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Mapping *per*- and polyfluoroalkyl substances contamination in England's surface waterbodies: Urban water cycle pathways and governance challenges

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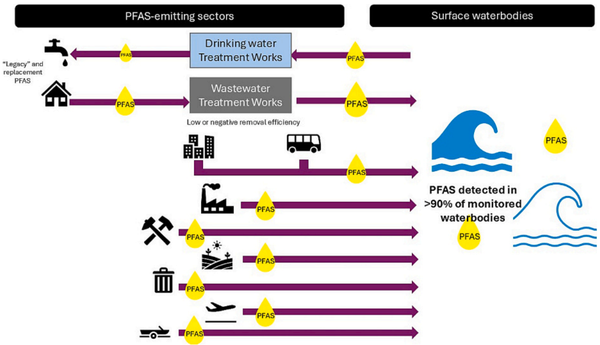
HIGHLIGHTS

- PFAS contamination is widespread, detected in 92% of England's monitored waterbodies.
- Water Industry/Domestic/General Public key pathways of PFAS leaching to waterbodies.
- PFOS routinely present in Wastewater Treatment Works effluents despite regulatory ban.
- Wastewater Treatment Works showed low or negative removal efficiencies for many PFAS.
- Stronger source controls and polluter-pays governance are essential for protection

GRAPHICAL ABSTRACT



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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) contamination has emerged as a major international environmental and regulatory challenge, with PFAS increasingly detected across freshwater systems worldwide. However, in countries with limited PFAS manufacturing, such as England, it remains unclear whether surface waterbodies contamination reflects diffuse consumer-driven pollution, sectoral pressures, or dominant point-source pathways of PFAS pollution, such as Wastewater Treatment Works (WWTWs). In this study, we address this gap by providing the first surface-waterbody-level characterisation of PFAS contamination across England, drawing on the Environment Agency 's 2024 national dataset. Linking PFAS detections with sectoral pressure classifications, the study makes the following contributions: 1) quantifies the associations between individual compounds and human activities, 2) assesses WWTWs as pathways for PFAS release, and 3) maps detected PFAS to sector-specific product applications. Our analysis reveals that 92% of monitored waterbodies contain at least one of thirty-four detected PFAS, with multiple compounds co-occurring (mean ~ 6.5) and PFOS frequently exceeding its Environmental Quality Standard. Water Industry/Domestic/General Public pressures showed strong positive associations with 11 PFAS compounds, with effect sizes of 2.9–9.9 (FDR < 0.05). After adjusting for overlapping sectoral influences, significant positive associations remained for PFHxS.L, PFBS, PFHpA, PFOS..B, PFOS..L and

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PFOS_combined, with odds ratios between 2.0 and 3.0 (FDR < 0.05). PFAS were also routinely present in WWTWs effluents, where removal efficiencies were often low or negative, indicating that WWTWs function as chronic point sources. Persistent PFOS detections in WWTWs effluents long after its restriction reflect that PFAS are now deeply embedded within the built environment, recirculating through the urban water cycle. These findings underscore the necessity for a comprehensive, system-level governance approach for PFAS that transcends single-compound restrictions and advocates for a fair allocation of mitigation responsibilities.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a wide group of synthetic, man-made organofluorine compounds manufactured since the 1940s, with their usage becoming prominent in the 1950s and remaining significant ever since (Buck et al., 2011; Gaines, 2023). The chemical structure of PFAS is characterised by the presence of one or more perfluoroalkyl moieties (CF_n). These grant them a series of unique properties, such as thermal and chemical stability, and hydrophobic and lipophobic surfactant properties. Because of these properties, PFAS have been widely used in almost all industrial sectors and in many consumer products. Common applications include textile impregnation, medical utensils, food packaging, personal care products, pesticides, pharmaceuticals, cleaning agents, and as chemical manufacturing additives (Curtzwiler et al., 2021; Gaines, 2023; Glüge et al., 2020).

The same characteristics that make PFAS valuable to industry and consumer goods also render them a significant environmental and public health hazard. The exceptional stability of the CF_n bond means that no known biotic or abiotic degradation mechanisms can fully mineralise PFAS (Brunn et al., 2023). Consequently, they persist in the environment (Cousins et al., 2020), and they are commonly referred to as “forever chemicals” (Cordner et al., 2021). Compounding this challenge, their widespread use across numerous industries and products has led to the continuous release of large quantities into the environment.

Owing to their persistence and widespread use, PFAS are now ubiquitously detected in all environmental matrices worldwide. They are found even in remote regions with minimal or no direct human activity and undisturbed from human pressures (Faust, 2022). PFAS have been reported in soil (Brusseau et al., 2020; Johnson, 2022), air (Faust, 2022), water (Kurwadkar et al., 2022), plants (Adu et al., 2023), wildlife (Giesy and Kannan, 2001; Remili et al., 2025) and humans (Sunderland et al., 2019). Their extensive prevalence has significant environmental and health consequences. These range from quality degradation in soil, air and water, disruptions in the food chain and bioaccumulation (Ahrens and Bundschuh, 2014; Chen et al., 2023), to adverse human health impacts such as ulcerative colitis, endocrine disruption, increased risk of metabolic diseases (Lai et al., 2018; Shittu et al., 2023; Steenland et al., 2018; Sunderland et al., 2019) and various types of cancer (Messmer et al., 2022).

As synthetic compounds, the ubiquitous presence of PFAS in environmental matrices is entirely attributable to anthropogenic activities, mainly through the production, use and disposal of PFAS-containing materials and products (European Chemicals Agency, 2025; Kwiatkowski et al., 2020). Fluorochemical companies have been widely implicated as major sources of PFAS contamination in aquatic environments (Bach et al., 2017; Gebbink et al., 2017; Pétré et al., 2021; Wang et al., 2015; Y. Zhang et al., 2024). Substantial PFAS inputs have also been linked to the industrial uses of PFAS, including the textile and electronics sectors and electroplating processes (Jia et al., 2023; Kim et al., 2021; Payne et al., 2024). Waste treatment facilities and landfills are also identified as significant sources of PFAS loading to groundwaters and surface waterbodies (Roy et al., 2025; Tolaymat et al., 2023; Yan et al., 2015). Airports constitute an additional well-established hotspot, primarily due to historical use and testing of aqueous film-forming foams used for fire-fighting purposes (Ahrens et al., 2015; Awad et al., 2011; Milley et al., 2018). Other sectors with potential PFAS contributions include mining (Barfoot et al., 2022), agriculture, specifically through the application of PFAS-containing pesticides and insecticides (Donley et al., 2024; Libenson et al., 2024), and transport, through the use of PFAS in hydraulic oils and lubricants (OECD, 2025; Zhu and Kannan, 2020).

Moreover, municipal wastewater represents another major pathway for PFAS release into the environment (Campo et al., 2014; Gallen et al., 2018; Salawu et al., 2025; Silver et al., 2023). Wastewater Treatment Works (WWTWs) receive influents containing PFAS from domestic, trade and stormwater sources. Although they employ combinations of physical, chemical and biological processes, conventional WWTWs are not designed to remove PFAS, resulting in low removal efficiencies, and many studies report higher PFAS concentrations in the effluents than in the influents (Cookson and Detwiler, 2022; Lenka et al., 2021, 2022; C. Zhang et al., 2015). Sludge generated as a by-product of wastewater treatment represents an additional pathway of release. Sludge can be applied to land as fertiliser (Johnson, 2022), serving as a source of PFAS to soils and crops (Fredriksson et al., 2022), whereas if incinerated or landfilled, it can introduce PFAS into the atmosphere (Winchell et al., 2024), soils, and groundwater (Johnson, 2022).

Internationally, regulatory frameworks for PFAS production, use and environmental release are evolving. In the United Kingdom, the main instruments used to regulate PFAS are the UK REACH and the Stockholm Convention on Persistent Organic Pollutants. UK REACH restricts the production and use of PFOA and identifies nine other PFAS as Substances of Very High Concern (Environment Agency, 2021). The Stockholm Convention on Persistent Organic Pollutants regulates PFOS, PFOA and PFHxS by prohibiting their production and use except in limited cases, such as in specific industrial applications or trace impurities (Secretariat of the Stockholm Convention, n.d.). In addition, statutory Environmental Quality Standards (EQS) remain limited, with PFOS currently being the only PFAS with a defined EQS value (6.5×10^{-4} µg/L) (European Union, 2008). In August 2024, the Drinking Water Inspectorate (DWI) issued guidance setting a non-statutory limit of 0.1 µg/L for the sum of 48 PFAS in drinking water (Drinking Water Inspectorate, 2025). Even though these guidelines are not yet enforced, the DWI has the power to place water companies which fail to comply with the recommendations in legally binding programs aimed at raising the quality of the water. This lack of enforceable thresholds represents a significant regulatory gap that may contribute to the continued prevalence of PFAS in aquatic environments. In the United States, recent legislative developments have held major manufacturers financially liable for PFAS remediation, signalling a shift towards extended producer responsibility (EPR).

The consistent detection of PFAS in municipal effluents with little or no industrial influence (Barisci and Suri, 2021; Gobelius et al., 2023; Moneta et al., 2023; Krlovic et al., 2024; Lenka et al., 2022), highlights the significance of domestic use and disposal of PFAS-containing consumer goods as an under-recognised source of leakage (Lenka et al., 2021). This raises a critical policy challenge, particularly in countries with limited PFAS industrial activity, such as England. While water companies are legally obligated to collect and treat all wastewater under the Urban Wastewater Treatment Directive, they have limited control over upstream PFAS inputs driven by consumer products. Therefore, determining whether consumer-driven sources, sector-specific activities, or point-source pathways such as WWTWs dominate contribution patterns is critical for guiding governance dynamics and responsibility allocation. Resolving this uncertainty would strengthen policy and mitigation strategies, as it will likely cascade responsibility upstream to producers, brands and product distributors, creating an imperative for regulatory, technical and infrastructural reforms across the PFAS value chain.

As a result, despite extensive documentation of PFAS production, uses and potential release pathways, a fundamental scientific gap remains unresolved: which human activities are associated with PFAS occurrence in surface waterbodies, and to what extent do diffuse consumer-driven releases, sectoral pressures, or point-source pathways such as WWTWs explain observed contamination patterns? Previous national assessments for England have focused on a limited number of legacy compounds (mainly PFOS, PFOA and a small subset of related acids) and have largely summarised monitoring up to 2020 at the sampling-location level, drawing on relatively few monitoring sites (Environment Agency, 2021; Environmental Agency, 2019). Meanwhile, the compound-specific contributions, co-occurrence patterns and sectoral attribution of PFAS contamination across waterbodies remain insufficiently characterised. Without national-scale, waterbody-level analyses and sector-resolved evidence, the explanatory strength of whether human activities and wastewater pathways are driving PFAS releases, and whether PFAS co-occur across waterbodies in ways that are associated with different pressures, cannot be properly assessed.

In this study, we address this critical gap by providing the first surface waterbody-level characterisation of PFAS contamination across England, drawing on the Environment Agency 's 2024 national dataset covering 51 PFAS compounds monitored at over 1000 sites. By linking PFAS detections with pressure categories and examining how contamination patterns vary across key human activities, we quantify the relative strength of association between individual PFAS compounds and sectoral pressures. We further assess the role of WWTWs as a major pathway for PFAS release, quantifying both their contribution to environmental loads and their capacity for removal. Together, these analyses offer the first comprehensive national overview of PFAS prevalence and sources in England's surface waters, providing an urgently needed evidence base to guide targeted monitoring, strengthen regulatory measures, and clarify the allocation of responsibility within an evolving policy landscape.

2. Methods

2.1. PFAS in surface waterbodies

The prevalence of PFAS in England's surface waterbodies during 2024 was investigated using open-access, national-scale surface water quality data obtained from the Environment Agency (EA) through the Water Quality Archive (Environment Agency, n.d). The dataset comprised 70,439 records from ~1000 sampling locations across rivers, lakes, estuaries, and coastal waters. Each entry included geographic coordinates, waterbody type, sampling date and time, analytical parameter code, Measured Environmental Concentration (MEC), units, and censoring flags (‘<’, ‘>’). Units were consistent across all entries (μg/L). An overview of the methodological workflow is provided in Supplementary Fig. 1.

Given the high proportion (~80%) of left-censored MECs (‘<’), Regression on Order Statistics (ROS) was applied using the NADA package in R (version 1.6–1.1) to estimate mean PFAS concentrations per compound at each sampling location, following established reporting practices (Helsel, 2005; Environment Agency, 2025). Diagnostic metrics included: the number and proportion of censored observations; the number of detections; the minimum detected concentration; and the maximum reporting concentration. For each combination of sampling location and PFAS compound (hereafter referred to as a “group”) with 0% left-censored results (i.e., no ‘<’ qualifiers), the arithmetic mean of the MECs was used to derive the mean concentration per compound at that location. For groups with a combination of MECs and left-censored values, ROS modelling was conducted to estimate the mean concentration per compound and sampling location, given that the following criteria were met: minimum sample size $n \geq 4$; minimum of detected observations ≥ 3; and maximum left-censored proportion $<$ 80%. The groups that did not meet the criteria were flagged accordingly and excluded from statistical summaries.

Sampling locations were spatially linked to their corresponding surface waterbodies in QGIS by overlaying monitoring points with Water Framework Directive (WFD) Cycle 3 waterbody designations (Environment Agency, 2025). This enabled the aggregation of monitoring data at the waterbody level, allowing identification of compounds detected within each waterbody, counts of unique compounds, and calculation of average ΣPFAS concentrations. The monitored waterbodies were broadly representative of national conditions in terms of the distribution of waterbody types (Supplementary Table 1).

2.2. Identification and mapping of sectoral pathways

To identify the sectors potentially contributing to PFAS contamination in England's surface water bodies, PFAS monitoring data (Section 2.1) were first converted into a binary detection format (1 = detected, 0 = not detected). For PFOS, detections recorded as PFOS, PFOS..B and PFOS..L were combined into a single binary variable (“PFOS_combined”) indicating the presence or absence of PFOS in any isomeric form. This binary dataset was then integrated with the Environment Agency's Challenges database. This database attributes waterbody failures to achieve “good status” under the WFD to specific sectors and activities (Environment Agency, 2025). The Agency does not currently assign PFAS-related failures to business sectors; as a result, the established sector-pressure-impact linkages were used as surrogates to examine whether PFAS detections spatially coincide with waterbodies already impacted by known anthropogenic activities. Records associated with identified point and diffuse sources were retained.

Each waterbody record contained both a sectoral category label and a more specific activity descriptor, which together defined the pressures attributed to that waterbody. Because several of the original sectoral categories (e.g., “Urban and transport,” “Local and Central Government,” “Recreation”) grouped heterogeneous activities with differing emission sources and wastewater pathways, individual activities were reassigned to nine harmonised typologies. These typologies were developed by consolidating Environment Agency (2025) activity descriptors and sector categories: Agriculture, Airports, Industry, Mining & Quarrying, Navigation, Transport, Urban, Waste Treatment & Disposal, and Water Industry/Domestic/General Public. For example, activities such as sewage discharges, misconnections, and private sewage treatment systems were reclassified under Water Industry/Domestic General Public; landfill leaching under Waste Treatment & Disposal; airport drainage under Airports; transport drainage, roads, and railways under Transport; and port or harbour authorities and inland waterways under Navigation. The remaining activities under the original Urban and transport category, including mainly Urbanisation - urban development, but also observations on contaminated land, incidents, and atmospheric sources, were retained under Urban. These nine typologies were mapped to PFAS occurrence data at the waterbody level to identify potential pollution pathways and cross-sectoral contamination patterns.

Pairwise associations between PFAS compounds and sectoral activities were first screened using Fisher's exact tests applied to 2 × 2 contingency tables contrasting PFAS detection status (detected vs. non-detected) against the presence or absence of each sectoral pressure within waterbodies. Odds ratios (ORs) and corresponding 95% confidence intervals were computed to represent the relative likelihood of PFAS detection under each activity type. Statistical significance was assessed using Fisher's exact test p-values adjusted for multiple testing via the Benjamini–Hochberg false discovery rate (FDR) procedure (FDR < 0.05).

To visualise the cross-sectoral association patterns, odds ratio estimates were summarised across PFAS compounds and activities in a hierarchical tree diagram. PFAS compounds showing significant associations in Fisher's tests were retained for subsequent analyses. To account for overlapping pressures (i.e., waterbodies influenced by multiple activities), multivariable logistic regression models were then fitted for each PFAS compound, estimating adjusted odds ratios for detection as a function of individual sectoral activities while including all remaining sectors as covariates.

Finally, a sensitivity analysis was conducted to assess the robustness of PFAS compounds' associations with Water Industry/Domestic/General Public pressures. Models were restricted to waterbodies where these activities were present, i.e., isolating cases of waterbodies influenced solely by them. This tested whether the observed relationship between PFAS detection and Water Industry/Domestic/General Public pressures persisted once other sectoral pathways were excluded.

2.2.1. *PFAS prevalence and fate at WWTWs*

To characterise the occurrence and discharge of PFAS from WWTWs in England, PFAS concentrations in influent and effluent were obtained from UKWIR (n.d.) under the UK Chemical Investigation Programme (CIP). The CIP brings together the 10 large water and wastewater companies in England and Wales with regulators (Defra, the Environment Agency and Natural Resources Wales) to investigate a range of chemical substances, often contained in many domestic products. Influent and/or effluent data were available for 10 PFAS compounds measured at 684 distinct WWTWs across England, from CIP's Phase 2 (2015–2020) and Phase 3 (2020−2022).

To quantify PFAS concentrations in treated effluents, the dataset was filtered to retain effluent records corresponding to the most recent monitoring year per WWTWs (Supplementary Table 2). Concentrations for all PFAS compounds measured in that year were then extracted. The data were summarised 1470 WWTWs x compound combinations: 1365 without left-censored results, 13 fully censored, and 92 mixed. For groups without censored data, the arithmetic mean of MEC was used to derive the mean per compound, following the same procedure described in Section 2.1. Groups containing censored data were assessed for suitability for ROS modelling using the same diagnostic criteria as in 2.1 (minimum $n \geq 4$; ≥ 3 detects; $< 80\%$ censored). ROS was applied to eligible groups to estimate mean concentrations, while non-eligible groups were flagged and excluded from statistical summaries.

The effectiveness of England's WWTWs for PFAS removal was assessed using matched influent-effluent concentration pairs for each PFAS compound and WWTWs, ensuring both samples originated from the same facility and sampling date. This pairing ensured comparability under identical operational conditions. Information on treatment technologies was obtained from the Environment Agency's Consented Discharges to Controlled Waters with Conditions data. Where duplicate influent or effluent measurements existed for the same WWTWs x sampling date x PFAS compound combination, these were reviewed individually, and the maximum concentration within each duplicate group was retained. Removal efficiency (%) was calculated as:

$$\textit{Removal efficiency}\ (\%) = \frac{C_{influent} - C_{effluent}}{C_{influent}} * 100 \tag{1}$$

To represent current performance, only data from the most recent sampling year per plant were retained. One extreme outlier (−118,065.78% removal at Saltford STW, 2021-02-04) was removed, and 2022 records for Cherry Burton STW were replaced with 2021 data due to evident measurement anomalies. Mean removal efficiencies were then calculated for each PFAS-WWTWs pair, followed by summary statistics aggregated by PFAS compound and treatment process type.

2.2.2. *Mapping of identified PFAS compounds to sector-specific product uses*

To link the PFAS compounds identified in the monitoring dataset to the relevant sectors we identified (Section 2.2), we used two primary sources: a PFAS application database developed by Glüge et al. (2020) the European Chemicals Agency (ECHA) Background Document on PFAS Restriction (European Chemicals Agency, 2025). Supplementary sources included the Food Contact Chemicals database (K. Groh et al., 2020; K. J. Groh et al., 2021), the Database on Migrating and Extractable Food Contact Materials (Geueke et al., 2023), the United States Environmental Protection Agency's ChemExpo Knowledgebase (Williams et al., 2017), and peer-reviewed screening studies, following the methodology outlined by Xiao and Watson (2019).

Each PFAS was then assigned to one or more usage categories mapped from Glüge et al. (2020) and European Chemicals Agency (ECHA) (2025). The categories we deemed relevant to investigate are: (1) Apparel; (2) Upholstery; (3) Carpets and floor coverings; (4) Floor polishes; (5) Cleaning compositions; (6) Cosmetics and personal-care products (PCP); (7) Cookware and baking ware; (8) Paper and food-contact packaging; (9) Additives to hydraulic fluids in transport vehicles; (10) Architectural coatings & paints / Wetting & levelling agents; and (11) Surface protection. Categories (1) to (8) correspond to consumer product applications, accounted for in the Water Industry and Domestic & General Public sectors; (9) pertains to Transport; (10) and (11) address the Urban sector. Additionally, we flagged degradation products (DP). The DP identifier flags PFAS, which are known to be degradation products from a parent fluorinated compound linked to that specific application.

3. Results

3.1. *PFAS in surface waterbodies*

In 2024, 51 PFAS compounds were monitored across 724 waterbodies in England (45 unique compounds plus isomers) (Supplementary Table 3). Four were monitored as linear and branched isomers: PFOS (PFOS..L and PFOS..B), in addition to records documented just as “PFOS”; PFHxS (PFHxS.L and PFHxS·B); EtFOSAA (EtFOSAA.L and EtFOSAA.B); and MeFOSAA, (MeFOSAA.L and MeFOSAA.B). PFOA was monitored as perfluorooctanoate anion (pFoctanoate) and as perfluorooctanoic acid (PFOA). Thirty-four compounds were detected at least once across 665 waterbodies (92% of waterbodies monitored). Specifically, 333 waterbodies had 1–5 PFAS compounds detected, 183 had 6–10, 124 had 11–15, 24 had 16–20, and one waterbody had 21 PFAS compounds detected. Moreover, 42 waterbodies had individual measurements above the limit of detection but insufficient data to compute concentrations (Fig. 1). In 59 waterbodies, all monitoring results were below the analytical Limit of Detection (LOD).

PFOS and its isomers were among the most prevalent compounds, detected in 517 waterbodies. PFOS was detected in 255 waterbodies (~100%), while the isomers PFOS..B and PFOS..L were detected in 317 (55%) and 311 (54%) waterbodies, respectively (Table 1). Most Perfluoroalkyl Carboxylic Acids (PFCAs) were also highly prevalent. PFOA was detected in 441 waterbodies (76%), PFBA in 438 (76%), and PFHxA in 412 (71%). PFOA in its anion form was monitored in fewer waterbodies but still showed high detection, occurring in 208 waterbodies (84%). PFHpA and PFPeA were detected in 40% and 53% of monitored waterbodies respectively, whereas PFNA, PFODA and PFDA showed low detection rates (<5.7%).

Among Perfluoroalkyl Sulfonic Acids (PFSAs), PFBS was found in 335 waterbodies (58%), while PFHxS isomers (PFHxS.L and PFHxS·B) were detected in 302 (52%) and 112 (19%) waterbodies, respectively (Table 1). Other sulfonic acids, such as PFPeS, PFecHS, and PFHpS, showed progressively lower prevalence, with detections in fewer than 15% of monitored waterbodies. PFDS was detected in only one waterbody.

The fluorotelomer sulfonic acids (FTSAs) were a less common group. 6:2 FTSA was detected in 222 waterbodies (38%), while 8:2 FTSA and 4:2 FTSA were rarely detected (≤1%) (Table 1). The Perfluoroalkane Sulfonamides (FASAs) were detected at varying frequencies. FBSA occurred in 180 waterbodies (31%), while FOSA was found in just 2

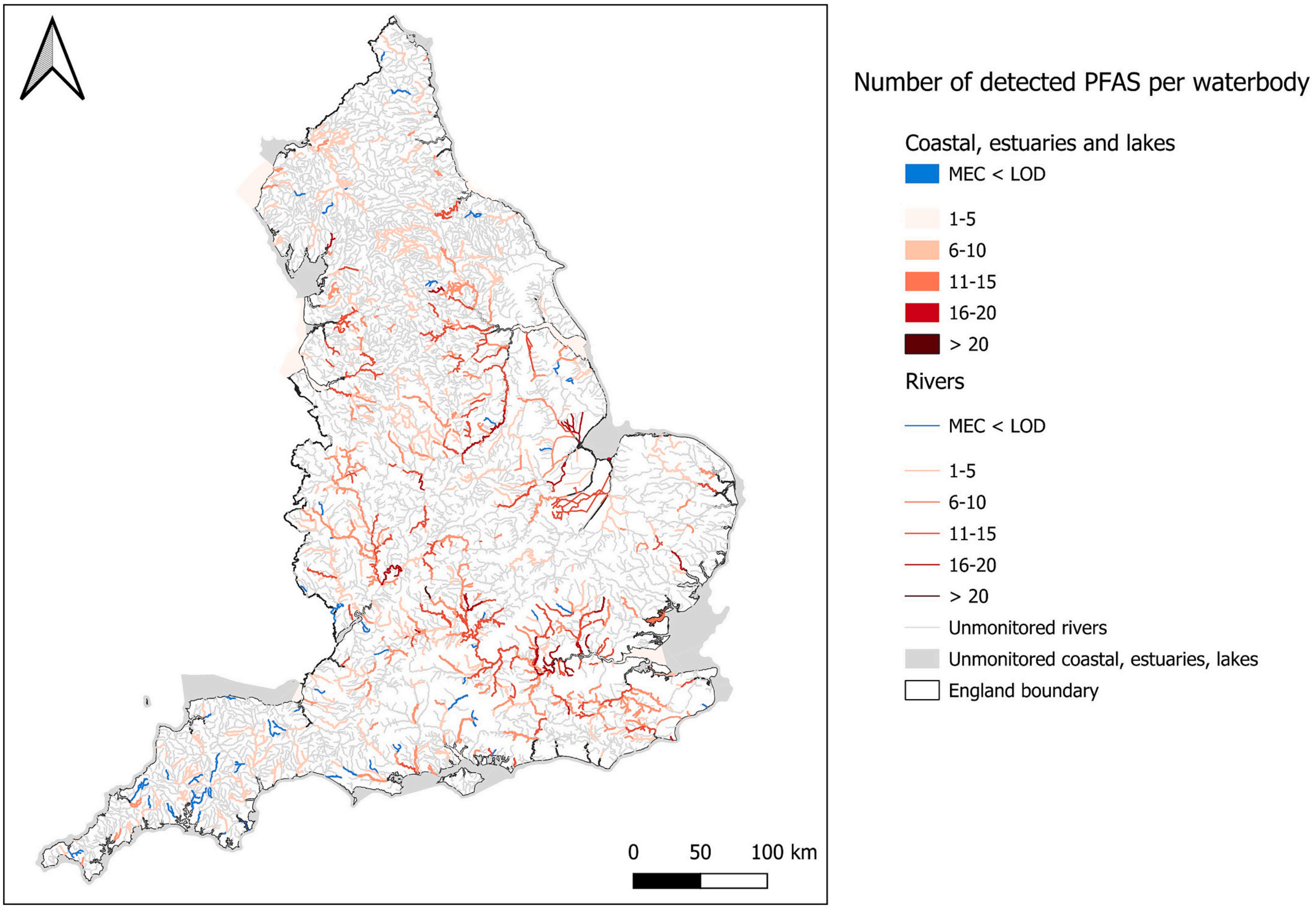

Fig. 1. Spatial prevalence of distinct PFAS compounds in England's surface waterbodies in 2024. Results are shown for rivers (lines) and for lakes, estuaries, and coastal waters (polygons). Red shading indicates the number of compounds detected above the analytical limit of detection (LOD), expressed as measured environmental concentrations (MEC).

waterbodies (<1%). MeFOSA/EtFOSA and their derivatives were not detected at all in 2024.

Emerging PFAS, including, HFPO-DA (GenX), and NFDHA, were detected in only one or two waterbodies (<1%). 17 PFAS compounds were not detected in 2024 in any of the sampling campaigns, including: 11Cl.PF3OUdS; 3.3.FTCA; 9Cl.PF3ONS; ADONA; EtFOSA; EtFOSAA.B; EtFOSAA.L; EtFOSE; MeFOSAA.B; MeFOSAA.L; MeFOSE; N.MeFOSA; PFDoS; PFEESA; PFMOPrA; PFTrDA; PFTeDA.

Among the PFAS compounds with sufficient data for concentration analysis, PFOS (unqualified) exhibited the highest maximum concentration at 630 ng/L, with a mean of 5.86 ng/L across 238 waterbodies (Table 1). PFOS isomers PFOS..B and PFOS..L were quantified in 304 and 285 waterbodies, respectively, with similar median concentrations (1.38 and 1.50 ng/L), though PFOS-L exhibited notably higher mean and maximum values (4.83 ng/L and 212.20 ng/L) compared to PFOS-B (2.73 ng/L and 65.60 ng/L). Collectively, PFOS and its isomeric forms surpassed the Environmental Quality Standard (EQS; 0. 65 ng/L) in 386 waterbodies, indicating widespread exceedances across England's surface waters.

Among PFCAs, PFOA and its anion form exhibited moderate median concentrations of 1.80 and 1.69 ng/L and moderate maxima of 85.50 and 60.67 ng/L, respectively. Short-chain PFCAs (i.e., PFBA, PFHxA, PFHpA, PFPeA) also showed moderate median concentrations (2.1–3.5 ng/L), with higher maxima (110.6–364 ng/L). Regarding PFSAs, the most notable compounds were PFHxS.L (max: 250 ng/L; mean: 3.71 ng/L) and PFBS (max: 93.52 ng/L; mean: 4.30 ng/L).

PFHxSA and 6:2 FTSA showed considerable variability (sd > 37 ng/L) and high maxima (190 and 446 ng/L) alongside elevated mean concentrations (i.e., 15.74 and 10.02 ng/L) (Table 1). The remaining compounds, exhibited either very low detection frequencies or very low median, mean and maximum concentrations (Table 1).

Across the 724 monitored waterbodies, 334 exhibited ΣPFAS concentrations between of 0.1 and 10 ng/L; 214 had concentrations of 10–50 ng/L; 57 had concentrations of 50–100 ng/L and 18 exceeded 100 ng/L. Notably, two waterbodies displayed concentrations >1000 ng/L (Fig. 2).

3.2. *Sectoral pathways mapping to PFAS contamination*

PFAS were detected in 665 waterbodies. According to the Environment Agency, 495 of these failed to achieve good status due to pollution from identified point and diffuse pollution sources. Although, as stated in Methodology (Section 2.2), these failures are not attributed to PFAS under current WFD classifications, they provide a useful basis for examining whether PFAS detection concurs spatially with pressures from business sectors.

Water Industry/Domestic/General Public pressures accounted for the highest number of failures (403 waterbodies) and exhibited high PFAS detection frequencies across several compounds, with values exceeding 0.80 for pFoctanoate, PFOA, PFBA, PFHxA, PFOS_combined. (Fig. 3). Agriculture was the sector with the second-highest number of failing waterbodies (363 waterbodies) and exhibited high PFAS detection frequencies (>0.80) for pFoctanoate, PFOA, PFBA and PFHxA. Urban and Transport sectors were linked to fewer waterbody failures (91

Table 1

Prevalence and concentrations of PFAS compounds in England's surface waterbodies (WB) in 2024. Shown are the number of WBs monitored, WBs with at least one detection (*), and WBs with PFAS detected with sufficient data (†). Detection rates are given as proportions of all monitored WBs (%). Summary statistics of measured environmental concentrations (MECs, ng/L) are reported for waterbodies with sufficient data. Similar statistics are provided on Supplementary Table 4 for the WB x PFAS compound groups that were excluded from ROS analyses.

PFAS compound	WBs Monitored (n total = 724)		WBs* with detections		WBs† with detections and sufficient data		MEC (ng/L)				
	(n)	(%)	(n)	(%)	(n)	(%)	Min	Median	Mean	Max	Sd
PFOA	577	79.7	441	76.4	413	71.6	0.44	1.80	4.06	85.50	8.54
PFBA	577	79.7	438	75.9	381	66.0	0.63	2.80	4.33	110.60	6.46
PFHxA	581	80.2	412	70.9	374	64.4	0.47	2.38	4.79	210.60	12.35
PFBS	581	80.2	335	57.7	300	51.6	1.00	2.60	4.30	93.52	8.29
PFOS..B	572	79.0	317	55.4	304	53.1	0.50	1.38	2.73	65.60	6.14
PFOS..L	577	79.7	311	53.9	285	49.4	0.40	1.50	4.83	212.20	18.83
PFPeA	580	80.1	305	52.6	263	45.3	1.00	3.50	7.12	364.00	23.36
PFHxS.L	580	80.1	302	52.1	281	48.4	0.50	1.50	3.71	250.00	15.85
PFOS	256	35.4	255	99.6	238	93.0	0.09	1.24	5.86	630.00	41.30
PFHpA	583	80.5	231	39.6	200	34.3	1.00	2.10	3.75	126.80	9.38
6.2.FTSA	581	80.2	222	38.2	152	26.2	0.86	3.58	10.02	446.00	37.65
pFoctanoate	248	34.3	208	83.9	181	73.0	0.50	1.69	3.19	60.67	5.57
FBSA	580	80.1	180	31.0	136	23.4	0.43	0.90	1.34	16.00	1.71
PFHxS.B	577	79.7	112	19.4	98	17.0	0.45	0.80	1.65	42.00	4.39
PFPeS	579	80.0	80	13.8	57	9.8	0.47	0.84	1.31	12.00	1.74
PFecHS	580	80.1	50	8.6	42	7.2	0.41	0.89	2.07	11.00	2.15
PFNA	582	80.4	33	5.7	24	4.1	0.94	1.42	3.00	23.59	4.74
5.3.FTCA	580	80.1	25	4.3	13	2.2	1.10	1.90	2.59	8.00	2.04
PFHxSA	581	80.2	23	4.0	17	2.9	0.58	1.50	15.74	190.00	45.42
PFHpS	580	80.1	20	3.4	12	2.1	0.63	1.10	2.26	11.00	2.96
7.3.FTCA	568	78.5	13	2.3	6	1.1	0.77	2.27	3.87	12.00	4.34
PFDA	585	80.8	9	1.5	4	0.7	2.30	7.35	6.74	9.95	3.29
8.2.FTSA	582	80.4	7	1.2	5	0.9	1.20	7.25	28.61	84.80	36.06
PFUnDA	585	80.8	3	0.5	0	NA	NA	NA	NA	NA	NA
4.2.FTSA	580	80.1	2	0.3	1	0.2	0.58	0.58	0.58	0.58	NA
FOSA	578	79.8	2	0.3	2	0.3	2.08	5.29	5.29	8.50	4.54
HFPO.DA	581	80.2	2	0.3	0	NA	NA	NA	NA	NA	NA
PFNS	581	80.2	2	0.3	0	NA	NA	NA	NA	NA	NA
PFUnDS	581	80.2	2	0.3	2	0.3	11.00	15.00	15.00	19.00	5.66
PFDoDA	584	80.7	2	0.3	0	NA	NA	NA	NA	NA	NA
NFDHA	568	78.5	1	0.2	0	NA	NA	NA	NA	NA	NA
PFDS	581	80.2	1	0.2	0	NA	NA	NA	NA	NA	NA
PFMOBA	560	77.3	1	0.2	0	NA	NA	NA	NA	NA	NA
PFODA	573	79.1	1	0.2	0	NA	NA	NA	NA	NA	NA

*WBs with at least one sampling event > LOD (i.e., including cases where the proportion of non-detects exceeds 80%).
†WBs with sufficient data.

and 48, respectively), but the waterbodies influenced by these sectors displayed a wider diversity of PFAS compounds and detection frequency, particularly for fluorotelomer sulfonates and sulfonamides. The Industry sector was linked to 48 failing waterbodies and exhibited high detection frequencies for pFoctanoate, PFOA, PFBA and PFHxA and PFOS_combined. Mining & Quarrying, linked to only 19 failing waterbodies, exhibited the lowest PFAS detection frequencies across most of the compounds: the highest detection frequency was ~0.69 and corresponded to PFBA. Waste Treatment & Disposal was linked to only seven failing waterbodies, and four compounds (pFoctanoate, PFBA, PFOS_combined, PFBS) displayed high detection frequencies. Although Airports and Navigation were the sectors with the highest detection frequencies across many PFAS compounds (19 for each sector), these detections occurred in one and four waterbodies, respectively, reflecting highly localised contamination patterns.

Fisher's exact tests with FDR correction revealed distinct sectoral patterns in PFAS compounds occurrence (Fig. 4, Supplementary Table 5). Specifically, significant positive sector-PFAS compound associations were observed for the Water Industry/Domestic/General Public (OR = 2.9–9.9; FDR-adjusted $p < 0.05$), Urban (OR = 2.4–19.0; FDR-adjusted $p < 0.05$) and Transport (OR = 2.6–6.0; FDR-adjusted $p < 0.05$) pressures, indicating that PFAS detections occurred more frequently in waterbodies influenced by these sector-specific activities. In contrast, significant negative sector-PFAS compound associations (OR < 0.5; FDR-adjusted $p < 0.05$) were consistently observed for Agriculture, and Mining and Quarrying, suggesting reduced PFAS detection in waterbodies affected by these land uses. No significant sector-PFAS compound associations (FDR > 0.05) were found for Industry, Waste Treatment & Disposal, Airports, or Navigation.

Building on the PFAS compound - sectoral associations identified through Fisher's exact tests (Fig. 4), multivariable logistic regression models were then used to evaluate the influence of the Water Industry/Domestic/General Public pressures, while adjusting for overlapping pressures from other sectors. Significant positive associations (FDR < 0.05) were observed for PFHxS.L, PFBS, PFHpA PFOS..B, PFOS..L and PFOS-combined, with odds ratios between 2.0 and 3.0 (Fig. 5, Supplementary Table 6). Two additional PFAS compounds (i.e., PFHxA, and PFPeA), showed elevated OR with nominal significance ($p < 0.05$), though these did not remain significant after FDR correction. Logistic regression was not applied to the Urban and Transport sectors because these pressures overlap with Water Industry/Domestic/General Public pressures in >90% of waterbodies, preventing statistical isolation of their effects.

To test the robustness of the Water Industry/Domestic/General Public pressures - PFAS compound associations, a targeted sensitivity analysis was carried out to test whether the association was maintained when considering waterbodies influenced exclusively by these two activities. This analysis compared 73 waterbodies affected exclusively by the Water Industry/Domestic/General Public pressures with 92 waterbodies failing to achieve good status due to pressures from other sectors. Fisher's exact tests with FDR correction revealed significant positive associations (FDR < 0.05) across eight PFAS compounds, with higher OR ranging from 3.1 to 5.2 (Fig. 6, Supplementary Table 7). Equivalent sensitivity analyses were not feasible for the Urban and Transport

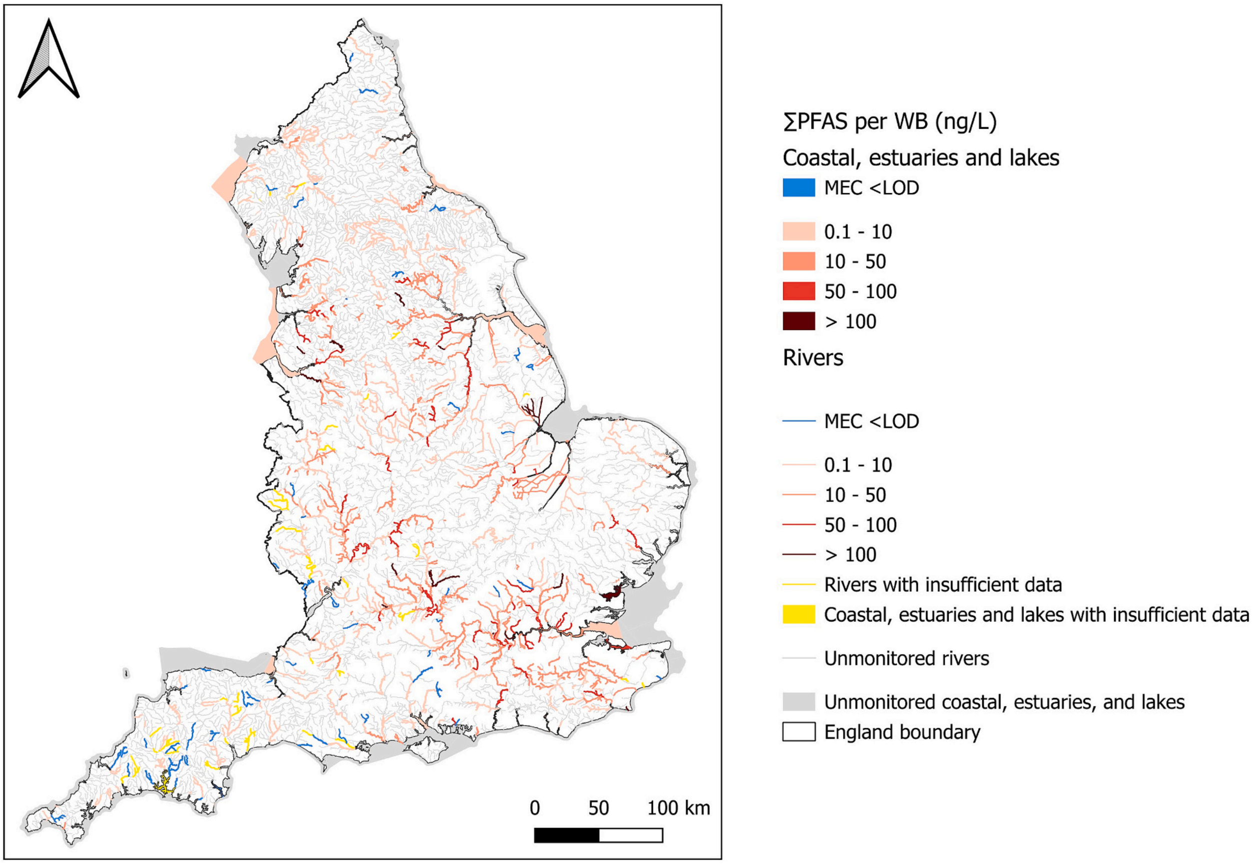

Fig. 2. Spatial distribution of summed PFAS concentrations (ΣPFAS, ng/L) in England's surface waterbodies in 2024. Results are shown for rivers (lines) and for lakes, estuaries, and coastal waters (polygons). Red shading represents measured environmental concentrations (MEC) above the analytical limit of detection (LOD), with darker shades indicating higher concentrations.

sectors because these pressures almost completely overlap with other sectoral activities.

3.2.1. *Fate of PFAS at WWTWs*

Ten distinct PFAS compounds were detected at frequencies that ranged between 94 and 100% and varying concentrations in treated effluents across the 684 WWTWs monitored in England under the CIP Phases 2 & 3 (Table 2). Perfluoropentane sulphonate exhibited the highest mean effluent concentration (~143 ng/L). Across compounds, median effluent concentrations spanned from about 1.9 to 143 ng/L. The maximum effluent concentrations recorded in the monitored WWTWs were 5711.82 ng/L for PFOS, followed by 636.43 ng/L for PFOA, with the other compounds being lower. Overall, sulfonates (PFHxS, perfluorobutane sulphonate, and perfluoropentane sulphonate) exhibit higher average or median concentrations than the carboxylic acids (PFPeA, PFHxA, PFHpA).

Across the 55 WWTWs with matched influent-effluent concentration pairs for PFAS compound PFAS removal efficiencies varied widely by compound and treatment type (Table 3). Yet a consistent pattern emerged: most treatment configurations failed to achieve meaningful removal, with many exhibiting net increases in PFAS concentrations post-treatment (Table 3). This systemic underperformance was particularly evident for Carboxymethyldimethyl-3-[[(3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl) sulfonyl]amino]propylammonium hydroxide, PFOA and PFOS with extreme minima down to −533.4, −485% and − 243% respectively. For PFOA, the mean removal was negative in 10/10 treatment categories, with medians from −81.9% to −4.4%. For PFOS, 8/10 categories showed negative or zero mean removal; only activated sludge and phosphate stripping had small positive means (15.1 and 0.9%). Short-chain PFCAs showed occasional positive means in specific treatment categories (e.g., PFHxA in tertiary biological 68.2%; PFPeA in tertiary biological 78.0%; PFHxA in phosphate stripping 17.8%), but most categories remained negative on average.

3.2.2. *Mapping of identified PFAS compounds to sector-specific product uses*

The PFAS compounds detected by the Environment Agency in 2024 were mapped to eleven product categories representing the four sectors associated with PFAS pollution in waterbodies in Section 3.1, namely, Water Industry, Public & Domestic use, Transport, and Urban. The product categories are: Apparel; Upholstery; Carpets and Floor Coverings; Floor Polishes; Cleaning compositions; Cosmetics and Personal Care Products (PCP); Cookware and baking ware; Paper and Food-Contact Packaging; Additives to hydraulic fluids; Architectural coatings & paints / Wetting & levelling agents; and Surface protection (Table 4).

Paper and Food-Contact Packaging and Apparel categories were each associated with 21 distinct PFAS compounds, making them the categories linked to the widest range of detected PFAS. Upholstery and Cosmetics/PCP were each associated with 16 PFAS, while Carpets and Floor Coverings and Floor Polishes were linked to 14 and 11 compounds, respectively. Fourteen (14) PFAS were associated with the Additives to hydraulic fluids category, either as a component or a degradation product of fluorinated compounds in them, and Architectural coatings & paints / Wetting & levelling agents applications accounted for eight PFAS. Cleaning Compositions: Cookware and baking ware; and Surface protection categories were linked to three PFAS each, (i.e., the fewest PFAS compounds associations). There are fewer categories associated with the Transport and Urban sectors (one and two, respectively) as compared to Water Industry, Public & Domestic use (eight categories

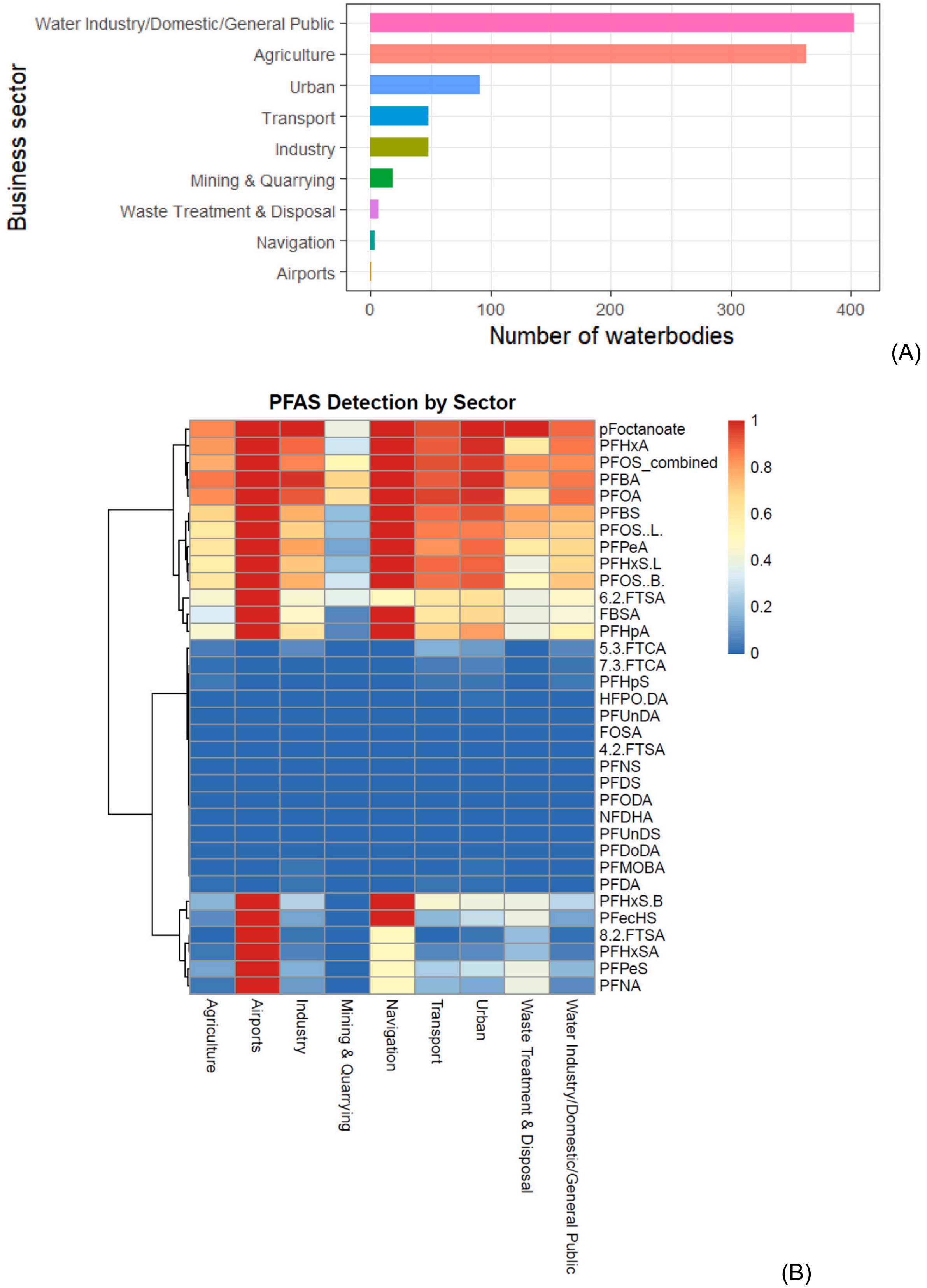

Fig. 3. Sector-specific waterbody failures to achieve “good status” under the WFD in England in waterbodies where PFAS were detected in 2024 (A), and the relative detection frequency of PFAS compounds across different sectors (B).

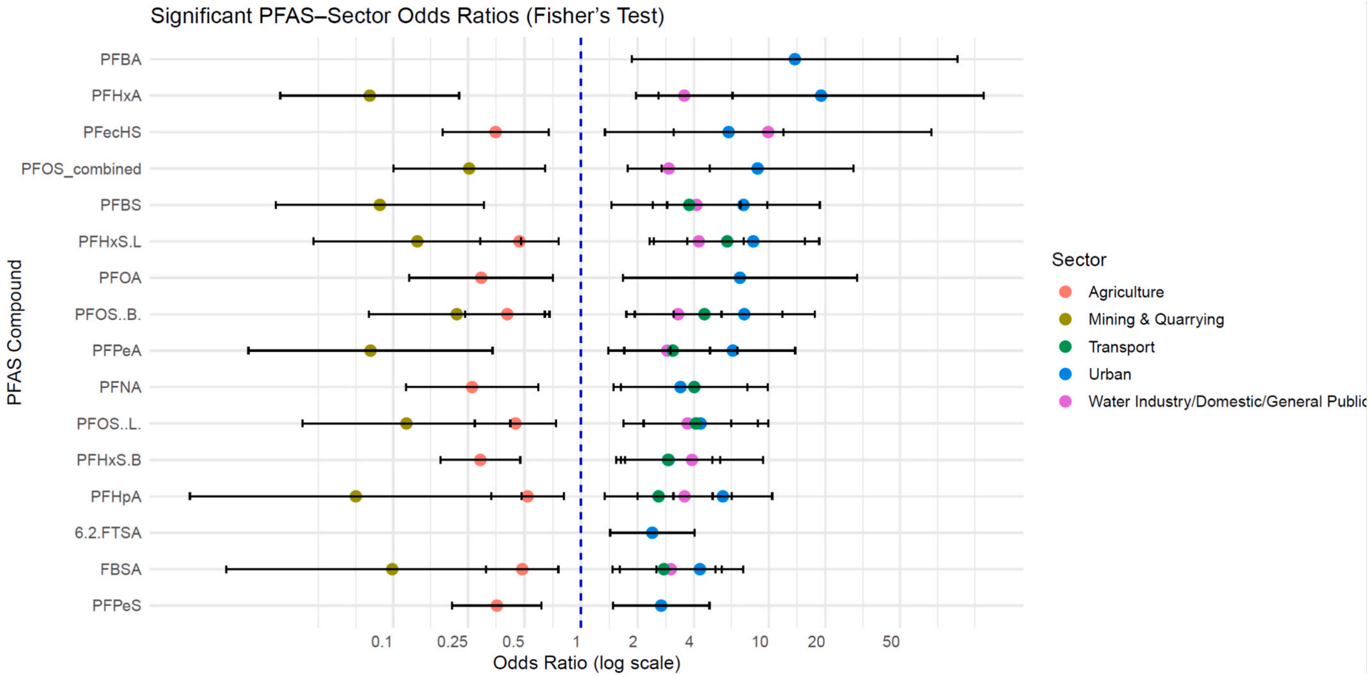

Fig. 4. Odds ratios (log scale) from Fisher's exact tests showing significant PFAS compounds- sector associations across England's waterbodies (FDR < 0.05).

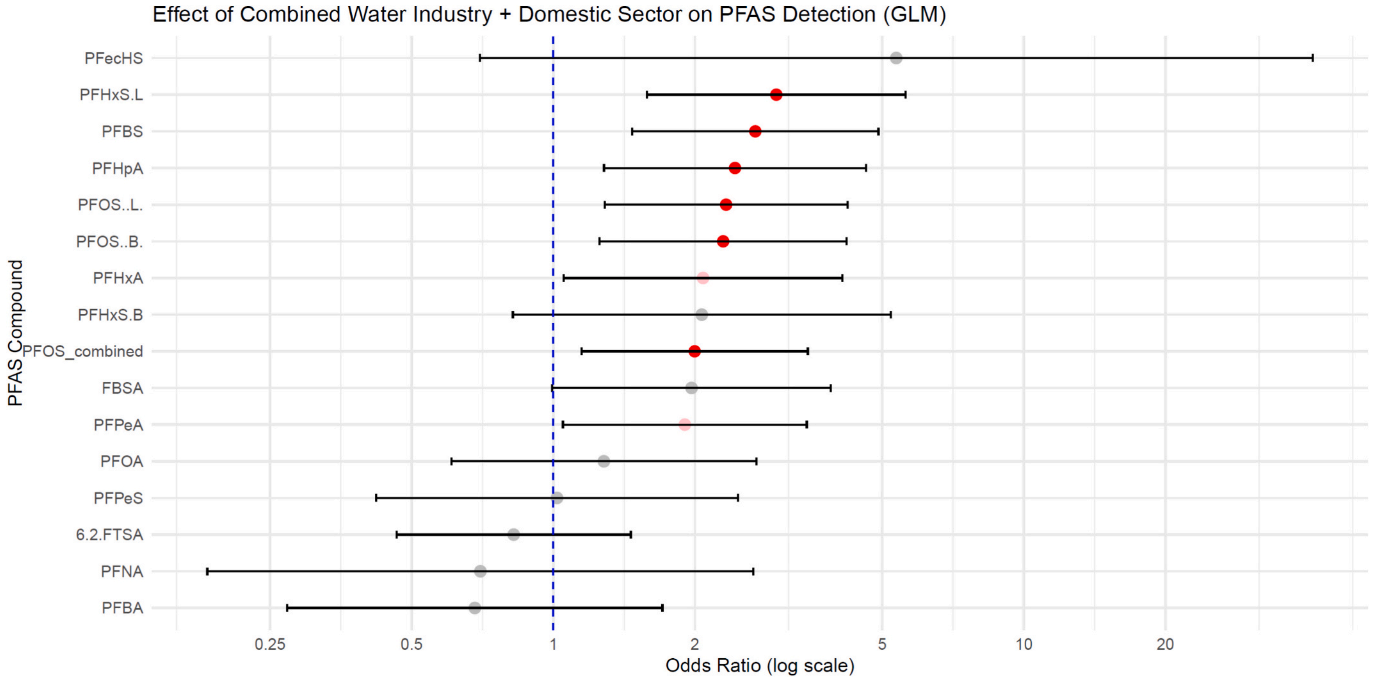

Fig. 5. Odds ratios (OR, log scale) from logistic regression models assessing the effect of combined Water Industry/Domestic/General Public pressures on PFAS detection across England's waterbodies. Red points indicate statistically significant associations after FDR correction (FDR < 0.05); pink points denote nominally significant associations ($p < 0.05$) that did not remain significant after correction; grey points represent non-significant associations ($p \geq 0.05$). Horizontal lines show 95% confidence intervals, and the dashed blue line at odds ratio = 1 marks the null effect threshold.

linked to the use of consumer products related to both sectors).

4. Discussion

This study provides the first surface waterbody-level characterisation of PFAS contamination across England, demonstrating that PFAS are widespread, chemically diverse, and strongly linked to urban water cycle pathways. Of the 51 PFAS compounds monitored in 2024, 34 were detected at least once in 665 waterbodies (92% of monitored waterbodies). These waterbodies contained an average of ~6.5 PFAS compounds, indicating the frequent co-occurrence of multiple PFAS species extending well beyond legacy PFOS and PFOA. PFOS frequently exceeded the defined EQS, and several other PFAS reached maximum concentrations in the 100–450 ng/L range. Detection patterns were broadly representative across waterbody types (Supplementary Table 9), and they were not influenced by variation in sampling intensity

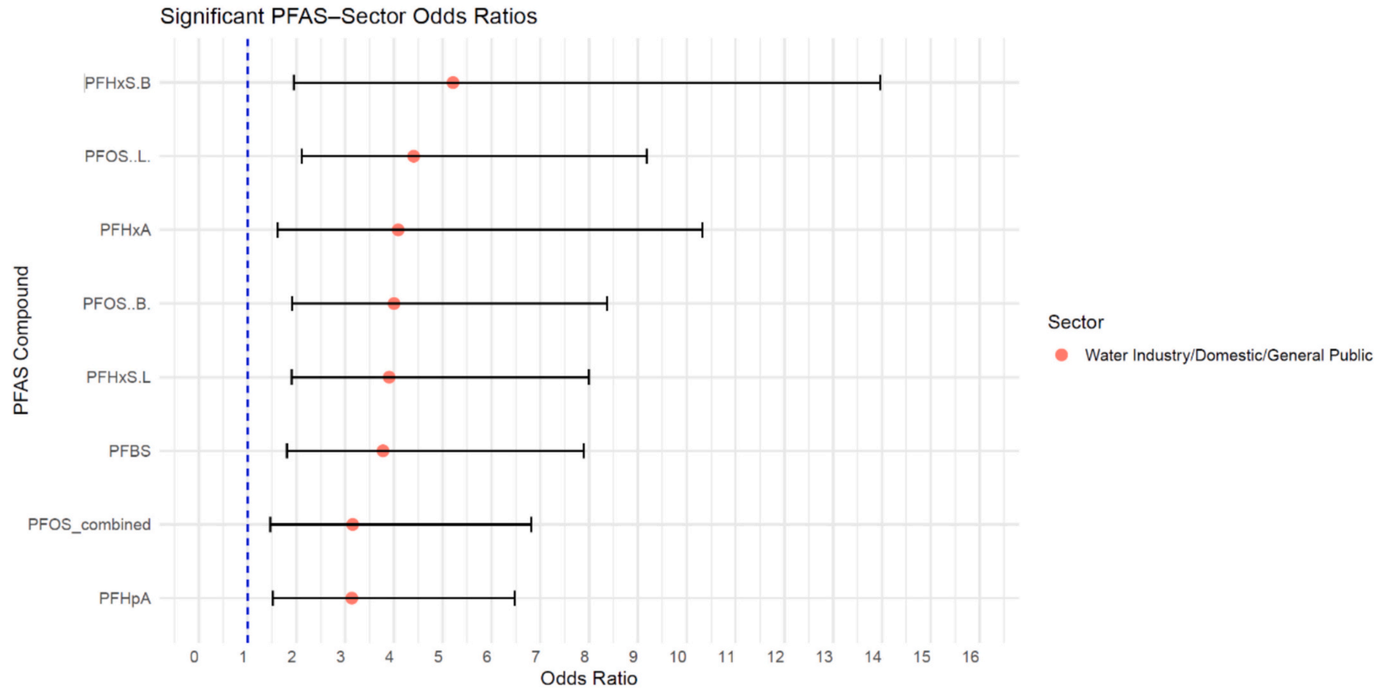

Fig. 6. Odds ratios from Fisher's exact tests showing significant PFAS associations with Water Industry/Domestic/General Public pressures (FDR < 0.05).

Table 2
PFAS detections and concentrations in the treatment effluent of 684 distinct WWTWs monitored under the Chemical Investigation Programme (CIP) Phases 2 & 3. Shown are the number of WWTWs monitored, WWTWs with detections of PFAS in their effluents (*), and WWTWs with PFAS detected with sufficient data (†). Summary statistics of concentrations (ng/L) are reported for WWTWs with sufficient data. Similar statistics are provided on Supplementary Table 8 for the WWTWs x PFAS compound groups that were excluded from ROS analyses.

PFAS Compound	WWTWs Monitored ($n = 684$)		WWTWs with detections*		WWTWs with detections and sufficient data†		Concentrations (ng/L)				
	(n)	(%)	(n)	(%)	(n)	(%)	Min	Median	Mean	Max	Sd
3,3,4,4,5,5,6,6,7,7,8,8,8- Tridecafluorooctanesulfonic acid	16	2.3	15	93.8	13	86.7	1.00	3.29	39.05	408.88	111.92
Carboxymethyldimethyl-3-[[(3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl)sulfonyl]amino]propylammonium hydroxide	16	2.3	16	100.0	15	93.8	1.67	17.53	41.78	138.64	46.60
PFOA	635	92.8	634	99.8	633	99.8	0.55	4.41	7.78	636.43	30.95
PFOS	668	97.7	660	98.8	654	99.1	0.48	4.09	15.77	5711.82	223.59
PFBS	37	5.4	37	100.0	37	100.0	0.86	15.00	34.09	170.00	38.71
PFHpA	19	2.8	18	94.7	18	100.0	0.62	1.89	2.85	8.18	2.30
PFHxS	31	4.5	29	93.5	27	93.1	1.00	21.00	32.37	116.50	34.85
PFHxA	28	4.1	28	100.0	28	100.0	1.19	4.52	6.23	25.50	5.56
Perfluoropentane sulfonate	2	0.3	2	100.0	2	100.0	56.00	143.00	143.00	230.00	123.04
PFPeA	18	2.6	18	100.0	18	100.0	0.57	4.43	5.69	13.47	3.45

(Supplementary Table 10).

Water Industry/Domestic/General Public pressures were the dominant contributors at the national scale, affecting ~77% of the waterbodies assessed. PFHxS.L, PFBS, PFHpA, PFOS..B, PFOS..L and PFOS_combined were the compounds most consistently associated with these pressures. They were found to be among the most frequently detected PFAS compounds in 2024 and linked to everyday consumer product categories. Importantly, 73 waterbodies influenced exclusively by Water Industry/Domestic/General Public pressures exhibited elevated PFAS detection across eight compounds. This demonstrates that wastewater pathways alone can generate strong PFAS signatures in receiving waters, even in the absence of additional pressures.

The strongest and most chemically diverse PFAS signatures were linked to the Urban and Transport sectors, which occur in only ~18% and ~ 10% of waterbodies studied, respectively. The compounds driving these associations (e.g., PFBS, PFHxS.L, PFOS..B) align closely with PFAS in additives, adhesives, coatings and paintings used across urban and transport applications (Kim-fu et al., 2024; M. Liu et al., 2025; Mahoney et al., 2022; Zhu and Kannan, 2020). These patterns indicate chemically possible source linkages between sector-specific uses and the mixture profiles observed in receiving waters. However, these sectoral pressures rarely occur independently. This suggests a cumulative effect of sectoral pressures on PFAS contamination.

Specifically, among the waterbodies that failed to achieve good status, 94% of those failing due to Transport, and 91% of those failing due to Urbanisation and Urban Development, also failed due to Water Industry/Domestic/General Public pressures. This near complete co-occurrence indicates that the PFAS compound signatures attributed to Urban and Transport are almost always embedded within Water Industry/Domestic/General Public contexts, suggesting wastewater-mediated delivery. Further evidence comes from the quality elements the Environment Agency attributes to Urbanisation/Urban Development and Transport in these waterbodies. These failures predominantly concern phosphates, ammonia, and dissolved oxygen-indicators of

Table 3
Removal efficiencies across WWTWs for six PFAS compounds. Summary statistics are reported per WWTWs treatment type and per PFAS compound. The total amount of WWTWs with matched influent-effluent concentration pairs for PFAS compounds is 55.

PFAS Compound	Treatment technique	WWTWs (n)	Removal Efficiency (%)				
			Min	Mean	Median	Max	Sd
3,3,4,4,5,5,6,6,7,7,8,8,8-Tridecafluorooctanesulfonic acid	Biological filtration	2	−50.0	−3.6	−3.6	42.9	65.7
	Chemical & biological	1	16.7	16.7	16.7	16.7	NA
	Phosphate stripping	2	−66.7	−36.2	−36.2	−5.7	43.1
Carboxymethyldimethyl-3-[[(3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctyl)sulfonyl]amino] propylammonium hydroxide	Biological filtration	3	−470.0	−265.7	−242.6	−84.4	193.8
	Chemical & biological	1	−88.1	−88.1	−88.1	−88.1	NA
	Phosphate stripping	4	−533.4	−323.9	−353.5	−55.2	229.5
	Tertiary biological	1	−64.3	−64.3	−64.3	−64.3	NA
PFOA	Activated sludge	4	−91.3	−33.8	−39.0	34.0	59.4
	Biological filtration	15	−485.8	−77.0	−16.5	39.4	150.4
	Chemical & biological	1	−44.2	−44.2	−44.2	−44.2	NA
	Phosphate stripping	13	−162.7	−36.1	−16.7	25.5	54.3
	Membrane filtration	1	−72.6	−72.6	−72.6	−72.6	NA
	None	1	−68.3	−68.3	−68.3	−68.3	NA
	Sand filtration	1	−4.4	−4.4	−4.4	−4.4	NA
	Tertiary biological	1	−40.5	−40.5	−40.5	−40.5	NA
	Uv disinfection	7	−120.3	−51.7	−39.3	−1.2	44.8
	Na	6	−156.3	−79.3	−81.9	−12.8	50.6
PFOS	Activated sludge	5	−12.1	15.2	4.0	80.5	37.3
	Biological filtration	15	−82.6	−11.6	−3.6	22.9	28.3
	Chemical & biological	1	−6.5	−6.5	−6.5	−6.5	NA
	Phosphate stripping	14	−77.1	0.9	6.9	62.1	40.7
	Membrane filtration	1	−22.5	−22.5	−22.5	−22.5	NA
	None	1	−6.2	−6.2	−6.2	−6.2	NA
	Sand filtration	1	−28.3	−28.3	−28.3	−28.3	NA
	Tertiary biological	1	−86.2	−86.2	−86.2	−86.2	NA
	Uv disinfection	7	−71.5	−3.5	5.8	42.9	41.6
	Na	6	−243.1	−38.9	0.7	51.1	106.8
Perfluorobutane sulphonate	Biological filtration	3	−112.8	−5.6	25.0	71.0	95.7
	Chemical & biological	1	−70.8	−70.8	−70.8	−70.8	NA
	Phosphate stripping	5	−102.4	5.9	6.4	87.1	74.6
PFHpA	Biological filtration	2	−66.7	−33.3	−33.3	0.0	47.1
	Chemical & biological	1	−30.8	−30.8	−30.8	−30.8	NA
	Phosphate stripping	4	−2.4	15.5	10.2	43.8	21.5
PFHxS	Biological filtration	1	−33.3	−33.3	−33.3	−33.3	NA
	Chemical & biological	1	−44.4	−44.4	−44.4	−44.4	NA
	Phosphate stripping	3	−28.6	−11.3	−3.3	−2.0	15.0
PFHxA	Biological filtration	3	−105.6	−57.6	−94.5	27.3	73.7
	Chemical & biological	1	−15.7	−15.7	−15.7	−15.7	NA
	Phosphate stripping	4	−9.4	17.8	15.1	50.4	25.9
	Tertiary biological	1	68.2	68.2	68.2	68.2	NA
PFPeA	Biological filtration	3	−100.8	−26.3	−50.0	71.8	88.7
	Chemical & biological	1	−51.9	−51.9	−51.9	−51.9	NA
	Phosphate stripping	4	−4.6	21.4	12.6	65.1	30.4
	Tertiary biological	1	78.0	78.0	78.0	78.0	NA

Table 4
Summary of PFAS applications. The numerical code goes as follows: (1) Apparel, (2) Upholstery, (3) Carpets and floor coverings, (4) Floor polishes, (5) Cleaning compositions, (6) Cosmetics and personal-care products (PCP), (7) Cookware and baking ware, and (8) Paper and food-contact packaging (9) Additives to hydraulic fluids in transport vehicles, (10) Architectural coatings and paints / Wetting & levelling agents, (11) Surface protection. The a,b,c identifiers denote the source used to assign each application category: (a) Glüge et al., (b) Food Packaging Forum databases, (c) peer-reviewed papers.. DP indicates relevance as degradation products. The sectors were abbreviated as follows: (WI/D/GP) Water Industry /Domestic / General Public, (U) Urban, (T) Transport.

PFAS subgroup	Compound	Uses related to consumer products	Uses related to the Transport sector	Uses related to the Urban sector	Is it a degradation product?	Significant sectoral associations (Fisher's exact tests, FDR < 0.05)	Significant in multivariable logistic regression (WI/D/GP sectors, FDR < 0.05)	Significant in WI/DG impacted only waterbodies (sensitivity analysis, FDR < 0.05)
Perfluoroalkyl Carboxylic Acids (PFCAs)	PFBA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 5^{a}, 6^{a}, 8^{a}	9	10^{DP}	Yes	U	–	–
	PFPeA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 6^{a}, 8^{a}	9	10^{DP}	Yes	WI/D/GP, U, T	–	–
	PFHxA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 6^{a}, 8^{a}	9	10^{DP}	Yes	WI/D/GP, U	–	Yes
	PFHpA	1^{a}, 2^{a}, 3^{a}, 6^{a}, 8^{a}	9 (also as a DP)	–	Yes	WI/D/GP, U, T	Yes	Yes
	PFOA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 5^{a}, 6^{a}, 7^{a}, 8^{a}	9 (also as a DP)	–	Yes	U	–	–
	PFNA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 6^{a}, 8^{a}	9	–	–	U, T	–	–
	PFDA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 5^{a}, 6^{a}, 8^{a}	9	–	–	–	–	–
	PFunDA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 6^{a}, 8^{a}	9	–	–	–	–	–
	PFDoDA	1^{a}, 2^{a}, 3^{a}, 4^{a}, 8^{a}	9	–	–	–	–	–
	PFODA	7^{c}, 8^{b}	–	–	–	–	–	–
Perfluoroalkyl Ether Carboxylic Acid (PFECAs)	HFPO-DA	6^{c}, 8^{b}	–	–	–	–	–	–
	PFMOBA	–	–	–	–	–	–	–
	NFDHA	–	–	–	–	–	–	–
Fluorotelomer Carboxylic Acids (FTCAs)	5:3 FTCA	8^{b}	–	10^{DP}	Yes		–	–
	7:3 FTCA	8^{b}	–	10^{DP}	Yes	–	–	–
Perfluoroalkyl Sulfonic Acids (PFSAs)	PFBS	1^{a}, 2^{a}, 3^{a}, 4^{a}, 7^{a}, 8^{b}	9^{DP}	10, 11	Yes	WI/D/GP, U, T	Yes	Yes
	PFPeS	6^{c}	–	–	–	U	–	–
	PFHxS*	1^{a}, 2^{a}, 3^{a}, 6^{c}, 8^{b}	9	–	Yes	WI/D/GP, U, T	Yes	Yes
	PFHpS	2^{a}, 8^{b}	–	–	–	–	–	–
	PFOS*	1^{a}, 2^{a}, 3^{a}, 8^{a}	9 (also as a DP)	–	Yes	WI/D/GP, U, T	Yes	Yes
	PFNS	1^{c}	–	–	–	–	–	–
	PFDS	1^{c}, 2^{a}, 4^{a}, 8^{b}	9	–	–	–	–	–
	PFUnDS	–	–	–	–	–	–	–
Cyclic Perfluorinated Acids (PFA)	PFecHS	–	–	10	–	WI/D/GP, U	–	–
Perfluoroalkane Sulfonamides (FASAs)	FBSA	1	–	10, 11^{DP}	Yes	WI/D/GP, U, T	–	–
	PFHxSA	–	–	–	–	–	–	–
	FOSA	1^{a}, 2^{c}, 6^{c}, 8^{b}	–	–	–	–	–	–
Fluorotelomer Sulfonic Acids (FTSAs)	4:2 FTSA	1^{c}, 6^{c}	–	–	–	–	–	–
	6:2 FTSA	1^{ac}, 3, 6^{ac}, 8^{b}	–	–	–	U	–	–
	8:2 FTSA	1^{a}, 6^{c}	–	–	–	–	–	–

urban wastewater influence rather than isolated land-runoff pathways. Most urban sewer networks in England are combined systems, known as combined sewer overflows (CSOs). During rainfall-driven surcharge events, CSOs can further mobilise PFAS compounds via untreated, but diluted (from stormwater) wastewater, which is discharged to receiving waters via storm overflows (Giakoumis and Voulvoulis, 2026). These mobilisations explain the high PFAS compound diversity associated with the Urban and Transport sectors, and highlight that addressing PFAS pollution requires a systemic perspective that accounts for overlapping sectoral influences rather than isolated mitigation measures.

Moreover, PFAS are routinely detected in WWTWs' effluents. Despite the UK's widespread ban and heavy restriction of PFOS and PFOS-related substances, enforced in 2011, measurable PFOS levels in WWTWs' effluents persist. This can be explained by its continuous release from legacy PFOS-treated consumer products across their lifecycle (i.e., use, laundering and end-of-life pathways) (Schellenberger et al., 2022; Supreeyasunthorn et al., 2016); the continuous use of precursor compounds in products (Buck et al., 2011; Kurwadkar et al., 2022; J. Liu and Mejia Avendaño, 2013); and the remobilisation of PFOS-laden sediments in sewer networks. This highlights a substantial temporal lag between regulatory restriction and observable reductions in effluent concentrations. PFOA, which is also a widely detected PFAS compound in England's WWTWs, has only recently been officially restricted, with a full ban on its use in firefighting foams taking effect in July 2025. Given its pre-ban prevalence, similar legacy-driven persistence to PFOS should be anticipated for PFOA but also for other PFAS compounds if future restrictions extend to short-chain alternatives or fluorotelomer precursors (Liwara et al., 2025; Ribes Ortega et al., 2025; Schellenberger et al., 2019).

Across treatment configurations and PFAS compounds, many WWTWs exhibited net increases in PFAS concentrations post-treatment, which is consistent with previous studies reporting low or even negative PFAS removal efficiencies in municipal WWTWs (Campo et al., 2014; Gobelius et al., 2023; Moneta et al., 2023; Kibambe et al., 2020; Lenka et al., 2022; C. Zhang et al., 2015). This accumulation is largely attributed to the transformation of precursor compounds into persistent PFAS compounds during the treatment process (Moneta et al., 2023; Liu and Mejia Avendaño, 2013; Tavasoli et al., 2021); effectively turning conventional WWTWs into chronic point sources that discharge higher PFAS concentrations in effluents as compared to the influents (Thompson et al., 2022; Moneta et al., 2023; Xiao et al., 2012). This creates a governance dilemma: water companies are legally obliged to collect and treat wastewater, but often lack scalable, cost-effective technologies to prevent PFAS discharge. This challenge is compounded by the fact that many waterbodies receiving treated wastewater effluents are often abstracted downstream for drinking water supply, creating a direct human exposure pathway (Sunderland et al., 2019). Concurrently, conventional drinking water treatment technologies or water blending techniques are largely ineffective at removing these chemicals, allowing PFAS to recirculate through the urban water cycle.

Advanced treatment processes capable of removing PFAS are under development, including destructive technologies, such as photodegradation, sonolysis, electrochemical degradation, advanced oxidation processes, thermal degradation, biodegradation and enzymatic degradation (Antonopoulou et al., 2024; Guo et al., 2025; Mekureyaw et al., 2025; Mirabediny et al., 2023; Niu et al., 2024; Sidnell et al., 2022). While promising at bench scale, their large-scale application remains uncertain due to high energy demands and associated carbon emissions, financial and environmental costs, and complex design requirements (Tshangana et al., 2025; Verma et al., 2024; Zang et al., 2025). Compared with conventional methods, these technologies also involve higher operational costs and variable effectiveness in removing PFAS precursor compounds (Hao et al., 2021; Singh et al., 2019).

The financial burden of cleaning up wastewater and drinking water contaminated with PFAS is borne by water companies, which also incur significant revenue losses from diverting PFAS-contaminated sludge to landfills at increasing landfill tax rates (Cordner et al., 2021). Ultimately, these costs are passed onto customers or taxpayers who routinely and disproportionately bear the long-term health and environmental consequences of PFAS exposure. Failure to remove PFAS effectively during the water treatment process allows these pollutants accumulate over time and recirculate in the urban water cycle (sludge, wastewater, drinking water); exacerbating exposure and environmental persistence (Fredriksson et al., 2022; Winchell et al., 2022).

From a governance perspective, this evidence exposes a dual challenge. First, diffuse, consumer-driven sources of PFAS are difficult to regulate and control downstream, indicating the need for upstream interventions in the product value chain (e.g., product substitution, EPR, and chemical regulation using a Safe and Sustainable by Design framework). Second, legacy hotspots and the long environmental residence time of PFAS mean that contamination persists long after the use of PFAS is restricted. This exemplifies a “tragedy of the commons” (Hardin, 2009), where chemical manufacturers and product developers historically benefited from PFAS's lucrative applications, while the costs of diffuse persistent pollution and health risks were externalised to the public, water companies and freshwater systems.

This dual challenge has already triggered major polluter-pays litigation internationally, yielding settlements between fluorochemical companies and environmental regulators, public water utilities, and government prosecutors across Europe and the United States throughout the 2020s (United States: (City of Camden, et al., v. 3M Company, 2024; City of Camden, et al., v. Tyco Fire Products LP, 2024; New Jersey Department of Environmental Protection, et al. v. E.I. Du Pont Nemours and Company, et al, 2025; New Jersey Department of Environmental Protection v. Solvay Specialty Polymers USA, LLC, 2024); Europe: *Flemish Government v. 3 M Belgium* (Flemish Government, 2022); (Corte d'Assise Di Vicenza – Miteni PFAS Criminal Judgment, 2025)). These settlements are used to finance critical interventions, including, monitoring and characterisation efforts in affected and suspected waterbodies (New Jersey Department of Environmental Protection v. Solvay Specialty Polymers USA, LLC, 2024); in-situ remediation efforts (Flemish Government v. 3M Belgium; New Jersey Department of Environmental Protection, et al. v. E.I. Du Pont Nemours and Company, et al, 2025; New Jersey Department of Environmental Protection v. Solvay Specialty Polymers USA, LLC, 2024); the development of remedial strategies to prevent further pollution in the natural environment, and treatment technologies for public water suppliers (City of Camden, et al., v. 3M Company, 2024; City of Camden, et al., v. Tyco Fire Products LP, 2024). This emerging governance pathway could offer an action model for the UK, complementing regulatory and product-control measures.

Notwithstanding, the international restrictions on PFOA, PFOS and a few other perfluorinated carboxylic and sulfonic acids have created a shift towards alternative PFAS, such as short- and ultra-short-chain representatives, *per*- and polyfluorinated oxo carboxylic acids, telomeric alcohols and acids (Brunn et al., 2023). While short-chain and ultra-short PFAS are generally understood to be less bioaccumulative than their long-chain analogues (Brendel et al., 2018; Chambers et al., 2021), their increased mobility and environmental persistence have led to widespread detection across aquatic and terrestrial systems (Environment Agency, 2021; Li et al., 2020; Neuwald et al., 2022; Zhi et al., 2024). This reactive, compound-by-compound substitution, often involving sparsely investigated substances, fails to address the systemic nature of chemical pollution. The detection of regulated (or outright banned) PFAS in England's waterbodies (i.e., PFOA, PFOS and PFHxS, all listed under the Stockholm Convention and UK REACH) evidences the shortcomings of the current approach. A precautionary, comprehensive, group-based regulatory approach, combined with lifecycle controls, monitoring, product bans for non-essential uses, and investment in scalable destructive technologies, would reduce the likelihood of repeated substitution cycles. The lag of approximately 14 years between PFOS and PFOA bans underscores the slow pace of PFAS phaseouts and the urgent need to align technical responses with governance.

5. Conclusion

PFAS contamination in England's surface waters is both pervasive and embedded within the urban water cycle. Nationally, Water Industry/Domestic/General Public pressures are the primary pathways of PFAS leaching into freshwater systems, with Urban and Transport sectors contributing the highest chemical diversity locally, almost always in combination with wastewater-mediated influences. These patterns reflect a cumulative effect driven by the role of sewerage networks and CSOs in mobilising PFAS from diffuse consumer and urban sources.

Effluent monitoring confirms that WWTWs routinely discharge PFAS, with PFOS and PFOA detected almost universally. Persistent PFOS detections more than a decade after restriction, and extensive PFOA detections ahead of the 2025 ban, highlight the enduring influence of legacy stocks in consumer products, buildings and sewer networks. These findings suggest that PFAS are now deeply embedded within the built environment and recirculate through the urban water cycle, reinforcing the need for a comprehensive, system-level governance approach for PFAS that moves beyond single-compound restriction. Although direct correlations between PFAS characteristics in water bodies and the removal efficiencies of PFAS in WWTWs could not be established due to the use of temporally and spatially divergent datasets, harmonised monitoring would enable a more detailed assessment of the magnitude and intensity of WWTW contributions, thereby strengthening evidence base for policy development.

Nevertheless, relying solely on monitoring harmonisation and regulatory bans is insufficient in view of legacy stocks, ongoing precursor use and widespread substitution to short- and ultra-short-chain alternatives. Current water and wastewater treatment technologies cannot prevent PFAS leakage, and the sector cannot absorb the financial and operational costs of action without dedicated fiscal and regulatory support. This emphasises the need for an effective mitigation strategy that combines coordinated upstream interventions across the PFAS value chain, with sustained fiscal investment in the development and deployment of scalable PFAS destruction and removal technologies.

Polluter-pays mechanisms, already introduced at the international level, offer a viable governance tool to support monitoring, remediation and infrastructural upgrades needed to address this systemic issue, which has long remained under-regulated. Without comprehensive systemic reforms, PFAS contamination will continue to accumulate in England's surface waters, exemplifying a classic “tragedy of the commons”, in which individual benefits are externalised as persistent and potentially irreversible environmental and human health costs.

CRediT authorship contribution statement

Ana García Herrera: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation. **Eleni Iacovidou:** Writing – review & editing, Validation, Supervision, Investigation, Conceptualization, Funding acquisition, Writing – original draft. **Theodoros Giakoumis:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, 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. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2026.181779.

Data availability

Data will be made available on request.

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