort- and Short-Chain perfluoroalkyl acids in US homes and people, Environ. Sci. Technol. 57 (2023) 15782–15793, https://doi.org/10.1021/acs.est.2c06715.
[26] M. Scheurer, K. Nödler, F. Freeling, J. Janda, O. Happel, M. Riegel, U. Müller, F. R. Storck, M. Fleig, F.T. Lange, A. Brunsch, H.-J. Brauch, Small, mobile, persistent: trifluoroacetate in the water cycle – overlooked sources, pathways, and consequences for drinking water supply, Water Res. 126 (2017) 460–471, https://doi.org/10.1016/j.watres.2017.09.045.
[27] K.A. Pike, P.L. Edmiston, J.J. Morrison, J.A. Faust, Correlation analysis of perfluoroalkyl substances in regional U.S. Precipitation events, Water Res 190 (2021) 116685, https://doi.org/10.1016/j.watres.2020.116685.
[28] T.M. Cahill, Assessment of potential accumulation of trifluoroacetate in terminal lakes, Environ. Sci. Technol. 58 (2024) 2966–2972, https://doi.org/10.1021/acs.est.3c08822.
[29] Metropolitan Sewer District of Greater Cincinnati. 2025 Permitted Facilities with Issued Pretreatment or Waste Hauler Permits (Including Case Manager).
[30] Y. Peng, W. Hu, X.-Z. Niu, Per-and polyfluoroalkyl substances removal in water and wastewater treatment plants: overall efficiency and performance of adsorption, Environ. Res Commun. 6 (2024) 092002, https://doi.org/10.1088/2515-7620/ad75ea.
[31] J. Kim, X. Xin, B.T. Mamo, G.L. Hawkins, K. Li, Y. Chen, Q. Huang, C.-H. Huang, Occurrence and fate of Ultrashort-Chain and other Per- and polyfluoroalkyl substances (PFAS) in wastewater treatment plants, ACS EST Water 2 (2022) 1380–1390, https://doi.org/10.1021/acsestwater.2c00135.
[32] P. Jacob, D.E. Helbling, Rapid and simultaneous quantification of Short- and Ultrashort-Chain perfluoroalkyl substances in water and wastewater, ACS EST Water 3 (2023) 118–128, https://doi.org/10.1021/acsestwater.2c00446.
[33] C. Zhang, S. Hao, N. Gonda, Y. Zhi, T.J. Strathmann, C.E. Schaefer, C.P. Higgins, Quantification of long–chain, short–chain, and ultrashort–chain liquid chromatography–amenable PFASs in water: evaluation of approaches and tradeoffs for AFFF–impacted water, J. Hazard Mater. 466 (2024) 133591, https://doi.org/10.1016/j.jhazmat.2024.133591.
[34] G. Xie, J. Cui, Z. Zhai, J. Zhang, Distribution characteristics of trifluoroacetic acid in the environments surrounding fluorochemical production plants in jinan, China, Environ. Sci. Pollut. Res. 27 (2020) 983–991, https://doi.org/10.1007/s11356-019-06689-4.
[35] T. Reemtsma, U. Berger, H.P.H. Arp, H. Gallard, T.P. Knepper, M. Neumann, J. B. Quintana, P.D. Voogt, Mind the gap: persistent and mobile organic Compounds—Water contaminants that slip through, Environ. Sci. Technol. 50 (2016) 10308–10315, https://doi.org/10.1021/acs.est.6b03338.