Marine Environmental Research xxx (xxxx) xxx

Contents lists available at ScienceDirect

Marine Environmental Research

journal homepage: www.elsevier.com/locate/marenvrev

PFAS contaminants in surface waters, effluents, sediments and coastal food webs off the Solent coastline UK

Henry E. Obanya [a,*], Francesca Ginley [b], Millicent R. Payne [a], Alex T. Ford [a,**]

[a] School of Biological Sciences, University of Portsmouth, Portsmouth, Hampshire, PO1 2DY, United Kingdom
[b] Marine Conservation Society, Ross-on-Wye, HR9 7UK, USA

ARTICLE INFO

Keywords:
Per- and polyfluoroalkyl substances (PFAS)
Coastal pollution
Wastewater effluent
Marine biota
Bioaccumulation
PFOS
Food webs
Environmental quality standards
UK coastal waters
Solent

ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) are persistent coastal contaminants. Their distribution across environmental compartments and food webs remains poorly understood. The Solent is a densely urbanised tidal strait representative of industrialised coastal systems. Inputs from wastewater treatment plants, combined sewer overflows, and historic landfills create multiple PFAS pathways to the marine environment. We integrated our collected field samples (including surface waters, n = 3; and multiple biota species collected in triplicate) with regulatory monitoring datasets (sediment and archived biota) and wastewater effluent records to quantify PFAS across surface waters, sediments, treated wastewater effluent, and marine biota from the Solent. In sediments, ΣPFOS was present at the highest mean concentrations, at 0.5 μg kg^{-1} dry weight. Surface waters and effluents contained a broader mixture of short- and long-chain PFAS. Mean ΣPFOS concentrations in surface water reached 8.5 ng L^{-1}, exceeding the UK/EU environmental quality standard for coastal waters (0.65 ng L^{-1}). Treated effluent contained several short-chain PFAS, including PFHxA, PFBA, and PFBS, at mean concentrations of 1.3–5.9 ng L^{-1}. PFOA (3.1–4.0 ng L^{-1}) and PFOS (5.7–7.2 ng L^{-1}) were also consistently detected. In biota, ΣPFOS exceeded the biota EQS (9.1 μg kg^{-1} wet weight) only in harbour porpoise liver, with a mean concentration of 341.39 μg kg^{-1}. However, when expressed as PFOA-equivalents using EU relative potency factors (RPFs), most of the biota samples exceeded the EFSA benchmark of 77 ng/kg. However, twenty-three samples from Langstone harbour had no detectable PFAS. Multivariate analyses showed that PFAS composition varied among species but was not associated with literature-based trophic grouping (PERMANOVA R^2 = 0.058, $p > 0.05$). Although ecological EQS exceedance was limited to PFOS in specific matrices, application of RPFs revealed widespread exceedance of mixture-based health benchmarks. These findings demonstrate that reliance on single-compound regulatory thresholds may underestimate cumulative PFAS risk in UK coastal systems. A mixture-based toxic equivalency approach exposes potential regulatory blind spots in current PFAS assessment frameworks.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a large group of synthetic chemicals used in a wide range of industrial and consumer products (Glüge et al., 2020; Wee and Aris, 2023a). Their resistance to heat, water, grease, and degradation makes them useful in everything from firefighting foams (Anderson et al., 2025) to waterproof textiles (Pervez et al., 2025) and non-stick cookware (Sajid and Ilyas, 2017). Unfortunately, these same properties mean they do not break down easily in the environment. As a result, PFAS have become some of the most persistent organic pollutants in global ecosystems and are often referred to as "forever chemicals (Buck et al., 2011; Wang et al., 2017). PFAS differ in their chain length and functional groups. This influences their environmental fate and bioaccumulation. Short-chain PFAS (e.g., PFBA, PFPeA) are generally more water-soluble and mobile (Ohoro et al., 2024). Long-chain PFAS (e.g., PFOS, PFOA, PFNA) exhibit stronger sorption to sediments and greater potential for biomagnification (Wee and Aris, 2023b). These compounds have been shown to build up in marine food webs (Garcia-Garin et al., 2023; Ibor et al., 2025) and may cause harm to wildlife. This could be through

* Corresponding author.
** Corresponding author.
E-mail addresses: henry.obanya@port.ac.uk (H.E. Obanya), alex.ford@port.ac.uk (A.T. Ford).

https://doi.org/10.1016/j.marenvres.2026.108094
Received 19 February 2026; Received in revised form 14 April 2026; Accepted 30 April 2026

immune (DeWitt et al., 2012; Lukić Bilela et al., 2023), liver (Sciancalepore et al., 2021), and neurological effects (Brown-Leung and Cannon, 2022; Ríos-Bonilla et al., 2024). Studies have detected PFAS in organisms at all levels of the marine food chain, including top predators such as dolphins (Foord et al., 2024), seals (Sait et al., 2023), and seabirds (Lu et al., 2024) [e.g., gannet eggs in the UK show high PFOS fractions with persistent long-chain PFCAs (Pereira et al., 2021)].

In the UK, awareness of PFAS contamination is growing. Monitoring programmes have detected PFAS in surface waters across a wide range of locations (Folorunsho et al., 2025; Megson et al., 2024). However, there is still relatively little information on how these compounds accumulate in marine organisms, especially those living near sources of wastewater discharge and landfills. This is a significant gap in our understanding, as some PFAS are now known to exceed regulatory limits (EQS and Norman) in UK surface waters (Ford and Ginley, 2024), sediments (Ma et al., 2025), and occasionally in biota (Environment Agency, 2019). UK and other European surveys of edible fish muscle report PFOS/PFOSA as frequent detections at levels relevant to dietary exposure (Fernandes et al., 2018). Biota PFOS monitoring data from seven European Member States also highlight recurring hotspots near airports, fire-training facilities, and industrial/waste sites (Georges and Johansson, 2025). These findings show that PFAS contamination is widespread across coastal environments and food webs. However regulatory frameworks largely assess PFAS on a single-compound basis, such as PFOS and PFOA (Kurwadkar et al., 2022; Simpson et al., 2021). This contrasts with real-world environmental exposure, where PFAS occur as complex mixtures of multiple compounds (Ahrens and Bundschuh, 2014; Wang et al., 2017). In response, increasing attention has been given to mixture-based approaches that aim to better capture cumulative exposure and associated risks (Reinikainen et al., 2024). One such approach applies RPFs to derive toxic equivalency metrics, such as PFOA-equivalents (Reinikainen et al., 2024; (EFSA CONTAM Panel et al., 2020). This enables different PFAS to be assessed within a common risk framework. Despite this, mixture-based approaches remain rarely applied in coastal environmental assessments. PFAS can reach the marine environment through several different pathways. These include industrial releases, leachate from historic landfills, treated wastewater, and discharges from combined sewer overflows (CSOs).

The Solent, a tidal strait on the south coast of England separating the Isle of Wight from mainland Hampshire (Fig. 1). It is bordered by numerous legacy landfill sites, many of which are situated within low-lying coastal floodplains (Nicholls et al., 2021). Recent regional assessments have identified multiple Solent habitats, including saltmarsh, seagrass, Oyster reefs, intertidal chalk and subtidal kelp beds, as being in unfavourable or declining condition (Solent Protection Society, 2024b; Solent Restoration News, 2024). Water quality impairment and pollution pressures are recognised as contributing stressors (Solent Restoration News, 2024). Landfilling in the UK was largely unregulated until the Control of Pollution Act (1974) (Legislation, 1974). The systematic requirements for waste characterisation and engineered containment were not introduced until the Waste Management Licensing Regulations (1994) (Legislation, 1994) and the Landfill (England and Wales) Regulations (2002) (Legislation, 2002), respectively. Where records exist, historic landfills typically contain heterogeneous mixtures of domestic and industrial waste (Wadey et al., 2019). This regulatory legacy means that the composition and containment integrity of many sites are not fully known. On the other hand, limited funding for proactive management has rendered remediation unlikely in the absence of demonstrable risks to public health or the environment (Wadey et al., 2019). An increasing body of evidence has identified landfills as significant sources of PFAS to surrounding soils, groundwater, and surface waters (Capozzi et al., 2023; Lang et al., 2017; Sabba et al., 2025). Compounding this pressure, the Solent relies on ageing sewer systems with a high number of CSO outfalls. Wastewater treatment plants (WWTPs) are not currently designed to remove these PFAS effectively (Coggan et al., 2019; Kim et al., 2024; Pan et al., 2016; Yadav et al., 2022). In addition, much of the UK's sewer system includes combined sewer overflows (CSOs), which allow untreated sewage to be released directly into rivers, estuaries, and the sea during periods of heavy rainfall or system overload. Many of these CSO outlets are located close to sensitive and legally protected marine areas (Perry et al., 2024).

A recent study in Langstone Harbour, a designated marine protected

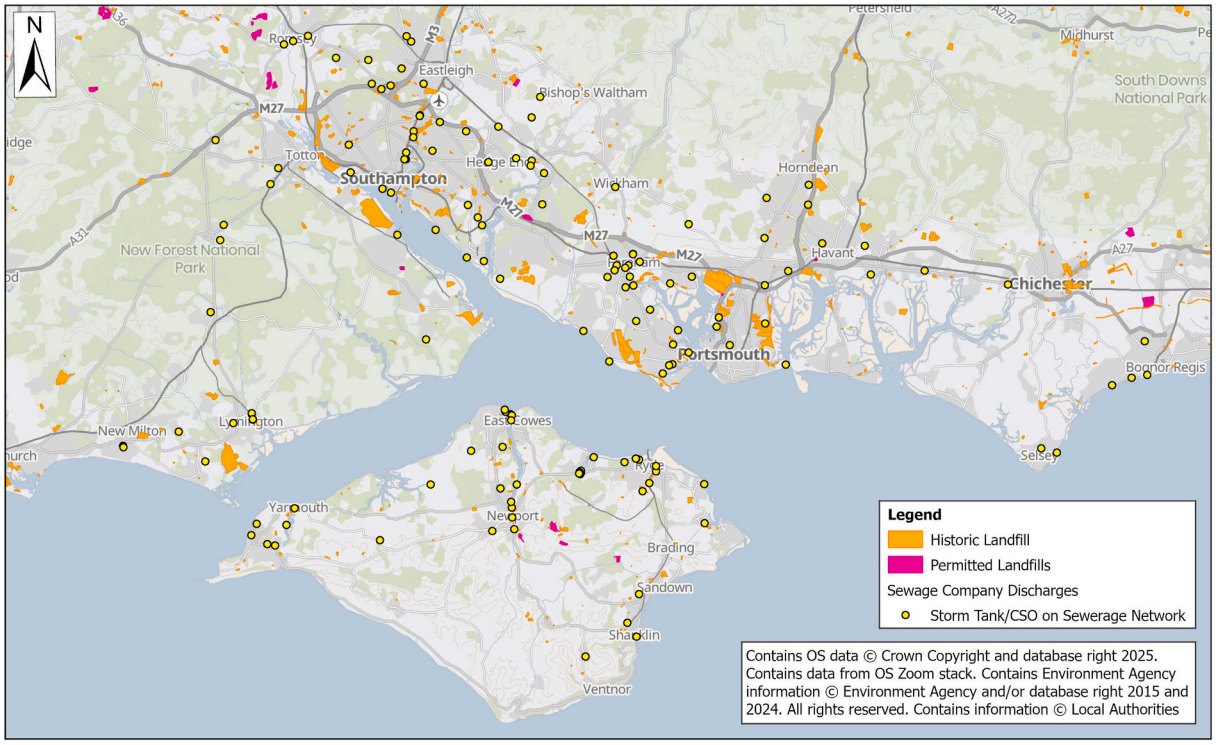


Fig. 1. Map of the Solent region in southern England, showing the main harbours, rivers, and coastal study areas.

area within the Solent, provided clear evidence of the impact of sewage discharges on PFAS contamination (Ford and Ginley, 2024). Seven PFAS compounds were detected in seawater after a CSO event, compared to just one compound before. Both linear and branched PFOS, along with PFOA, exceeded EQS limits for surface waters (Ford and Ginley, 2024). The study also found high levels of PFBA in macroalgae (*Fucus vesiculosus*), with Bioaccumulation factor (BAF) exceeding 6000. This suggests that short-chain PFAS might build up significantly in marine algae and may be passed through the food web. Consistent with broader coastal evidence, nearby estuarine systems on the continent show short-chain PFAS dominance in rivers and pronounced seasonal peaks linked to runoff (Zhao et al., 2015).

Between 2020 and 2023, over 25,000 CSO discharges released more than 10 million minutes (~19 years) of untreated wastewater into the South East coastline monitored by Southern Water (Ford et al., 2025). While designed for storm relief, discharges also showed daily peaks between 7 and 10 a.m., linked to domestic water use (Ford et al., 2025). This challenges assumptions about their timing and potential contaminant load. For persistent pollutants like PFAS, which resist dilution and degradation, such patterns highlight a critical oversight in current risk frameworks. Comparable continental studies show substantial PFAS riverine fluxes to the North Sea, estimated at 335 ± 100 kg yr^{-1} from the Elbe and 102 ± 22 kg yr^{-1} from the Weser (Zhao et al., 2015). An earlier study has reported up to ~510 kg yr^{-1} from the Elbe in 2006 (Ahrens et al., 2010).

This study presents a regional-scale integration of PFAS occurrence across surface water, sediment, treated effluent, and marine biota within a UK coastal system. We built on our previous assessment of surface waters in the Solent (Ford and Ginley, 2024), by expanding the scope to include a broad range of algae, fish and invertebrate species. This was done alongside PFAS measurements from two major wastewater treatment plants obtained through Environmental Information Requests and publicly available sediment and biota monitoring datasets. Rather than attempting to quantify mechanistic transfer among environmental compartments, we synthesise these independently generated datasets. This was done to evaluate (i) species-level differences in PFAS composition, and (ii) whether mixture-based toxic equivalency approaches alter regulatory interpretation relative to single-compound environmental quality standards (EQS). To assess cumulative exposure, RPFs were applied to derive total PFOA-equivalent (ΣPFOA-eq) concentrations in biota. This enables comparison with the European Food Safety Authority (EFSA) health-based benchmark of 77 ng/kg wet weight. Concentrations were additionally interpreted against current UK/EU EQS for PFOS in water and biota (water AA-EQS: 0.65 ng/L inland; biota EQS: 9.1 μg/kg ww) (European Environment Agency, 2024). Through this integrative framework, we evaluate whether reliance on single-compound thresholds adequately captures PFAS mixture exposure within a coastal food web.

2. Methods

2.1. Research location

The Solent is a coastal strait located on the southern coast of England, separating the Isle of Wight from the mainland (Fig. 1). It is approximately 32 km long and up to 8 km wide, with extensive estuarine and intertidal habitats (May et al., 2023). This includes seagrass beds, saltmarsh, mudflats, and shellfish grounds (Foster et al., 2014; Hinder et al., 2011; Tubbs and Tubbs, 1983). The region is of high ecological and conservation value, containing multiple Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC), Special Protection Areas (SPA), and RAMSAR wetlands (SEMS, n.d.). The Solent is also part of the Isle of Wight UNESCO Biosphere Reserve and supports commercial fisheries and aquaculture (Solent Forum, n.d.).

The Solent is a tidal strait that separates the Isle of Wight from mainland Hampshire and is commonly divided into the East and West Solent. The map includes some environmental features: historic landfills (Orange shading) represent former landfill sites; permitted waste sites (Purple shading) show locations with issued or superseded permits for waste processing or disposal; sewage company discharges (yellow dots) mark storm tanks or combined sewer overflows. They also represent locations on the sewerage network where stormwater or combined sewage is released. There are approximately 194 “Storm Tank/CSO on Sewage Network” and 546 “Historic Landfills”, and 69 “Permitted Landfills” within close proximity to the Solent (Fig. 1 - data downloaded 26/10/2025).

2.2. Biota sample collection

A diverse range of marine organisms was collected in triplicate from intertidal and subtidal habitats of the Solent during winter 2024 and summer 2025 and subsequently frozen at −20 °C until analysis. Sampling targeted different trophic levels and functional groups within the coastal ecosystem. Three major categories were represented: primary producers, invertebrates, and fish. Collection sites were selected to overlap with regions of previous surface water sampling (Ford and Ginley, 2024). All samples were collected within the Solent, except fish, which were obtained from commercial fishers on the southern side of the Isle of Wight. Primary producers included the seaweeds Bladderwrack (*Fucus vesiculosus)*, Gut weed (*Ulva intestinalis*), Sea lettuce (*Ulva lactuca*) and Knotted wrack (*Ascophyllum nodosum*). Invertebrate species included the common cuttlefish (*Sepia officinalis*), edible crab (*Cancer pagurus*), ragworm (*Hediste diversicolor*), common periwinkle (*Littorina littorea*), oyster (*Ostrea edulis*) and amphipod (*Marinogammarus marinus*). Data for bladder wrack (*Fucus vesiculosus*) and oyster (*Ostrea edulis*) were obtained from the earlier Solent study reported by Ford and Ginley (2024). Fish species included black bream (*Spondyliosoma cantharus*), plaice (*Pleuronectes platessa*), red gurnard (*Chelidonichthys cuculus*), and whiting (*Merlangius merlangus*). Trophic positions were assigned using published ecological literature and established knowledge of species feeding behaviour and ecological niche (www.fishbase.se/search.php; Supplementary Table S8). This literature-based approach was used because stable isotope data were not available and is consistent with approaches recommended for contaminant studies under the EU Water Framework Directive (Kidd et al., 2019). However, it does not provide site-specific trophic resolution and may not capture local dietary variability or temporal shifts in feeding behaviour. These species represent a broad cross-section of the Solent's marine food web. They were selected to support ecological analysis and to improve understanding of trophic transfer.

2.3. PFAS analysis

All samples were submitted to an accredited commercial laboratory (RPS, a Tetra Tech Company), which also performed PFAS analysis for the previous study. Analytical procedures followed identical validated methods. Briefly, 1 g of homogenised tissue or algal material was weighed into PFAS-free centrifuge tubes. Samples were spiked with a suite of 13C- and 2H-labelled internal standards (26 compounds in total) to cover the target analytes. Method blanks and sample spikes were included with each batch.

Samples were extracted using a modified QuEChERS/dSPE method with 10 mL ultrapure water and 10 mL acetonitrile. Extracts were evaporated to incipient dryness, reconstituted in 500 μL of methanol: water (80:20), and analysed using an Agilent 1290 Infinity II UPLC system coupled to an Agilent 6495C triple quadrupole mass spectrometer (negative electrospray ionisation, dynamic multiple reaction monitoring mode). Calibration curves were prepared with ISO Guide 34 certified reference materials over the range of 0.02–20 μg/L.

A total of 54 PFAS compounds were targeted, including linear and branched isomers of PFOS and PFOA, alongside a suite of short- and long-chain perfluoroalkyl acids (PFAAs). Full details of the target

analytes, internal standards, and instrument parameters are provided in Supplementary Tables 1 and 2 All solvents, reagents, and consumables were pre-screened for PFAS contamination. Limits of detection (LOD) and quantification (LOQ) were identical to those used in the previous study (Ford and Ginley, 2024).

2.4. Environmental monitoring datasets

PFAS concentration data for wastewater effluents were obtained via an Environmental Information Regulations (EIR) request to Southern Water Services Ltd (Reference: 2309). These datasets included post-treatment measurements from pilot recycling trials at Budds Farm (Portsmouth) and Peel Common (Fareham) wastewater treatment plants. Both Budds Farm and Peel Common WWTPs serve approximately 400,000 and 250,000 people, respectively. For untreated discharges, a targeted regional analysis published by the Solent Protection Society (Solent Protection Society, 2024a) was used to isolate combined sewer overflows (CSOs) discharging directly into the Solent. This includes Portsmouth, Langstone and Chichester Harbours, Gosport, and the Isle of Wight. The SPS analysis was based on Environment Agency data originally published via Southern Water's flow and spill reporting portal.

PFAS concentrations in surface waters across the Solent were obtained from our previous study (Ford and Ginley, 2024). Samples were collected from Langstone Harbours and analysed for 54 PFAS by an accredited laboratory (RPS, a Tetra Tech company). These data provided baseline aqueous concentrations for comparison with effluent, sediment, and biota datasets.

PFAS concentrations in sediments were obtained from the Cefas environmental monitoring database. Grab samples were collected by the RV Solent Guardian in November 2020 at sites within Southampton Water and Portsmouth. Reported parameters included individual PFAS compounds (μg/kg dry weight). For the present study, only sampling sites located within the Solent were extracted.

Data on the aquatic mammal and fish were compiled from the Cefas biota monitoring archive (https://data.cefas.co.uk/) and visualised through the Watershed Investigations portal (www.watershedinvestigations.com). To increase sample coverage, we extracted individual records from the Solent and adjacent coastal waters (eastward to Brighton and southward to the Isle of Wight). These include PFAS concentrations (μg/kg wet weight) in mammal liver tissue (Harbour porpoise) and whole-fish samples (in plaice).

2.5. Statistics

All statistical analyses were performed in R (version 4.3.2; (R Core Team, 2023). Data processing, visualization, and statistical testing were carried out using the tidyverse collection of packages (Wickham et al., 2019). Rstatix (Kassambara, 2022) was used for statistical testing, ggpubr (Kassambara, 2023) for charts, vegan (Oksanen et al., 2001) for community and multivariate analyses. FactoMineR (Lê et al., 2008) was used for principal component analysis and related multivariate techniques. PFAS data were harmonized across sample types (surface water, sediment, effluent, and biota). Analyte names were standardized and all concentrations were converted to common units. This study integrates datasets from multiple sources (including newly collected samples, regulatory monitoring data, and wastewater effluent records). Hence some variation in analytical methods, detection limits, and sampling design exists among matrices (Supplementary Table S10). Comparisons among matrices are interpreted as indicative of general patterns rather than strict quantitative equivalence. Relative potency factors (RPFs) were applied to estimate PFOA-equivalent (ΣPFOA-eq) concentrations in biota, enabling comparison against health-based benchmarks. Values below detection limits were assumed to be zero in the calculation of ΣPFOA-eq, consistent with a conservative, lower-bound exposure estimate. Differences among PFAS analytes in surface water and sediment samples were evaluated using the Kruskal–Wallis test with Dunn's post hoc correction and Bonferroni adjustment for multiple comparisons (Ryan, 1959). The Mann–Whitney *U* test (Rosner and Grove, 1999) was used for pairwise comparisons between short and long-chain PFAS groups within matrices. Normality and homogeneity of variance were assessed using the Shapiro–Wilk and Levene tests (shapiro.test () and car:leveneTest ()) (Fox J, 2019). When test assumptions were not met, non-parametric alternatives were applied. Compositional differences were examined using multivariate analyses. Principal component analysis (PCA) was conducted on centered and scaled molar fractions using the FactoMineR::PCA function (Lê et al., 2008) to assess similarities in PFAS fingerprints. Permutational multivariate analysis of variance (PERMANOVA) (vegan:adonis 2, 999 permutations) (Oksanen et al., 2001) was used to test for significant differences in PFAS composition across trophic levels. Non-metric multidimensional scaling (NMDS) based on Bray–Curtis dissimilarities was performed to visualize compositional relationships among samples. Results were considered statistically significant at $p < 0.05$. Because no significant grouping by trophic position was detected using PERMANOVA, supervised discriminant approaches were not done to avoid overfitting and artificial group separation.

2.6. Spatial analysis and mapping

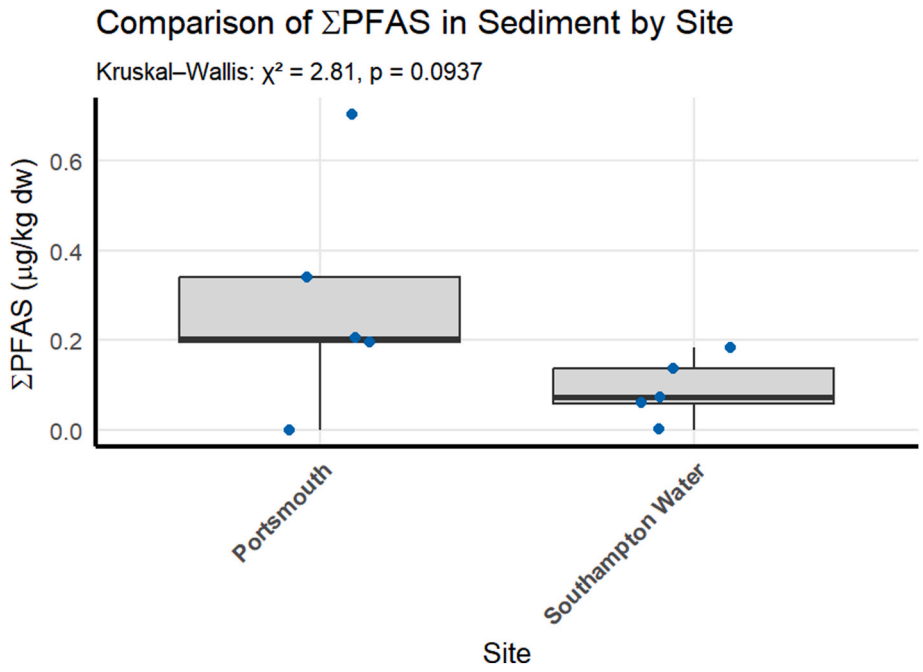
All spatial analyses and maps were produced using ArcGIS Pro (version 3.6). Sample locations were georeferenced and visualised using Ordnance Survey Zoomstack basemaps, which are provided as open data within ArcGIS Pro. Spatial datasets representing historic landfill sites, permitted landfill sites, wastewater treatment works, and combined sewer overflows (CSOs) were obtained from publicly available UK Environment Agency and local authority data portals. Historic landfill locations were sourced from the Environment Agency Historic Landfill dataset (https://environment.data.gov.uk/dataset/7a955570-d465-11e4-a37c-f0def148f590), while permitted landfill sites were obtained from the Environment Agency Permitted Waste Sites dataset (https://environment.data.gov.uk/dataset/692eaecf-d465-11e4-ac2e-f0def148f590). Data on consented discharges to controlled waters, including wastewater treatment works and CSOs, were obtained from the UK government open data portal (https://www.data.gov.uk/dataset/55b8eaa8-60df-48a8-929a-060891b7a109). PFAS concentration data were linked to spatial coordinates and displayed using graduated symbology to illustrate spatial patterns and potential contaminant sources across the Solent region.

3. Results

3.1 PFAS in Sediment: Sediment PFAS concentrations differed significantly among individual analytes (Tukey post-hoc test, $p < 0.05$) (Supplementary Material Table S2 and Fig. S2). ΣPFOS exhibited significantly higher mean concentrations than both short- and long-chain perfluoroalkyl carboxylic acids (PFCAs) and most other perfluoroalkyl sulfonic acids (PFSAs). Concentrations of ΣPFOS ranged from 0.028 to 0.501 μg kg^{-1} dry weight. The highest ΣPFOS concentrations were observed in Portsmouth Harbour, with multiple sites exceeding 0.35 μg kg^{-1} dw and one reaching 0.50 μg kg^{-1} dw. In contrast, sites in Southampton Water, including those near the mouths of the Rivers Itchen and Test, recorded much lower ΣPFOS concentrations, mostly below 0.09 μg kg^{-1} dw, with three sites showing 0.028 μg kg^{-1} dw. These areas are closer to the open Solent channel. Despite these spatial differences, replicate-level ΣPFAS concentrations did not differ significantly between Portsmouth and Southampton Water (Fig. 2; Kruskal–Wallis, $\chi^2 = 2.81$, $p = 0.094$, Supplementary Material Table S12).

3.1. PFAS in surface water and effluent

PFAS in Surface Water: Surface water samples (n = 3) collected



Figuree 2. Comparison of ΣPFAS concentrations in sediment between sampling sites
(ΣPFAS concentrations (µg kg^{-1} dry weight) in sediment samples from Portsmouth and Southampton Water. Boxplots represent the median and interquartile range, with individual replicate samples shown as points. No significant difference in ΣPFAS concentrations between sites was observed (Kruskal–Wallis, $\chi^2 = 2.81$, $p = 0.094$).

following a CSO event contained multiple short- and long-chain PFAS at comparable orders of magnitude (Supplementary Material Table S2; Fig. 3). Short-chain PFAS, including PFHxA and PFPeA, occurred at concentrations of 3–4 ng/L, while PFOA and PFHpA were detected at 2.5–3 ng/L. PFHxS and PFPeS were present at lower concentrations. There was limited number of environmental replicates, hence differences among analytes are presented descriptively rather than subjected to formal hypothesis testing. Mean ΣPFOS (8.5 ng/L) exceeded the UK/EU Environmental Quality Standard (EQS) for coastal waters (0.65 ng/L).

PFAS in Effluent: Data obtained from Southern Water from treated effluent at Peel Common and Budds Farm wastewater treatment plants, reveal a broader suite of PFAS compounds, encompassing both short- and long-chain substances (Supplementary Material Table S2 and Fig. 4). A total of 12 analytes at Budds Farm and 11 analytes at Peel Common were quantified above their respective LODs. At Budds Farm, these included: PFBA, PFHxA, PFHpA, PFPA, PFOA, PFBS, PFHxS, PFHxS (branched), ΣPFOS, 6:2 FTSA, FBSA, and PFOSA. At Peel

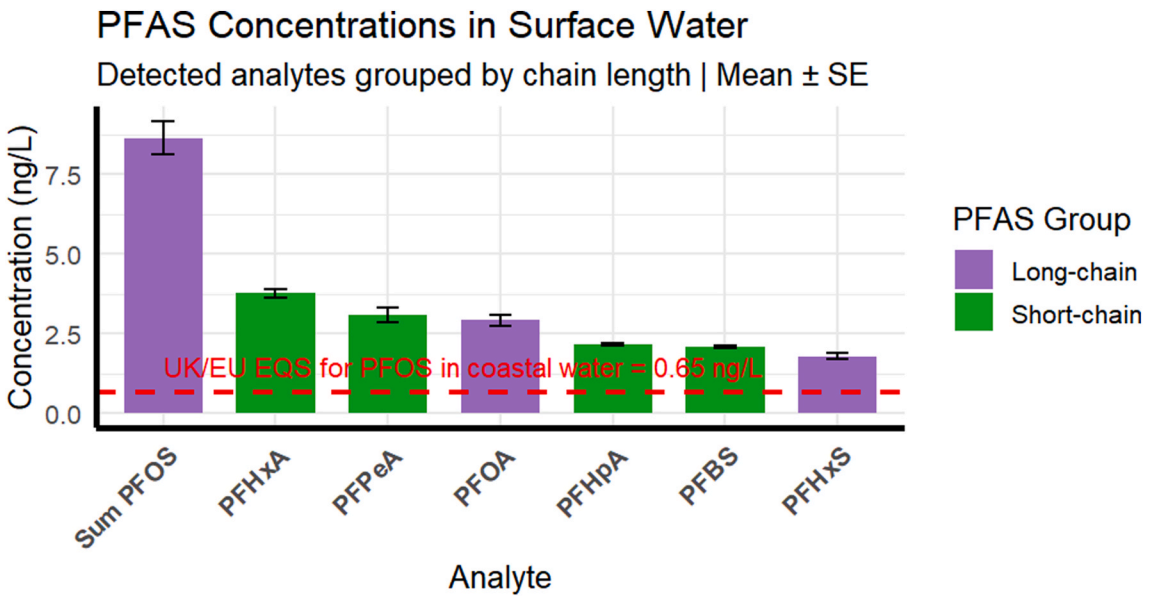


Fig. 3. PFAS concentrations (mean ± SE) in surface water samples, grouped by chain length.
(Data are from field sampling conducted in Langstone Harbour in 2023, originally reported by Ford and Ginley (2024). Colours indicate chain length classification: purple = long-chain PFAS, green = short-chain PFAS. The red dashed line marks the UK/EU Environmental Quality Standard (EQS) for PFOS in coastal waters (0.65 ng/L). ΣPFOS concentrations exceeded this regulatory threshold.)

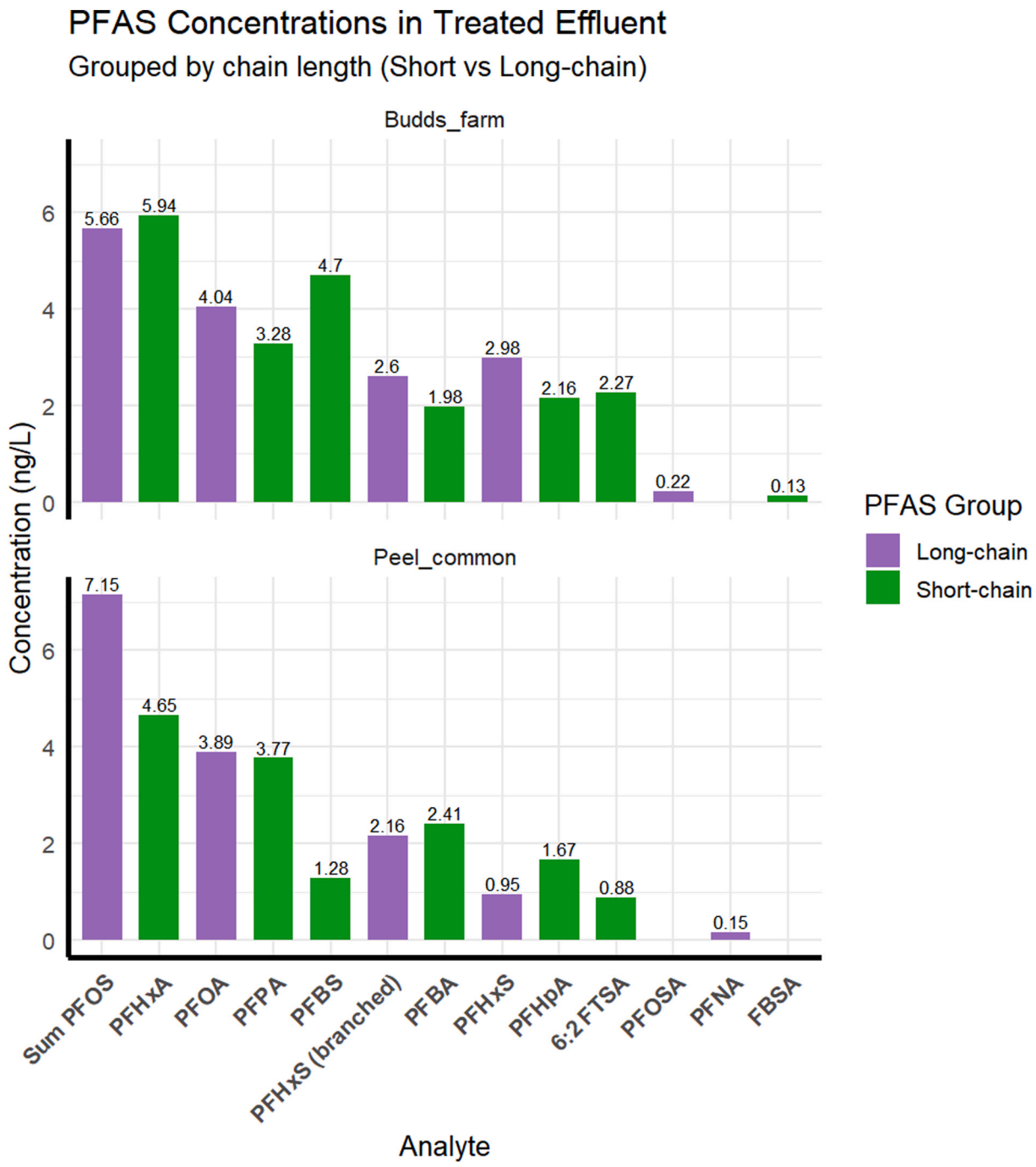


Fig. 4. PFAS concentrations (ng/L) in treated effluent from two wastewater treatment plants: Budds Farm (top) and Peel Common (bottom).
(To determine what concentrations of PFAS might be entering into the Solent region, PFAS data from two of the largest WWTP in the local vicinity were reviewed. Data represent PFAS concentrations in post-treatment effluent from Budds Farm and Peel Common WWTPs, obtained via Environmental Information Request to Southern Water (Reference: 2309). Sampling was conducted as part of pilot recycling trials in 2023. Detected analytes are grouped by chain length (purple = long-chain PFAS; green = short-chain PFAS). ΣPFOS was the dominant compound at both sites. Values represent mean concentrations based on above-detection limit samples.)

Common, the analytes above LOD were: PFBA, PFHxA, PFHpA, PFPA, PFOA, PFBS, PFHxS, PFHxS (branched), ΣPFOS, 6:2 FTSA and PFNA. Short-chain PFAS (including PFHxA, PFBA, and PFBS), were observed, with mean concentrations ranging from 1.28 to 5.94 ng/L. PFHxA was consistently present in effluent from both facilities, with average concentrations of 4.65 ng/L at Peel Common and 5.94 ng/L at Budds Farm. Similarly, the long-chain compound PFOA was frequently detected, with mean concentrations of 3.89 ng/L and 4.04 ng/L, respectively. PFOS was also present in treated effluent, with average concentrations of 7.15 ng/L at Peel Common and 5.66 ng/L at Budds Farm.

3.2. PFAS in marine biota

3.2.1. PFAS composition and group profiles across species

Total PFAS concentrations varied widely across species (Supplementary material. Table S1) (Fig. 5, left panel). Harbour porpoises' liver (*Phocoena phocoena*) showed the highest concentrations, with sum total levels exceeding 8000 μg/kg wet weight (ww). All other

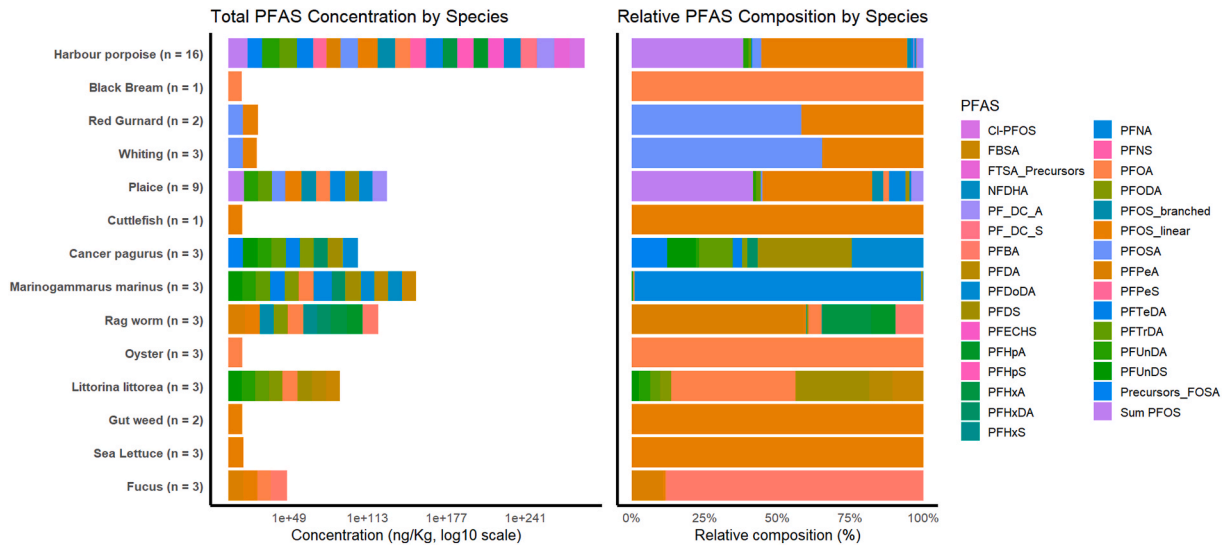


Fig. 5. Total and relative PFAS concentrations by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight). Right: Relative PFAS composition (%) by individual compound.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. The left panel is presented on a $\log_{10}$ scale; therefore, stacked bar segments are not visually additive.)

species which were primarily made up of muscle tissues or whole organism samples had much lower concentrations (generally below 100 μg/kg ww). ΣPFOS concentrations (μg kg^{-1} wet weight) were quantified in *Sea Lettuce* (0.60 ± 0.11), *Gut weed* (0.12 ± 0.04), Seaweed (0.17 ± 0.0), *Rag worm* (0.86 ± 0.0), *Cuttlefish* (0.24 ± 0.0), *Whiting* (0.19 ± 0.04), *Plaice* (0.86 ± 0.12), *Red Gurnard* (0.43 ± 0.15), and *Harbour porpoise* (341.39 ± 49.77). Among these, ΣPFOS exceeded the UK/EU environmental quality standard of 9.1 μg kg^{-1} ww only in *Harbour porpoise* liver tissues, with all other taxa remaining below the regulatory threshold. No PFAS compounds were detected in *Ascophyllum nodosum*.

When applying EU RPFs (Fig. 6; Appendix C), ΣPFOA-eq was below detection in all three replicates (3/3) of *Ascophyllum nodosum*, Cuttlefish, Gut weed, Red Gurnard, Sea Lettuce, and Whiting. Partial non-detections occurred in Black Bream and Oyster (2/3 replicates each), and in Plaice (1/3 replicates). For *Plaice*, all other replicates from different sampling locations showed detectable ΣPFOA-eq, and exceeded the EFSA benchmark of 77 ng/kg ww. Several additional species also exceeded the EFSA benchmark including *Cancer pagurus* (edible crab), *Hediste diversicolor* (ragworm), *Littorina littorea* (periwinkle),

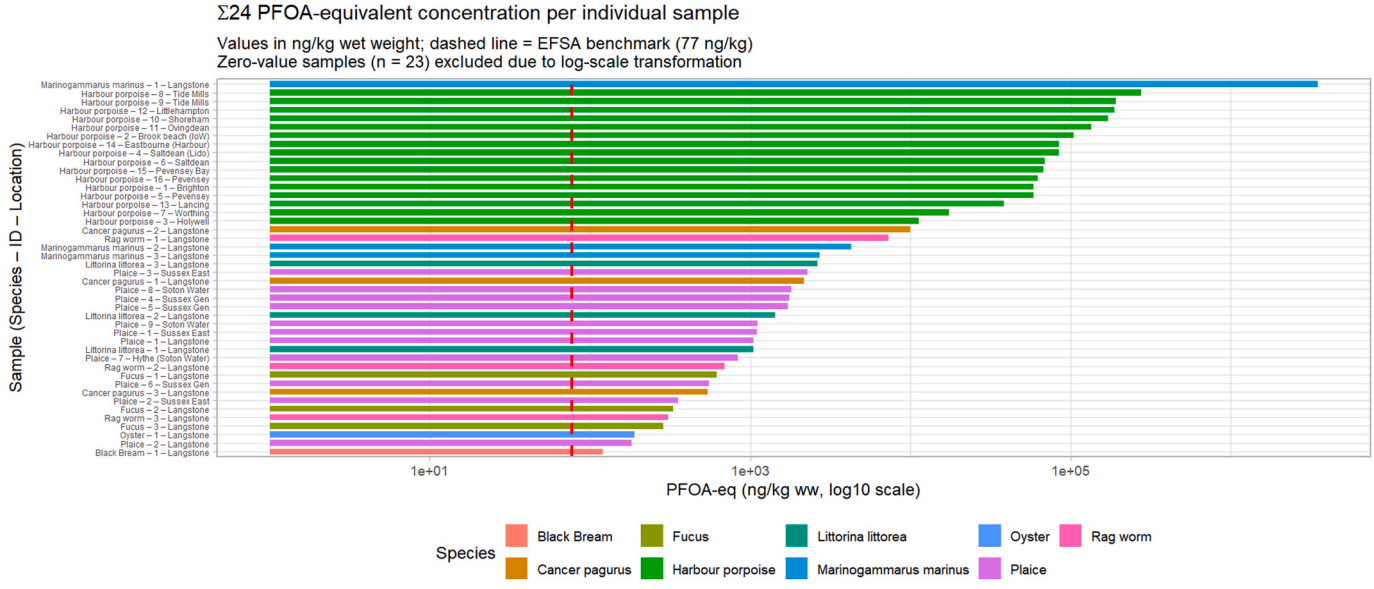


Fig. 6. Summed PFOA-equivalent concentrations (Σ24 PFOA-eq).
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. Each bar represents individual biota samples across species and locations, converted to ng/kg wet weight and displayed on a $\log_{10}$ scale. Each bar represents one species–location replicate, and the red dashed line indicates the EFSA benchmark of 77 ng/kg ww. Samples with ΣPFOA-eq equal to zero (n = 23) were excluded from the plot. These included replicates of Ascophyllum nodosum, gut weed, sea lettuce, and oyster collected from intertidal habitats at the Solent; cuttlefish and plaice collected from subtidal zones of the Solent; and black bream, red gurnard, and whiting obtained from commercial fishers off the southern coast of the Isle of Wight.)

Marinogammarus marinus (amphipod), and *Fucus vesiculosus* (bladderwrack). Elevated ΣPFOA-eq concentrations were also observed in *Phocoena phocoena* (harbour porpoise) liver samples from the Cefas archive.

PFAS composition varied across species (Figs. 7 and 8; Supplementary Material Tables S3 and S4). Vertebrates, including fish and marine mammals, were dominated by sulfonic acids, particularly PFOS. Several invertebrates and algal species contained a wider range of PFAS. These species showed greater representation of carboxylic acids and precursor compounds compared to sulfonates. For example, *Marinogammarus marinus* (amphipod) contained multiple carboxylic acids. These included PFBA (detected in 1 out of 3 samples), PFOA (2/3), PFDA (2/3), PFPeA (1/3), PFODA (2/3), PFTrDA (3/3), and PFUnDA (1/3). Precursors were also detected, including FTAB compounds (1/3) and N-EtFOSA-related compounds (1/12). Rag worm (*Hediste diversicolor*) also showed a mixture of short-chain carboxylic acids and precursors. These included PFHxA (3/3), PFBA (2/3), PFHpA (2/3), and PFOS_branched (1/3). Among macroalgae, *Fucus vesiculosus* had consistent detections of short-chain PFAS. These included PFBA (3/3) and PFPeA (3/3). It also contained FOSE precursors (6/6) and N-EtFOSA-type compounds (6/12). In contrast, PFOS and other sulfonates were less frequently detected in these taxa. For example, PFOS_linear was found in only 1 out of 3 *Fucus* samples, and not at all in *Ascophyllum nodosum*.

A similar pattern was observed when PFAS were grouped by chain length (Fig. 8; Supplementary material. Table S7). Long-chain compounds dominate in vertebrates. Short-chain PFAS were more common in macroalgae and some invertebrate species. For example, *Fucus* and *Rag worm* were composed almost entirely of short-chain PFAS.

3.3. Trophic and multivariate patterns

Principal Component Analysis (PCA): Principal component analysis summarised variation in PFAS composition across species (Fig. 9). The first principal component explained 54.9% of the total variance, and the second explained 17.4%, accounting for 72.2% of the overall variance (Supplementary Material, Appendix E). Loadings on the first axis were highest for several long-chain PFAS and selected precursor compounds. Loadings on the second axis were highest for shorter-chain carboxylates and sulfonates. Species were distributed across both axes. Harbour porpoise, gurnard, and whiting aligned positively along the first axis, consistent with higher proportions of long-chain PFAS and precursor compounds in their profiles. In contrast, rag worm and some other invertebrates (e.g., *Marinogammarus marinus* and *Cancer pagurus*) loaded higher on the second axis, reflecting enrichment in short-chain carboxylates and sulfonates. Primary producers, such as Fucus, sea lettuce, and gut weed, were located near the origin or on the negative side of the first axis, indicating lower overall PFAS concentrations or less distinctive profiles.

Non-metric Multidimensional Scaling (NMDS): Non-metric multidimensional scaling produced a two-dimensional ordination with low stress (0.051; Fig. 9). Species positions were separated across ordination space. No consistent ordering of species with trophic position was observed along either NMDS axis. PERMANOVA indicated that the trophic position explained a small proportion of variance in PFAS composition (Model $R^2 = 0.058$; $p > 0.05$).

4. Discussion

This study examined the occurrence and composition of PFAS in surface water, sediment, treated effluent, and marine biota within the Solent. It positions this urbanised UK estuary as a representative example of PFAS dynamics in densely populated coastal regions. The aim is to understand how these compounds are distributed across environmental compartments and incorporated into a coastal food web. Globally, temperate estuaries receiving treated wastewater and storm overflow discharges are increasingly recognised as convergence zones for both legacy long-chain PFAS and emerging short-chain substitutes (Ahrens and Bundschuh, 2014; Armitage et al., 2009; Zhao et al., 2015; Brendel et al., 2018; Wang et al., 2017; Pan et al., 2016). Differences in PFAS composition were observed among matrices, with sediments dominated by PFOS, while surface waters and treated effluent contained a broader mixture of compounds, including short-chain PFAS such as PFHxA and PFBA. These contrasting patterns suggest that PFAS exposure in the Solent reflects the integration of multiple input pathways rather than a single dominant source. While wastewater discharges and combined sewer overflows are likely contributors, other sources such as historic landfill sites, diffuse urban runoff, atmospheric deposition, and existing contamination within sediments may also play important roles. For example, our mapping of CSOs (n = ~194) and historical landfills (n = ~546) has revealed many potential PFAS sources into this coastal region. Historical landfills are known to be sources of PFAS contamination into surface and groundwaters (Lang et al., 2017; Capozzi et al., 2023; Roy et al., 2025). Furthermore, our analysis of wastewater

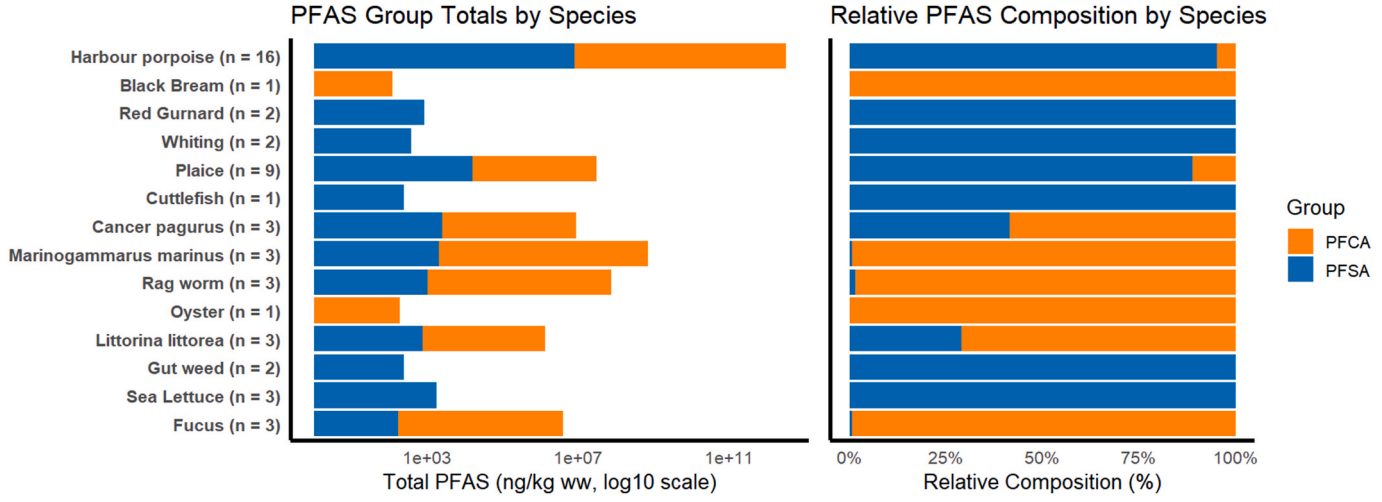


Fig. 7. PFAS group composition (PFCAs vs PFSAs) by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight); Right: Relative PFAS composition (%) by functional group.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. The left panel is presented on a $\log_{10}$ scale; therefore, stacked bar segments are not visually additive.)

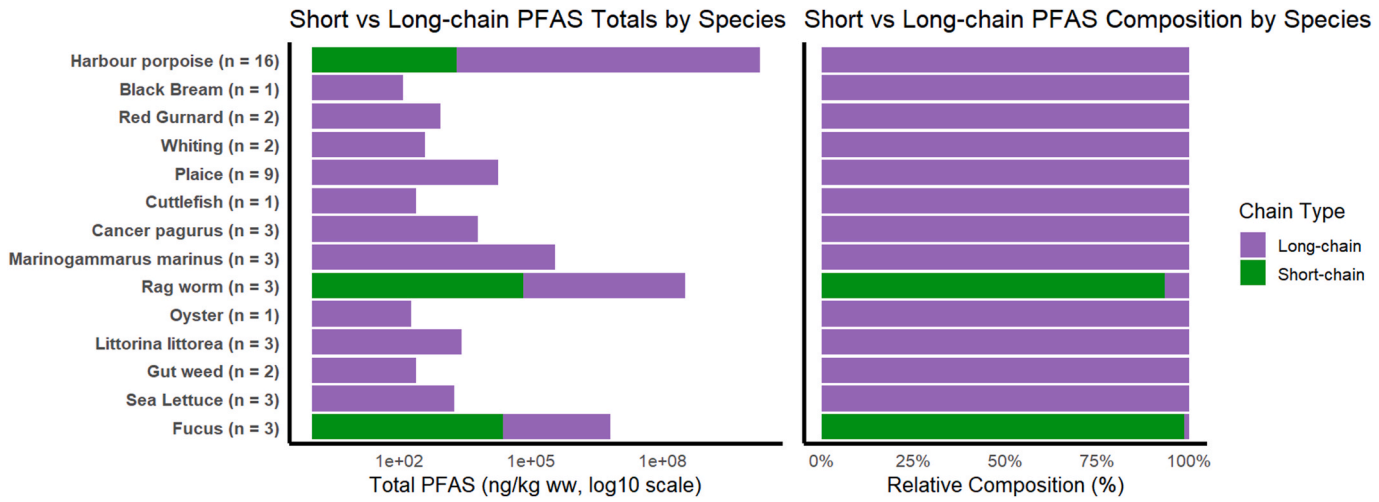


Fig. 8. Short-chain vs long-chain PFAS composition by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight); Right: Relative (%) PFAS composition by chain length.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. The left panel is presented on a $\log_{10}$ scale; therefore, stacked bar segments are not visually additive.)

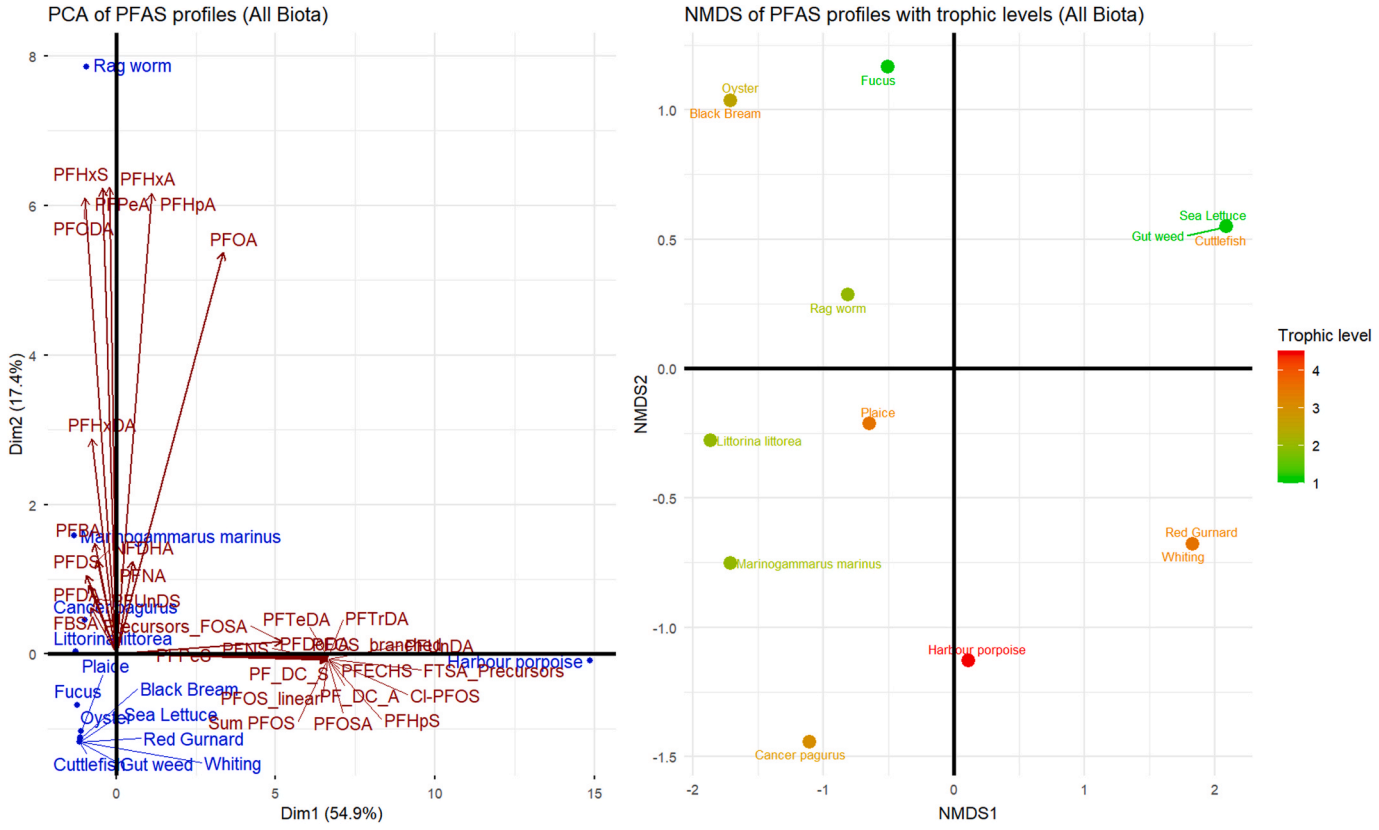


Fig. 9. PCA and NMDS of PFAS Profiles for all biota
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. Principal component analysis (left) and non-metric multidimensional scaling (right) summarise variation in PFAS composition among species after aggregation across study periods. PCA axes are labelled with the percentage of variance. NMDS is based on Bray–Curtis dissimilarities (stress value = 0.051). Points represent species, coloured by trophic position.).

effluents from two local WWTPs has revealed high concentrations of PFAS. Stormwater overflows carry untreated wastewater which make them potential sources of PFAS at concentrations potentially greater than treated effluent (Ford et al., 2025).

The co-occurrence of long-chain and short-chain PFAS, including known precursor compounds, makes it difficult to distinguish between recent inputs and degradation products of historically used substances such as PFOS and PFOA. Although spatial proximity to wastewater and landfill infrastructure suggests plausible source contributions, no formal source apportionment (e.g., fingerprinting or mixing models) was

undertaken. Therefore, compositional patterns should be interpreted as consistent with mixed urban inputs rather than evidence of source dominance. Hence, the relative contributions of individual sources cannot be quantified. In addition, air–sea exchange processes may contribute to the redistribution of PFAS within the coastal environment (Xie et al., 2013). Together, these findings highlight the complexity of PFAS exposure in the Solent and emphasise the importance of a multi-matrix approach when assessing sources, transport, and potential risks.

Elevated ΣPFOS concentrations were observed at inner-harbour sites; however, differences between sites were not statistically significant. These concentrations at inner-harbour sites could suggest localised enrichment within semi-enclosed harbour environments. In this location (e.g Portsmouth Harbour) restricted tidal exchange and long water residence times limit dilution and enhance contaminant retention Wu et al. (2024); Zhang et al. (2025); Singh et al. (2025) demonstrated through high-frequency estuarine measurements that ΣPFAS concentrations can vary by four-to seven-fold over tidal cycles, with seawater influx during high tide diluting PFAS levels and low-tide conditions capturing peak contamination. PFOS exhibits sorption to organic-rich, fine-grained sediments that accumulate in sheltered harbours and estuaries Higgins and Luthy (2006); Chen et al. (2012) demonstrated that the sorption of PFOS in seawater to marine sediments was evident (≈10 times higher than that in freshwater). In the Solent, inner harbours such as Portsmouth Harbour and Southampton Water may be influenced by dense urban catchments and complex wastewater infrastructure. These include multiple combined sewer overflows and storm tanks that discharge untreated sewage during rainfall events (Ford et al., 2025). This is alongside continuous inputs from treated effluent, where PFOS removal is inefficient (Ford and Ginley, 2024; Weinberg et al., 2011; Eriksson et al., 2017 detected a broad range of PFAS including PFCAs, PFSAs, FTCA/FTUCAs, FTSAs and PFPA/PFPiA in filtered effluents from three WWTPs in Sweden. Additional contributions are likely associated with port, naval, and industrial activities. They include the historical use of aqueous film-forming foams at dockyards (Ahmadireskety et al., 2021; Filipovic et al., 2015) and fire-training facilities (Cao et al., 2022), which are well-established sources of PFOS in coastal environments. Reinikainen et al., 2022 showed that the extensive use of nowadays restricted or substituted PFAS, particularly PFOS, are still often the predominant compounds detected at aqueous film forming foams-impacted sites. The proximity of historic landfills and permitted waste sites further suggests diffuse PFAS inputs via leaching and runoff. Roy et al., 2025 showed evidence that groundwater plumes from legacy landfills can discharge PFAS to adjacent surface waters. This produced persistently elevated concentrations in sediments and measurable downstream contamination despite dilution in the water column.

Not surprisingly, treated effluent contained a wider range of PFAS than surface waters, including both short-chain and long-chain compounds. Short-chain PFAS, including PFPA, PFBA, 6:2FTSA, were consistently detected in effluent from both wastewater treatment plants alongside PFOA and PFOS. Whereas surface waters were characterised majorly by elevated ΣPFOS, with mean concentrations exceeding the UK/EU environmental quality standard for coastal waters (0.65 ng/L) (European Environment Agency, 2024). However, interpretation of surface water concentrations should be treated with caution due to the limited number of samples (n = 3) and restricted temporal coverage. Several PFAS detected in treated effluents were not detected in surface water samples, including PFPA, PFBA, and 6:2 FTSA at both sites, PFNA in effluent from Peel Common, and PFOSA and FBSA in effluent from Budds Farm. Future comparisons across matrices should consider harmonized detection thresholds to better assess differences in PFAS profiles. The presence of overlapping compounds across effluent and surface water (e.g PFOA, PFBS) is consistent with continuity in PFAS inputs to the coastal system (Du et al., 2022; Topaz et al., 2024). Differences in relative concentrations between matrices suggest post-discharge processes such as dilution and enrichment (Liu et al., 2024; Sarti et al., 2025). Additionally, these distributions are consistent with established differences in physicochemical properties among PFAS, including chain-length-dependent sorption (Higgins and Luthy, 2006; Wang et al., 2018) behaviour and aqueous mobility (Ahrens and Bundschuh, 2014; Armitage et al., 2009).

PFAS concentrations and compositions differed among taxa. However, it is important to note that biota data were compiled from multiple sources, including archived monitoring datasets and new field collections. They were each analysed using different methods and detection limits. These variations may have influenced the range of compounds detected and could partly explain differences in PFAS composition across species. Porpoise data obtained from the Cefas biota monitoring archive showed the highest total PFAS burdens. In harbour porpoise liver, ΣPFAS exceeded 8000 μg kg^{-1} wet weight, with profiles largely composed of sulfonic acids, particularly PFOS. According to the approach of (van den Heuvel-Greve et al., 2025), whole-body PFOS concentrations is around ten times lower than liver concentrations in harbour porpoises. Hence, the values measured in our study correspond to estimated whole-body PFOS concentrations of 34.1 μg kg^{-1} wet weight. Similar scaling factors have been applied previously in cetaceans (Houde et al., 2006). Fish species showed intermediate PFAS concentrations, with profiles characterised by a predominance of PFOS and other long-chain PFAS. Similar compositional patterns have been widely reported in marine and estuarine fish, where sulfonates tend to dominate tissue burdens relative to carboxylates (Galatius et al., 2013; Munoz et al., 2017; van den Heuvel-Greve et al., 2025). Within our fish data, PFAS composition varied among species but did not show a consistent benthic–pelagic pattern. Plaice included detections of longer-chain carboxylates such as PFTrDA and PFDoDA, and red gurnard showed PFOSA and precursor compounds; however, whiting also exhibited consistent PFOSA detections, while black bream showed fewer overall compounds. Long-chain PFUnDA was observed only in the previous plaice dataset, indicating variability among species and between study periods. In contrast, invertebrates and primary producers exhibited lower total PFAS concentrations but more heterogeneous profiles that included short-chain carboxylates and precursor compounds. This is consistent with earlier speculation (Ford and Ginley, 2024) that grazing species might reflect the PFAS profiles of macroalgae such as Fucus. In the present dataset, both *Marinogammarus marinus* and *Littorina littorea* contained short-chain PFAS and precursors also detected in Fucus, including PFBA, PFPeA, and FOSE-type compounds. This suggests a possible dietary exposure pathway, although further targeted studies would be needed to confirm direct transfer. PFAS mixtures at lower trophic levels have been observed in multiple coastal systems and are often attributed to direct exposure from water and sediments rather than trophic transfer alone (Estment et al., 2025; Munoz et al., 2019). The prominence of short-chain PFAS in macroalgae and some benthic invertebrates is consistent with their higher water solubility and mobility (Adeogun et al., 2024; Brendel et al., 2018). Interestingly, *Fucus vesiculosus* and *Ascophyllum nodosum* were collected from the same intertidal location adjacent to a sewage pipe, yet exhibited very different PFAS profiles. Fucus showed consistent detections of short-chain PFAS and precursors, while Ascophyllum had no detectable PFAS above the method detection limit. This may reflect species-specific differences in uptake mechanism. It could also be influenced by seasonal variation, given that samples were collected at different times. Similar enrichment of short-chain PFAS in primary producers has been documented under conditions of sustained environmental input (Blaine et al., 2013; Brendel et al., 2018; Gkika et al., 2025). This indicates that these compounds can enter food webs at their base despite lower bioaccumulation potential relative to long-chain PFAS.

Comparison with regulatory benchmarks indicates that ΣPFOS concentrations exceeded the UK/EU biota EQS (European Environment Agency, 2024) only in harbour porpoise liver, while remaining species were below the regulatory limit. These benchmarks address different endpoints; EQS values are designed to protect aquatic ecosystems,

whereas EFSA thresholds relate to human dietary exposure. When applying RPFs to derive Σ24 PFOA-equivalent concentrations, many biota samples exceeded the EFSA health-based benchmark of 77 ng/kg wet weight (EFSA CONTAM Panel et al., 2020). Species with ΣPFOA-eq concentrations consistently below the EFSA benchmark included *Ascophyllum nodosum*, Cuttlefish, Gut weed, Red Gurnard, Sea Lettuce, and Whiting. These taxa may reflect limited capacity for bioaccumulation under local environmental conditions. In these calculations, non-detects were assigned a value of zero, representing a conservative lower-bound estimate of exposure. This approach likely underestimates total PFOA-equivalent burdens but avoids overestimation of risk. Additionally, these results highlight the compound- and species-specific nature of regulatory exceedances. Differences likely reflect varying exposure routes, tissue distribution, and elimination capacity (Georges and Johansson, 2025). Comparable patterns have been reported internationally: in Belgian coastal biota, PFOS and long-chain PFCAs dominate (Cara et al., 2022); in Danish top predators (seals and cetaceans) (Galatius et al., 2013); and in Arctic food webs, where PFOS biomagnifies to seabirds while some substitutes accumulate in invertebrates (Ali et al., 2021). It is important to note that most fish analysed here were muscle tissues representative of edible portions, and harbour porpoise data were derived from liver tissue. Additionally, a subset of archived fish samples (e.g. Cefas datasets), consisted of composite tissues. Nonetheless, the 2020 EFSA PFAS-4 tolerable weekly intake of 4.4 ng/kg body weight per week (EFSA CONTAM Panel et al., 2020) suggests that even fish below the PFOS EQS may contribute meaningfully to cumulative dietary exposure when mixtures are considered. Together, these results emphasise the importance of toxicological equivalency approaches when evaluating mixture risk. They also demonstrate that compliance with single-compound ecological standards does not necessarily imply negligible human exposure risk. This regulatory contrast is relevant in the Solent. Recent assessments in this region report that several coastal habitats are in unfavourable or declining condition (Solent Protection Society, 2024a,b). Pollution and water quality pressures are identified as major stressors (Obanya, 2025; Solent Protection Society, 2024a,b). Current restoration efforts focus largely on habitat extent and broad water quality indicators (Preston et al., 2025; Solent Restoration News, 2024). Persistent contaminants such as PFAS are not yet routinely incorporated into these ecological status frameworks. Our results indicate that mixture-based PFAS metrics could add an important chemical dimension to ongoing conservation and restoration planning.

The application of RPFs to derive ΣPFOA-equivalent concentrations provides a useful screening-level approach for assessing mixture exposure. However, this method has some uncertainties. RPFs are derived from a limited set of toxicological endpoints, most commonly liver-related effects observed in experimental animal studies (Bil et al., 2021). They also assume that PFAS share similar dose–response relationships and act in an additive manner. In reality, PFAS exhibit diverse toxicokinetic and toxicodynamic behaviours (Fischer et al., 2025; Juhasz et al., 2025). Different compounds can vary in their target organs, modes of action, and bioaccumulation potential (Arnesdotter et al., 2025). In addition, toxicological data remain limited for many PFAS, and some RPF values are based on read-across approaches or constrained datasets (Bil et al., 2021; EFSA CONTAM Panel et al., 2020). The assumption of concentration additivity further introduces uncertainty because it does not account for potential synergistic or antagonistic interactions within PFAS mixtures (Ojo et al., 2020). As such, ΣPFOA-equivalent concentrations should be interpreted with caution. They are best viewed as an indicator of relative mixture burden rather than a precise quantitative estimate of risk.

Neither ordination nor PERMANOVA identified a significant relationship between PFAS composition and trophic position. The low explanatory power (R^2) further indicates that trophic position accounts for only a small proportion of the observed variability. However, trophic positions were derived from published ecological literature rather than site-specific stable isotope measurements. As such, this analysis does not constitute a formal test of biomagnification or trophic magnification. It also cannot resolve local dietary variability or seasonal shifts in feeding behaviour. Additionally, the absence of an isotopic framework limits inference regarding trophic transfer. Therefore, observed patterns are interpreted as compositional differences among taxa rather than evidence for or against trophic magnification. Similar patterns have been reported in other marine systems, where PFAS profiles diverge among taxa. These have been due to differences in metabolism, habitat use, and exposure pathways rather than following a simple biomagnification gradient (Houde et al., 2006; Kelly et al., 2009; Munoz et al., 2017). Comparable patterns have been reported in Belgian coastal ecosystems, where PFOS and long-chain PFCAs dominate biota with estuarine hotspots (Cara et al., 2022), and in Danish marine predators, where PFAS profiles differ between seals and cetaceans (Galatius et al., 2013). Arctic food-web studies similarly show enrichment of PFOS in seabirds alongside accumulation of some substitutes in invertebrates (Ali et al., 2021).

This study integrates PFAS data across environmental matrices with a varied limit of detections (Supplementary Material - Table S10) that were collected under different sampling designs and time frames. Additionally, the analytical focus was restricted to targeted PFAS, and additional precursors or transformation products were not captured. Despite these constraints, the multi-matrix approach provides a basis for identifying compound and species-specific patterns of PFAS exposure in a coastal food web. Overall, this integrated assessment suggests that PFAS distribution in the Solent is influenced by compound-specific physicochemical behaviour, multiple urban inputs, and plausible differences in enrichment among environmental matrices. No detectable relationship was observed using literature-derived trophic positions. Long-chain sulfonates, particularly PFOS, dominate sediments and higher trophic levels. In contrast, short-chain acids and precursor compounds are more common in effluent, surface waters, macroalgae, and some invertebrates. These differences reflect contrasting mobility and retention within the coastal system. Regulatory interpretation further highlights this complexity. Exceedance of ecological EQS thresholds was limited, yet mixture-based PFOA-equivalent concentrations suggest wider relevance for human exposure assessment. Together, these findings support the need for compound-specific monitoring that integrates water, sediment, and biota. Future work should include site-specific stable isotope analysis to better understand trophic transfer. Passive sampling and non-target screening would also improve understanding of precursor inputs and transformation pathways in UK coastal environments.

5. Conclusions

This study provides an integrated assessment of PFAS across surface water, sediment, treated effluent, and marine biota in a UK coastal system. Distinct PFAS patterns were observed among matrices. These patterns reflect a combination of wastewater inputs, environmental enrichment, and compound-specific behaviour. Sediments functioned as long-term sinks for legacy PFAS, particularly PFOS. In contrast, surface waters and effluents contained more varied PFAS profiles that included short-chain and emerging analytes. In biota, PFAS accumulation differed mainly among species and compounds rather than trophic position. The highest concentrations occurred in marine mammals, while lower trophic levels showed more heterogeneous PFAS profiles. Regulatory exceedances were seen in surface water and Harbour porpoise. However, lower-level PFAS exposure was widespread across the food web. When total PFAS burdens were expressed as PFOA-equivalents, most biota samples exceeded the EFSA benchmark of 77 ng/kg wet weight, suggesting the potential for dietary exposure even in samples below individual compound thresholds. Together, these results show that PFAS exposure in coastal ecosystems arises from multiple, overlapping pathways. Sediments and biota integrate these inputs over time. These

findings also indicate that regulatory reliance on single-compound EQS may underestimate mixture-based exposure in coastal food webs. Incorporating toxic equivalency frameworks into coastal monitoring programmes would provide a more precautionary and ecologically relevant basis for PFAS risk assessment.

CRediT authorship contribution statement

Henry E. Obanya: Data curation, Formal analysis, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **Francesca Ginley:** Data curation, Funding acquisition, Investigation, Project administration, Resources, Supervision, Validation, Visualization, Writing – review & editing. **Millicent R. Payne:** Data curation, Formal analysis, Investigation, Methodology, Software, Writing – review & editing. **Alex T. Ford:** Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing.

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.

Acknowledgements

The authors gratefully acknowledge financial support from the Marine Conservation Society, which supported the field sampling and data analysis associated with this study. We thank Southern Water for providing wastewater monitoring data via Environmental Information Request, and Cefas for access to sediment and biota monitoring datasets. We are also grateful for the support of Watershed Investigations in obtaining environmental data through Freedom of Information requests. We are also grateful to local fishers for assistance with specimen collection, and to laboratory staff at RPS(a Tetra Tech company) for analytical support.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marenvres.2026.108094.

Data availability

Data will be made available on request.

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