Research Square

Preprints are preliminary reports that have not undergone peer review.
They should not be considered conclusive, used to inform clinical practice,
or referenced by the media as validated information.

Unidentified fluorine and TFA dominate the fluorine mass balance in European drinking water

Ian Cousins

ian.cousins@aces.su.se

Department of Environmental Science, Stockholm University https://orcid.org/0000-0002-7035-8660

Zongzhe He

Fudan University

Xing Chen

Fudan University

Bénilde Bonnefille

Stockholm University https://orcid.org/0000-0001-5141-7111

Jonathan Zweigle

University of Copenhagen

Merle Plassmann

Stokcholm University

Anna Roos

Greenland Institute of Natural Resources

Mohammad Sadia

Mid Sweden University https://orcid.org/0000-0003-2797-8757

Håkon Langberg

Akvaplan niva AS

Rita Binetti

SMAT Research Centre Laboratories

Tony Venelinov

University of Architecture, Civil Engineering and Geodesy https://orcid.org/0000-0002-4394-6096

Martin Cornelsen

Cornelsen Umwelttechnologie GmbH

Juan Santos

Universidad de Sevilla

Andreas Kaiser

Environment Agency Austria

Dzintars Zacs

Animal Health and Environment “BIOR”

Adrian de la Torre

CIEMAT
Mélanie Ourgaud
University Perpignan Via Domitia
Dvorakova Darina
University of Chemistry and Technology
Alexander Haluska
University of Tübingen
Hanna Joerss
Helmholtz-Zentrum Hereon
Paweł Struciński
National Institute of Public Health NIH
Daniel Zahn
Helmholtz Centre for Environmental Research - UFZ https://orcid.org/0000-0003-3038-9371
Stoimir Kolarevic
University of Belgrade
Roesch Philipp
German Environment Agency
Rosie Williams
Zoological Society of London
Marco Parolini
University of Milan
Mingliang Fang
Fudan University https://orcid.org/0000-0002-2204-9783
Jonathan P. Benskin
Stockholm University

Article

Keywords: Extractable fluorine, Inorganic fluorine, PFAS Total, Trifluoroacetic acid, Tap water

Posted Date: July 1st, 2026

DOI: https://doi.org/10.21203/rs.3.rs-10104926/v1

License: This work is licensed under a Creative Commons Attribution 4.0 International License.
Read Full License

Additional Declarations: There is **NO** Competing Interest.

Unidentified fluorine and TFA dominate the fluorine mass balance in European drinking water

Zongzhe He[1,2,#,*], Xing Chen[1,#], Bénilde Bonnefille[3], Jonathan Zweigle[4], Merle Plassmann[2], Anna Roos[5], Mohammad Sadia[6], Håkon Austad Langberg[7], Rita Binetti[8], Tony Venelinov[9], Martin Cornelsen[10], Juan Luis Santos[11], Andreas Marius Kaiser[12], Dzintars Zacs[13], Adrian de la Torre[14], Mélanie Ourgaud[15], Dvorakova Darina[16], Alexander Haluska[17], Hanna Joerss[18], Paweł Struciński[19], Daniel Zahn[20], Stoimir Kolarevic[21], Roesch Philipp[22], Rosie Williams[23], Marco Parolini[24], Mingliang Fang[1,*], Ian T. Cousins[2,*], Jonathan P. Benskin[2,*]

[1]Department of Environmental Science and Engineering, Fudan University, Shanghai, 200438, China

[2]Department of Environmental Science, Contaminant Chemistry Unit, Stockholm University, 10691 Stockholm, Sweden

[3]Department of Environmental Science, Exposure and Effects Unit, Science for Life Laboratory, Stockholm University, Stockholm 10691, Sweden

[4]Analytical Chemistry Group, Department of Plant and Environmental Science, Faculty of Science, University of Copenhagen, Thorvaldsensvej 40, 1871, Frederiksberg C, Denmark

[5]Greenland Institute of Natural Resources, Kivioq 2, PO Box 570, 3900, Nuuk, Greenland

[6]Department of Natural Science, Design and Sustainable Development (NDH), Mid Sweden University, SE-85170, Sundsvall, Sweden

[7]Akvaplan niva AS, Havnegata 9, Trondheim 7010, Norway

[8]SMAT Research Centre Laboratories, Corso Unità d'Italia 235/3 10127, Torino, Italy

[9]Faculty of Hydraulic Engineering, University of Architecture, Civil Engineering and Geodesy, 1046, Sofia, Bulgaria

[10]Cornelsen Umwelttechnologie GmbH, Graf-Beust-Allee 33, D-45141, Essen, Germany

[11]Departamento de Química Analítica, Escuela Politécnica Superior, Universidad de Sevilla, E–41011, Seville, Spain

[12]Environment Agency Austria, Spittelauer Lände 5, A-1090, Vienna, Austria

[13]Institute of Food Safety, Animal Health and Environment "BIOR", Lejupes iela 3, Riga, LV-1076, Latvia

[14]Unit of POPs and Emerging Pollutants in the Environment, CIEMAT, Madrid, Spain

[15]University Perpignan Via Domitia, CNRS, CEFREM, UMR 5110, Perpignan, France

[16]University of Chemistry and Technology (UCT), Prague, Faculty of Food and Biochemical Technology, Department of Food Analysis and Nutrition, Technicka 5, 166 28, Prague, Czech Republic

[17]Department of Geosciences, University of Tübingen, Schnarrenbergstraße 94-96, 72076, Tübingen, Germany

[18]Department of Organic Environmental Chemistry, Helmholtz-Zentrum Hereon, Geesthacht, 21502, Germany

[19]Department of Food Safety and Health Risk Assessment, National Institute of Public Health NIH – National Research Institute, Chocimska 24, 00-791, Warsaw, Poland

[20]Helmholtz Centre for Environmental Research - UFZ, Permoserstrasse 15, 04318, Leipzig, Germany

[21]Institute for Biological Research "Siniša Stanković" - National Institute of the Republic of Serbia, University of Belgrade, Bulevar despota Stefana 142, 11060, Belgrade, Serbia

[22]German Environment Agency, Waste Water Analysis and Monitoring Methods, Schichauweg 58, 12307, Berlin, Germany

[23]Institute of Zoology, Zoological Society of London, Regent's Park, London NW1 4RY, U.K.

[24]University of Milan, Department of Environmental Science and Policy, via Celoria 26, I-20133, Milan, Italy

[#]Z.H. and X.C. contributed equally to this work.

[*]To whom correspondence may be addressed. Email: hezz@fudan.edu.cn, mlfang@fudan.edu.cn, Ian.Cousins@aces.su.se, or Jon.Benskin@aces.su.se

Abstract

Human exposure to per- and polyfluoroalkyl substances (PFAS) via drinking water is of concern and likely underestimated because current monitoring frameworks cover only a fraction of PFAS. Here we provide a comprehensive assessment of PFAS contamination by applying a fluorine mass balance (FMB) approach to tap water from 34 European cities across 18 countries, quantifying ultra-short-, short-, and long-chain PFAS, inorganic fluorine (IF) species, and unidentified extractable fluorine (UEF). Trifluoroacetic acid (TFA) was the most abundant PFAS (detection frequency [DF] 95%; median 45.7 ng L^{-1}; range 8.29–300 ng L^{-1}). Three PFAS regulated under the Stockholm Convention, perfluorooctanoic acid (PFOA; DF 83%; median 0.767 ng L^{-1}), perfluorohexanesulfonic acid (PFHxS; DF 90%; median 0.432 ng L^{-1}), and perfluorooctanesulfonic acid (PFOS; DF 73%; median 1.15 ng L^{-1}), remain ubiquitous. Recovery-adjusted PFAS Total concentrations ranged from 42.0–4020 ng L^{-1} and exceeded the 500 ng L^{-1} threshold in ten cities, affecting approximately 27 million inhabitants. FMB results showed that >60% of extractable fluorine (EF) remained unidentified in most sampled cities, highlighting a knowledge gap in current PFAS monitoring. Importantly, concentration-based metrics did not fully reflect toxicological risk, with 21 of 34 cities exceeding the potency-equivalent risk quotient benchmark, primarily attributed to PFOS and perfluorononanoic acid (PFNA). These findings underscore the need to complement EF-based screening with compositional analyses

and potency-based approaches to support evidence-driven drinking water safety standards.

Keywords: Extractable fluorine; Inorganic fluorine; PFAS Total; Trifluoroacetic acid; Tap water.

Introduction

Safe drinking water is foundational to public health and recognized as a human right under United Nations General Assembly Resolution 64/292[1]. However, contamination by per- and polyfluoroalkyl substances (PFAS) poses a persistent challenge to drinking water safety worldwide[2-4]. Water serves as a major environmental reservoir for PFAS due to their exceptional persistence and mobility in aquatic systems[5]. Conventional drinking water treatment processes, including coagulation, sedimentation, and biological filtration, are largely ineffective at removing PFAS. Even with advanced treatment technologies such as activated carbon filtration and ion-exchange resins, short- and ultra-short-chain polar PFAS remain poorly removed and therefore frequently detected in finished tap water[6-9]. In Europe, tap water is supplied for direct consumption; residual PFAS in finished water therefore represents a direct route of population exposure and a potential public-health concern[10,11].

Concerns over PFAS in drinking water have grown as advances in analytical chemistry have revealed far greater structural diversity than previously recognized. As legacy PFAS such as perfluorooctanoic acid (PFOA), perfluorooctanesulfonic acid (PFOS), and perfluorohexanesulfonic acid (PFHxS) have been phased out under the Stockholm Convention on Persistent Organic Pollutants (POPs)[12], numerous substitutes have entered use, such as hexafluoropropylene oxide dimer acid (HFPO-DA) and chlorinated polyfluoroalkyl ether sulfonates such as F-53B[13-15]. With >10,000 PFAS currently in commercial use and/or detected in the environment[16], both regulatory oversight and

analytical monitoring face substantial scalability challenges. Conventional targeted analytical methods that identify and quantify individual substances cannot keep pace with the expanding recognition of PFAS used in commerce[17-19], while substance-by-substance regulation invites ongoing cycles of regrettable substitution, as restricted compounds are replaced by structurally similar alternatives[20].

To address this challenge, the revised EU Drinking Water Directive (Directive (EU) 2020/2184) introduced a regulatory limit of 500 ng L^{-1} for "PFAS Total", defined as the totality of PFAS in drinking water[21]. This legislation will enter the implementation phase in 2026. Measurement of extractable fluorine (EF) by combustion ion chromatography (CIC) has been proposed as an enforcement method for this limit, providing a sum-parameter approach that captures both known and unknown PFAS without requiring individual substance identification[22]. Despite this regulatory advancement, no systematic surveys of PFAS Total in European drinking water have been conducted to date, and the composition of PFAS Total remains poorly characterized.

We hypothesize that PFAS contamination in European drinking water has been systematically underestimated by conventional monitoring strategies, and consequently that the risks to human health remain poorly understood. To address this, we carried out a fluorine mass balance (FMB) analysis of drinking water consumed by approximately 37 million residents across 34 European cities (~5% of the European population), including regions lacking prior monitoring data. Our objectives were to:

(1) quantify EF and target PFAS, including ultra-short, short, and long-chain substances; (2) determine the fraction of unidentified extractable fluorine (UEF) in each city and its contribution to the FMB; (3) evaluate PFAS Total concentrations relative to the EU regulatory limit of 500 ng L^{-1}; and (4) assess the potential health risk associated with PFAS in drinking water. These results provide critical baseline data to support implementation of European PFAS regulations and underscore the need for advanced analytical approaches to ensure drinking water safety.

Results and Discussion

Widespread occurrence of PFAS in European drinking water

A total of 40 fluorinated substances were monitored in 41 tap water samples collected from 34 cities in 18 European countries (Supplementary Fig. 1 and Supplementary Table 1). Because our study largely relies on single grab samples, the findings represent a snapshot in time and are not intended to evaluate regulatory compliance or the safety of individual drinking water supplies. Full compound names and abbreviations are provided in Supplementary Table 2. Targets included 14 perfluoroalkyl carboxylic acids (PFCAs, C2–C15), 10 perfluoroalkane sulfonic acids (PFSAs, C1–C10), 4 perfluorooctane sulfonamides (FOSAs, namely FOSAA, FOSA, MeFOSAA and EtFOSAA), 3 fluorotelomer sulfonic acids (FTSAs; namely 4:2 FTS, 6:2 FTS and 8:2 FTS), 4 fluorinated alternatives (HFPO-DA, DONA, 9Cl-PF3ONS and 11Cl-PF3OUdS), 3 fluorotelomer carboxylic acids (FTCAs; namely FPrPA, FPePA, and

FHpPA) and 2 inorganic fluorine (IF) species (BF_4^- and PF_6^-). Among these, 27 fluorinated compounds were quantified above the limit of quantification (LOQ) in at least one of the 41 tap water samples (Supplementary Table 3). Individual PFAS concentrations varied widely (0.006–300 ng L^{-1}), spanning nearly five orders of magnitude (Fig. 1a,b and Supplementary Table 4), with higher abundance compounds generally detected more frequently.

To assess how representative our grab samples were, we examined temporal variability in Stockholm (two samples collected from the same tap one year apart) and replicate variability in Berlin and Marseille (three samples collected at the same time from the same tap). Most PFAS and EF measurements were broadly consistent across these comparisons, although several low-abundance targets showed greater variability (e.g., PFNA, PFHpS, and FPePA), as expected near quantification limits[23] (Supplementary Text 4 and Supplementary Table 5). With the exception of compounds near the LOQ, temporal and replicate variability remained below 23% and 39% relative standard deviations (RSD), respectively, for both EF and target PFAS, indicating relatively low fluctuations over time and across repeated measurements. These results suggest that, in the absence of significant short-term contamination events and within the constraints of our sampling design, our "snapshot" of Europe provides a reasonable approximation of PFAS exposure over the timescales examined, with good repeatability at individual sampling locations.

Legacy compounds such as PFOA (median 0.767 ng L^{-1}; range 0.193–14.1 ng L^{-1}), PFHxS (0.432 ng L^{-1}; 0.0617–8.84 ng L^{-1}), and PFOS (1.15 ng L^{-1}; 0.0864–7.67 ng L^{-1}) were ubiquitous but contributed only a minor fraction to Σ_{38}PFAS in European drinking water. PFAS with even longer chains (e.g., PFDA, PFUnDA, PFDoDA, 11Cl-PF3OUdS) generally occurred at sub-ng L^{-1} concentrations. This legacy PFAS burden is substantially lower than that reported in the United States and China, where an estimated 18–80 million people receive tap water with combined PFOA and PFOS ≥10 ng L^{-1} [24-26]. This difference likely reflects stricter regulatory frameworks in Europe, where comprehensive restrictions on legacy PFAS and other POPs have contributed to the protection of drinking water supplies. However, industrial point sources remain problematic. Our data showed relatively high concentrations of C5-C8 PFCAs in the Lyon-2 sample, a pattern consistent with profiles previously reported in regions influenced by fluoropolymer manufacturing, which is known to occur in the industrial area south Lyon[27,28].

Several emerging PFAS (4:2 FTS, 6:2 FTS, EtFOSAA, FPePA, FHpPA, and 11Cl-PF3OUdS) were detected at trace levels, suggesting either that pollution control measures for these alternatives are currently effective or that they have not been emitted in sufficient quantities over time for levels to accumulate. Despite the generally low exposure to PFAS alternatives in our dataset, localized risks remain near manufacturing sites, as evidenced by periodically elevated HFPO-DA levels reported in the Netherlands[29].

Short- and ultrashort-chain PFAS, including PFCAs (PFPrA 5.91 ng L^{-1}, PFBA 3.23 ng L^{-1}, PFPeA 3.64 ng L^{-1}, PFHxA 1.23 ng L^{-1}; [median]) and PFSAs (TFMS 1.29 ng L^{-1}, PFBS 1.37 ng L^{-1}; [median]), were detected in more than 50% of samples, highlighting their widespread occurrence in European tap water. TFA was frequently the most dominant compound, with a median of 45.7 ng L^{-1} (range 8.29–300 ng L^{-1}), and was the primary contributor to Σ_{38}PFAS in European drinking water. This finding aligns with previous global studies reporting that median TFA concentrations in drinking water are typically one to two orders of magnitude higher than those of other PFAS, although European levels appear lower than the global median (~160 ng L^{-1})[30-32]. Notably, TFA was not detected in the Copenhagen and Nuuk-2 samples, which suggests limited influence from anthropogenic sources[31]. The common presence of TFA at other cities likely reflects multiple emission sources, including industrial discharges, transformation products from pharmaceuticals/agrochemicals, and atmospheric deposition from refrigerant degradation[31].

IF anions PF_6^- and BF_4^- were inevitably co-extracted with ultra-short chain PFAS, with detection frequencies of 49% and 39%, and concentration ranges of 2.00–107 ng L^{-1} (PF_6^-) and 1.78–560 ng L^{-1} (BF_4^-), respectively. Limited studies have reported elevated environmental levels of these anions in drinking water from Germany and China[33,34]. The high detection frequencies and concentrations suggest that PF_6^- and BF_4^- occur ubiquitously in European tap water samples[35]. Their presence may be associated with

industrial activities and emissions from electrolyte usage, highlighting the need for further investigation into potential health implications[36].

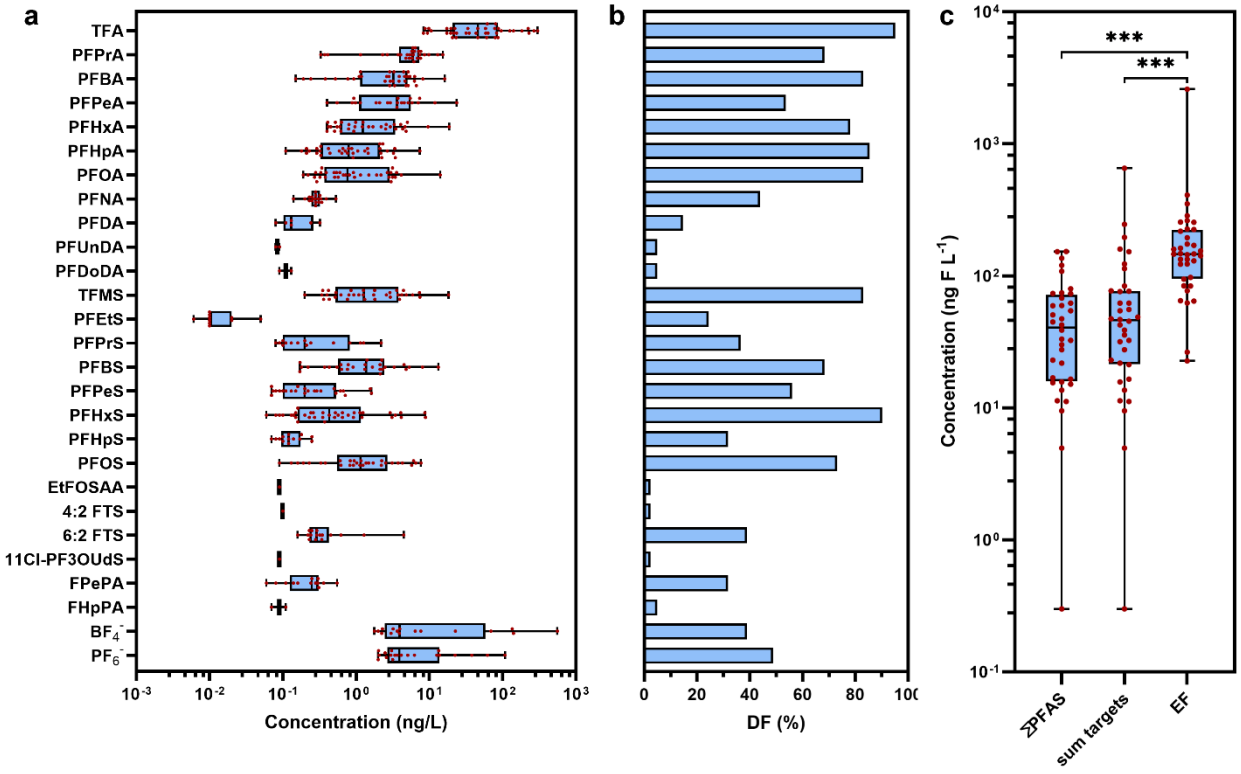


Fig.1 | Concentrations, detection frequencies, and fluorine mass balance of fluorinated analytes quantified in European tap water. Shown are the 27 analytes quantified above the limit of quantification (LOQ) in at least one sample. a, Boxplots of individual compound concentrations (ng L^{-1}), with red dots representing individual samples. The central line denotes the median, boxes represent the interquartile range (IQR), and whiskers indicate the minimum and maximum values. b, Detection frequencies (DF, %) of each compound across all samples. c, Comparison of fluorine-equivalent concentrations (ng F L^{-1}) of targeted PFAS (Σ_{38}PFAS), total target fluorine

(sum targets), and EF across samples. Statistical significance was assessed using paired Wilcoxon signed-rank test; *$p < 0.05$, ***$p < 0.001$.

Most EF in European tap water is unidentified

Fluorine equivalent concentrations (ng F L^{-1}) of summed targets (i.e., Σ_{38}PFAS + PF_6^- + BF_4^-) were significantly lower than EF across European tap water samples (Fig, 1c, $p<0.001$), with medians of 42.4 ng F L^{-1} (range: 0.302–655 ng F L^{-1}) versus 145 ng F L^{-1} (range: 22.7–2590 ng F L^{-1}), respectively (Supplementary Table 6). The considerable difference between these values points to high levels of UEF (median 98.9 ng F L^{-1}; range: 13.2–1930 ng F L^{-1}), accounting for >60% of EF in most sampled cities. This UEF cannot be attributed to fluoride ions, which are effectively removed during extraction[23]. Potential contributors include unidentified PFAS but also organic and inorganic substances that fall outside the OECD PFAS definition (e.g. certain pharmaceuticals and pesticides). Next to UEF (median 69%), TFA accounted for the second highest fraction of EF in most cities (17%; Fig. 2a, Supplementary Fig. 2, and Supplementary Table 7). Other short-chain PFCAs were a consistent secondary component (median 4%), followed by trace-level PFSAs (median <2%). FOSAs, FTSAs, FTCAs, and fluorinated alternatives were generally below 1% in most samples, but locally elevated in some sites. IF species (BF_4^- and PF_6^-) were typically minor (median <2%) but exhibited high variability among cities (maximum 53%).

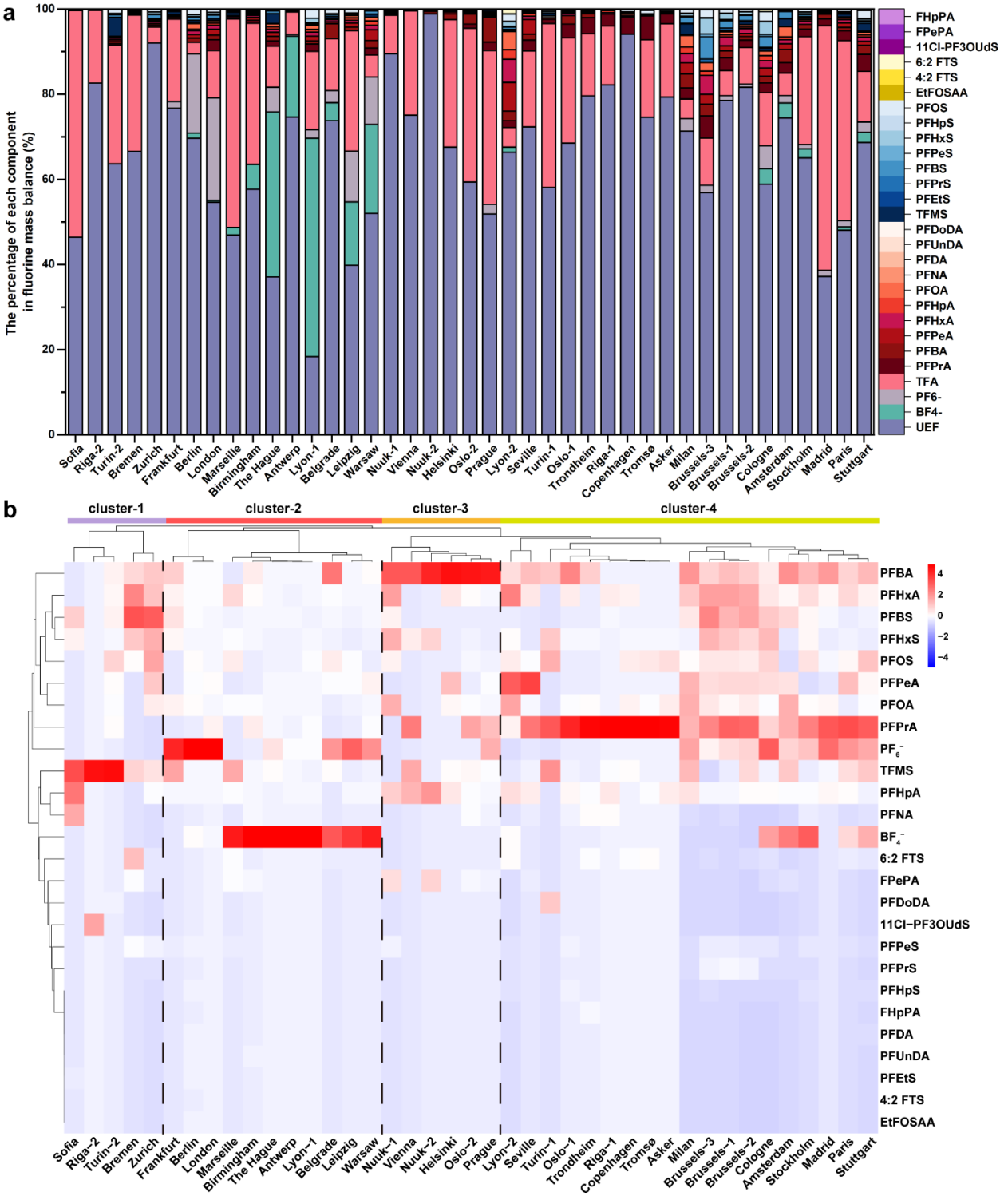


Fig.2 | Compositional distribution and clustering of PFAS and inorganic fluorinated substances in European tap water. a, Stacked bar plots showing the relative contribution of identified PFAS, UEF, and IF (BF_4^- and PF_6^-) across 41

European tap water sampling cities, normalized to 100% on a fluorine-equivalent basis. b, Hierarchical clustering heatmap based on normalized PFAS composition (UEF and TFA excluded), with column-wise normalization to highlight relative enrichment of individual PFAS across sites. Warmer colors indicate higher relative abundance within each city. Dashed lines delineate the four composition-based clusters, which are labeled as Cluster-1 to -4.

Geographically disparate cities can have similar patterns of contamination

We classified European drinking water samples into four major groups based on hierarchical clustering of normalized PFAS profiles (Fig. 3 and Supplementary Table 8; Fig. 2b). Cluster-1, containing Sofia, Riga-2, Turin-2, Bremen, and Zurich, was characterized by the occurrence of the ultra-short-chain short-chain perfluoroalkyl acids (PFAA), TFMS, and for some samples, PFBS. Most samples in this cluster contained relatively low levels of TFA and EF, with the exception of Sofia, which displayed elevated levels of both, despite containing some of the lowest concentrations of longer chain length target PFAS.

Cluster-2 included cities from western and central Europe, and was characterized by the occurrence PF_6^- and/or BF_4^-. Many of the cities with the highest EF burdens appeared in this cluster, including Leipzig, London, The Hague, Lyon-1, and Antwerp, where EF concentrations frequently exceeded 200 ng F L^{-1}. The elevated EF burden was largely attributed to UEF (66.9-1930 ng F L^{-1}; Fig. 3), consistent with contributions

from multiple anthropogenic sources of fluorinated compounds. The most notable city in Cluster-2 was Antwerp, which exhibited the highest EF value in our dataset (EF: 2,590 ng F L^{-1}), driven largely by UEF (1,930 ng F L^{-1}). This site highlights the importance of including EF measurements along with targeted analyses: based on target analysis alone, the elevated TFA (135 ng F L^{-1}) and trace levels of legacy PFAS, would classify Antwerp as moderately contaminated. EF determination, however, revealed that targeted analytes accounted for only 25% of total EF, leaving 1,930 ng F L^{-1} unidentified (i.e., UEF). Previous studies in Antwerp have not reported PFAS contamination at magnitudes comparable to those observed in our study for both drinking water and human serum despite known nearby industrial sources[37,38]. However, a non-targeted analysis focused on wastewater in Antwerp identified high-concentration discharges of $C_8H_3F_{11}O_2$ and $C_{10}H_3F_{15}O_2$[39], which may be among the source-related candidates driving UEF. This discrepancy highlights the potential of EF for flagging samples containing novel fluorinated contaminants, which would otherwise be missed by targeted analysis.

Cluster-3 contained several Nordic samples (e.g., Nuuk-1 and 2, Helsinki, and Oslo-2) together with samples from central Europe (Vienna, Prague), and was characterized by the occurrence of PFBA. Overall, the Nordic samples in this cluster exhibited relatively low EF burdens. For example, Nuuk-2, collected in downtown Nuuk, showed an EF level of 26.4 ng F L^{-1}, representing a background level with limited anthropogenic influence. In contrast, despite exhibiting a broadly similar PFAS pattern, Nuuk-1

(sampled from the airport during construction) showed a substantially elevated EF burden (130 ng F L^{-1}), primarily driven by UEF (116 ng F L^{-1}). Local observations documented open waste burning near the drinking water supply at Kangerlussuaq (Greenland's former international airport), which may have contributed to the elevated UEF levels observed at this site[40].

Cluster-4 was characterized by the prevalence of short-chain PFAS, in particular PFPrA. Cities appearing in Cluster-4 were geographically diverse, ranging from the Nordic region (e.g. Tromsø, Oslo-1, Trondheim), central Europe (e.g. Brussels-1 and -2, Amsterdam, Paris), and southern Europe (e.g. Milan, Madrid, Seville). Among these cities, Lyon-2 was particularly notable due to its complex and site-specific contamination and exhibited the highest 6:2 FTS concentrations among all European cities surveyed (2.57 ng F L^{-1}), along with elevated PFCAs (PFPeA: 15.4 ng F L^{-1}, PFHxA: 12.4 ng F L^{-1}, PFHpA: 5.04 ng F L^{-1}, PFOA: 9.70 ng F L^{-1}). The chemical industry in Lyon has undergone a significant evolution in its PFAS manufacturing history, with the production of polychlorotrifluoroethylene (PCTFE, using perfluoroalkyl telomers) in the late 1950s, polytetrafluoroethylene (PTFE, using PFOA) in the 1960s, or polyvinylidene fluoride (PVDF, using 6:2 FTS, PFNA, PFunDA, PFtrDA, and PFOA) from the 1970s[27,28]. This led to the discharge of PFAS into the Rhône River over time, with recent reports of 6:2 FTS at ppm-level concentrations released in its waters until September 2022[27,41]. In addition, a 1987 port fire involving around 200 m^3 aqueous film-forming foam (AFFF) use contaminated local

groundwater[28]. The elevated concentration of 6:2 FTS in Lyon's drinking water (2.57 ng F L^{-1}) likely reflects both contamination in the source water and limited removal during treatment. Indeed, comparable 6:2 FTS concentrations were detected in Lyon source water and finished drinking water[28], indicating the known difficulty in removing certain short-chain PFAA during treatment. Recent studies have reported poor removal of 6:2 FTS by advanced treatment technologies such as anion exchange resins[42,43], and similarly low EF removal (<25%) has been observed even with advanced treatment[44]. These lines of evidence support prioritizing upstream source control (i.e., preventing industrial fluorinated discharges to surface waters) over relying on treatment alone for persistent fluorinated compounds.

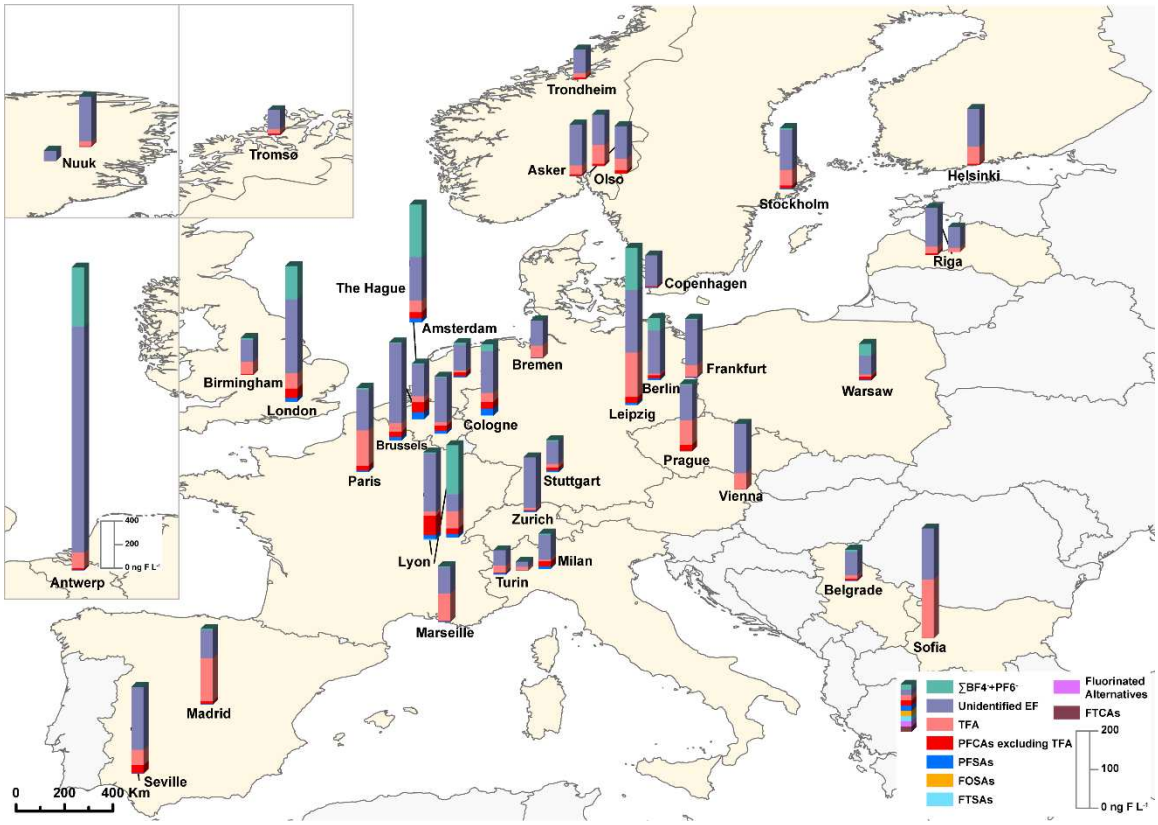


Fig.3 | Spatial distribution of EF in European tap water. Map of Europe showing EF concentrations (ng F L^{-1}) across 41 tap water sampling sites. Stacked bars indicate the absolute fluorine contributions from TFA, PFCAs excluding TFA, PFSAs, FOSAs, FTSAs, fluorinated alternatives, FTCAs, UEF, and IF species (BF_4^- and PF_6^-). Given the markedly higher total EF in Antwerp, its stacked bar is displayed using a separate bar-height scale in the inset panel. Countries with sampling sites are highlighted.

Fractionation offers clues to the nature of UEF

Our SPE method employed a polarity-based fractionation strategy to separate most ultra-short-chain (Fraction I) from longer chain length (Fraction II) PFAS, improving EF estimation and UEF accuracy[23]. A closer investigation of the relative distribution of UEF between fractions (Supplementary Table 9) therefore offers insight into the physical chemical properties of the UEF. In most samples, UEF was detected in both fractions, with slightly higher UEF contributions in Fraction II, suggesting a broad polarity range in European drinking water. For example, Leipzig exhibited substantial UEF in both fractions (Fraction I: 75.3 ng F L^{-1}; Fraction II: 88.2 ng F L^{-1}), suggesting the occurrence of both mobile and more strongly retained fluorinated compounds. Antwerp represented a notable exception, with UEF predominantly distributed in Fraction I (Supplementary Table 9). This pattern suggests a comparatively greater contribution from highly polar and mobile fluorinated compounds at this site.

UEF drives EU PFAS Total exceedances not captured by targeted Σ_{38}PFAS analysis

Given that fluoride removal during extraction caused substantial losses of ultra-short-chain PFAS (approximately half), recovery correction was applied using analyte-specific recovery determined from QC samples to obtain recovery-corrected PFAS concentrations for each city.

PFAS regulations have evolved toward increasing stringency and broader scope[45,46]. This evolving regulatory landscape can lead to different conclusions about drinking water quality, as countries regulate different PFAS, apply different grouping strategies, and set varying threshold values. Several European countries have established stricter limits for the sum of four PFAS (Σ_4PFAS: PFOA, PFNA, PFHxS, and PFOS) in drinking water, relative to the EU, which is based on 20 substances. As shown in Supplementary Fig. 3 and Supplementary Table 10, A comparison of recovery-corrected PFAS data (Supplementary Text 3) to guideline levels revealed that the frequency of exceedance was highly dependent on the regulatory threshold applied. Under the most stringent Danish Σ_4PFAS limit of 2 ng L^{-1}, 15 European cities were in exceedance, whereas under the less stringent Italian and German guidance value of 20 ng L^{-1} (effective in 2026 and 2028, respectively)[47,48], only Lyon (26.8 ng L^{-1}) was in exceedance. Although all samples from the 34 cities were below the EU's Sum of PFAS limit of 100 ng L^{-1} (Fig. 4a and Supplementary Table 10), this comparison represents a

lower-bound estimate because PFUnDS, PFDoDS, and PFTrDS were not included due to analytical constraints.

The aforementioned Σ_4PFAS and Σ_{20}PFAS limits are suitable for flagging drinking water contaminated with known PFAA, but fall short in addressing contamination by other PFAS, both known and unknown. To address the potential for targeted limits to overlook certain substances, Directive (EU) 2020/2184 introduced a class-based approach, establishing a 500 ng L^{-1} limit for PFAS Total[22], which is defined as the totality of PFAS. After converting our results to this regulatory metric (Supplementary Text 5), PFAS Total ranged from 42.0 to 4020 ng L^{-1} and exceeded the 500 ng L^{-1} threshold in 10 cities (Fig. 4b). Targeted PFAS monitoring accounted for only 2.6–88% of PFAS Total, with a median contribution of 55% (Supplementary Table 11). In Antwerp, Sofia, Leipzig, Madrid, and Paris, TFA alone approaches or exceeds the regulatory limit, disproportionately contributing to the FMB. Among the 37 million people supplied by the sampled cities, approximately 27 million (71%) were exposed to drinking water with potential PFAS Total exceedance after recovery correction (for results without recovery correction, see Supplementary Fig. 4,5).

Correlation analyses (Supplementary Text 6 and Supplementary Fig.6) further contextualize these findings. PFCAs and PFSAs exhibit strong intra- and inter-class correlations, suggesting common industrial sources (e.g., fluoropolymer manufacturing, AFFF) and similar removal inefficiencies in drinking-water treatment. In contrast, TFA

shows no correlation with legacy PFAS, suggesting that TFA originates from multiple contamination pathways that differ from those captured by legacy PFAS[31].

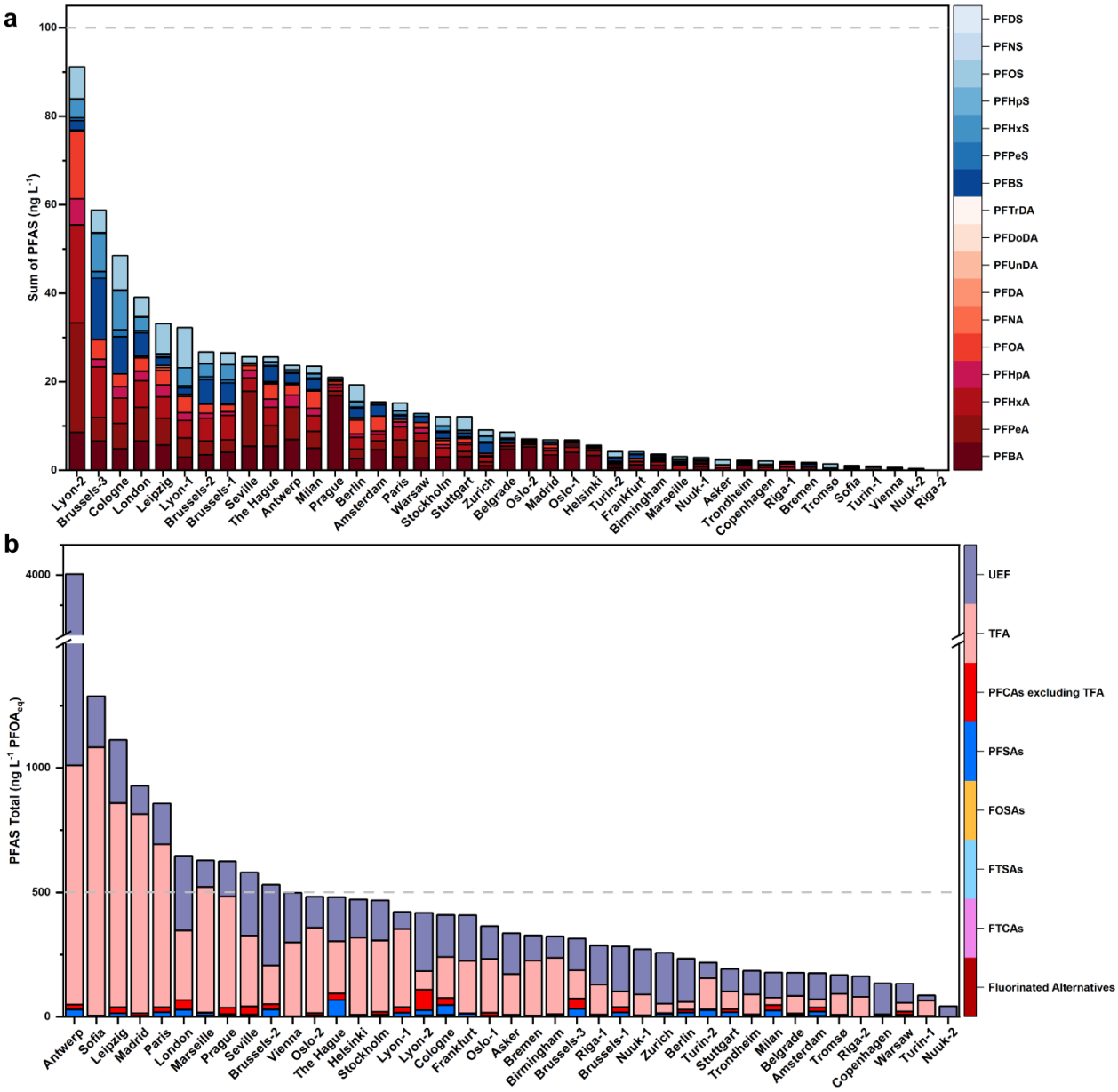


Fig.4 | Concentration-based regulatory metrics of PFAS in European tap water. Stacked bar plots illustrate city-level PFAS mixtures under two concentration-based regulatory metrics: a, Sum of PFAS; b, PFAS Total. The Sum of PFAS metric was calculated using 17 of the 20 PFAS listed in the EU Drinking Water Directive (EU)

2020/2184 (PFUnDS, PFDoDS and PFTrDS) were not included due to analytical constraints. Horizontal dashed lines indicate the EU reference levels of 100 ng L^{-1} (Sum of PFAS) and 500 ng L^{-1} (PFAS Total). All concentrations shown are recovery corrected.

Potency-weighted assessment highlights widespread PFAS mixture risk in European drinking water

Potency-equivalent risk quotient (RQ_{peq}) was evaluated by weighing each recovery-corrected PFAS according to its relative potency factor (RPF) referenced to PFOA, and by comparing the resulting mixture potency with drinking-water health-based guidance derived from the European Food Safety Authority (EFSA) scientific opinion. As shown in Fig. 5 and Supplementary Table 12, 21 of 34 cites exceeded this benchmark, indicating that PFAS mixtures in European drinking water frequently surpass the precautionary risk threshold. The highest RQ_{peq} value was observed at Lyon–2 (≈11), primarily driven by elevated PFOS and PFOA, which together contributed more than 60% of total PFOA-equivalents. Several western and central European cities including Leipzig, Lyon–1, Cologne, Brussels–3, London, and Berlin, showed pronounced exceedances (RQ_{peq} = 4-8). A second group showed moderate exceedances (RQ_{peq} = 1-4), comprising 18 samples from 16 cities, including The Hague, Milan, Antwerp, Brussels–1, Seville, Stuttgart, and Zurich. In contrast, northern and eastern sites (e.g., Tromsø, Warsaw, and Belgrade) generally exhibited $RQ_{peq} < 1$, consistent with their

lower total PFAS concentrations and reduced contribution from high-potency compounds. PFOS (RPF = 2) and PFNA (RPF = 10) were the dominant toxicological drivers across all samples, together defining the majority of potency-weighted burdens in nearly every region. PFOA and PFHpA (both RPF = 1) contributed consistently as secondary components, while the importance of PFHxS (RPF = 0.6) was highly site-specific (e.g., Brussels–3 and Cologne). As shown in Supplementary Fig. 7, results without recovery correction were comparable to recovery-corrected results, although four fewer cities exceeded the RPF benchmark without correction.

Comparison of potency-weighted outcomes with EF burdens revealed a clear decoupling between concentration-based metrics and toxicological risk. Antwerp exhibited the highest EF (2,590 ng F L^{-1}) but a moderate RQ_{peq} of approximately 3. In contrast, Lyon-2 showed an 11-fold lower EF (226 ng F L^{-1}) yet a 3.5-fold higher toxicological risk. These differences reflect mixture composition rather than total fluorine load. TFA accounts for approximately 34% of median EF yet contributes less than 1% to potency-weighted risk due to its low RPF (0.002). This result should be interpreted with caution, as TFA is extremely persistent and highly mobile[31], and a recent ECHA Risk Assessment Committee opinion recommended its classification as toxic to reproduction Category 1B, as well as PMT and vPvM[49]. These developments suggest that current mixture-risk frameworks may underestimate the toxicological relevance of TFA, and future revisions of TFA toxicity values could alter, and potentially increase, its relative importance in PFAS mixture-risk assessment.The UEF

fraction remains toxicologically uncharacterized and therefore cannot be meaningfully included in potency-weighted calculations. Notably, UEF was predominantly associated with Fraction I, suggesting that much of the unidentified fluorine consists of highly polar, ultra-short-chain substances, with limited bioaccumulation potential. However, their toxicological relevance remains uncertain. In contrast, toxicological risk is disproportionately determined by a small subset of high-potency PFAS concentrated in Fraction II. For example, in Lyon-2, PFOS, PFNA, and PFOA collectively represented only 17% of EF but accounted for more than 95% of PFOA-equivalent burden (Supplementary Table 12). The fractionation-based approach employed in this study enables differentiation of toxicologically relevant PFAS (Fraction II: longer-chain, less polar, higher bioaccumulation potential) from lower toxicological potency species (Fraction I: ultra-short-chain, highly polar, lower toxicity). Our results show that compliance with EU thresholds (Sum of PFAS = 100 ng L^{-1}; PFAS Total = 500 ng L^{-1}) does not guarantee acceptable toxicological risk when high-potency compounds dominate the mixture. Conversely, elevated total EF may overstate health risks when the mixture is dominated by TFA or other low-potency species.

In conclusion, EF-based monitoring and RQ_{peq} provide complementary insights into PFAS exposure and risk. EF-based monitoring can capture both known PFAS and UEF fractions, whereas RQ_{peq} provides a potency-weighted assessment for PFAS with available toxicological data. Therefore, EF-based monitoring should be combined with target and non-target analysis, ideally coupled with fractionation, to better identify the

composition of UEF and its contribution to overall toxicity. However, elevated IF concentrations are not included in RQ_{peq} because they are not classified as PFAS and currently lack health-based toxicological benchmarks, leaving a remaining gap in risk characterization. Effect-directed analysis combined with further fractionation could help identify the observed biological effects and clarify the contribution of UEF to overall toxicity. These limitations in chemical characterization and toxicological data have important implications for water management, as many fluorinated compounds are difficult to remove from drinking water, placing the burden on suppliers despite their limited control over contamination. This underscores the importance of source-control measures to prevent PFAS release and limit their accumulation in the water cycle.

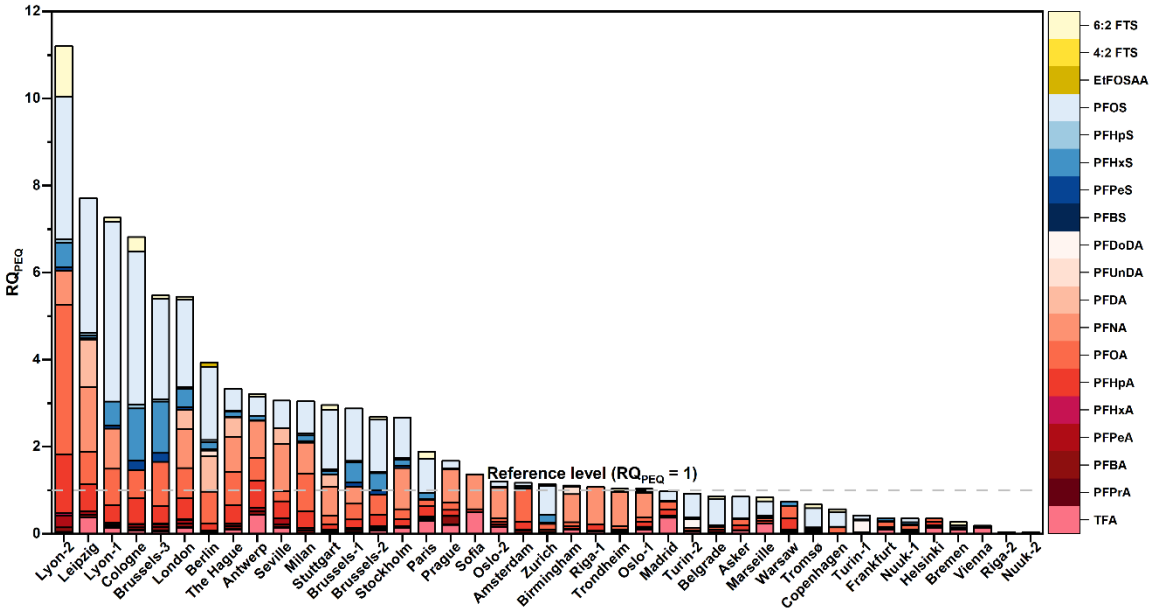


Fig.5 | RQ_{peq} of PFAS mixtures in European tap water. Stacked bars display the site-specific RQ_{peq} values across 41 European tap water sampling sites, arranged in descending order. Each bar segment represents the proportional contribution of

individual PFAS to the total PFOA-equivalent burden. The dashed horizontal line denotes the health-based reference level (RQ_{peq} = 1). Colors correspond to PFAS chemical classes as indicated in the legend. All data shown are recovery corrected.

Methods

Sampling and sample handling

Our sampling strategy aimed at obtaining drinking water samples representative of the largest possible number of consumers. To this end, major European cities were initially prioritized for sampling based on population size and number of drinking water treatment plants, using the population served per treatment plant as an indicator of potential consumer coverage (Supplementary Table 1). Cities that could not be sampled were subsequently excluded from the final sampling plan. Ultimately, tap water was collected from 34 cities across 18 European countries (Supplementary Fig. 1). The sampling campaign was carried out between January 2022 and January 2023, predominantly during October-December 2022, with most sites sampled once. Sampling was performed by collaborating partners following a standardized protocol (Supplementary Text 1) to minimize contamination, including requirements for sampling location selection, sampler clothing restrictions, and field blank collection. All samples were stored at -20°C prior to analysis and were analyzed unfiltered.

Sample extraction was performed using a previously validated WAX SPE protocol, which produces two fractions[23]. Extracts were analyzed for target PFAS by UHPLC-

MS/MS using dual chromatographic methods (anion exchange for short- and ultra-short-chain compounds; reversed-phase C18 for long-chain PFAS) and for EF by CIC, following our previously reported protocols[23]. Complete analytical details and QA/QC procedures are provided in the Supporting Information (Supplementary Text 2 and Text 3).

Data analysis

PFAS concentrations (in units of ng L^{-1}) determined by UHPLC-MS/MS were converted to fluorine equivalent concentrations (i.e., ng F L^{-1}) using the weight percentage of fluorine in each substance. The UEF, reported as fluorine equivalents (ng F L^{-1}), was converted to PFOA mass equivalents (ng F L^{-1}) because the Directive expresses PFAS Total in mass units rather than fluorine units, and PFOA is used as the reference compound for unidentified species captured by fluorine-based measurements[22].

After fractionation, the EF value measured by EF_{CIC} was determined using Eq 1.

$$\textbf{Eq 1.}\quad EF_{CIC} = EF_{CIC,\ FrI} + EF_{CIC,\ FrII}$$

Where $EF_{CIC,\ FrI}$ and $EF_{CIC,\ FrII}$ represent the fluorine concentrations determined by CIC in fractions I and II, respectively.

The IF concentration (IF) was defined according to Eq 2.

$$\textbf{Eq 2.}\quad IF = IF_{LC-MS,\ FrI} + IF_{LC-MS,\ FrII}$$

Where $IF_{LC\text{-}MS,\ FrI}$ and $IF_{LC\text{-}MS,\ FrII}$ represent the sum concentration (in fluorine equivalents) of inorganic fluorinated substances (i.e., BF_4^- and PF_6^-) in Fraction I and II, respectively.

The EF without interferences of BF_4^- and PF_6^-, which served as a proxy for PFAS Total, and was calculated using Eq 3.

Eq 3. $$EF = \max\left(0, EF_{CIC,\ FrI} - IF_{LC-MS,\ FrI}\right) + \max\left(0, EF_{CIC,\ FrII} - IF_{LC-MS,\ FrII}\right)$$

Finally, the unidentified extractable fluorine (UEF) determined using Eq 4:

Eq 4. $$UEF = \max\left(0, EF_{CIC,\ FrI} - EF_{LC-MS,\ FrI}\right) + \max\left(0, EF_{CIC,\ FrII} - EF_{LC-MS,\ FrII}\right)$$

Where $EF_{LC\text{-}MS,\ FrI}$ and $EF_{LC\text{-}MS,\ FrII}$ represent the EF concentrations determined from UHPLC-MS/MS analysis of fractions I and II, respectively.

The presence of "n.d." indicates certain substances were not detected in the samples. For "<LOQ" values, these substances were detected but their concentrations were below the quantification limit and therefore not quantified. In both cases, these values were excluded from statistical analyses.

Hierarchical clustering analysis

Hierarchical clustering was applied to explore compositional similarities among European tap water samples based on fluorine mass–normalized PFAS profiles. The analysis was performed after normalization to EF (set as 100%), excluding TFA and

UEF. Missing values ("n.d." or "<LOQ") were excluded during computation. Bidirectional clustering of samples and compounds was conducted using the *correlation* distance metric and *average* linkage method, and the clustered heatmap was generated in R (version 4.5.1).

Geographic patterns of EF across Europe

The spatial distribution and composition of EF across Europe were visualized using ArcGIS Pro 3.4.3 (Esri, USA). All geographic layers were projected to WGS 1984, and sampling coordinates were imported from Supplementary Table 1. Each city was georeferenced according to its latitude and longitude, with a stacked bar chart placed at the corresponding map location to represent the EF composition.

Potency-weighted risk calculation

The potency-weighted risk quotient (RQ_{peq}) of PFAS mixtures in European tap water was assessed using the RPF framework outlined by the Dutch National Institute for Public Health and the Environment (RIVM) in its knowledge note "Werkwijze toetsing PFAS in drinkwatermonsters" [50,51]. For each measured PFAS i, its concentration (C_i, ng L^{-1}) was multiplied by the corresponding RPF_i to express its contribution as PFOA-equivalents (PEQ_i) using Eq 5.

$$\textbf{Eq 5.}\ PEQ_i = C_i \times RPF_i$$

The summed PFOA-equivalents for each sample were then computed using Eq 6.

$$\textbf{Eq 6.}\ \sum PEQ = \sum (C_i \times RPF_i)$$

∑PEQ values were directly compared to the drinking-water benchmark value of 4.4 ng PFOA-eq L^{-1}, which corresponds to the EFSA TWI of 4.4 ng PFOA-eq kg^{-1} bw wk^{-1} established for the sum of PFOA, PFNA, PFOS, and PFHxS, converted to a water-based threshold for human exposure[52,53].

The resulting risk quotient was expressed according to Eq 7.

$$\textbf{Eq 7.}\ \mathrm{RQ_{peq}} = \frac{\sum \mathrm{PEQ}}{4.4\ \mathrm{ng\ L^{-1}}}$$

Using the RPF framework, where each PFAS is assigned a potency value relative to PFOA (RPF = 1), mixture toxicity was expressed as PFOA-equivalent, and the corresponding risk quotient (RQ_{peq}) was calculated with a threshold of 1, representing the benchmark for acceptable PFAS chronic exposure. RQ_{peq}>1 indicates exceedance of the health-based guidance level. Only RPFs derived from direct oral exposure (drinking-water ingestion) were applied, as recommended by RIVM. RPF values were taken from the latest RIVM database (https://www.rivm.nl/pfas/rpf). Not all measured PFAS have assigned RPFs; therefore, PEQ and RQ_{peq} were calculated only for compounds with available RPFs.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (22506019). We acknowledge Richard Sempéré (Ocean Sciences Institute), the MIO marine chemistry technical platform for their support, as well as the European Regional Development Fund (MIO-Project 1166-39417). This work was supported by the

Ministry of Science, Technological Development and Innovations of the Republic of Serbia, grants No. 451-03-33/2026-03/ 200007. The content of this study is in line with following the United Nations (UN) Sustainable Development Goals, under the frame of the UN Agenda for Sustainable Development (Transforming our World: The 2030 Agenda for Sustainable Development – UN, 2015): Good health and well-being and Clean water and sanitation. ITC acknowledges funding from the EU Horizon 2020 research and innovation program (project ZeroPM, grant agreement no. 101036756).

Notes

The authors declare no conflicts of interest.

Data availability

All data are included in the manuscript and/or the Supplementary Information.

References

1. Assembly, U.G. The human right to water and sanitation: resolution / adopted by the general assembly. 3 (United Nations, New York, 2010).
2. Tokranov, A.K.*, et al.* Predictions of groundwater PFAS occurrence at drinking water supply depths in the United States. *Science* **386**, 748-755 (2024).
3. Ferraro, P.J. & Prasse, C. Reimagining safe drinking water on the basis of twenty-first-century science. *Nat. Sustain.* **4**, 1032-1037 (2021).
4. Ackerman Grunfeld, D.*, et al.* Underestimated burden of per- and polyfluoroalkyl substances in global surface waters and groundwaters. *Nat. Geosci.* **17**, 340-346 (2024).
5. Prevedouros, K., Cousins, I.T., Buck, R.C. & Korzeniowski, S.H. Sources, fate and transport of perfluorocarboxylates. *Environ. Sci. Technol.* **40**, 32-44 (2006).
6. Sun, M.*, et al.* Legacy and Emerging Perfluoroalkyl Substances Are Important Drinking Water Contaminants in the Cape Fear River Watershed of North Carolina. *Environ. Sci. Technol. Lett.* **3**, 415-419 (2016).

7. Belkouteb, N., Franke, V., McCleaf, P., Köhler, S. & Ahrens, L. Removal of per- and polyfluoroalkyl substances (PFASs) in a full-scale drinking water treatment plant: Long-term performance of granular activated carbon (GAC) and influence of flow-rate. *Water Res.* **182**, 115913 (2020).
8. Shi, Q.*, et al.* Strategy to minimize plant accumulation of contaminants of emerging concern: alternation between treated wastewater and freshwater irrigation. *Environ. Sci. Technol. Lett.* **10**, 302-307 (2023).
9. Scheurer, M.*, et al.* Small, mobile, persistent: trifluoroacetate in the water cycle – overlooked sources, pathways, and consequences for drinking water supply. *Water Res.* **126**, 460-471 (2017).
10. Tosun, J., Scherer, U., Schaub, S. & Horn, H. Making Europe go from bottles to the tap: Political and societal attempts to induce behavioral change. *WIREs Water* **7**, e1435 (2020).
11. Rosario-Ortiz, F., Rose, J., Speight, V., Gunten, U.v. & Schnoor, J. How do you like your tap water? *Science* **351**, 912-914 (2016).
12. Cousins, I.T., Johansson, J.H., Salter, M.E., Sha, B. & Scheringer, M. Outside the safe operating space of a new planetary boundary for per- and polyfluoroalkyl substances (PFAS). *Environ. Sci. Technol.* **56**, 11172-11179 (2022).
13. Solomon, K.R.*, et al.* Sources, fates, toxicity, and risks of trifluoroacetic acid and its salts: Relevance to substances regulated under the Montreal and Kyoto Protocols. *J. Toxicol. Environ. Health B Crit. Rev.* **19**, 289-304 (2016).
14. Guo, W., Hao, W. & Xiao, W. Emerging perfluorinated chemical GenX: environmental and biological fates and risks. *Environ. Health (Wash)* **3**, 338-351 (2025).
15. Munoz, G., Liu, J., Vo Duy, S. & Sauvé, S. Analysis of F-53B, Gen-X, ADONA, and emerging fluoroalkylether substances in environmental and biomonitoring samples: A review. *Trends Environ. Anal. Chem.* **23**, e00066 (2019).
16. OECD. Reconciling terminology of the universe of per- and polyfluoroalkyl substances: recommendations and practical guidance. (OECD Publishing, Paris, 2021).
17. Wang, X.*, et al.* Machine learning-enhanced molecular network reveals global exposure to hundreds of unknown PFAS. *Sci. Adv.* **10**, eadn1039 (2024).
18. EPA. SW-846 test method 8327: per-and polyfluoroalkyl substances (PFAS) by liquid chromatography/tandem mass spectrometry (LC/MS/MS). (EPA, 2025).
19. Dauchy, X.*, et al.* Poly- and perfluoroalkyl substances in runoff water and wastewater sampled at a firefighter training area. *Arch. Environ. Contam. Toxicol.* **76**, 206-215 (2019).
20. Forster, A.L.B., Zhang, Y., Westerman, D.C. & Richardson, S.D. Improved total organic fluorine methods for more comprehensive measurement of PFAS in industrial wastewater, river water, and air. *Water Res.* **235**, 119859 (2023).
21. European Union. Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption. in *Gazzetta Ufficiale dell'Unione Europea L* (Official Journal of the European Union, Brussels, Belgium, 2020).
22. European Commission. Technical guidelines regarding methods of analysis for monitoring of per- and polyfluoroalkyl substances (PFAS) in water intended for human consumption (C/2024/4910). (Official Journal of the European Union, Brussels, Belgium, 2024).

23. He, Z., Plassmann, M., Cousins, I.T. & Benskin, J.P. A novel fluorine mass balance method for improved characterization and quantification of extractable (Organo)fluorine in drinking water. *Environ. Sci. Technol. Lett.* **12**, 73-78 (2025).
24. Andrews, D.Q. & Naidenko, O.V. Population-wide exposure to per- and polyfluoroalkyl substances from drinking water in the United States. *Environ. Sci. Technol. Lett.* **7**, 931-936 (2020).
25. Liu, L., Qu, Y., Huang, J. & Weber, R. Per- and polyfluoroalkyl substances (PFASs) in Chinese drinking water: risk assessment and geographical distribution. *Environ. Sci. Eur.* **33**, 6 (2021).
26. Smalling, K.L.*, et al.* Per- and polyfluoroalkyl substances (PFAS) in United States tapwater: comparison of underserved private-well and public-supply exposures and associated health implications. *Environ. Int.* **178**, 108033 (2023).
27. Dauchy, X. Evidence of large-scale deposition of airborne emissions of per- and polyfluoroalkyl substances (PFASs) near a fluoropolymer production plant in an urban area. *Chemosphere* **337**, 139407 (2023).
28. Teymoorian, T., Delon, L., Munoz, G. & Sauvé, S. Target and suspect screening reveal PFAS exceeding European Union guideline in various water sources south of Lyon, France. *Environ. Sci. Technol. Lett.* **12**, 327-333 (2025).
29. Dalmijn, J.*, et al.* HFPO-DA and other PFAS in air downwind of a fluoropolymer production plant in the Netherlands: measurements and modeling. *Environ. Sci. Technol.* **59**, 8662-8672 (2025).
30. Neuwald, I.J.*, et al.* Ultra-short-chain PFASs in the sources of German drinking water: prevalent, overlooked, difficult to remove, and unregulated. *Environ. Sci. Technol.* **56**, 6380-6390 (2022).
31. Arp, H.P.H., Gredelj, A., Glüge, J., Scheringer, M. & Cousins, I.T. The global threat from the irreversible accumulation of trifluoroacetic acid (TFA). *Environ. Sci. Technol.* **58**, 19925-19935 (2024).
32. Zhi, Y.*, et al.* Environmental occurrence and biotic concentrations of ultrashort-chain perfluoroalkyl acids: overlooked global organofluorine contaminants. *Environ. Sci. Technol.* **58**, 21393-21410 (2024).
33. Neuwald, I.J.*, et al.* Occurrence, distribution, and environmental behavior of persistent, mobile, and toxic (PMT) and very persistent and very mobile (vPvM) substances in the sources of German drinking water. *Environ. Sci. Technol.* **56**, 10857-10867 (2022).
34. Jiao, E.*, et al.* Further insight into extractable (Organo)fluorine mass balance analysis of tap water from Shanghai, China. *Environ. Sci. Technol.* **57**, 14330-14339 (2023).
35. Tisler, S.*, et al.* Nontarget and suspect screening of fluorinated ionic liquids and PFAS in european wastewaters using supercritical fluid chromatography. *Environ. Sci. Technol.* **59**, 21300-21311 (2025).
36. Rensmo, A.*, et al.* Lithium-ion battery recycling: a source of per- and polyfluoroalkyl substances (PFAS) to the environment? *Environ. Sci.: Processes Impacts* **25**, 1015-1030 (2023).
37. van Larebeke, N.*, et al.* Per- and polyfluoroalkyl substances (PFAS), thyroid hormones, sexual hormones and pubertal development in adolescents residing in the neighborhood of a 3M factory. *Environ. Health* **24**, 34 (2025).

38. Cappelli, F., Ait Bamai, Y., Van Hoey, K., Kim, D.-H. & Covaci, A. Occurrence of short- and ultra-short chain PFAS in drinking water from Flanders (Belgium) and implications for human exposure. *Environ. Res.* **260**, 119753 (2024).
39. Jeong, Y*., et al.* Occurrence and contamination profile of legacy and emerging per- and polyfluoroalkyl substances (PFAS) in Belgian wastewater using target, suspect and non-target screening approaches. *J. Hazard. Mater.* **437**, 129378 (2022).
40. Kallenborn, R*., et al.* Industrial and public infrastructure as local sources of organic contaminants in the Arctic. *Environ. Sci.: Adv.* **5**, 304-347 (2026).
41. Saliu, T.D., Pepin, D., Delon, L., Fontaine, J. & Sauvé, S. Occurrence of PFAS in Lyon soils: contribution and impact of industrial activities. *Environ. Sci.: Adv.* **4**, 1746-1766 (2025).
42. Gagliano, E., Sgroi, M., Falciglia, P.P., Vagliasindi, F.G.A. & Roccaro, P. Removal of poly- and perfluoroalkyl substances (PFAS) from water by adsorption: Role of PFAS chain length, effect of organic matter and challenges in adsorbent regeneration. *Water Res.* **171**, 115381 (2020).
43. Jiang, X*., et al.* Pilot-scale removal of PFAS from chromium-plating wastewater by anion exchange resin and activated carbon: adsorption difference between PFOS and 6:2 fluorotelomer sulfonate. *Chem. Eng. J.* **481**, 148569 (2024).
44. Miller, A.J*., et al.* Global Inventory of Fluoropolymer Production Plants and Their Associated PFAS Environmental Contamination. *Environ. Sci. Technol.* **60**, 11688-11701 (2026).
45. Post, G.B. Recent US state and federal drinking water guidelines for per- and polyfluoroalkyl substances. *Environ. Toxicol. Chem.* **40**, 550-563 (2020).
46. EPA. Fifth unregulated contaminant monitoring rule (UCMR 5). (U.S. Environmental Protection Agency, Washington, D.C., USA, 2021).
47. Wasser, T.D.-T. PFAS in drinking water: Legal regulations are only partially realisable for water analysis. in *TZW News* (2024).
48. Italian Government. Legislative Decree No. 102 of 19 June 2025: Supplementary and corrective provisions to Legislative Decree No. 18 of 23 February 2023 implementing Directive (EU) 2020/2184 on the quality of water intended for human consumption. in *Serie Generale No. 153* (Italian Government, Rome, Italy, 2025).
49. SCAHT. ECHA Scientific Committee Supports Classification of TFA as Toxic to Reproduction. Vol. 2026 (Swiss Centre for Applied Human Toxicology, 2026).
50. Hartman J, B.B., Smit CE. Werkwijze toetsing PFAS in drinkwatermonsters [Method of testing PFAS in drinking water samples]. (RIVM, Bilthoven, The Netherlands, 2023).
51. Smit CE, V.E., Bokkers BGH. Aanvulling RPF's en RBF's voor PFAS. (RIVM, 2025).
52. EFSA. Outcome of a public consultation on the draft risk assessment of perfluoroalkyl substances in food. *EFSA Supporting Publications* **17**, 1931E (2020).
53. EFSA*, et al.* Risk to human health related to the presence of perfluoroalkyl substances in food. *EFSA Journal* **18**, e06223 (2020).

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Dataset.xlsx
- SI.docx