From:	[redacted]
To:	Hampshire Water Project
Subject:	HWTWRP - Water Environment Hearing - Prof Alex Ford research papers
Date:	02 November 2026 22:24:12
Attachments:	PFAS contaminants_Solent coast_Obanya et al.pdf PFAS contaminants before and after sewage Ford and Ginley 2024.pdf Harbour contaminant profiling_Uhlhorn et al 2025.pdf Storm overflow_EDM data Southern_Ford et al 2025.pdf Water Industry disinformation_Ford et al 2025.pdf

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HWTWRP - WA010002 – Examination Process – Water Environment Hearing – XXX

Dear sir/ madam,
I have registered Professor Alex Ford to speak on behalf of the Water Matters Community at the Water Environment Hearing on 12 November. Alex has suggested that I share with you 5 published research papers that provide background information for the matters he will cover at the hearing

1. PFAS contaminants in surface waters, effluents, sediments and coastal food webs off the Solent coastline UK (Obanya et al)
2. Insights into PFAS contaminants before and after sewage discharges into a marine protected harbour (Ford & Gineley)
3. Chemical profiling of surface water and biota in protected marine harbours impacted by combined sewer overflows (Uhlhorn et al)
4. A comprehensive analysis of combined stormwater overflow (CSO) discharges to coastal waters and data communications by a UK water company between 2021 and 2023 (Ford et al)
5. Water industry strategies to manufacture doubt and deflect blame for sewage pollution in England (Ford et al)

Alex does not anticipate asking for any of the papers to be displayed at the hearing, but we felt that it was important to provide them to substantiate the research that will be referenced at the hearing.

Yours sincerely,

[redacted]

On behalf of Water Matters and other community representatives
PINS registration identification number is: [redacted]

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Full length article



Chemical profiling of surface water and biota in protected marine harbours impacted by combined sewer overflows

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ABSTRACT

Few studies exist that focus on contaminants of emerging concern (CECs) in transitional and coastal waterbodies. This study presents chemical profiling of two protected marine harbours on the South coast of the UK sampled in 2022. Across 21 sites, 105 unique compounds were detected (0.05 ng L^{-1} —1798 ng L^{-1}, median: 11 ng L^{-1}) in water samples and biota, including 67 pharmaceuticals, 29 pesticides and nine recreational drugs. There were significant differences between campaigns with increased chemical numbers and concentrations that coincided with increased rainfall and combined sewer overflow (CSO) discharges. The comparison with CSO discharges revealed that they were an important source for loading of specific chemicals with concentrations increasing for some cases by three-orders of magnitude. High relative risks were estimated for sites sampled during recorded CSO discharges for five compounds with risk quotients (RQs) ranging from 1.1 up to 9.3, with the highest risk from the neonicotinoid, imidacloprid. To understand the exposure in biota, six species; one macroalgae (*Fucus vesiculosus*) and five fauna (*Hediste diversicolor*, *Patella vulgate*, *Crassostrea gigas*, *Carcinus maenas*, *Echinogammarus marinus)* were analysed (n 5/species) at a CSO-impacted site. Between eight to 18 compounds were detected with *Fucus vesiculosus* (seaweed) showing the highest accumulation with mean cumulative burdens reaching up to 343 ± 71 ng g^{-1}. Surface water contamination did not correlate with body burdens. Overall, the work highlights the complexity of the chemical space present in a transitional waterbody showing dynamic contamination patterns that are further influenced by tide, rainfall and salinity. CSOs demonstrated an important but compound specific role for CEC input and pulsing into receiving waters.

1. Introduction

Anthropogenic contaminants including pharmaceuticals, pesticides, industrial chemicals are prevalent in the environment with contaminants typically found in parts per billion (ppb) to parts per trillion (ppt) range across various compartments including in solid matrices such as sediment (Amos Sibeko et al., 2019; Wilkinson et al., 2018), soils (Gworek et al., 2021; Sabzevari and Hofman, 2022) and sludge (Mejías et al., 2021) as well as water types including wastewater (Khasawneh and Palaniandy, 2021; Rivera-Jaimes et al., 2018) surface water (Galindo-Miranda et al., 2019; Wilkinson et al., 2022) groundwater (Bunting et al., 2021; Lapworth et al., 2012), marine water (Ojemaye and Petrik, 2019) and in organisms inhabiting these spaces (Belenguer et al., 2014; Świacka et al., 2022; Miller et al., 2018). However, chemical monitoring of aquatic environments has often focused on freshwater ecosystems whilst transitional, coastal and marine waters have remained understudied (Branchet et al., 2021). These surface water bodies have changing salinity gradients that could affect chemical fate, and represent unique habitats for a wide range of aquatic fauna and flora where there is limited ecotoxicity data available for these species' assemblages. Current environmental quality standards are often exceeded in transitional and marine waters with further uncertainty from the limited ecotoxicity information that is extrapolated from freshwater data (Gustavsson et al., 2017; Ghekiere et al., 2013).

Pharmaceuticals encompass a wide range of unique chemicals that are designed to be readily absorbed and elicit a desired physiological effect, which are considered contaminants of emerging concern (CECs). Due to frequent use, these compounds are commonly detected in surface

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water primarily entering through wastewater treatment plants (WWTPs). Similarly, recreational drugs which can overlap with pharmaceuticals via their misuse may also enter waterbodies through WWTP discharges. Removal efficiencies for these and other micropollutants can vary widely between WWTPs (Fernández-López et al., 2016; Munro et al., 2019; Rapp-Wright et al., 2023) leading to a significant point source for chemical contamination in the aquatic environment. For other contaminants, such as pesticides, input is typically considered from diffuse sources such as leaching and run-off from agricultural applications (Challis et al., 2018). However, more recently it has been shown that some specific pesticides such as companion animal parasiticides (*e.g.* imidacloprid, fipronil) have input from wastewater stemming from domestic use on pets leading to a significant down-the-drain emission pathway (Perkins et al., 2024), as well as from treated animals swimming activity (Yoder et al., 2024). To add to the complexity of source emissions, waterbodies are often further impacted by combined sewer overflows (CSOs) which prevent the capacity of sewer systems being exceeded during periods of rainfall. However, CSOs lead to influent being directly input into surface waters which presents a hazard to aquatic life and public health (Munro et al., 2019; Botturi et al., 2021). Current knowledge gaps surrounding CSOs are related to the volume and composition of spills (Perry et al., 2024) further limiting our understanding for source apportionment (Chrapkiewicz et al., 2024).

The aim of this work was to determine chemical profiles crossing a range of unique and important habitats associated with two harbours (Chichester Harbour and Langstone Harbour), both designated as Sites of Special Scientific Interest (SSSI) for biology that have been impacted by pollution from wastewater (Ford and Ginley, 2024) hence the focus of CECs in the present study. To address the aim, surface water samples were collected from 21 sites impacted by WWTP and CSO discharges and analysed using a targeted quantitative LC-MS/MS method for several classes of CECs (148 analytes). Surface water was collected in March (spring) and November (autumn) where input related to wastewater was expected to be increased from storm events. A qualitative screening method that could tentatively identify an additional 350 contaminants was applied to further characterise the broader chemical space. Finally, biological samples were also analysed from Langstone Harbour in the region which included five different species of fauna (*Hediste diversicolor*, *Patella vulgate*, *Crassostrea gigas*, *Carcinus maenas*, *Echinogammarus marinus*) and a macroalgae (*Fucus vesiculosus*) to determine the potential for accumulation. To improve our understanding of risk in the marine environment, it is critical that chemical exposure and hazard are better characterised and pathways that drive exposure are identified.

2. Materials & methods

2.1. Chemicals & Reagents

A total of 148 chemicals were targeted for quantitative analysis in this study by liquid chromatography-tandem mass spectrometry (LC-MS/MS). All non-labelled chemical standards were of analytical purity (≥97 %) and sourced from Merck Life Sciences (Dorset, UK). Additionally, 36 stable isotopically-labelled internal standards (SIL-IS) were of certified analytical purity (≥97 %) and purchased from Merck Life Sciences or QMX (Essex, UK). The full chemical standard list and details are presented in Table S1 of the Supplementary Information (SI). For instrumental analysis, all solvents including methanol (MeOH) and acetonitrile (MeCN) used were of high-performance liquid chromatography (HPLC) grade (or LC-MS grade) sourced from Fisher Scientific (Loughborough, UK). Ultra-pure water was obtained from a Merck Millipore Milli-Q water purification system with a specific resistance of 18.2 MΩ cm or greater. Ammonium formate (HPLC grade), acetic acid (HPLC grade) and formic acid (LC-MS grade) were sourced from Fisher Scientific. All stock solutions were prepared in MeOH or MeCN at 1 mg mL ^{1} and stored in unsilanised amber glass vials (20 mL). Working solutions were prepared from stocks daily in ultra-pure water or MeOH, as required. All solutions were stored at 20 °C in amber vials to reduce possible degradation.

2.2. Sample location & collection

The study was focused on chemical occurrence in Langstone and Chichester Harbours which have been reported to have declining water quality (Environment Agency, 2024; Natural England, 2021). All sites were selected with assistance from citizen scientists from the Clean Harbour Partnership (CHP) that covered tributaries flowing into the respective harbours and samples from the harbours themselves. The CHP advised on sites that were near known discharge points (WWTP or CSO), that were accessible and used by local water users (*e.g.* sailing clubs). Surface water samples were collected across 21 different sites in the region in March (Fig. 1 & Table S2). Of these 21 sites, four sites were re-sampled in November (Sites 6, 10, 11 and 16) and selected based on the proximity to known WWTP effluent and CSO discharge points in the two harbours and increased rainfall in the region compared to March. Rainfall data for the region was taken from https://environment.data.gov.uk/hydrology/station/125b2167-6fe0-4e51-8976-50a733f3d690. Additionally, one site (Site 16) was sampled in August 2022 as an *ad hoc* sampling event in response to an observed CSO discharge by members of the CHP. All samples were collected for 5 consecutive days (triplicate) in March, August and November unless otherwise stated (Table S2).

Biota samples ($n = 30$ total, $n = 5$ per species) were collected only in March from two locations in Langstone Harbour and included seaweed (*Fucus vesiculosus*), ragworms (*Hediste diversicolor*), limpets (*Patella vulgata*), oysters (*Crassostrea gigas*), crabs (*Carcinus maenas*) and shrimp (*Echinogammarus marinus*). Surface water samples were collected by members of the CHP in Nalgene bottles (60 mL) following a training session on sampling. Samples were stored in insulated cool bags on ice during collection and then frozen at 20 °C as soon as possible on the same day. After each sampling event the frozen samples were transported to the lab on ice within 6 h and then stored at 20 °C. Eight of the 21 sites were located within 1 km downstream of WWTPs and storm overflow discharge points (Fig. 1 & Fig. S1). All sites were tidally influenced and subsequently all samples were collected on the ebb tide on consecutive days to ensure consistency between sampling intervals and locations. Full specification of sampling dates, times and sites can be found in Table S2.

2.3. Sample preparation

For biota, the sample preparation followed methods described previously (Miller et al., 2019; Miller et al., 2021). Both limpets and oysters were de-shelled before extraction but the crab samples were not de-shelled due to the small size (<2 cm in diameter). Briefly, 20 mg of lyophilised solid sample material was weighed into a 2 mL Eppendorf tube. These were then spiked with SIL-IS at 50 ng g ^{1} and extraction was performed using 2 mL of 3:1 MeCN:H_2O acidified with 0.1 % acetic acid. The sample was vortexed briefly (30 s), then sonicated for 15 min and finally centrifuged for 5 min (4 °C at 14,000 rpm). The supernatant (1.9 mL) was diluted with 50 mL 10 mM ammonium acetate in H_2O. The preconcentration and clean-up step was performed using SPE with a Strata® Alumina-N cartridge (1 g, 6 mL; Phenomenex Ltd) and an Oasis HLB cartridge (200 mg, 6 mL, Waters Ltd) configured in series. Cartridges were conditioned with 6 mL methanol followed by 6 mL 10 mM ammonium acetate solution. Liquid sample extracts were subsequently loaded and washed with 1 mL H_2O. The alumina cartridges were discarded. The HLB cartridges were dried separately for 20 min and then stored at –20 °C until required for analysis. The cartridges were eluted with 5 mL MeOH and dried using a TurboVap (Biotage, Hengoed, UK) at 40 °C until the solvent was completely evaporated. The sample was reconstituted in 0.1 mL mixture of MeOH:MeCN:H_2O (5:5:90, v/v). The reconstituted sample was filtered using a 0.2 μM centrifuge filter (hydrophilic-PTFE membrane) and transferred to a 0.3 mL glass insert held



Fig. 1. Map showing locations of selected sites (diamonds) for chemical profiling across Langstone and Chichester Harbours in March and November 2022. Blue diamonds indicate the four sites (6, 10, 11, 16) that were re-sampled in November. Triangles represent local WWTPs and circles represent all associated sewer overflow outfalls in the region. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

within a 2 mL autosampler vial with a PTFE/silicone septum cap.

Surface water samples were processed similarly and as described in Miller et al., 2019 and 2021 (Miller et al., 2019; Miller et al., 2021). Water samples were thawed overnight in the fridge at 4 °C and filtered through a 0.2 µm PTFE syringe filter. A 10 mL aliquot from each sample was measured in a volumetric flask. The sample was subsequently loaded onto a conditioned HLB cartridge (conditioned with 6 mL MeOH, 6 mL H_2O) and washed (1 mL H_2O). The cartridge was dried and stored as described above. For analysis the cartridges were eluted with 5 mL MeOH and dried. The samples were reconstituted in 300 µL H_2O:MeCN (95:5 v/v) and transferred to amber glass (non-silanised) screw cap vials.

2.4. Instrumental analysis

For biota extracts, the instrumental analysis followed the method published by Egli et al., 2021 (Egli et al., 2021). A LCMS 8060 (Shimadzu Corporation, Kyoto, Japan) was used for targeted quantitative analysis using at least two MRM transitions. Separations were performed on a Shimadzu Nexera X2 ultra high-pressure LC (Shimadzu Corporation, Kyoto, Japan) configured with a short 5.0 × 3.0 mm, 2.7 µm particle size Raptor™ biphenyl cartridge (Thames Restek, Saunderton, UK) housed within an EXP® Direct Connect Holder. Multiple reaction monitoring (MRM) was performed with polarity mode switching, and quadrupoles Q1 and Q3 were set to unit resolution. Full MS conditions are provided in the SI (Table S3). The injection volume was 10 µL and MeCN was used as a wash solvent between injections.

For water samples, A SCIEX Triple Quad™ 7500 (Sciex, Framingham, MA, USA) was used for targeted analysis. Separations were performed on a Sciex ExionLC™ configured with a 100 x 2.1 mm, 3 µm particle size Luna® Omega Polar C_{18} cartridge (Phenomenex, Macclesfield, UK). MRM was performed with polarity mode switching, and quadrupoles Q1 and Q3 were set to unit resolution (SI Table S3). The injection volume was 20 µL and MeCN:MeOH:IPA:Water (1:1:1:1) was used as wash solvent between injections. A flow rate of 0.5 mL min^{-1} was used for all analysis. Mobile phases A (0.1 % formic acid in H_2O + 5 mM ammonium formate) and B (0.1 % formic acid in MeOH + 5 mM ammonium formate) were used for separation. The full gradient profile and LC conditions are given in the SI (Table S4).

In addition to the quantitative methods described above, a previously validated LC-MS/MS method for drinking water was applied as a targeted qualitative screening method for an additional 350 analytes as the certified reference material was not available (Stahl-Zeng et al., 2020). This method included pharmaceuticals, pesticides, biocides and industrial chemicals. The screening method used the same instrument (SCIEX Triple Quad™ 7500) and conditions described above and given in the SI (Table S4). The screening method applied tentatively identified a compound based on retention time (t_R) within a ± 0.65 min window and two MRM transitions. The qualitative targeted screening method was applied to the samples collected in November only and prioritised due to expected increases in input from CSO sources.

2.5. Quantification and targeted screening

Quantification of surface water samples was performed using pooled matrix-matched calibration curves prepared from 1 500 ng L^{-1}. Pooling was performed by taking a 10 mL aliquot from each replicate of selected sites across the sampling intervals and combining them. The pooled samples were then measured volumetrically (10 mL) and spiked with the target analytes. Background correction was done using pooled neat samples. Triplicate extraction blanks were included in the analysis. Quantifications were conducted when linearity was acceptable, defined as $R^2 \geq 0.98$ for a linear or quadratic regression. For several analytes the sensitivity of the method saturated the detector response at high calibrant concentrations (>500 ng L^{-1}). For these compounds they are reported as > 500 ng L^{-1} as the concentrations could not be reliably estimated from the calibration curves. Analytes were reported as below the lower limit of quantification (LOQ) when the peak had a signal-to-noise (S/N) ratio below 10:1 and the limit of detection (LOD) was defined as peaks with a S/N ratio of ≥ 3:1.

For biota, species-specific matrix-matched calibration curves were prepared by pooling samples into a bulk mass and weighing out 20 mg of dried material to prepare each calibrant. Pooling was done by taking 100 mg of each sample (n = 5) for each species and then mixed using a vortex. Biota samples were spiked with SIL-IS at a concentration of 50 ng g^{-1} dry weight (dw). Pharmaceuticals and recreational drugs were

spiked to prepare the calibration curve at concentrations of 10, 25, 50, 100 and 200 ng g ^{1} dw. Isotope-dilution using SIL-IS was used to correct for variability and background correction was performed using data from unspiked, pooled sample extracts. Quantification was considered reliable where linearity was acceptable ($R^2 \geq 0.98$) and measured LC-MS/MS signals were above a S/N of 10. All quantified data in biota is presented on a dw basis.

3. Results & discussion

3.1. Overview of CEC Occurrence in Surface Water Samples from Langstone and Chichester Harbours

Of 148 target compounds in the quantitative LC-MS method, a total of 105 unique compounds were detected across both sampling periods (339 samples: 67 pharmaceuticals, 29 pesticides and nine recreational drugs). The most frequently detected pharmaceuticals (Fig. S1) included carbamazepine (100 %), tramadol (99.7 %), trimethoprim (98.8 %), venlafaxine (98.5 %), sulfapyridine (96.8 %), sulfamethoxazole (95.3 %), diclofenac (93.5 %), memantine (92.9 %), valsartan (92.6 %), and lidocaine (89.1 %). Of all 67 pharmaceuticals detected, 22 were detected at more than 50 % of the sites. The most frequently detected recreational drugs were nicotine (92.6 %) and ketamine (83.8 %) and the metabolite of cocaine, benzoylecgonine (BZE, 94.7 %) which is in line with previous work concerning recreational drugs in UK surface waters (Miller et al., 2019; Miller et al., 2021; Egli et al., 2021).

Across both harbours, 67 compounds were detected in March and 90 during November. The sampling in spring followed an unusually dry period in the region with three-month cumulative rainfall levels defined as 'notably low' (Environment Agency, 2022) and total monthly rainfall in Havant was 32 % lower than compared to the long-term average (LTA) (Fig. S2). In contrast, during autumn there was above average rainfall (Sept-Oct 2022), with November itself receiving more than double the LTA (Fig. S2).

For pesticides, the compounds with higher detection frequencies (>50 %) included fenuron (94.7 %), simazine (82.0 %), propamocarb (79.6 %), clothianidin (68.4 %) and imidacloprid (59.9 %). Fenuron was quantified at an average concentration of 102 ± 99 ng L^{-1} (maximum > 500 ng L^{-1}). This chemical has been previously detected in several UK sites reaching up to 169 ng L^{-1} (Miller et al., 2019; Miller et al., 2021; Ng et al., 2020). Considering it has not been approved in agricultural applications since 2009, sources remain unclear and LC-MS/MS data from injections of both mobile phase and method extraction blanks were clean (Fig. S3). It was suggested that it is unlikely to be related to agriculture but may stem from use in other sectors such as building and construction (Ng et al., 2020; ECHA, 2024). Similarly, simazine has also not been approved for more than a decade and was detected across 278 samples. This compound which is a WFD priority substance has been detected in WWTP effluent from rural and urban areas in Ireland but not in receiving waters (Rapp-Wright et al., 2023). In a separate study simazine was also detected across several waterways in London (Egli et al., 2023).

Most of the neonicotinoids, which had partial bans since 2018, were completely banned for all outdoor use on any crops from 2020 (Suchenia, 2022). However, regulations for emergency use have seen approval for the compound thiamethoxam for the past four years to combat Yellows Virus in sugar beet (HM Government, 2021). Despite the restricted use, imidacloprid and clothianidin were quantified up to 63.1 and 35.7 ng L^{-1}, respectively. Interestingly, thiamethoxam was not detected at any site but clothianidin is also the major metabolite of thiamethoxam which has been demonstrated in humans, insects and plants (Liu et al., 2018; Nauen et al., 2003; Wrobel et al., 2022) so its presence may be related to the emergency use of thiamethoxam. Moreover, an important source of input for imidacloprid has been reported to be related to its use in pet parasiticides (Perkins et al., 2024; Yoder et al., 2024).

Given that several of these compounds are no longer approved for use for agricultural purposes with 3 substances banned since 2002 (Table S5), it is concerning to find them so widely in these transitional waterbodies at high detection frequencies and at relatively higher concentrations when compared to the range of concentrations measured in this study. The presence may stem from legacy use but for specific compounds (*e.g.* imidacloprid) could be further related to market repositioning following usage restrictions in other sectors (Barron et al., 2024).

3.2. Comparison of chemical input between spring and autumn following recorded CSO discharges

For samples collected in March, there were ten analytes detected across all 21 sites including carbamazepine, lidocaine, sulfapyridine, tramadol, trimethoprim, venlafaxine (pharmaceuticals), fenuron (pesticide), BZE, ketamine and nicotine (recreational drugs). The number of detected compounds ranged from 9 to 44 within any single sample. Mean concentrations of chemicals in samples from March ranged from 0.12 ± 0.06 ng L^{-1} (propamocarb) up to 129 ± 108 ng L^{-1} (citalopram), with cumulative mean concentrations for all compounds across all replicates reaching up to 995 ng L^{-1} (Fig. S4).

All but three sites (2, 5, 8) showed consistent measured concentrations with inter-day standard deviation below 100 ng L^{-1} across all measured CECs, suggesting source input was less variable during dry weather when input from diffuse sources (*e.g.* leaching and run-off) and storm events (*i.e.*, CSO input) is expected to be minimal. Site 5 showed the largest differences in measured concentrations with cumulative concentrations ranging from 26 ± 14 ng L^{-1} to 1769 ± 493 ng L^{-1} dependent on the day (Fig. S5A). The concentrations were elevated on the first two sampling days and statistically significant compared to the last three sampling days (Mann-Whitney U, $p = <0.001$). *Post hoc* testing using Tukey's test showed that the second day was significantly different compared to the last three sampling days ($p = \leq 0.01$). The sampling location was next to a WWTP (Bosham) outlet and close to CSO discharge points. There are two CSOs for this WWTP which include a settled storm overflow (SSO) and a combined emergency overflow (CEO). However, no recorded discharge events occurred from the Event Duration Monitoring (EDM) during or prior to the sampling period and effluent flow for Bosham WWTP was consistent (1909 ± 30 m^3 d^{-1}) across the sampled days. The data indicates that chemical input increased on the first two sampled days but the reason for the pulse is not clear.

Compared with the pharmaceuticals, the measured concentrations of pesticides were lower with average concentrations ranging from; fenuron (102.1 ng L^{-1}), imidacloprid (26.8 ng L^{-1}), clothianidin (8.3 ng L^{-1}), simazine (0.3 ng L^{-1}) and propamocarb (0.1 ng L^{-1}). Of these quantified pesticides (Fig. S3), none are currently approved for agricultural use except propamocarb. Furthermore, out of 14 pesticides detected in at least one sample, nine are no longer approved for agricultural use in the UK and EU. The ongoing detection of non-licensed pesticides is similar to a Danish study which measured 17 out of 22 pesticides not approved for use (McKnight et al., 2015). In addition to alternative uses for some pesticides (*e.g.* veterinary use) previous works have highlighted the importance of groundwater in the transport and fate of pesticides in the environment (McKnight et al., 2015; Stuart et al., 2006; McManus et al., 2017) with occurrence of legacy pesticides closely associated with mobility in aquifers (McKnight et al., 2015). Therefore, future work should look at long term persistence and transport through soil-groundwater dynamics for these banned substances.

Four sampling sites (Site 6, 10, 11 & 16) were re-sampled in November 2022 after rainfall had increased for the region and where chemical input was expected to increase (Fig. S2 & 7). These four sites were selected as they were in the harbours and close to known point sources. Chemicals separated by major class across the four sites were significantly different (Mann-Whitney U, $p = <0.001$) when compared

between the spring and autumn (Fig. S8). An additional 43 compounds were detected in the autumn that were not previously detected during spring at those sites, which include 27 pharmaceuticals, 14 pesticides, and 2 recreational drugs. During autumn as rainfall increased, both the number of compounds and the concentrations determined were higher compared to the sampling in March and the mean concentrations increased by two-fold in November.

This overall trend was not observed for the pesticides which showed a decline in concentrations which varied by site (Fig. S9 and S10). Pesticides showed a significant decline in concentrations at Site 6 that was driven by fenuron. The trends may be expected here as input of pesticides is typically from diffuse sources rather than point sources associated with the sewerage network (McKnight et al., 2015; Bach et al., 2001; Vryzas et al., 2009; Holvoet et al., 2007). Moreover, pesticide applications for many types of crops in the UK often take place between early Spring to early Autumn. With the low levels of rainfall leading up to March and the collection of samples in November this may further explain the lack of similar trends when compared to pharmaceuticals and recreational drugs. A previous long-term study of pesticide application in Greece (>10 years) demonstrated that the highest concentrations of pesticides were detectable from the first period of rainfall after application (Vryzas et al., 2009). Therefore, although there was increased rainfall in November, the timing of sampling from respective pesticide applications could be an important factor to better understand spatiotemporal trends.

The higher rainfall in the latter half of 2022 increased the number of CSO events recorded by the EDM data and this was reflected from the measured environmental concentrations in this study. There were additional samples collected in August as a discharge was noted by the CHP and confirmed by EDM data after (Table S6). This reactive sampling took place at Site 16 (near Budds Farm WWTP) which had samples collected in March and November as well. Budds Farm WWTP is the largest facility operated in the study area which serves 366,725 population equivalents (PE), with the other 4 WWTPs in the study area serving between 2,000 – 21,000 PE (Water and Catchment, 2021). Overall, at this site, the mean concentrations compared to March increased up to 43-fold and number of compounds increased by up to 3-fold on the sampling days that coincided with recorded CSO discharges (Fig. 2). Moreover, the increase in chemical concentrations on sampling days that coincided with a recorded CSO discharge were significantly higher than sampling days where no CSO event occurred in both August and November (Mann-Whitney U, $p = <0.001$).

Concentrations peaked on the day of the CSO discharge before returning to baseline concentrations, similar to those recorded during March. The rapid decline in concentrations after the spill event in August is likely related to dilution given the tidal influence of these transitional waters. The concentrations for chemicals that increased relative to the CSO discharge returned to the baseline within 24 h and agrees with observed recovery timeframes from a previous study (Munro et al., 2019). However, the spill in August was short-lived lasting only 5 h (Table S6). The last sampling day for November coincided with a spill event that lasted for 55 h, so there may be a potential for longer spills to

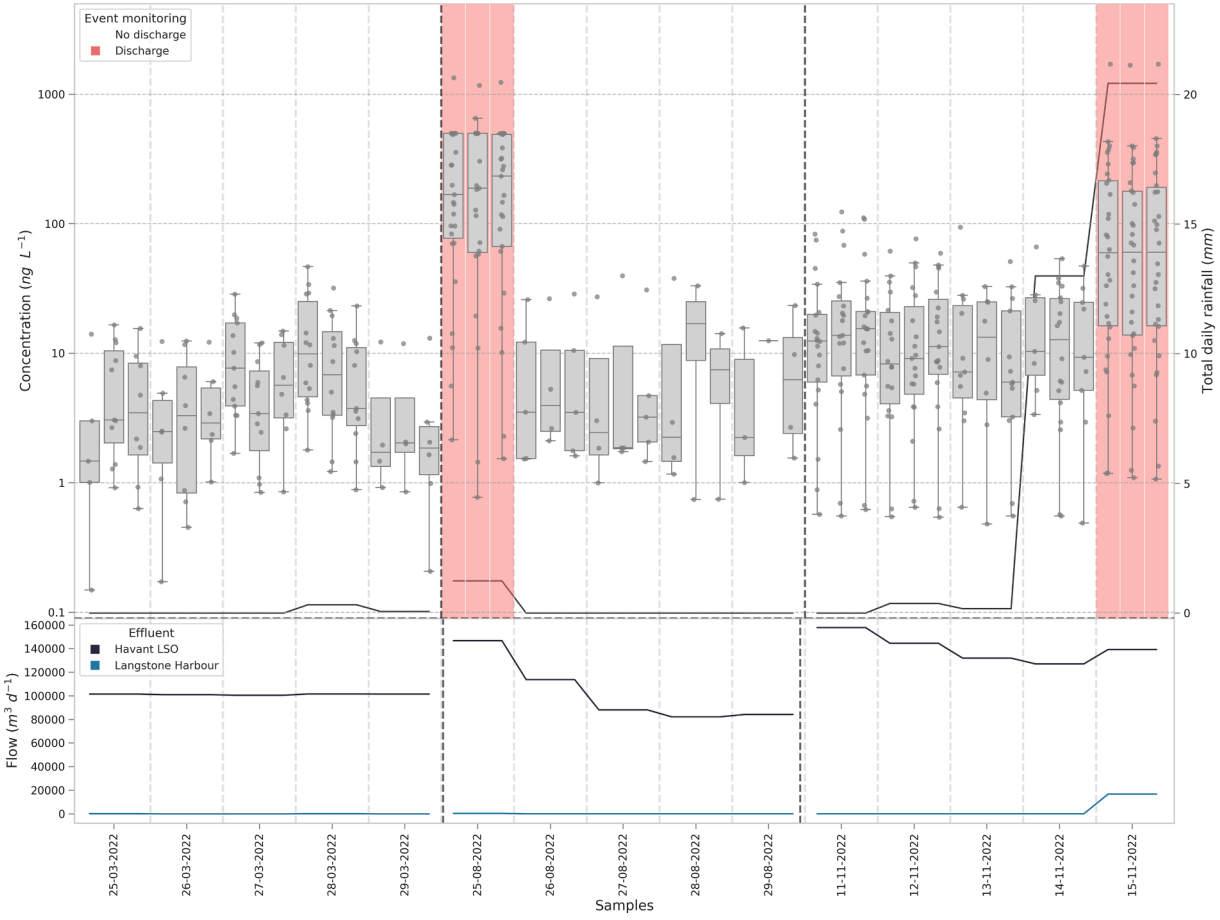


Fig. 2. Measured chemical concentrations across consecutive days at Site 16 (near Budds Farm WWTP) in March, August and November 2022 compared to total daily rainfall (top panel). Red panels indicate the dates that a CSO spill event was recorded by the EDM dataset. Effluent outfall flows per day from Budds Farm WWTP, Havant long sea outfall (LSO) discharges to the Solent and the second outfall discharges during heavy rainfall into Langstone Harbour (bottom panel). Line is the median of the data, boxes represent the 25th and 75th percentile, whiskers represent first or third quartile ± 1.5*interquartile range (IQR). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

have a slower return to baseline concentrations. Whilst duration can be a useful indicator for monitoring CSOs events, the volume of discharges would further increase our understanding of the input and recovery timeframes. Moreover, the pulsing of chemical load into the transitional waters could potentially increase risk for fauna and/or flora that are exposed depending on the recovery period between pulses (Zhao and Newman, 2006; Ashauer et al., 2007; Boxall et al., 2013), where frequent and prolonged pulses may increase sensitivity of organisms.

For Site 16, the average daily concentration of chemicals quantified in March was 6.6 ± 2.7 ng L^{-1}, this increased to 63.9 ± 108.6 ng L^{-1} in August and 48.2 ± 59.5 ng L^{-1} in November, showing a minimum increase of 7-fold. On the individual days where a CSO discharge occurred these averages increased further to 281.1 ng L^{-1} (August) and 167.2 ng L^{-1} (November) showing chemical burden increasing by 25 to 43-fold from baseline concentrations in March. Individual compounds showed considerable increases in chemical burden (Fig. 3). For example, valsartan in March was determined at a mean of 11.6 ± 21 ng L^{-1} which increased to 122.4 ± 209 ng L^{-1} August and 245.1 ± 199 ng L^{-1} in November.

Given the increase in chemical concentrations determined, a comparison to thresholds for predicted no effect concentrations (PNECs) taken from the NORMAN Ecotoxicology Database (Ecotoxicology Database, 2024) (Table S7) was performed for the spring (March) and autumn (November) sampling periods (Fig. 4). The four compounds that exceed freshwater PNECs most frequently were imidacloprid and clothianidin (neonicotinoids) and naproxen and diclofenac (two pharmaceuticals). Quantified concentrations exceeded the lowest PNEC in both sampling periods but were more frequently exceeded during November and potentially related to CSOs. However, there were increased concentrations for multiple compounds in November that stemmed from one site (Site 10).

One antibiotic (azithromycin) exceeded the PNEC in freshwater for one measurement, a further 15 measurements exceeded the marine PNEC value. The lowest PNEC for azithromycin was derived from the cyanobacteria *Microcystis aeruginosa*. For toxicity testing of antibiotics, generally the most sensitive species is microalgae or cyanobacteria in comparison to metazoa such as fish or invertebrates (Fu et al., 2017; Le Page et al., 2017). Moreover, cyanobacteria were equally sensitive when compared to other bacteria including clinically relevant species (Le Page et al., 2017). The risk posed by antibiotics can not only impact ecologically important bacteria and microalgae species but also lead to the spready of antimicrobial resistance (AMR). Previous work has shown that PNECs for surface water ($PNEC_{SW}$) are not always protective of selection pressures that favour AMR, although the PNEC for azithromycin was 10-fold lower than the theoretical PNEC for resistance ($PNEC_{R(T)}$) (Le Page et al., 2017; Bengtsson-Palme and Larsson, 2016). However, knowledge gaps and issues remain for antibiotic risk assessment which are related to the limited number of bacterial species used in testing, lack of testing on marine species and limited data for chronic exposures (Le Page et al., 2017; Sharma et al., 2021).

Imidacloprid exceeded the PNEC in every quantified measurement (apart from in two samples) across both periods with mean risk quotients (RQs) estimated as 3.9 (March) and 3.3 (November) and reaching up to 9.3 across all samples. Other compounds demonstrated RQs reaching up to 7.0 (diclofenac), 3.6 (clothianidin), 1.9 (propranolol) and 1.1 (naproxen). The frequent exceedance of PNECs is a cause for concern and based on these measured environmental concentrations demonstrates that communities are predicted to be negatively affected. Furthermore, the PNECs and RQs are based on individual compounds and do not account for chemicals present as mixtures likely altering their combined risk.

The RQs estimated used PNECs in freshwater as the availability of marine based PNECs is limited and those from the NORMAN Ecotoxicology Database have been estimated by dividing the freshwater PNEC by an assessment factor of 10. Based on these marine PNECs, estimated RQs would increase substantially for the measured chemicals. However,

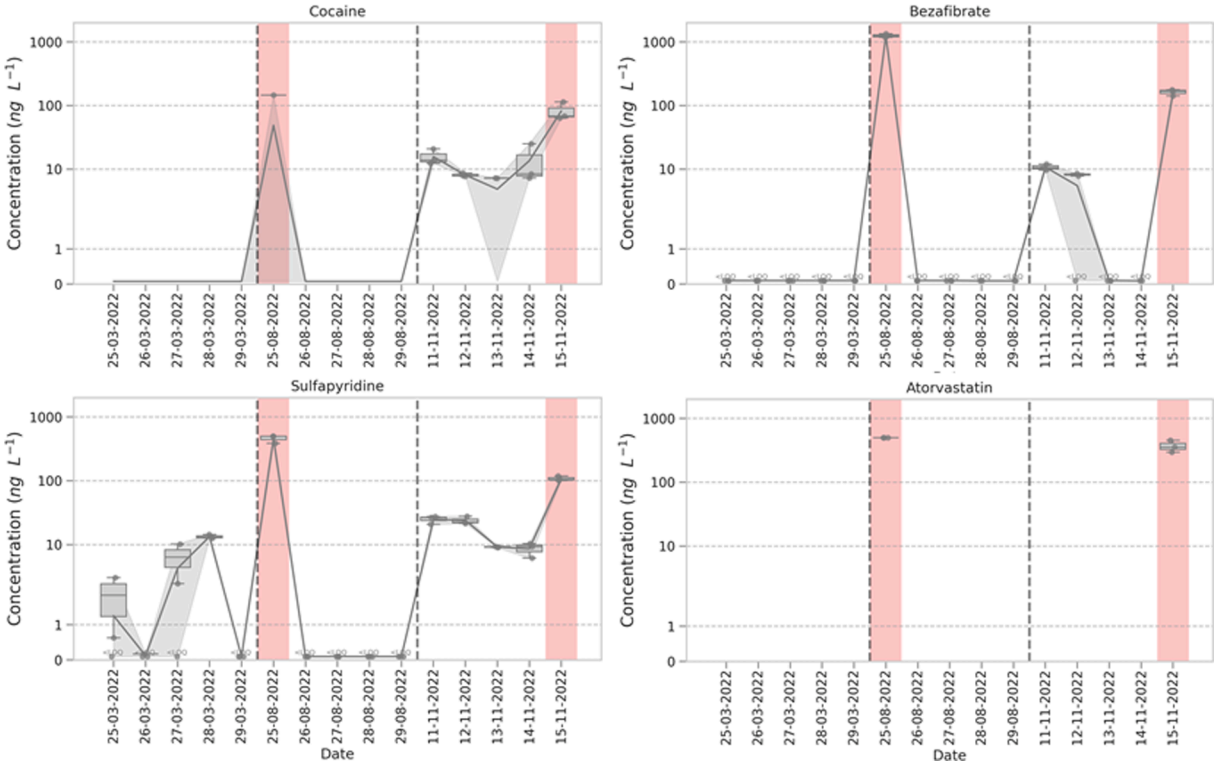


Fig. 3. Examples of concentration profiles for individual chemicals; cocaine (top left) bezafibrate (top right), sulfapyridine (bottom left) and atorvastatin (bottom right) measured at Site 16 (near Budds Farm WWTP) in March, August and November 2022. Red panels indicate the date that a CSO spill event was recorded by the event duration monitoring dataset. Line is the median of the data, boxes represent the 25th and 75th percentile, whiskers represent first or third quartile ± 1.5*IQR. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

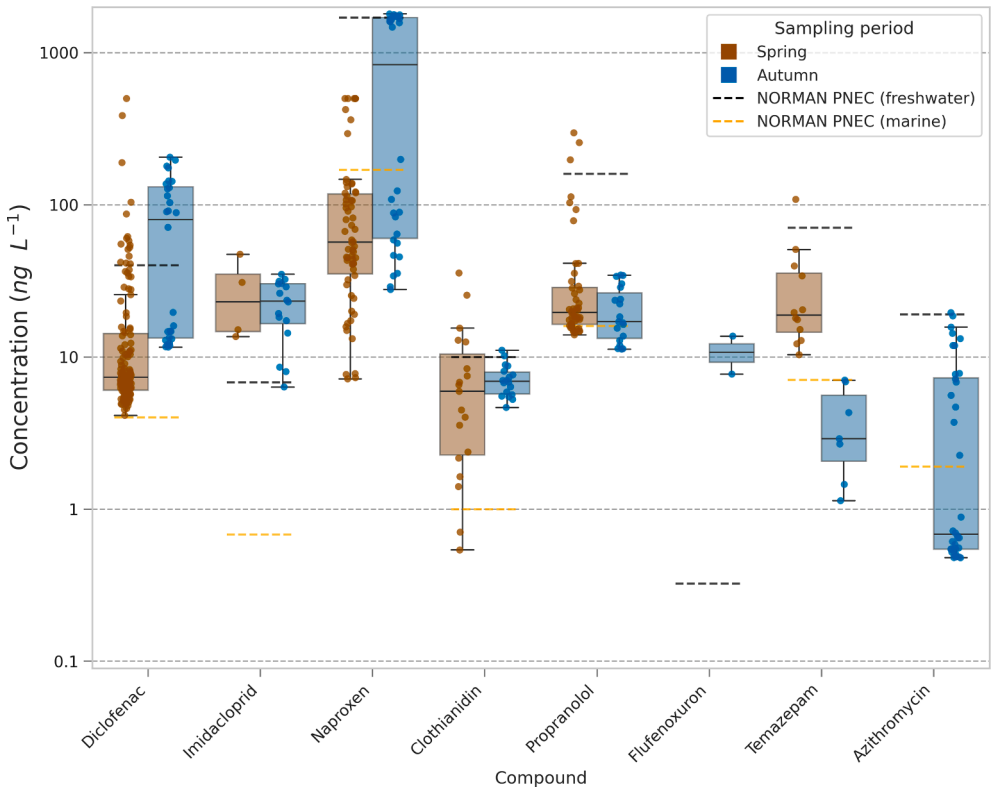


Fig. 4. Comparison of quantified chemical measurements between the spring (March) and autumn (November) sampling periods to both freshwater PNECs (black dashed line) and marine PNECs (yellow dashed line). Line is the median of the data, boxes represent the 25th and 75th percentile, whiskers represent first or third quartile ± 1.5*IQR. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

freshwater PNECs are based on either experimentally derived values from specific species or extrapolated from QSAR approaches that limit applicability. There is clear uncertainty regarding the risk for these estuarine and marine environments, and whilst the assessment factor serves as a conservative approach to risk it could potentially underestimate or overestimate the true risk of chemicals measured at these sites.

A PCA was used to visualise the samples between the different sampling time points across all sites (Fig. 5). The latent variables described 56 % of the data and revealed that generally samples clustered together despite the rainfall increase in November. The exception to this were samples that coincided with recorded CSO spills which separated distinctly from the remaining samples. This indicated that CSOs are an important source for specific compounds in the region. There was not a clear separation between samples collected from sites in March or November indicating that rainfall had less impact on these sites and baseline contamination is similar. However, samples collected from Site 5 clustered more closely with sites influenced by a CSO discharge which is in line with our observations described above. Importantly, all the samples collected from Site 10 (across all consecutive days) in November grouped with samples that were affected by the recorded CSO

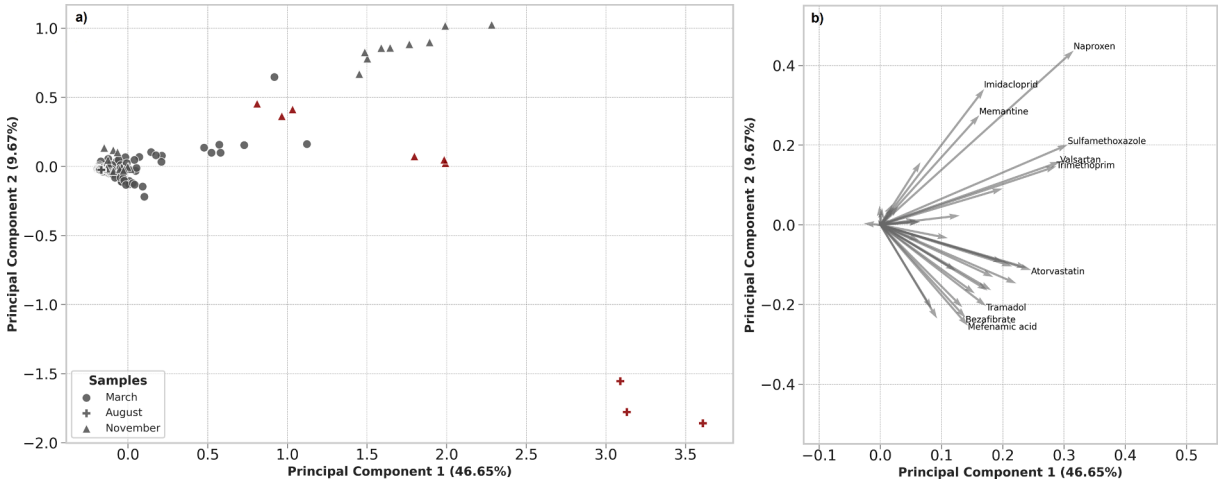


Fig. 5. Principal component analysis of quantifiable chemical measurements from all sampling timepoints in the monitoring campaign. (a) Scores plot showing March samples (circles) and November samples (triangles). Red symbols highlight the samples that were collected during recorded CSO events. (b) loading plot showing chemical features with the strongest influence on the clustering. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

events (Site 10 and Site 16) despite there being no recorded CSO discharge at this site. The two other sites in the November sampling (Site 6 and Site 11) clustered with the remaining samples in March.

The chemical signatures that drove separation were similar except for samples collected from Site 16 which coincided with a CSO event and clustered distinctly from the remaining samples. Closer inspection of the chemical profiles for Site 16 correlated with known CSO markers including BZE which increased by ~ 20-fold and cocaine which increased by ~ 15-fold (Munro et al., 2019). Caffeine was not part of the quantitative method but was included in the qualitative method where peak areas were noted to increase by two orders of magnitude on the day that coincided with a CSO discharge (for both Site 16 and Site 10). Similarly, other potential CSO markers (Munro et al., 2019) increased such as bezafibrate (<2.4 ng L^{-1} up to 176.7 ng L^{-1}) and sulfapyridine (<0.3 ng L^{-1} to > 500 ng L^{-1}). We also considered other potential CECs as CSO markers from a previous study that evaluated removal of pharmaceuticals across 45 WWTPs in the UK (Comber et al., 2019). Two compounds (atorvastatin and clarithromycin) showed high removal (median: 85 % and 68 %, respectively) from influent and could potentially be useful as a CSO marker (Comber et al., 2019). Concentrations of atorvastatin were determined < LOD or < LOQ in all samples except at Site 16 and Site 10, with measured concentrations ranging to > 500 ng L^{-1}. Clarithromycin followed the same trend remaining < LOQ or < LOD but increasing up to 316 ng L^{-1} from samples coinciding with a recorded CSO discharge or the Site 10 samples.

The clustering may indicate that the WWTP at Site 10 potentially had a lower removal efficiency stemming from lower hydraulic retention time due to increased rainfall and/or groundwater infiltration. However, there had been no rainfall for ~ 48 h before sample collection and effluent flow on the sampled days was significantly lower (Welch's Test, $p = <0.05$), at 72 % of the average for the month (November average: 20350 m^3 d^{-1}). Alternatively, the data might suggest that there was a CSO discharge across the sampled days at Site 10 due to the increased concentrations of relevant CSO markers which had not been captured by the EDM (Fig. S5D). The issue of CSOs in sewerage systems is well known in the UK, but evidence gaps still exist particularly related to the volume and composition of discharges (Perry et al., 2024). The data presented here suggests pollutant loads from CSO discharges are increased significantly for specific compounds. The risk posed by CSOs discharging into transitional waters is not well established but pulsed exposures in addition to the increased chemical loading as also noted in previous studies (Ford and Ginley, 2024; Phillips et al., 2012; Kay et al., 2017) may exacerbate pressures on these unique ecosystems.

3.3. Application of a targeted screening method for tentative identification of chemical presence

A qualitative targeted screening method was applied to screen the surface water samples collected in November 2022 for additional chemicals where it was expected that chemical input would be increased. The application tentatively identified an additional 50 compounds from these samples which covered pharmaceuticals, biocides, industrial chemicals, pesticides and metabolites (Fig. 6, S11). The most common chemical class tentatively identified was pesticides (n = 23) covering predominantly herbicides (n = 17) or fungicides (n = 5). The most frequently detected (>80 %) compounds were tentatively identified as acetylsulfamethoxazole, bentazone, iopamidol, 4-methyl-1H-benzotriazole, atenolol, caffeine, diatrizoic acid, gabapentin, iohexol, iomeprol, and paracetamol.

The tentative identification of several of these compounds is concerning given that many have biocidal properties and noted to be very toxic in the aquatic environment under ECHA classifications, for example, malachite green, 4-methyl-1H-benzotriazole and methylisothiazolinone. Moreover, of the 23 pesticides tentatively identified, 8 are no longer approved for use in the UK. However, the tentative identification of these compounds is based on the presence of two known MRM transitions and a retention window but are not confirmatory due to the lack of availability of reference materials, so further work would be needed to fully understand the occurrence of substances that are no longer approved for use.

Other compounds of interest included 1,3-diphenylguanidine which is used in rubber production and could indicate the input from tyre-wear particles and road run-off. This compound has been frequently and widely detected across different matrices including soil (Li et al., 2023) drinking water (Marques dos Santos and Snyder, 2023) and surface waters (Johannessen et al., 2021; Zahn et al., 2019; Zhang et al., 2023) but there is still limited understanding of risk from chemicals in tyre-wear which stems from challenges in analytical measurement, lack of routine monitoring, reliability of emission data, mechanisms of toxicity and regulatory approaches (Khan et al., 2024). Contrast agents used in medical imaging were also frequently detected across all sites with 4 compounds having ≥ 90 % detection frequency (diatrizoic acid, iohexol,

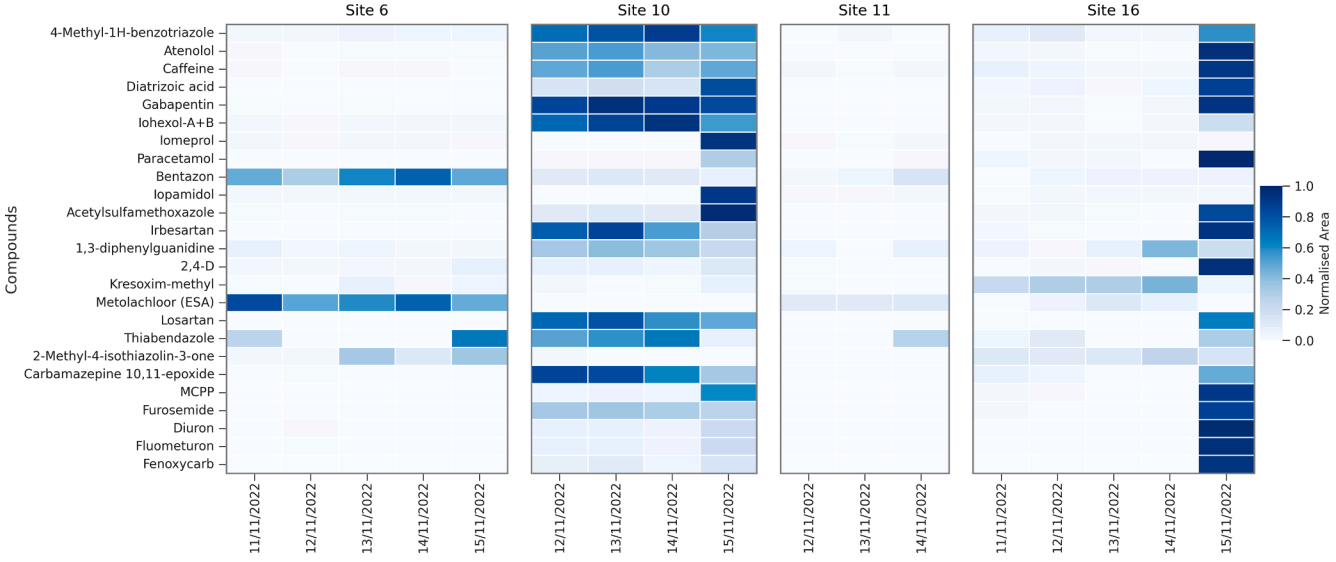


Fig. 6. Heatmap showing relative peak areas (normalised by min–max scaling) of the top 25 most frequently detected chemicals tentatively identified using the targeted screening method from daily samples collected from the four sites in November 2022. Tentative identifications were based on the presence of two MRM transitions and a retention time window within ± 0.65 min of expected t_R.

iomeprol andiopamidol). Two main hospitals are located in the area which are treated by Budd's Farm WWTP (near Site 16) and Apuldram (Chichester) WWTP (near Site 11). Contrast agents have been again frequently found in the environment as they are stable and persistent but are considered low risk based on hazard assessment (Steger-Hartmann et al., 1999) however some research has suggested that the toxicity of transformation products could be increased in comparison to the precursor compounds (Fabbri et al., 2016; Nowak et al., 2020).

The screening method together with the quantitative method applied in this study highlights the complexity of chemical mixtures found in the environment which furthermore does not account for potential transformation or degradation products. The characterisation of the true chemical space will be important to determine hazard and subsequent environmental risk but will depend on nontarget approaches in chemical monitoring and effect assessment such as effect directed analysis (EDA).

3.4. *Chemical burden in biological samples collected from Langstone Harbour*

Several species of biota were collected from Langstone Harbour during the March sampling period and analysed to better characterise uptake (*i.e.*, bioavailability) and potential for accumulation of chemicals in harbour waters. Fauna included crabs, ragworms, shrimp, limpets and oysters, and additionally a common species of seaweed was sampled. Seaweed, ragworms and shrimp were collected by the CSO outfall pipe at Site 16 whereas the oysters, limpets and crabs were collected ~ 2 km south in the main channel, further into Langstone Harbour. The species showed differing levels of chemical burden (Fig. 7, S12). The number of chemicals detected decreased from seaweed (n = 18 compounds) > ragworms (13) > shrimps (10) > oysters (9) > limpets (8) > crabs (8).

The species with the highest body burden corresponded to the seaweed samples which had an average burden of 32.4 ng g ^{1} and an average cumulative burden of 343.2 ng g ^{1}. Seaweed has typically been a focus for inorganic analysis with several previous studies showing contamination by metals (Squadrone et al., 2018; Corrias et al., 2020). Only a few studies have looked at organic contaminants in seaweed (Ford and Ginley, 2024; Álvarez-Muñoz et al., 2015; Pacheco-Juárez et al., 2019; Hahn et al., 2022). One study analysed for 6 benzotriazole UV stabilisers (personal care products) that were found to reach up to 115 ng g ^{1} dw (Pacheco-Juárez et al., 2019). A second investigation screened for 35 pharmaceuticals and detected only four compounds that were all below the LOQ (Álvarez-Muñoz et al., 2015) which contrasts with the results observed here. Given that these macroalgae had the highest average burden and high cumulative burdens, this species, which is widespread, may provide a useful bioindicator of chemical exposure in transitional and marine waters.

Individual chemical burden in ragworms averaged 15.7 ± 26.8 ng g ^{1} with an average cumulative burden of 166.4 ± 100.4 ng g ^{1}. This was followed by the shrimp which chemical burden averaged 12.9 ± 20.7 ng g ^{1} and an average cumulative burden of 67.1 ± 19.8 ng g ^{1}. The relatively higher chemical burden of ragworms compared to other species agrees with our previous study which found that ragworms showed higher body burden compared to a species of shrimp and snail (Miller et al., 2021). The higher body burden in species like ragworms may be related to species traits in that they are an infaunal generalist that burrow into sediments which may potentially be an important exposure route. Sediment as an exposure pathway for emerging contaminants is understudied but previous investigations have shown that this could play an important role for accumulation (Wilkinson et al., 2018; Miller et al., 2021; Celis-Hernandez et al., 2021).

The remaining species showed relatively lower body burdens with average burdens in the order of crab (73.1 ± 110.9 ng g ^{1}) > limpets (10.8 ± 8.1 ng g ^{1}) > oyster (2.4 ± 2.2 ng g ^{1}). The average burden in crab samples was driven by temazepam reaching up to 428.9 ng g ^{1}. Removing these values the average chemical burden reduced to 3.4 ng g ^{1}. Temazepam was only measured below the LOQ during March but has been previously found in surface waters and detected in wastewater effluents (Rapp-Wright et al., 2023; Lei et al., 2021). Removal efficiencies are not well reported but a previous study indicated a mean removal of 78 % (Wu et al., 2015). A yearlong study in Ireland showed incomplete removal with ~ 35 % remaining in effluent samples from both rural and urban areas (Rapp-Wright et al., 2023). However, concentrations were increased in only 2 out of 5 samples and varied widely in other species (ranging from < LOD to 41.8 ng g ^{1}). Bivalves have often been a selected species in monitoring studies (McEneff et al., 2014) as they are filter feeders which filter large volumes of water where the focus has often been on the exposure of contaminants via surface water. Some studies have shown relatively low contamination with individual

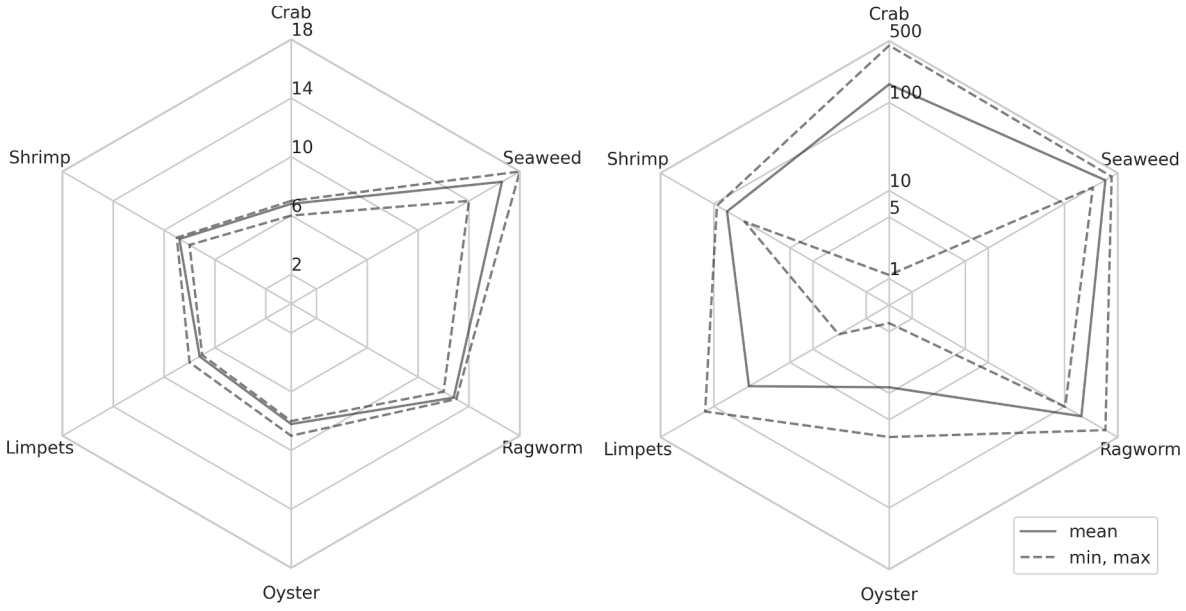


Fig. 7. Radar plot of chemicals measured from biota collected from Langstone Harbour March 2022. The mean number of chemicals detected across species with dashed lines representing the minimum and maximum for each (left). The mean cumulative chemical burden (ng g^{-1}) determined across species with dashed lines representing the minimum and maximum cumulative chemical burdens for each (right).

chemicals typically determined below 10 ng g ^{1} (Álvarez-Muñoz et al., 2015; McEneff et al., 2014; Dodder et al., 2014). However, other studies have shown higher measured concentrations reaching several hundred ng g ^{1} but is likely compound and site specific (Rodil et al., 2019; Pintado-Herrera et al., 2020). Crabs have been less studied and to the author's knowledge only one study has previously determined organic micropollutants (organophosphates) in crabs that were detected below the LOQs of 1–2 ng g ^{1} (Villegas et al., 2021).

Four compounds were detected in all samples across all species which included cocaine, its metabolite BZE, nicotine and oxazepam. This is similar to our previous work in the UK which detected BZE and cocaine at high frequency (Miller et al., 2019; Miller et al., 2021). However, concentrations were higher in this work with averages of 2.9 ± 4.3 ng g ^{1} and 27.7 ± 23.4 ng g ^{1}, respectively. The ratio of cocaine to BZE was 9.5, and the higher level of cocaine agrees with previous studies which showed ratios of > 2.5 in biota (Miller et al., 2019; Miller et al., 2021). At Site 16 (near Budds Farm WWTP) the surface water concentration for these same compounds were typically low, where cocaine was < LOD, BZE was < LOQ (100 % frequency), oxazepam < LOQ (53 % frequency) and nicotine was determined at an average of 11.6 ng L^{-1} (100 % frequency). Measurements compared to surface water samples did not translate well indicating that monitoring studies should incorporate sampling of other compartments within an aquatic system. Without this more holistic overview of exposure our understanding of risk will remain limited.

Individual compound concentrations showed differences between the species sampled. For example, cocaine was determined at higher concentrations in both seaweed and shrimp samples compared to the other species. Ketamine was determined at high concentration in seaweed samples (mean: 198.5 ± 27.8 ng g ^{1}) but was comparably low or not detected in other species. Venlafaxine, tramadol and levamisole were also higher in seaweed samples whereas betaxolol, oxazepam and temazepam showed higher concentrations in the ragworms. The species were collected from two sites within Langstone Harbour. The site 2 km further into the harbour (and further away from the CSO outfall) may account for the lower body burden measured in oysters, limpets and crabs where they were sampled highlighting a potential spatial risk for species in close proximity to CSO outfalls. Therefore, the differences observed in occurrence could be related to spatiotemporal exposure history, species specific traits and differences in toxicokinetic rates which has been demonstrated in controlled exposure studies with several macroinvertebrate species to be related to biotransformation and organism morphology (Rubach et al., 2010; Dalhoff et al., 2020).

4. Conclusions

A study into chemical pollution in a transitional water body revealed widespread contamination. A total of 105 compounds were detected and an additional 50 compounds tentatively identified from collected surface water samples. A total of 9 pesticides not approved for use in agriculture were confirmed in surface water samples, with some not approved for over a decade. Sources for these remain unclear but could stem from uses beyond agriculture or input from legacy pollution. There were differences observed between the sampling periods in terms of the number of compounds detected and concentration ranges measured, but this was related to sites close to a CSO outfall recorded to have discharged during the sampling. The coincidence with CSOs was clear and we observed a significant increase in chemical exposure with daily averages increasing up to 43-fold related to these discharges. The risk posed by chemicals present in CSO discharges into receiving waters is limited and further studies should investigate the composition, volume and frequency of discharges to better understand the pressure placed on these ecosystems. The biota analysis showed fewer chemicals detected when compared to the surface water samples reflecting bioavailability and accumulation processes. The variation in chemical burden between species is likely reflect exposure history and species traits, highlighting the importance of species selection in biomonitoring studies, and that routine monitoring for example in (inter)national programmes should monitor across species.

Overall, the study demonstrates the complexity of chemical mixtures found in a transitional and coastal water system in the UK. A limited number of studies have focused on these waters but given that they are complex human-environmental systems, it is critical that monitoring should be directed to better understand exposure as the true risk of chemical mixtures is likely to be underestimated especially for marine species which often lack ecotoxicity data.

5. Data Access Statement

The data underpinning this publication can be accessed from Brunel University London's data repository, Brunelfigshare here under a CCBY licence: https://doi.org/10.17633/rd.brunel.28513478.

CRediT authorship contribution statement

Jasmin Uhlhorn: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Keng Tiong Ng:** Methodology, Investigation, Formal analysis, Data curation. **Leon P. Barron:** Writing – review & editing, Writing – original draft, Resources, Methodology, Investigation. **Alex T. Ford:** Writing – review & editing, Investigation, Funding acquisition, Conceptualization. **Thomas H. Miller:** Writing – review & editing, Writing – original draft, Supervision, Resources, 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.envint.2025.109417.

Data availability

Data will be made available on request.

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Insights into PFAS contaminants before and after sewage discharges into a marine protected harbour

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HIGHLIGHTS

- Seven PFAS concentrations higher after stormwater discharges.
- PFOS and PFOA above EQS within legally protected marine areas.
- Very high bioconcentration (BAF >6000) of PFBA in macroalgae.
- Ecosystem and human health implications discussed.

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ABSTRACT

Per and polyfluoroalkyl substances (PFAS) and their degradation products are a concern to human and ecosystem health. Wastewater treatment plants are not efficient at removing PFAS compounds and are thought to be a major source of these compounds to marine environments. The sewerage infrastructure in the UK, has over 20,000 combined stormwater overflows (CSOs). These CSOs are relief values whereby untreated wastewater can discharge under permit from the Environment Agency with exceptional rain/snowfall conditions. CSOs discharged 3.6 million monitored hours of untreated wastewater into English rivers and coasts in 2023. Concerns have been raised about the proximity of these CSO discharges to highly protected marine habitats. This study is the first to determine that PFAS concentrations are elevated in a highly protected marine bay (Langstone Harbour, England) following recent sewage releases compared to an extended period without discharge. Analysis was carried out into a suite of 54 PFAS compounds of which only one (PFHpA) was detectable above LOD prior to discharges but 8 afterwards. These included banned PFOS (Linear and Branched 8.6 ng/L ∓ 0.90) and PFOA (2.9 ng/L ∓ 0.29) which were above annual average EQS for inland and 'other' surface waters. Most of the PFAS compounds detected doubled in concentration above LODs. These two-fold increases we discuss are likely conservative estimates based on the use of LODs and tidal conditions. Additional Oysters (*Crassostrea gigas*) and Seaweed (*Fucus vesiculosus*) were taken revealing high concentrations of the shorter chain PFBA (6.99μg/kg ∓ 2.42 ww) in seaweed samples. These seaweeds were calculated to have conservative bioaccumulation factors (BAF) > 6000 for PFBA indicating these algae might be an important reservoir of some PFAS contamination. We discuss these results in the context of the largescale discharges of untreated wastewater nationally and globally, and call upon a need for a better understanding of the transfer of PFAS contaminants into marine food chains.

1. Introduction

There is growing concern surrounding Per and polyfluoroalkyl substances (PFAS) due to the mounting body of evidence that they can harm wildlife and humans (De Silva et al., 2021). Furthermore, there is currently considerable attention on sewage discharges in the UK following the reports of substantial quantities of untreated wastewater to rivers and coasts, raising concerns about human health for bathers and impacts of wildlife (EA, 2024). Whilst much of the public concern has focussed on human health from faecal pathogens, these typically have a survivorship of several days in saline waters. PFAS compounds on the other hand are thought to survive decades in some environments

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with some estimates into the thousands of years (Russell et al., 2008; Washington et al., 2009)

Per and polyfluoroalkyl substances (PFAS) are a family of several thousand synthetic chemicals that are defined by the OECD as "*fluorinated substances that contain at least one fully fluorinated methyl or methylene carbon atom (without any H/Cl/Br/I atom attached to it) i.e., with a few noted exceptions, any chemical with at least a perfluorinated methyl group (–CF3) or a perfluorinated methylene group (–CF2–)*" (Wang et al., 2021)'. They span multiple industries as a result of their resistance to heat, water, grease and chemicals, making them suitable for over 200 uses: including as waterproof and stain proof materials; firefighting foams; non-stick kitchenware and electronics (Gluge et al., 2020). Despite the range of properties that different PFAS possess, the majority share the attribute of persistence (or the ability to degrade into other persistent PFAS), earning them the label 'forever chemicals' (Cousins et al., 2020). As a result, PFAS compounds are increasingly attracting attention due to their global presence in the environment and their impacts on human health and wildlife.

It has previously been established by Cousins et al. (2022), that the planetary boundaries for PFAS have been exceeded, primarily because of their global spread, poor reversibility of their pollution and the effects associated with their exposure. In the marine environment specifically, PFAS are ubiquitous, present in water, sediment (White et al., 2015), sea ice (Hartz et al., 2023) and shown to be concentrated in sea spray, ultimately leading to secondary pollution on land (Sha et al., 2024). In addition, PFAS are present across all trophic levels in the marine environment, from plankton, the building blocks of the ecosystem (Casal et al., 2017), to apex predators including harbour porpoise (Trimmel et al., 2021), seals (Van de Vijver et al., 2005), otters (O'Rourke et al., 2022) and sharks (Chynel et al., 2021). They are known to bioaccumulate (Munoz et al., 2022) in wildlife and have also been noted to biomagnify in the environment. Androulakakis et al. (2022) concluded that ΣPFAS concentrations were up to four orders of magnitude higher in apex predators than prey species. Furthermore, a range of PFAS have been shown to transfer into the eggs of gannets from two UK colonies (Pereira et al., 2021).

The studies on the specific impacts of PFAS on marine wildlife are somewhat limited, but those that do exist have shown several immunotoxic effects as well as impacts on blood, liver and kidney function in bottlenose dolphins (Fair and Houde, 2018) and impacts on the brain in polar bears (Greaves et al., 2013). Immunotoxicity has been widely reported for legacy PFAS in fish, seabirds and marine mammals (DeWitt et al., 2012; Soloff et al., 2017), but more recently, effects on immunity have been reported for emerging PFAS compounds as well (Alexandrino et al., 2022). Alongside the impacts on wildlife in the marine environment, there is the inevitable impact on human health through consumption of contaminated seafood (Fernandes et al., 2018).

PFAS have numerous routes into the environment, either directly (e.g. firefighting foam) or indirectly through release from wastewater and other waste management sources (Environment Agency, 2021). They have in a number of consumer and industrial uses and therefore, multiple pathways for these chemicals into the sewerage system (Kurwadkar et al., 2022).

Sewage must be treated before being released into the environment, to remove contaminants, pathogens and nutrients. However, some sewage is discharged, untreated, into rivers, estuaries and seas through storm overflows. Sewer pipes in the UK are mostly combined, collecting rainwater and sewage in the same pipes. Storm overflows were therefore designed to reduce the chance of sewage flooding homes and streets during periods of exceptional rainfall when the sewage network reaches capacity. Therefore, a combination of factors including climate change, leading to increased rainfall, and inadequate investment in the sewerage network, has led to the discharge of untreated sewage into rivers, estuaries and seas (Giakoumis and Voulvoulis, 2023; Environment Agency, 2024). In 2021, 1651 storm overflows spilled untreated sewage within 1 km of a Marine Protected Area (MPA) in England for 263,654 hours in total (Unearthed, 2023). Stormwater overflow discharges following exceptional rainfall were considered 'safe' or 'safer' for the environment because of the dilution factors. However, due to their persistence, capacity to bioaccumulate and regular discharge to the environment, PFAS compounds present may damage the ecosystem health of coastal areas.

This study therefore set out to ascertain whether PFAS can be detected in a highly marine protected marine area following sewage discharges from combined sewer overflows (CSOs). The aims of this study were twofold. Firstly, to determine the concentrations of PFAS before and after wastewater discharges to a semi-enclosed marine bay. Second, we investigated whether PFAS compounds were detectable in commercial bivalve species and a ubiquitous macroalgae with close proximity to sewage discharges. It has been estimated that 80% of wastewater is released to aquatic ecosystems untreated (UNESCO, 2017). Therefore, studies such as these will help society understand the scale of the impact discharging forever chemicals can have on coastal and offshore ecosystem health.

2. Methods

2.1. Research location

Langstone Harbour is approximately 20 km^2 natural harbour situated on the southern coast of England, close to the city of Portsmouth (Fig. 1). The harbour was known for extensive oyster, seagrass and saltmarsh habitats and is a designated shellfish water although these are degraded due to pollution, overfishing and invasive species. The harbour has a number of national and international protected designations including being a Site of Special Scientific Interest (SSSI), Special Area of Conservation (SAC), Special Protection Area (SPA), has several local nature reserves and is a RAMSAR site for the protection of migratory birds. The harbour is known as a nursery area of juvenile fish species and has resident common and grey seal populations. Langstone Harbour sits in between and interconnects with Portsmouth and Chichester Harbours which collectively adjoin 'The Solent' which is a coastal region which stretches for about 32 km long and approximately 8 km at the widest post. The Solent also forms part of the Isle of Wight UNESCO Biosphere Reserve. On the north side of Langstone Harbour has the region's main Wastewater Treatment Plant (WWTP) called Budds Farm and run by Southern Water Ltd. Budds Farm treats both commercial and domestic wastewater for over 400,000 people from the city of Portsmouth and surrounding area. Wastewater undergoes tertiary treatment with additional nitrate stripping before being discharged via a long sea outfall 5.7 km south of Portsmouth in the Solent. Under exceptional rainfall conditions, when Budds Farm WWTP can no longer cope with the influx of rainwater, the water company is permitted under licence from the Environment Agency to discharge untreated wastewater via stormwater overflows (CSOs) into Langstone Harbour. The main CSO outfall into Langstone Harbour is a 115 m direct from Budds Farm WWTP (Fig. 2).

2.1.1. Sample collection

Water samples (n = 3) of seawater collected at low tide from a scour pool in Langstone Harbour (Portsmouth, England) at the end of Budds Farm Wastewater Treatment plant (WWTP) Number 2 stormwater overflow (CSO) on the 15th (before; Low Tide 10.02am) and 29th (after; Low Tide 08.31am) of April 2024 (Fig. 2). This CSO pipe is approximately 115 m long and completely submerged at high tide and uncovered at low tide (Fig. 2a). The scour pool is created by the force of the untreated effluent discharging from the overflow is approximately 2 m high by 2 m wide (Fig. 2b). The discharge flow from this CSO is unknown but this WWTP can treat flows up to 108,853 cubic metres (m3) per day. Therefore, when these CSOs activate is when flows are in excess of these values due to excess rainfall.

When 'before' samples were taken there had not been any reported



Fig. 1. Aerial photo of Langstone Harbour facing south with the city of Portsmouth on the Island of Portsea (Top right) and Budds Farm Wastewater Treatment plant in the bottom left (red box; www.bing.com/maps). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

sewage discharges from this CSO for 8 days. This equates to 16 full tidal cycles or opportunities for contaminants to be flushed out of the area on outgoing tides into the Solent region of coastal water. However, between the 1st and 7th April there has been approximately 150 h of discharges to Langstone Harbour. The 'after' sample was taken immediately after there had been over 50 h of continuous effluent discharged between the 27th and 29th April. These water samples were taken 7 h 20 min after the discharge had stopped (02.41am) prior to our sample taken on the 29th. Therefore, there had been one full outgoing tide for discharged contaminants to be diluted and flushed out of the harbour.

2.1.2. Biota samples

Triplicate samples of seaweed (*Fucus vesiculosus*) were collected on the April 15, 2024 from upon Budds Farm No.2 pipe itself and triplicate samples of Pacific oysters (*Crassostrea gigas*) were collected from the old oyster house (50.825341, 1.022200) which is approximately 2.3 km miles south west of the discharge point in the middle of Langstone Harbour.

2.1.3. PFAS analysis

All samples were immediately frozen and sent to an accredited commercial laboratory for the analysis (RPS a Tetra Techcompany) of 54 PFAS compounds. For extraction 1 g of the homogenised sample is weighed directly into a PFAS free centrifuge tube. The samples are then spiked with internal standards (a mix of 13C- and 2H-labeled internal standards, 26 internal standards in total to cover the range of PFAS). Sample spikes/blanks are also extracted at the same time following the same procedure. Water is added so the total volume is 10 mL and followed by 10 mL of Acetonitrile. QuEChERS and dSPE extraction is performed prior to an aliquot of Acetonitrile being taken and blown down to incipient dryness and reconstituted with 500 μL of 80:20 Methanol:Water ready for analysis. The calibration range was 0.02 μg/l to 20 μg/L performed using ISO guide 34 certified reference materials. The analysis was conducted on an Agilent PFC-free 1290 Infinity II UPLC coupled to Agilent Triple Quadrupole 6495C in negative Electro Spray Ionisation (ESI) and dynamic Multiple Reaction Monitoring mode (dMRM). All reagents, chemicals and consumables having been pre-screened to ensure they are PFAS free prior to use. Limits of Detection for each compound can be found in Supplementary Tables 1 and 2

3. Results

The 'before' sewage discharge samples were all below LOD (from 1 to 4 ng/L; see Supplementary Table 1) for all 54 PFAS compounds except PFHpA (1.38 ng/L ∓ 0.08) and one single detection of PFHxA (2.4 ng/L) when the untreated effluent discharges had not occurred for approximately 8 days. Eight PFAS compounds were detected in the 'after' samples following sewage discharges to Langstone Harbour (see Table 1; Supplementary Table 1; Fig. 3).

3.1. Biota samples

Four PFAS compounds found above limits of detection (0.1–0.5ug/kg) in seaweed which were perfluoro-*n*-butanoic acid (PFBA 6.99μg/kg ∓ 2.42), perfluoro-*n*-pentanoic acid (PFPeA 0.85μg/kg ∓ 0.72), perfluoro-*n*-octanoic acid (PFOA; 1 sample above LOD at 0.11μg/kg) and perfluoro-1-octanesulfonic acid (L-PFOS; 1 sample above LOD at 0.17μg/kg; Supplementary Table 2). Biota samples were taken at the same time as 'before' sewage discharge samples when all but one PFAS concentrations were below LOD. Therefore, bioaccumulation factors (BAF = concentration in biota/water) were calculated conservatively using the concentrations detected after sewage discharges. These gave a BAF of 6990 for PFBA, 39 for L-PFOS and 38 for PFOA. The only PFAS compound detected above LODs in the oyster samples was for perfluoro-*n*-octanoic acid (PFOA) at 0.19μg/kg representing a BAF of ~66. Unfortunately, matrix effects raised the LODs for PFBA from 0.1 to 5.0 μg/kg for these biota samples.

4. Discussion

This study represents the first monitoring of PFAS compounds in UK coastal waters following known sewage discharge events. In this instance, there had been over 2 days of continuous discharges from over 400,000 domestic properties. Seven PFAS compounds change from below levels of detection with no sewage discharges for 8 days to being detectable in seawater. These compounds represent a likely minimum of 100% increase in concentration from wastewater effluent if one takes a conservative estimate that 'before' samples were at the LOD. It is possible that the 'after' samples are an underestimate of PFAS



Fig. 2. a) Aerial photo of Budds Farm Wastewater Treatment plant (www.bing.com/maps) with drone footage of stormwater overflow discharging into the harbour (Inset Photo: Chris Pearsal); b) LIDAR photo of sampling area highlighting the CSO scour pool (red circle; Environment Agency ARCHI MAPS www.archiuk.com/). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

concentrations given discharges stopped 1 h before high tide, and over 7 h prior to sampling at low tide. Therefore, the more concentrated effluents had more time to be diluted and washed away on an outgoing tide. Discharges from Budds Farm No.2 are predominantly freshwater and warmer than the receiving seawater therefore it is also likely that sewage discharges form on the surface layer unless there are more turbulent conditions. Comber et al. (2021) when studying WWTPs in the UK found that despite downstream concentrations being on average higher, the concentrations of PFOS and PFOA upstream of WWTW discharges already exceed the PFOS EQS by a large margin. On average, the concentration of PFOS is 19% higher downstream of WWTW compared with upstream samples and for PFOA, downstream concentrations are 35 % higher (Comber et al. (2021).

A report into PFAS in sediments from an adjacent harbour (Portsmouth Harbour) found that SUM PFAS ranged between 68 and 814 ng/kg dw in sediments (DEFRA, 2021) with the proportions of PFOA, PFOS and other PFAS compounds varying spatially within the harbour. Portsmouth Harbour received discharges from untreated wastewater and has several historical landfills plus also firefighter training facilities for the British Navy since the 1990s.The UK EA reports that shorter chain PFAS plus PFOS and PFOA are detected most frequently around the UK coasts and estuaries (EA, 2021). They highlight that "*the relative rank order of substances is different in saline waters, with PFPeA and PFOS being reported most frequently at sites sampled (88% and 81% of sites respectively), followed by PFHxA (79% of sites); PFBS (70% sites), PFHxS (68%) and PFOA (61%)*" (EA, 2021). The concentrations of PFOS reported by the UK Environment Agency in estuarine and coastal waters ranging from 0.00004 μg/l to 0.0095 μg/l (EA, 2021).

Table 1
Eight PFAS compounds detected above LODs following untreated sewage discharges from stormwater overflows to Langstone Harbour (England, UK).

PFAS compound	LOD (ng/L)	Mean Concentration ng/L ∓ SD
perfluoro-*n*-pentanoic acid (PFPeA)	2.0	3.1 ∓ 0.41
perfluoro-*n*-hexanoic acid (PFHxA)	2.0	3.8 ∓ 0.24
perfluoro-*n*-heptanoic acid (PFHpA)	1.0	2.2 ∓ 0.04
perfluoro-*n*-octanoic acid (PFOA)	2.0	2.9 ∓ 0.29
perfluoro-1-butanesulfonic acid (PFBS)	1.0	2.1 ∓ 0.08
perfluoro-1-hexanesulfonic acid (PFHxS)	1.0	1.8 ∓ 0.17
perfluoro-1-octanesulfonic acid (L-PFOS)	2.0	4.4 ∓ 0.51
perfluoro-1-octanesulfonic acid (br-PFOS)	2.0	4.3 ∓ 0.39
perfluoro-1-octanesulfonic acid (PFOS Sum L and br)	4.0	8.6 ∓ 0.90

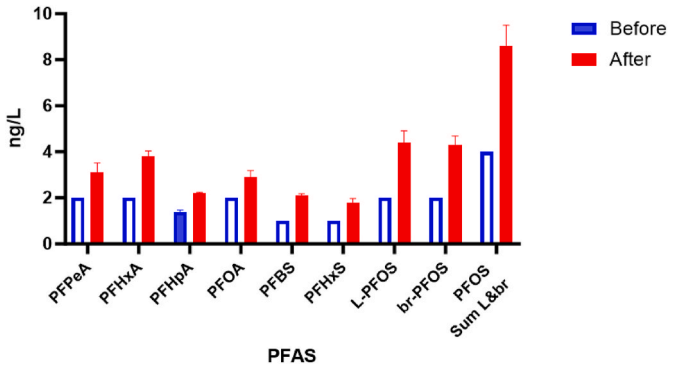


Fig. 3. Concentrations of PFAS compounds (ng/L) before (blue) and after (red) sewage discharges into Langstone Harbour. Unfilled blue bars represent values which were below limit of detection (LOD) and the maximum unknown concentrations. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Mean measured concentrations of PFOA in estuarine and coastal waters ranged from less than the minimum reporting value (MRV 0.0001μg/L) up to 0.0076 μg/l. The annual average (AA) EQS for PFOS in inland surface waters is 6.5×10^{-4} μg/l (0.00065 μg/L) and the AA EQS for 'other' surface waters is 1.3×10^{-4} μg/l (0.00013 μg/L). This study recorded PFOA around 0.003μg/L and PFOS 0.008μg/L in Langstone Harbour. These are on the upper end of the scale for mean values in coastal waters by the EA, and above AA EQS for inland and 'other' surface waters.

Langstone and connecting Chichester Harbour are home to common and grey seals which overall are increasing in numbers in the Solent region (Castles et al., 2021), although numbers are marginally decreasing in Langstone Harbour itself. Castles et al. (2021) was not able to conclude with any certainty whether reductions in numbers over 20 years around Langstone Harbour were impacted by low sample numbers or anthropogenic disturbances. No published data exists on the concentrations of PFAS compounds on these populations however given they are feeding in the surrounding waters it is conceivable they contain elevated concentrations of PFAS similar to other north east Atlantic and North Sea populations. PFAS compounds are known to be immunotoxins (Jacobson et al., 2010; Fair et al., 2013; Soloff et al., 2017; Fair and Houde, 2018; Tang et al., 2023), therefore PFAS contamination may not necessarily manifest itself through impacts on reproduction but through the immune system and these localised seal populations and their ability to cope with national and regional disease outbreaks.

This study as far as we are aware is the first record of PFAS in UK macroalgae. No EQS exists for seaweed, although the EQS for PFOS in Biota is 9.1ug/kg which was derived primarily from fish species (EA, 2021). We observed only 1 seaweed sample from three replicates where PFOS was above the LOD (0.17μg/kg). Interestingly concentrations of perfluoro-*n*-butanoic acid (PFBA) in seaweeds were 6.99μg/kg ∓ 2.42. Noteworthy was that PFBA was not detected above the LOD (1 ng/L) in the before, or after water samples, and a comprehensive survey only found PBFA above detectable levels in 1 or 44 locations (2%) around UK coasts and estuaries (EA, 2021). We calculated conservative BAF values $>$ 6000 for these macroalgae suggesting they might have the capacity to bioconcentrate and be a bio-reservoirs these short chain PFAS compounds. Data on PFBA concentrations in marine biota are more limited however this short chain PFAS has been found to be elevated in effluent discharges (Campo et al., 2014; Kimbanbe et al., 2020; Rodríguez-Varela et al., 2021). Some papers suggest that the shorter chain PFBA are less toxic (Wasel et al., 2021; Ma et al., 2023) although adverse effects to a variety of endpoints have been recorded (Xiao et al., 2022; Ivantsova et al., 2023; Wasel et al., 2022).

Hahn et al. (2022) reviewed a wide range of contaminants accumulating in macroalgae and highlighted that data on PFAS compounds was lacking. They also expressed concerns over the increased use of macroalgae in foods and personal care products. Griffin et al. (2023) found that red and green macroalgae plus seagrass bioaccumulates PFAS compounds, suggesting that it might be useful bio monitoring species. A number of species depend on seaweeds as a food resource. For example, a variety of herbivorous crustaceans and marine snails graze on *F. vesiculosus* and therefore could be a trophic link to predatory fish and birds. Jacobson et al. (2010) exposed sediment dwelling amphipods to PFOS for 5 weeks and observed reduced reproduction, altered frequencies of mature males and increased parasite infection with microsporidian. Interestingly, Botelho et al. (2021) reported on low population densities and unusually low sperm counts in amphipods (*Echinogammarus marinus*) collected close to this study in Langstone Harbour although no causal factor could be identified. PFAS compounds have been known to impact sperm quality and quantity in a variety of species (Sun et al., 2023; Gasparini et al., 2024). The primary food source of *E.marinus* are macroalgae including *F. vesiculosus* used in this study (Martins et al., 2014). *E.marinus* have been recorded consuming 0.29g/day of *F. vesiculosus* which would give them a daily dose of approximately 0.05 ng/day PFOS and 2 ng/day PFDA. Although no data currently exists on the concentrations of PFAS compounds in *E. marinus*. Crustaceans are known to bioaccumulate PFAS greater than that of bivalve molluscs. *E. marinus* conceivably could be a route to higher trophic levels in short chain PFAS due to its importance as a food source to fish and wading birds.

The only PFAS compound detected above LODs in the oyster samples was for perfluoro-*n*-octanoic acid (PFOA) at 0.19μg/kg. Unfortunately, the limits of quantification needed to be raised substantially for PFBA (<5ug/kg) which was detected in the seaweeds and seawater following sewage discharges. No PFOS was detectable in a survey on mussels (*Mytilus edulis*) around the UK between 2016 and 2018 (EA, 2021). Fernández-Sanjuan et al. (2010) in a broad scoping analysis of marine and freshwater organisms observed that no bivalves (mussels and oysters) accumulated PFAS compounds and considered it might be due to their detoxification mechanisms. Lemos et al. (2022) was able to detect PFAS compounds in oysters around the Florida coasts in the USA. The sites with the highest recorded levels of PFAS also had the highest recorded concentrations of phthalate esters (PAE) with these toxicants correlating negatively with health biomarkers in the oysters. The authors suggested that this could be due to multiple sources of PFAS and PAEs including effluent discharges, landfills, agriculture, industry and nearby airports. Wang et al., 2022 similarly was able to detect 28 PFAS compounds in oysters from an industrialised and urbanised Greater Bay Area of China. Total PFAS concentrations ranged from 0.969 to 1.98μg/kg concluding they might make effective and sensitive biomonitoring species for these compounds. The oyster beds where samples were collected in our study were further away from the discharge point compared to the seaweed samples which also could have accounted for low levels of detection. Aquilina-Beck et al. (2020) reported that oysters

(*Crassostrea virginica*) were able to depurate 96% of PFOS when exposed at 0.3 and 3 mg/L within 2 days which could be another explanation for low detection. Our biota samples were taken when there hadn't been recorded sewage discharges for 8 days.

Globally, 80% of wastewater is released to aquatic ecosystems untreated (UNESCO, 2017). Between 2019 and 2023 there were 12.7 million monitored hours of untreated wastewater discharged into English rivers and coasts (EA, 2024), making wastewater discharges a likely large contributor to coastal and offshore PFAS pollution. Sewage sludge is also used extensively in the UK for agriculture with 87% of treated sludge being distributed to land for fertiliser (Water UK, 2024). With many agricultural biocides also known to contain PFAS compounds (Lasee et al., 2022) there is also the potential for diffuse runoff additional sources to the marine environment. O'Rourke et al. (2022) found that sources of PFAS into the food chain to top freshwater predators (Otters) can depend on locality. While some most PFAS concentrations positively correlated with factory production of these compounds, most PFAS concentrations in otters were positively associated with load entering wastewater treatment works (WWTW) and with arable land leading the authors to suggest that WWTW effluent and sewage sludge-amended soils are significant pathways of PFASs into freshwaters (O'Rourke et al., 2022). Therefore, the sources of PFAS compounds to the marine environment likely vary considerably by location. Langstone Harbour is surrounded by large numbers of historic coastal landfills, receives considerable quantities of untreated wastewater and diffuse pollution from agriculture. Here we provide some evidence that PFAS loading into a legally protected marine bay is higher following recent discharges of untreated wastewater. Furthermore, we highlight that macroalgae might be an important trophic link for short chain PFAS compounds. We recommend that more effort is made to differentiate the sources of PFAS compounds to marine environments and the impacts on coastal ecosystems.

CRediT authorship contribution statement

Alex T. Ford: Conceptualization, Formal analysis, Investigation, Methodology, Project administration, Validation, Visualization, Writing – original draft, Writing – review & editing. **Francesca Ginley:** Funding acquisition, Investigation, 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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.chemosphere.2024.143526.

Data availability

Data will be made available on request.

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

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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

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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

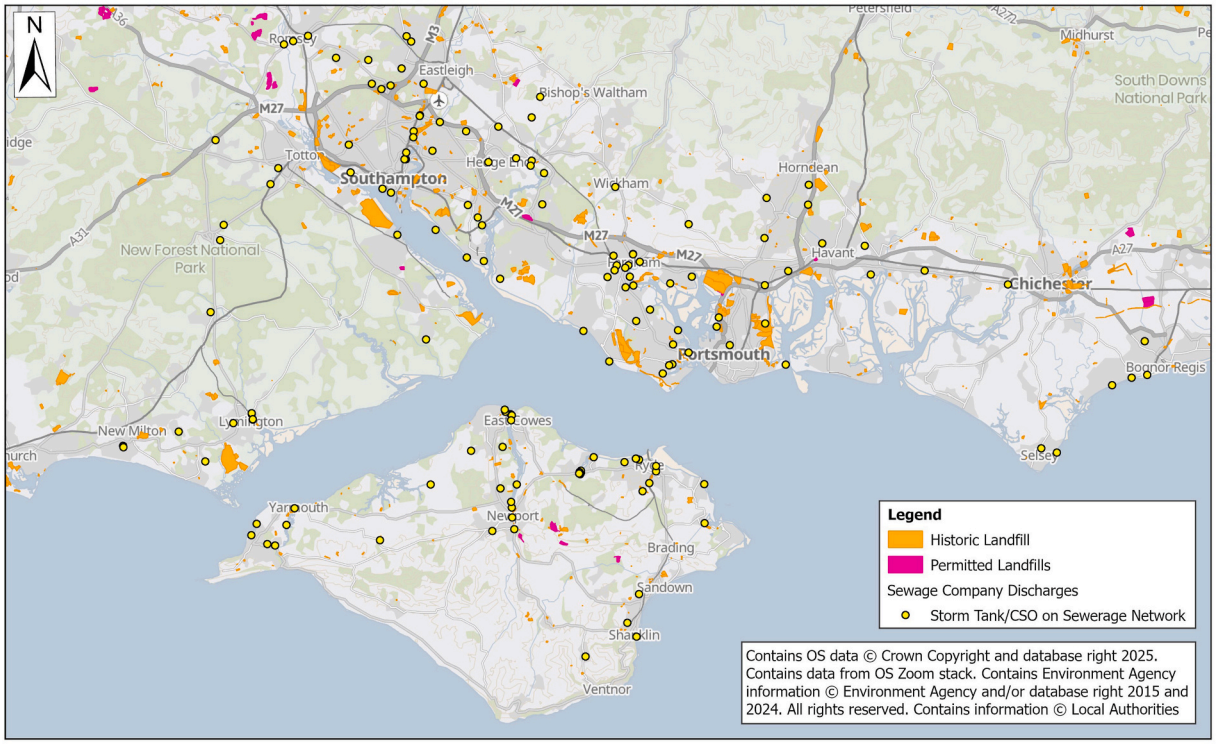


Fig. 1. Map of the Solent region in southern England, showing the main harbours, rivers, and coastal study areas.

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

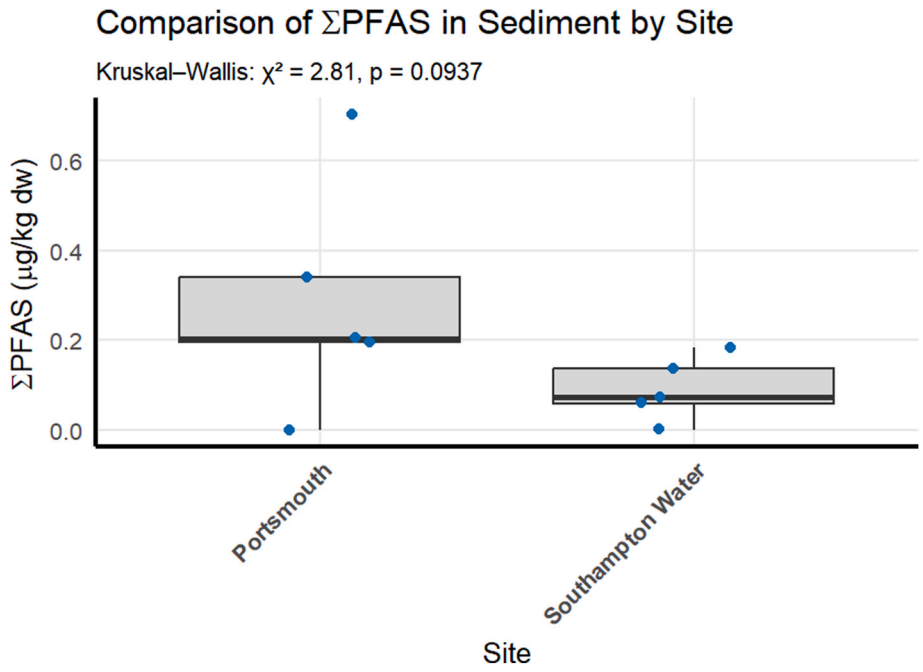
All spatial analyses and maps were produced using ArcGIS Pro (version 3.6). Sample locations were georeferenced and visualised using Ordnance Survey Zoomstack basemaps, which are provided as open data within ArcGIS Pro. Spatial datasets representing historic landfill sites, permitted landfill sites, 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, χ^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

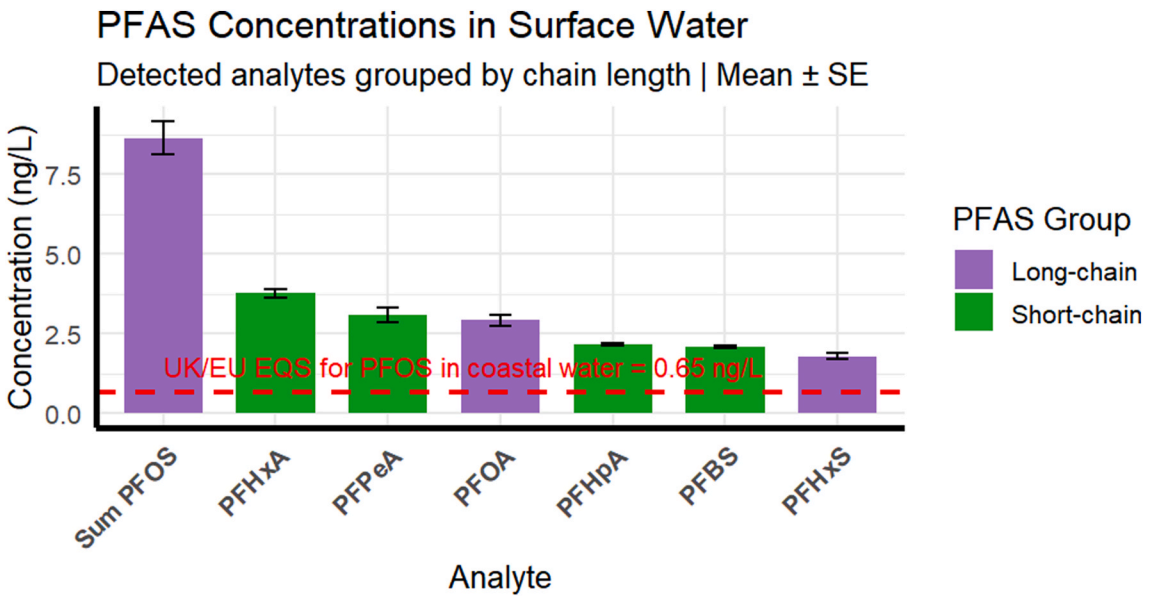


Fig. 3. PFAS concentrations (mean ± SE) in surface water samples, grouped by chain length.
(Data are from field sampling conducted in Langstone Harbour in 2023, originally reported by Ford and Ginley (2024). Colours indicate chain length classification: purple long-chain PFAS, green short-chain PFAS. 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.)

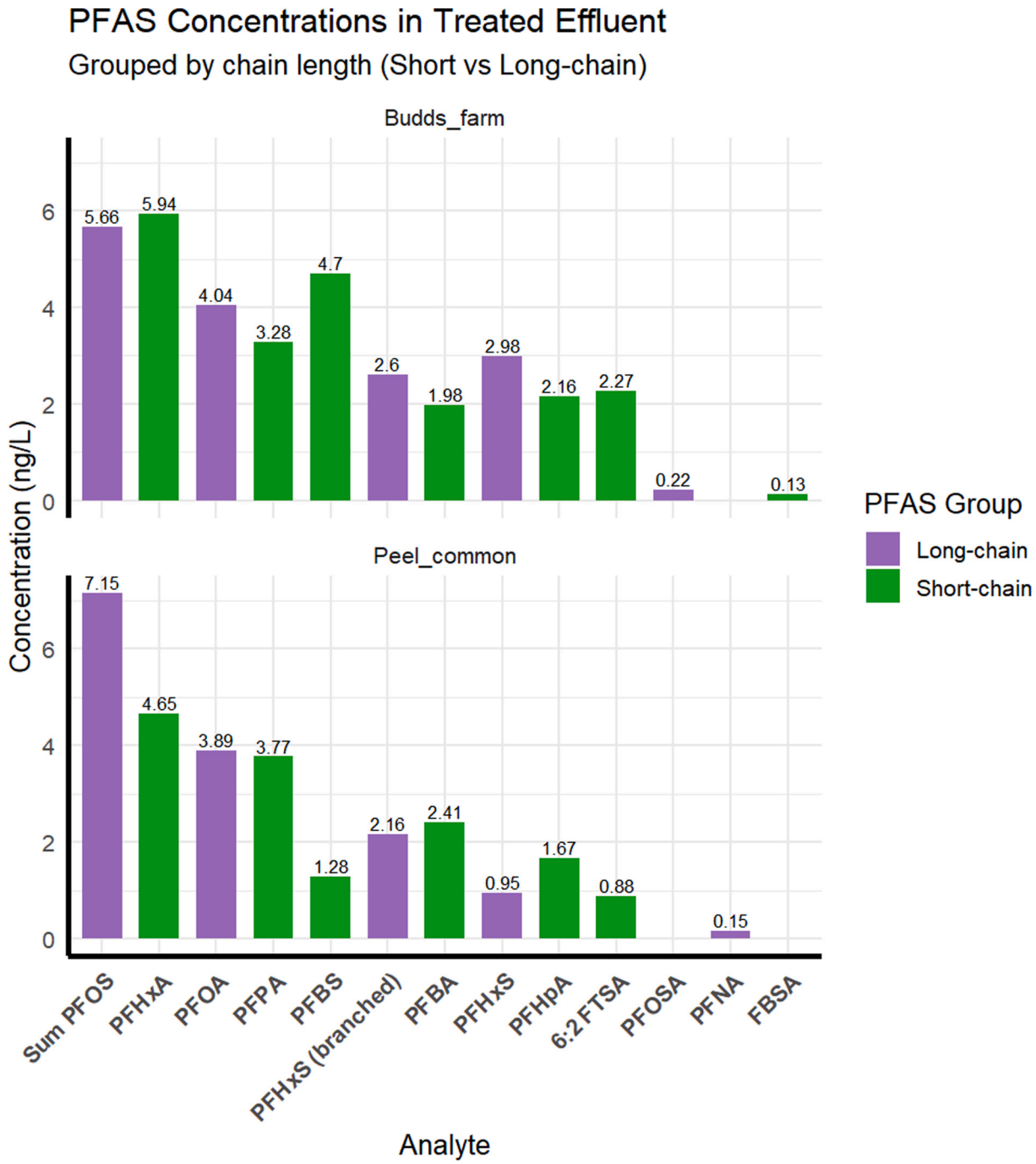


Fig. 4. PFAS concentrations (ng/L) in treated effluent from two wastewater treatment plants: Budds Farm (top) and Peel Common (bottom).
(To determine what concentrations of PFAS might be entering into the Solent region, PFAS data from two of the largest WWTP in the local vicinity were reviewed. Data 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

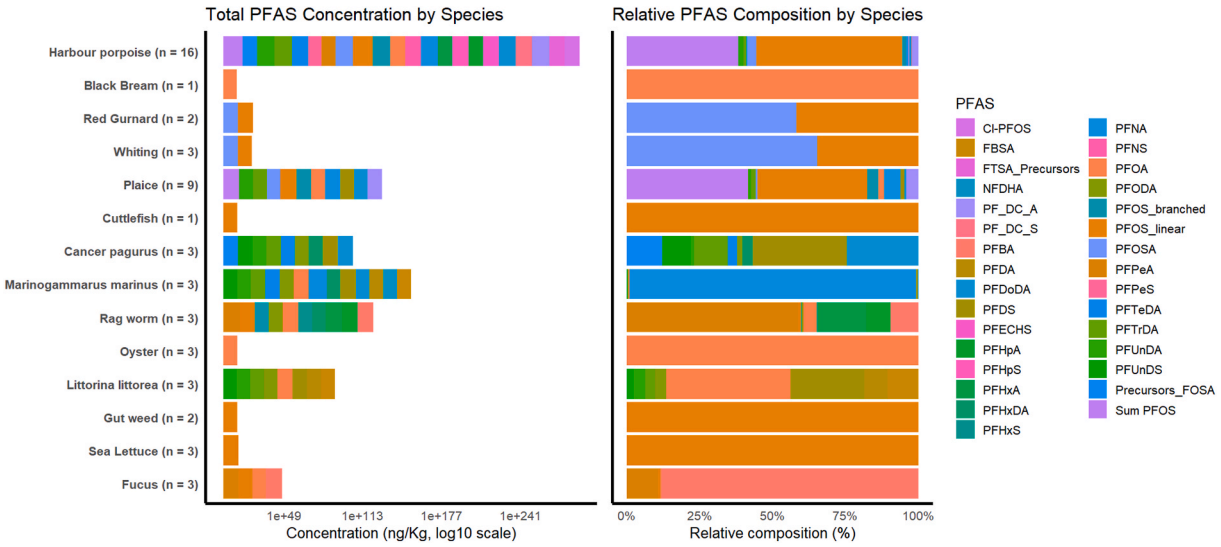


Fig. 5. Total and relative PFAS concentrations by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight). Right: Relative PFAS composition (%) by individual compound.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and 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),

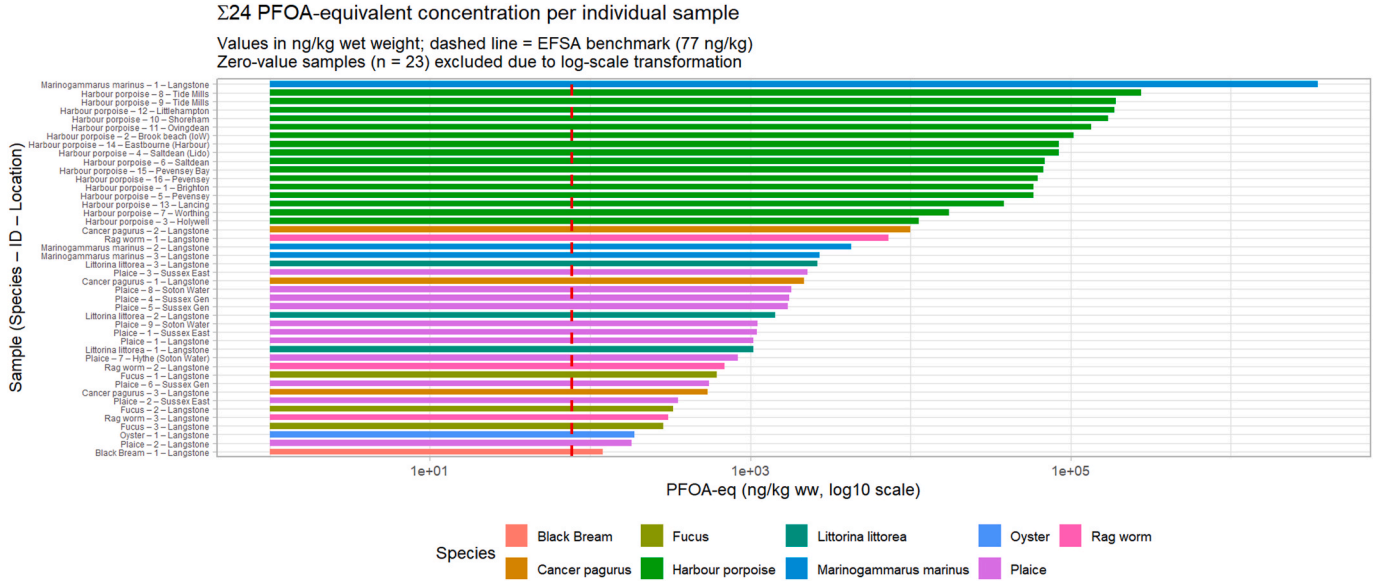


Fig. 6. Summed PFOA-equivalent concentrations (Σ24 PFOA-eq).
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. 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

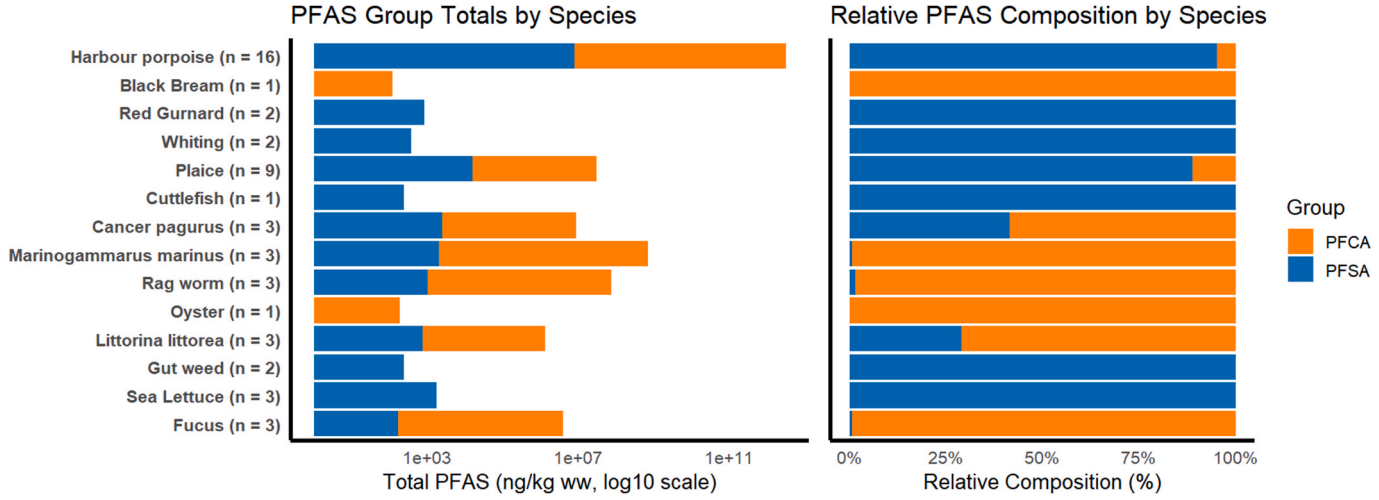


Fig. 7. PFAS group composition (PFCAs vs PFSAs) by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight); Right: Relative PFAS composition (%) by functional group.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and 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.)

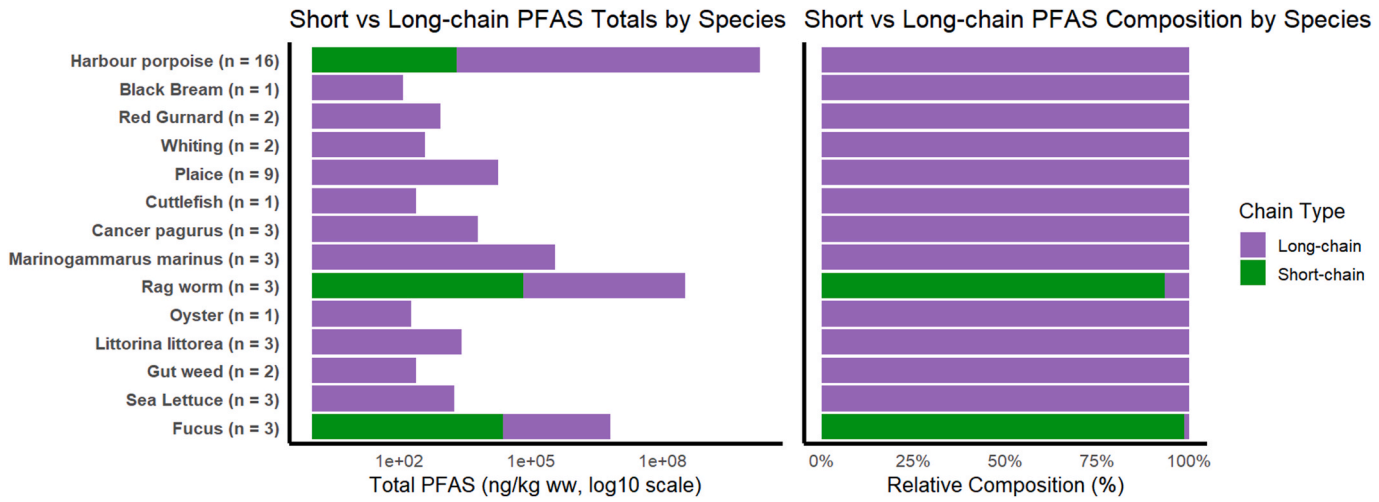


Fig. 8. Short-chain vs long-chain PFAS composition by species. Left: Total PFAS concentrations (ng/kg, $\log_{10}$ scale wet weight); Right: Relative (%) PFAS composition by chain length.
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and 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.)

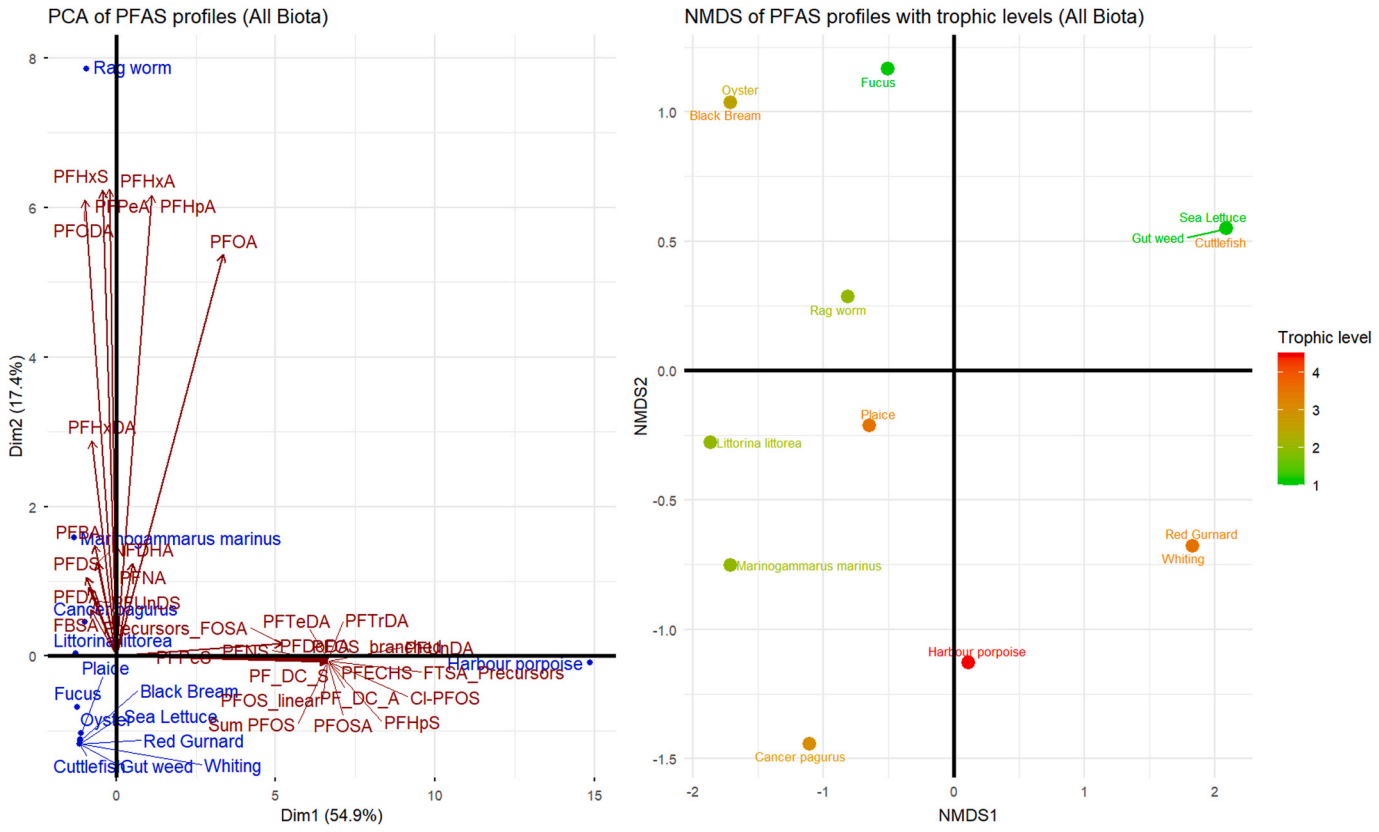


Fig. 9. PCA and NMDS of PFAS Profiles for all biota
(Biota samples were collected across the Solent during winter 2024 and summer 2025. Harbour porpoise samples and a subset of plaice samples were obtained from the Cefas biota monitoring archive, accessed via the Watershed Investigations portal. Principal 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.

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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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A comprehensive analysis of combined stormwater overflow (CSO) discharges to coastal waters and data communications by a UK water company between 2021 and 2023

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Sewage discharges to aquatic environments present a real danger to human and ecosystem health. Event duration monitors (EDMs) from combined stormwater overflows (CSOs) are now fitted to over 90% of storm overflows in England and Wales. These have transformed our understanding of consented and non-consented discharges of sewage and wastewater from UK water companies. In 2018, Southern Water Services Ltd launched "Beachbuoy" which is an online 'near' real-time platform to inform customers when EDMs have been activated at particular CSOs and bathing water sites. Since April 2022, this water company categorized CSO discharges as genuine, genuine but non-impacting, and not genuine (false alarms by EDMs). We analyzed Beachbuoy data to provide an overview of CSO discharges and EDM activity and performance in the region. Across all assets, between December 2020 and February 2023 there were 7164 656 genuine (impacting and non-impacting) minutes of discharges of which 19% overall were regarded as non-impacting of bathing water locations. Non-impacting discharges from all assets often persisted beyond multiple tidal cycles suggesting the impacts on bathing waters may need to be reevaluated. Discharges classed as 'not genuine' (false alarms) were highly variable between CSO for which some recorded false discharges 100% of the time. There were very strong correlations between the triggering of genuine and not genuine discharges and time of the day. Overall, 39% of all total minutes discharged and 14% of discharge events were classified as not genuine. Sewage releases from CSOs were more likely to happen between 7–10 am indicating that earlier morning patterns in human behaviours are substantially impacting the infrastructure's ability to tackle increased capacity in the system through precipitation. We discuss the appropriateness of classifying sewage discharges as 'non-impacted' and whether data should also be obtained on false negative discharges (EDMs not activating) as well as false positive discharges (not genuine). We call for better transparency of the data and models used by the water industry and regulation of how this information is presented to the public.

Water impact

Event Duration Monitors (EDMs) have revolutionised our understanding of the extent to which wastewater discharges occur into aquatic environments. We reveal the frequency and duration of sewage discharges to coastal environments plus the variability in their performance from one UK privatised water company. We highlight how EDM data can be better communicated to the public.

Introduction

Sewage discharges to aquatic environments present a risk to human and ecosystem health.[1 3] Not only do discharges contain human pathogens but also a vast array of toxic chemical contaminants including pharmaceuticals, plastic additives, flame retardants, surfactants, metals and PFAS compounds. Because many of the pathogens are only expected to last in seawater for relatively short periods of time (days/weeks) and are often linked with visible signs of discharge such as wet wipes and sanitary products, the impacts of longer-lasting but invisible chemical contaminants are often overlooked. For example, sewage is a major source of PFAS compounds[4 6] and these are toxic, bioaccumulate and classed as the most environmentally

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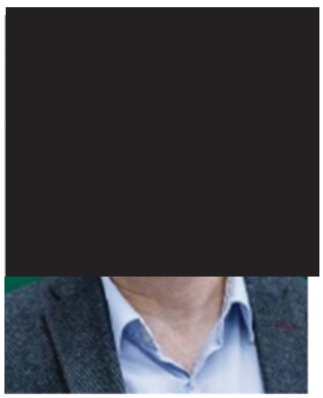
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persistent substances among organic chemicals.[7] Classed as 'forever chemicals' some have put estimates of over 1000 years on their breakdown and we are already starting to see these compounds bioaccumulating in our offshore wildlife.[8 10] Therefore, we must understand the amounts of sewage countries are discharging into coastal environments as their impacts like those of other persistent chemicals (*e.g.* PCBs) may be seen for generations after they are banned.

The water industry in the United Kingdom is publicly owned in Northern Ireland and Scotland, a private not-for-profit organization company in Wales and is made up of nine privately owned water and sewerage companies in England plus a smaller number which provide water only services.[11] Much of the sewerage infrastructure pre-1960s is a combined system which takes foul and surface water runoff. Built into this infrastructure were combined stormwater overflows (CSOs) which, under license from the Environment Agency, allows water companies to legally discharge untreated effluent into water courses during periods of extreme weather (rain or snowmelt). Between 2020 and 2022, there were an estimated 7.5 million hours of sewage discharges into watercourses in England and Wales.[12] These untreated discharges despite being diluted present an impact to the health of water courses and a human health risk to water users.[1,13,14]

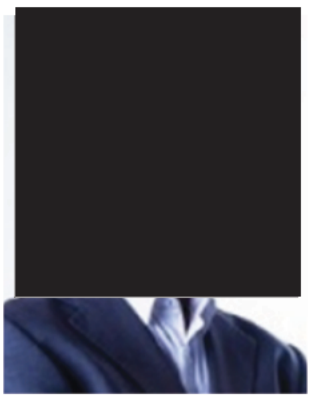
There has been considerable controversy over the frequent discharges from combined stormwater overflows (CSOs) in England prompting critics to suggest a failure in environmental and financial regulation.[15] Currently, there are ongoing environmental and financial investigations into these water companies by the regulators and investigations into the regulators themselves by a newly formed Office of Environment.[15 18] Furthermore, there are several judicial inquiries into the government in the UK High Court by NGOs and businesses plus civil actions against water companies for rebates on water bills. In 2021 Southern Water pleaded guilty to 51 counts of discharging untreated sewage into coastal waters between 2020 and 2025 which was estimated to be 16–21 billion liters.[19] As a consequence, they were fined £90 million and the financial regulator forced the company to discount bills for their poor performance.

There are estimated to be 15 000 CSOs in the UK. In 2013 the Department of Fisheries, Agriculture and Rural Affairs (DEFRA) insisted that the water industry had a better understanding of the performance of their wastewater assets and insisted on a program of fitting Event Duration



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Mike Owens

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Monitors.[12] These EDMs work by either measuring flow or using non-contact radar, ultrasonic, pressure devices and floats.[20] The Environment Agency hopes that the data from EDMs will "inform water company management of storm overflows under current and future pressures; and when they aren't meeting the bar, we can take robust action based on sound evidence to secure improvements, using enforcement if necessary". Details of where and how best to fit EDMs have been published by the Chartered Institute of Water and Environmental Management.[20] They report that special consideration should be taken over the location of EDMs to ensure accuracy (with minimum noise and interference), repeatability (consistently return the same measurement data for comparable spill events), reliability (minimize the likelihood of damage, make maintenance or repairs easier) and safety. Environment Agency data highlights the rapid 14-fold increase in the coverage of monitors, from 862 in 2016 to 12 393 in 2021;[12] (Fig. S1). The take-up of these monitors varies considerably between companies with Anglian Water having 54.3% CSOs listed as having EDMs and Southern Water having 98%;[12] (Table S1). Southern Water Services Ltd covers the South and Southeast coast of England, treating the wastewater and supplying water services to 2.6 million customers. The assets cover 700 miles of coastline, 84 bathing water sites where they have just under 200 CSOs which discharge directly to the sea.

In 2018, Southern Water launched "Beachbuoy" which is an online platform which is meant to inform customers when EDMs have been activated at particular CSOs and bathing water sites (**https://web.archive.org/web/20230330145232/https://www.southernwater.co.uk/water-for-life/our-bathing-waters/beachbuoy**). This initially focused on two locations Langstone Harbour close to Portsmouth (UK) and the Isle of Wight before being expanded across their region. How the data has been presented on Beachbuoy has evolved since its initiation launch from originally having EDMs displaying under review before being categorized as genuine or not genuine *via* human validation. Initially, all

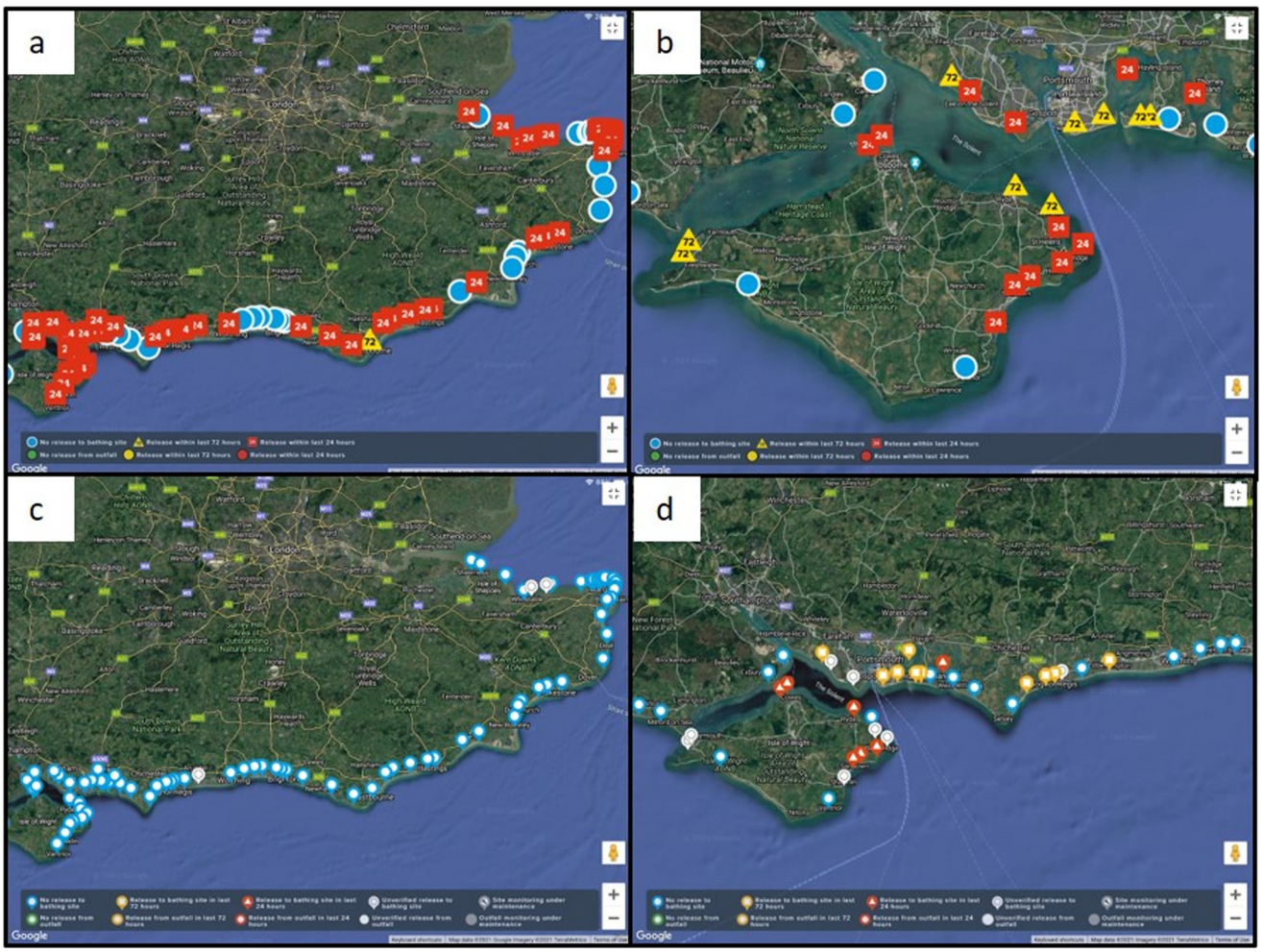


Fig. 1 Screenshots of Southern Water's Beachbuoy app utilizing Google Earth to display a and b) red squares and yellow triangles when a release had occurred in the past 24 and 72 hours respectively (2018 2022) and c and d) the more recent displays when icons would stay white circles until a verification exercise had been conducted and then change to red or yellow icons indicating releases in the past 24 and 72 hours respectively (2022 to date).

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discharges were considered genuine and the public were alerted. In April 2022 Southern Water made changes to the way the data was presented by differentiating between 'impacting' and 'non-impacting' discharges through what they refer to as dynamic modelling (Fig. 1). Discharges which are not expected to impact the water quality at bathing water locations are categorized as non-impacting and are meant to consider unique location-based hydrographic models and the tides and weather coupled with the known degradation rates of faecal coliform bacteria. The time it takes to validate spills as genuine or not genuine can vary spatially and temporally.

Southern Water point out that "Following feedback, we released an upgrade in September 2022 that enables Beachbuoy to now account for tidal conditions. This means that the tool also considers the impact a release has on a local bathing water based on the location of the outfall, the duration of the release and tidal conditions at the time. While previously the Beachbuoy alert cautiously took the worst-case scenario when it was flagged on the app, this could inadvertently cause unnecessary worrying for the public and the tourism industry alike".

The Environment Agency generates summary data on the discharge frequencies from storm overflows and how they compare to the sector averages. The current data for Southern Water is presented in Table S1 however these do not look at EDM performance in terms of false alarms, temporal trends or categorised such as non-impacting discharges. Therefore, this study aimed to investigate the publicly available data from the Beachbuoy website and determine the extent of false alarms and proportions of discharges categorized as non-impacting discharges across bathing sites, season and time. The EDM data is totally new to the wider community of environmental scientists, and as yet there is little academic appraisal of its reliability. This is appraisal is important because the data is already being used by industry, regulators and even scientists in quantitative calculations about sewage impact on the environment.[21] The objective therefore of this paper was to make the data available in a format that trends, patterns and general observations could be summarized and help inform those studying the impacts on human and ecosystem health to understand the temporal and geographic extent of spills.

Methods

An Environmental Information Regulation (EIR) request was put to Southern Water Ltd on the 6th March 2023 which is a UK Statutory Instrument (2004) right of access to environmental information held by UK public authorities (ERI 1751). The data requested were for all the event ID's, site numbers, bathing sites, outfalls, activation start and end times plus activity fields from 2021-01-01 to 2023-03-06 as a csv file. On the 3rd April 2023 the water company reported they required more time to provide a comprehensive response and this would take a further 20–40 days. This EIR (1751) request was refused on the 12th April 2023 on the basis that the data was publicly available on the BEACHBUOY website (previously available *via* **https://www.southernwater.co.uk/water-for-life/beachbuoy**). Southern Water response stated that "*Under the EIR Regulations and guidance issued by the Information Commissioner's Office, we are not required to create information in order to answer a request and as we do not already create, process or record this information, it cannot be said that we "hold" this information for the purposes of Regulation 12(4)(a) EIR. To clarify, Southern Water do not have a downloadable excel file for Beachbuoy, however, you are able to do this yourself. Just select an outfall you are interested in, and you can the copy and paste the data from the Historical Release Table into an excel spreadsheet*". The team were directed to Southern Water's official flow and spill reporting data however this was not suitable as it no longer includes discharges categorized as not-impacting or not genuine (**https://web.archive.org/web/20231212162144/https://www.southernwater.co.uk/our-performance/environmental-performance/flow-and-spill-reporting**). Southern Water are known to use a variety of different EDM sensors (optical, radar and float) across their assets although the types use at a particular location or not public knowledge.

The Beachbuoy website provides a list of all storm release activity relating to coastal bathing waters. Information that appears on the site is displayed in chunks of 25 spills/discharges. Anyone wishing to get a copy of the complete dataset of spills would need to spend many hours manually requesting every one of more than 800 pages, then copying and pasting 25 rows of data from each. Furthermore, because the data presented is dynamic with discharges constantly changing category from unverified to verified, and assigned genuine or not, this means the data is in constant flux making overviews of the data logistically challenging. As a result citizen science software developers wrote programs to automate this labour-intensive process which were optimised, and the results of each program were cross-checked for accuracy (**https://github.com/csodata/tools**). These yielded over 21 000 rows of data (one row per spill event) and over half a million data items. We have made these programs available *via* GitHub to enable others to replicate our analysis.

Results

Between December 2020 and March 2023 there were 25 294 discharges categorized on Beachbuoy (Fig. 2; Tables 1 and 2) across 77 classified bathing water areas and 186 outfalls. Presented alphabetically by CSO name and on a log 10 scale, the median, interquartile ranges and minimum and maximum for each CSO are presented in Fig. 2. The median discharge duration for a typical CSO is between 10 and 1000 minutes with some extending beyond 100 000 minutes.

Data entries scrutinized during this study found there were many incidences where one event ID had multiple entries. This sometimes could be explained by instances

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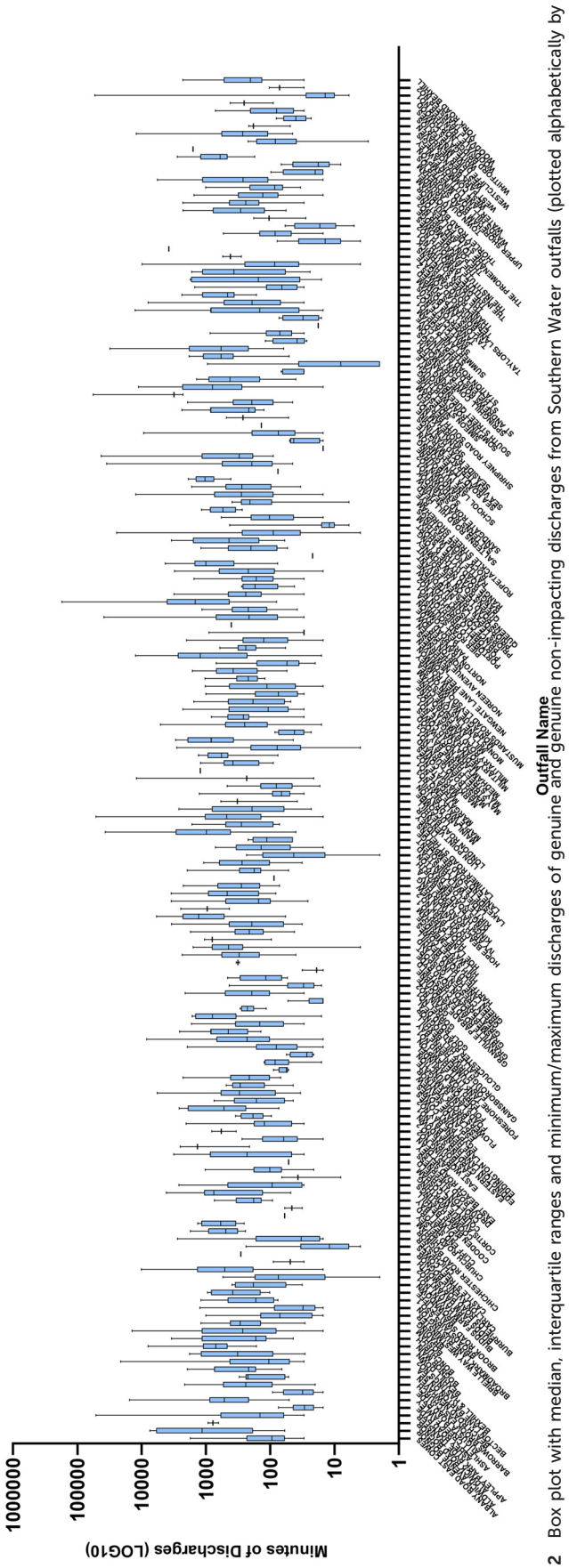


Fig. 2 Box plot with median, interquartile ranges and minimum/maximum discharges of genuine and genuine non-impacting discharges from Southern Water outfalls (plotted alphabetically by discharge location).

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Table 1 Frequency of discharges events between December 2022 and March 2023 categorised as genuine, genuine not-impacting and not genuine from 176 Southern Water outfalls

Year	Row labels	Genuine	Genuine – non impacting	Not genuine	Under review	Under review – non impacting	Grand total
2020	2020	1					1
	Qtr4	1					1
2021	2021	10 439					10 439
	Qtr1	3236					3236
	Qtr2	1679					1679
	Qtr3	1974					1974
	Qtr4	3550					3550
2022	2022	5708	3765	1502	147	270	11 392
	Qtr1	1746	1	39			1786
	Qtr2	451		50			501
	Qtr3	637	390	168			1195
	Qtr4	2874	3374	1245	147	270	7910
2023	2023	1152	1511	472	178	149	3462
	Qtr1	1152	1511	472	178	149	3462
Grand total		17 300	5276	1974	325	419	25 294

Table 2 Duration of discharges (minutes) between December 2022 and March 2023 categorised as genuine, genuine not-impacting and not genuine from 176 Southern Water outfalls

Year	Row labels	Genuine	Genuine – non-impacting	Not genuine	Under review	Under review – non-impacting	Grand Total
2020	2020	339					339
	Qtr4	339					339
2021	2021	2 612 472					2 612 472
	Qtr1	1 085 528					1 085 528
	Qtr2	298 399					298 399
	Qtr3	400 613					400 613
	Qtr4	827 932					827 932
2022	2022	2 315 123	950 533	2 571 231	14 368	22 508	5 873 763
	Qtr1	286 098	68	363			286 529
	Qtr2	48 954		66 249			115 203
	Qtr3	100 699	39 923	30 269			170 891
	Qtr4	1 879 372	910 542	2 474 350	14 368	22 508	5 301 140
2023	2023	883 788	402 401	359 939	12 176	15 423	1 673 727
	Qtr1	883 788	402 401	359 939	12 176	15 423	1 673 727
Grand total		5 811 722	1 352 934	2 931 170	26 544	37 931	10 160 301

whereby an event needed to be verified and subsequently had its category changed. For some event IDs there were over 1000 logs which probably highlights faulty monitoring devices which were allocated the same ID. Other incidents occurred where one event ID number was attributed to several locations.

In total, these data represented over 10 million minutes of discharges (10 095 826 minutes or 7010 days) and are presented in yearly quarters in Tables 1 (event counts) and 2 (event duration) and in descending order in Fig. 3. The top 20 outfalls which between them discharged 5 389 965 minutes of genuine (including non-impacting) are presented in Table S2.

The largest proportion of discharges would take place in the wetter months October to February with the amount discharging in the winter periods varying over the winter months of 2020/2021, 2021/2022 and 2022/2023 (Fig. 4). The winter period of 2022/2023 highlights visually the proportion of discharge categories as not impacting or not genuine since the new classification came into consideration in April 2022.

Excluding the discharges categorized as under review the total number of discharges was broadly similar in 2021 (10 439 events) compared to 2022 (10 975 events; Table 1). However, the total number of minutes reported discharged was substantially more in 2022 (5 836 887 minutes) compared to in 2021 (2 612 472 minutes; Table 2). Discharges were further categorized as non-impacting or not genuine from April 2022 onwards. The total number of minutes discharged in 2022 was broken down into 40% classed as genuine (2 315 123 min), 16% as genuine non-impacting (950 533 min) and 44% as not genuine (2 571 231 min). For comparison from the 1st quarter of 2023 for which data was captured 54% classed as genuine (883 788 min), 24% as genuine non-impacting (402 401 min) and 22% as not genuine (359 939 min). In 2022, based on the number of discharge events (counts), 52% were considered genuine, 34% were classed as genuine non-impacting and 14% not genuine based on the counts of discharge events (Table 1). Therefore, 'not genuine' discharges or false alarms make up 14% of discharges as classified by counts, but 44% of the total minutes discharged. The average

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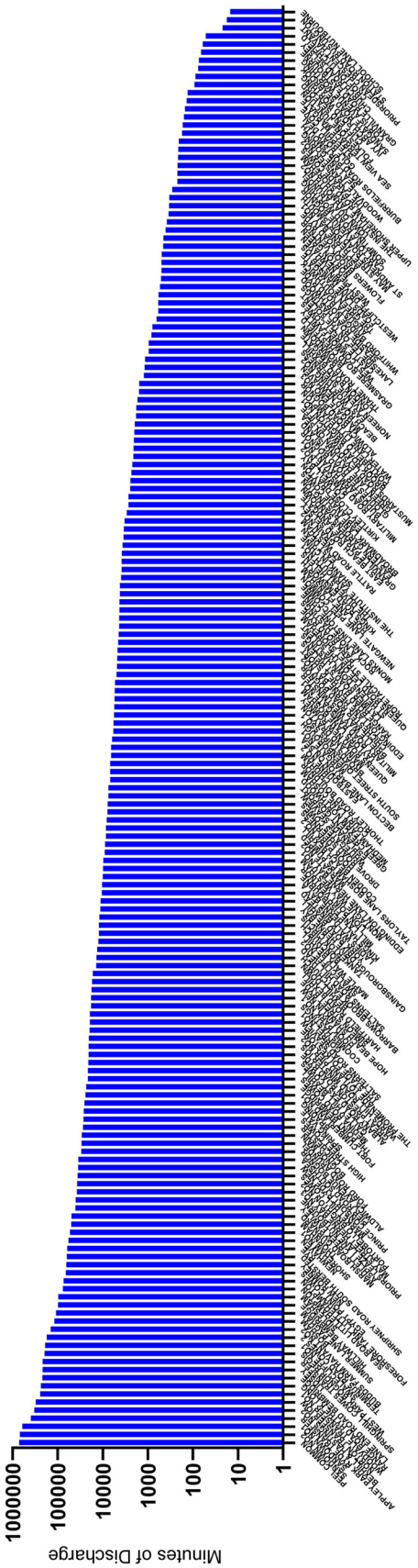


Fig. 3 Sum (minutes – log10) of genuine and genuine non-impacting discharges from Southern Water outfalls.

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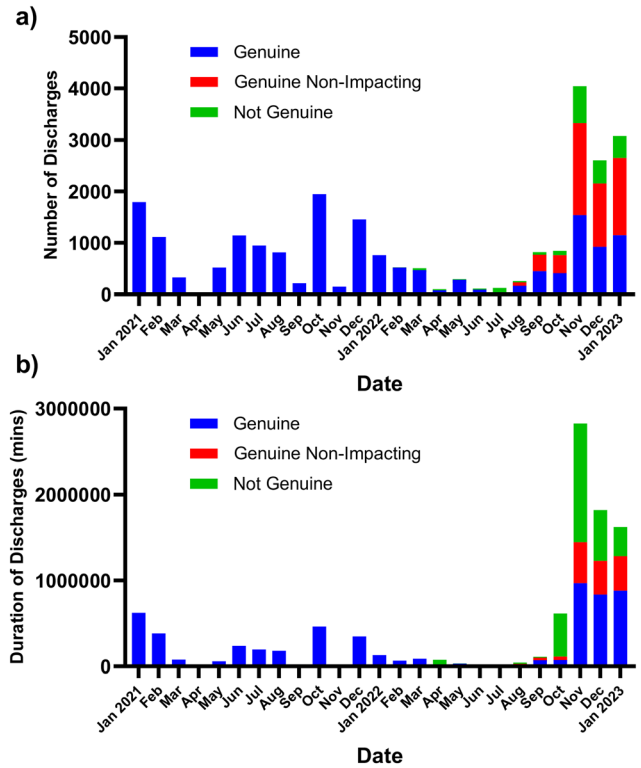


Fig. 4 The (a) number and (b) sum duration (minutes) of discharges from Southern Water outfalls between January 2021 and January 2023 categorized by genuine, genuine non-impacting and not genuine.

duration (min) of genuine, genuine non-impacting and non-genuine discharges were 336, 256 and 1485 minutes respectively. Those classified as not genuine would, on average, last 3 times longer in duration than genuine discharges.

Eighteen percent of the specific outlet locations have discharges categorized as not genuine for more than 50+% of the occasions (Fig. S2a). Twenty two percent of the specific outlet locations have discharges categorized as not genuine for more than 50+% of the overall minutes (Fig. S2b). A number of the outlets had 100% of their discharges classed as non-genuine. These were Albion Groyne Brighton, Black Rock Marine Drive Brighton, Cinque Ports Way Hastings No. 1, Foads Hill Cliffs End, Hurst Point View Totland, Roedean Close Bexhill, Shoreham No. 2, Sunnyfield Road Barton On Sea, Village Road Alverstoke, Military Road Alverstoke, Military Road Ramsgate No. 2 and Roedean Close Bexhill (Fig. S2). Results of the Spearman correlation indicated that there is a significant positive relationship ($r(185) = 0.885$, $p < 0.001$) between the proportion of the spills classified as not genuine and the proportion of the total overall minutes from outfalls (Fig. S3). However, there were many exceptions to the relationship. For example, many outlets had less than 5% of the discharges categorized as not genuine based on the counts but over 90% of the discharges based on the total minutes.

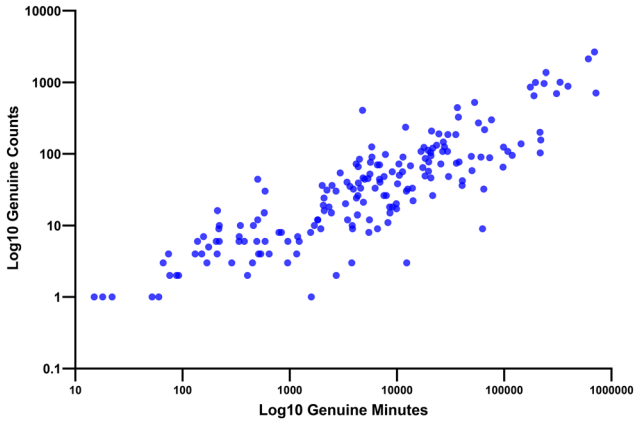


Fig. 5 Correlation between discharges categorised as either genuine (including non-impacting) based on the number of discharge events (log 10 counts) and duration (log 10 minutes) between Dec 2020 and March 2023.

There was a clear and strong positive correlation between the outfalls which discharge most frequently (counts) and the ones which discharged for the longest duration (min) (Spearman rank, $r(185) = 0.875$, $p < 0.001$; Fig. 5).

Discharges by time of day

Discharges irrespective of classification would occur significantly more frequently during the morning (6–10 am) compared to other hours of the day with the exception of 2–3 am (Kruskal–Wallis $X = 146.1$, treatments = 24, values = 12 206, $p < 0.0001$; Fig. 6).

When broken down into 5-minute time bins and plotted as sum duration minutes over 24 hours over there was a clear correlating daily patterns between those discharges classified as genuine (impacting and non-impacting) and those deemed to be not genuine (false alarms) (Fig. 7). The lowest point for

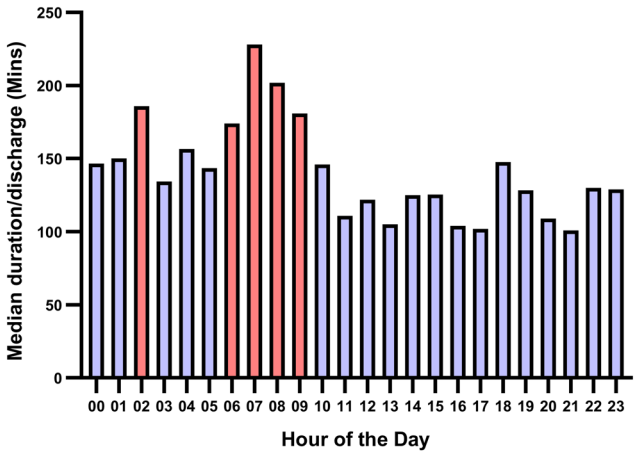


Fig. 6 Median number of discharges (minutes) categorized by hours of the day between December 2020 and February 2023. Those coloured red had significance differences from those coloured blue. Error bars were removed to aid interpretation.

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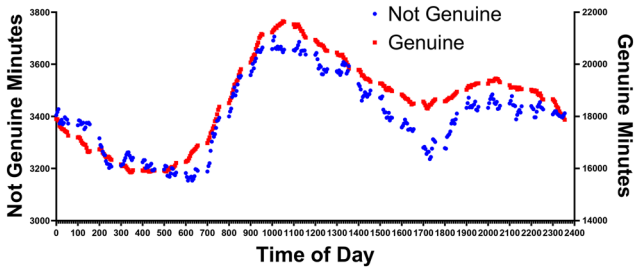


Fig. 7 Sum of minutes binned by 5-minute intervals across a 24-hour period discharged across all CSOs between Dec 2020 and March 2023 categorised as either genuine (including non-impacting) and not genuine.

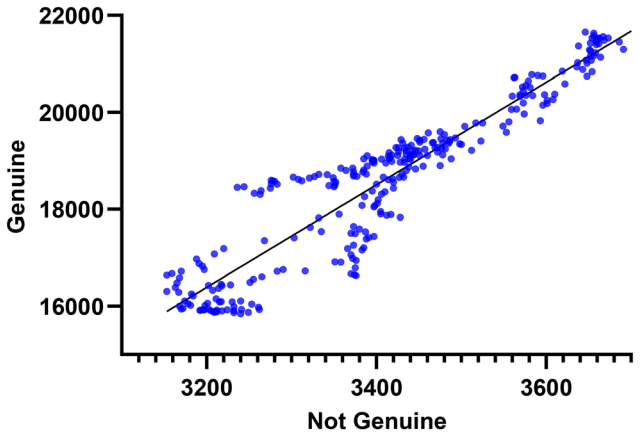


Fig. 8 Correlation between discharges categorised as either sum genuine (including non-impacting) minutes and not genuine minutes the between Dec 2020 and March 2023.

minutes discharged was between 3–6 am but from 7 am onwards through to 11 am there would be increasingly higher numbers of wastewater discharges. Plotting the two variables (genuine and not genuine) together revealed clear and significant positive relationships between the times of the day when genuine and not genuine spills occurred (linear regression (F = 1996; DFn, DFd 1, 286; $p < 0.0001$) $Y = 10.57 \times X - 17\,435$ $R^2 = 0.8747$; Fig. 8).

In 2022 Southern Water decided to categorize discharges not only on whether they were genuine or not but whether they were 'impacting' or not. Between Jan 2022 and Jan 2023, there were 5276 discharges categorized as non-impacting representing over 1.3 million minutes. The rank order of those CSOs reported as discharging the most non-impacting discharges in terms of events and duration are presented in Fig. S4.

Fig. 9 is a box plot of log 10 median, interquartile range and minimum/maximum duration of discharges (minutes) discharged to every bathing water area across the South East region covered by Southern Water. Most bathing water sites have discharges which last a median time of between 100 and 1000 minutes (1.6 h to 16.7 h). However, we determined that many bathing water areas have discharges categorized as non-impacting that proceed the length of 1 and often many 4+ full tidal cycles. Some locations had discharges lasting a maximum of 100 000 minutes (~70 days) and were classed as non-impacting.

Discussion

Water companies in England have come under intense public and political scrutiny after it has been revealed that untreated wastewater regularly discharges into rivers and coasts, often during periods of no or low rainfall.[15] As a result of this pressure, the government mandated the need for EDMs to be fitted into all WWTPs and CSO discharge points. Leading the early rollout of the EDM has been Southern Water whose area covers the South East coast of England. Here we highlight that over 2 years between Dec 2020 and March 2023 there were 25 294 untreated discharges including those still being reviewed representing 10 160 301 minutes (7076 days) registered on the Beachbuoy database.

Are all EDM false alarms really false alarms?

The company changed the way it displayed and represented the data during that period introducing categories such as non-impacting discharges and not genuine discharges for those considered false alarms (or false positives). We highlight that "not genuine" discharges are quite frequent occurrences with EDM devices and 14% discharges by event counts and 44% duration in minutes. Several CSOs have only been presented as having 'not genuine' discharges and have never purportedly had a real stormwater discharge event. What we know is devices are reporting false readings over 14% of the time by event count, what we don't know is the percentage of the time they do not detect discharges when real events are occurring (false negatives). Neither the water companies nor the government agencies provide data on the performance of EDM technologies. During this study, we also highlight very strong correlations in daily readings of not genuine discharges to those occurring that are considered genuine which brings into question whether some false alarms are indeed false but true discharges. The occurrence of genuine and not genuine increasing and decreasing at the same periods throughout the day leaves two plausible explanations. Firstly, whatever causes the genuine discharges also triggered false alarms or secondly, some of the alarms reported as false are indeed real.

Also interesting is the timing of the genuine and not genuine discharges which are most likely to occur during early morning 6 onwards through to about 11 am. This indicates that the triggering of EDMs is not also just a component of excess rainfall but also the daily patterns of the general public such as showering, setting washing machines/dishwashers and flushing toilets before leaving for school or work. This supports Giakoumis and Voulvoulis[21] who recently analysed EDM data and concluded that their findings reveal the chronic under capacity of the English

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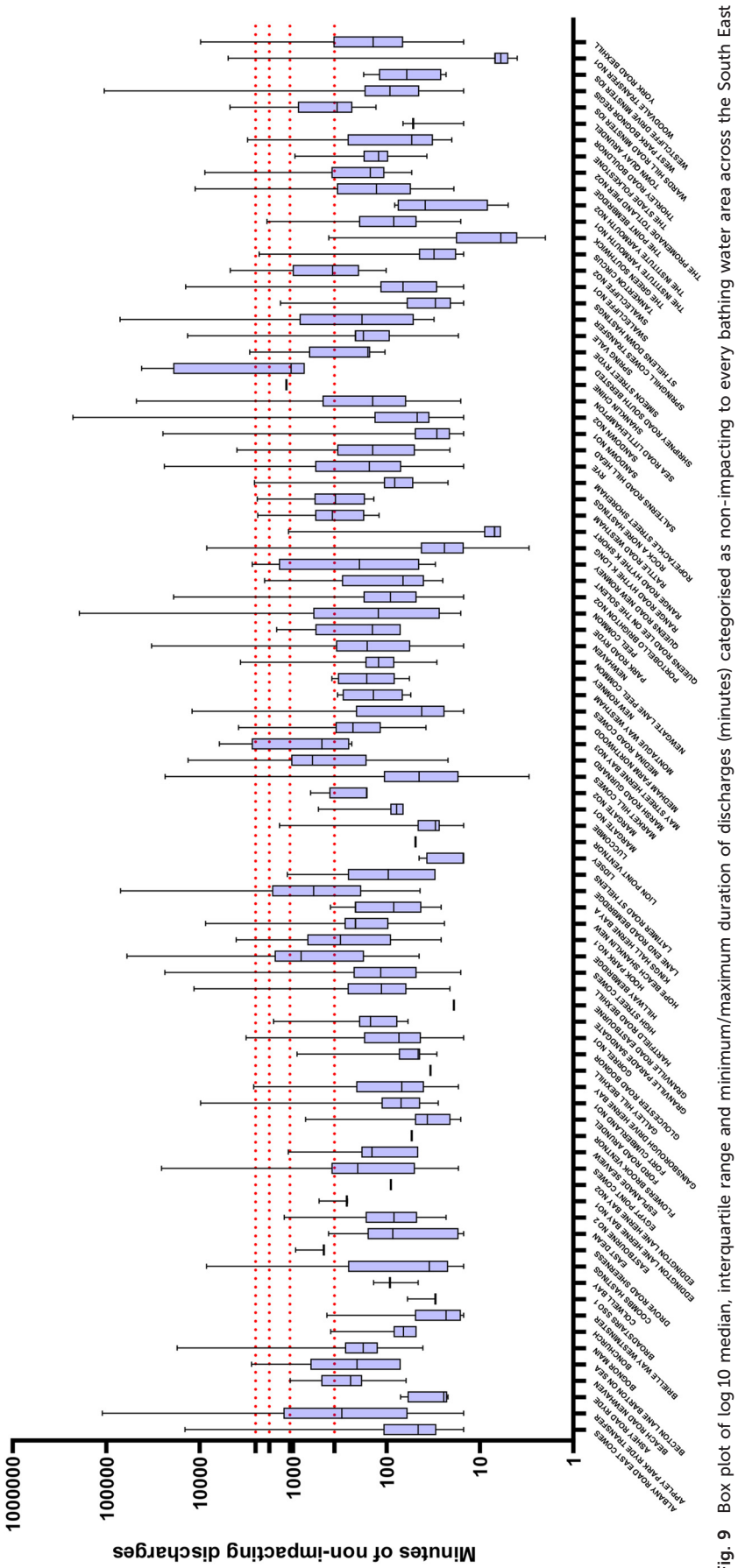


Fig. 9 Box plot of log 10 median, interquartile range and minimum/maximum duration of discharges (minutes) categorised as non-impacting to every bathing water area across the South East region covered by Southern Water between April 2022 and March 2023. The red dashed line indicates one complete tidal cycle (1 × low and 1 × high tide).

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wastewater systems as a fundamental cause behind the increased frequency and duration of CSO spills.

Are non-impacting discharges really 'non-impacting'?

Since spring 2022 when discharges were categorised as 'genuine impacting' or 'non-impacting', there was approximately 35% reported as non-impacting in 2022 and over 50% for the 1st quarter of 2023. In total, there had been 1 352 934 minutes (940 days) of non-impacting discharges over the study period. We have established that the median discharge duration for a typical CSO to the coastal environment is between 100 and 1000 minutes however many of these outfalls have discharges which exceed several tidal cycles and some lasted over 70 days.

In 2022, Southern Water decided to change the way it displayed and presented its discharge data. The iconography on the Beachbuoy portal was changed so that alerts didn't immediately appear as red icons but white icons until discharged had been characterized as genuine or not or impacting or non-impacting. At the time the water company received criticism for the approach with the UK broadsheet newspaper running with the headline "Southern Water alters pollution alert tool to curb automatic red alerts"[22] and the BBC "Southern Water's sewage discharge app changes provokes row"[23] in September 2022. A spokesperson for Southern Water at the time said "This is an important step for the tool, ensuring we provide accurate, fair and clear information to wild swimmers, kayakers, paddleboarders and all users of the beautiful beaches across our region. Beachbuoy is leading the way in providing near real-time data about storm releases, but we must ensure it goes further to inform the public about the impacts to the watercourse and not limit their enjoyment of their local bathing water. We'll never hide data though, with all releases still available on the website".

From April 2022 discharges considered to be genuine were further categorized as to whether they were genuine or genuine non-impacting. This also received criticism for several reasons. Firstly, the algorithms and oceanographic models from which the data was based were not available for scrutiny plus the term 'non-impacting' if the models are correct only provides data for designated bathing water areas and estimates 'risk' of human pathogens in seawater. However, these do not consider chemical contaminants in wastewater and ecological impacts, or discharges modelled currents flowing in the opposite direction to designated bathing water beaches.[24] In July 2023, on the back of this criticism Southern Water changed the terminology from "non-impacting" to "non-impacted" to better represent the condition of the specific bathing water location. However, Ford *et al.*[15] pointed to the fact that these 'non-impacting' discharges or indeed 'non-impacted' discharges could well be 'impacting' water quality for human and ecological health in areas not close to designated bathing water locations. We therefore suggest that if algorithms are based on the tidal cycles, wind and ocean currents this data be made available and fully scrutinized to determine the validity of those statements.

Summary

We have highlighted that the CSOs served by Southern Water discharged over 7000 days of untreated effluent into coastal environments in just over 2 years between December 2020 and March 2023. This is the equivalent of over 19 years of continuous discharges to the South Coast of England of untreated wastewater containing chemical contaminants and pathogens to the English Channel and Southern North Sea. Investigations by the BBC[25] and Unearthed[26] found that these discharges often occur during dry weather and directly or close to Special Areas of Conservation (SACs), Sites of Special Scientific Interest (SSSIs), RAMSAR sites for the protection of migratory seabirds, UNESCO World Heritage Sites and protected shellfish grounds. Therefore, the information presented by the water companies must be accurate and in a form that is easily understood to fully understand the impacts on human health and the environment. The data presented within was denied to the researchers *via* a statutory Environmental Information Request (EIR) because the data was publicly available online. However, in this instance, the data was not available in a way that is or was easily scrutinized. Water Companies in England have been criticised for not making environmental data easily available to researchers and environmental organizations. For example, the BBC during their dry spills investigation was denied data from 6 or 9 major water companies in England because they said "they couldn't provide information because they were already being investigated for potential illegal spilling by industry regulator Ofwat and the EA".[25] However, OFWAT the financial regulator of the water industry has recently warned water companies they are endangering public trust by withholding data.[27] In response, a senior lawyer for South West Water is reported as saying in an article for the Times newspaper that "it is the regulators and not the press or public" who should be able to view information regarding "complex technical and regulatory matters".[27]

We have highlighted that many EDMs on CSOs are faulty and provide many false alarms which at some specific locations can be 100% percent of the time. We have also highlighted an worrying correlation between discharges classified as genuine and not genuine which warrants further investigation as they collectively occur at similar times of the day. Water Companies in England self-report their discharges to the environmental regulator. It is important therefore that the regulators and public get accurate information on discharges. We have highlighted that discharges formerly classified as "non-impacting" before recently being renamed "not impacted" occur with considerable frequency and duration. The water companies may have modelled the risks posed based on death rates of bacteria in seawater and tidal currents. These frequent and lengthy discharges are likely to have an impact on the aquatic environment. It is expected

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that more water companies will make real-time or near real-time monitoring available to the public. We encourage all water companies to make their data and modelling open source. We also suggest that regulators serving the environment (EA), finances (OFWAT) and information commissioner to provide guidance on how EDM data by water companies are presented to the public.

Conflicts of interest

ATF has in the past received funding from water companies. JR and MO work for organizations which promote the impacts of sewage into aquatic environments.

Data availability

Supplementary information: Supplementary material contains data on a) numbers of event duration monitors commissioned b) Percentages of sewage discharges considered not genuine by outfall c) Relationship between not genuine counts and not genuine minutes d) Percentages of non-impacting discharges by outfall e) summary data on event duration performance across the sector and f) worst performing CSOs by minutes discharged. See DOI: **https://doi.org/10.1039/D4EW00693C**.

All data is publicly available and/or available on request.

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Analysis

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Water industry strategies to manufacture doubt and deflect blame for sewage pollution in England

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Check for updates

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The water and sewerage companies (WaSCs) in England are majority-owned by a range of global investors. The industry is under intense scrutiny for widespread failure in its environmental performance, discharging 12.7 million monitored hours of untreated wastewater and sewage into English waterways between 2019 and the end of 2023. At the time of writing, multiple investigations by environmental and financial regulators are in progress, and regulatory oversight is under review by the recently formed Office for Environmental Protection. While limited monitoring hid the full extent of underperformance, we argue that the WaSCs have prolonged this environmental disaster through strategies that mirror those of other large polluting industries in the past. We test this hypothesis for the nine major WaSCs in England against a published framework of 28 'greenwashing/deception' tactics of large industries. We identified 22 of these tactics that could be seen as disinformation, greenwashing and manufacturing doubt. The financial exploitation of water resources in England, alongside long-term degradation of infrastructure and ineffective regulation, raises globally important issues around water security, ethics and environmental stewardship. Much greater scrutiny of both industry performance and industry communication is required.

The disinformation playbook[1,2] has been used by numerous industries to stymy regulation and litigation and maintain profits[3–6]. Researchers have reported commonalities between industries in how they and their sponsored lobbyists distract, delay and disrupt the spread of information[1,2,5–8]. Commercial success does not have to mean environmental degradation, but these strategies often arise owing to conflicts of interest[9], tipping the balance in favour of profit margins over those of human and ecosystem health. When challenged on their environmental performance, polluting industries have been shown to employ tactics such as denial, disruption, distraction, distortion, deflection, casting doubt and ultimately delay through the impacts of this disinformation[10,11].

Several frameworks have been proposed for assessing an industry's capacity to 'spin the science'[5,6]. For example, Legg et al.[5] examined the literature on corporate attempts to influence science and the use of science in policy from eight sectors that pollute and impact human health, including the alcohol, tobacco, chemical, food/drink, fossil fuel, gambling, pharmaceutical and medical technology industries. The authors identified common strategies employed by these industries, including 5 macro-, 19 meso- and 64 micro-strategies[5]. Their results include common strategies such as 'activities to influence the science', including manipulation of scientific methods; reshaping criteria for establishing scientific 'proof'; threats against scientists; and clandestine promotion of policy reforms that increase reliance on the

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industry evidence. Five sectors used the same five macro-strategies, while the remaining three sectors had four of five macro strategies in common. The authors concluded that their analysis demonstrated how these strategies maximize the volume, credibility, reach and use of industry-favourable science while minimizing the same aspects of industry-unfavourable science. In a similar study, Goldberg and Vandenberg[6] identified 28 discrete tactics employed to combat scientific evidence or promote narratives that favour big corporate enterprises, such as coal, oil, sugar, tobacco and pesticide industries, and non-governmental organizations associated with spreading doubt over climate change. Five tactics were commonly used across these sectors, including attacking research study design, enlisting the support of 'reputable' individuals, misrepresenting information, employing hyperbolic or absolutist language and influencing government agencies or laws. A further ten commonly used tactics relied on logical fallacies—a well-established form of rhetorical manipulation.

This article examines the communications and behaviours of the water and sewerage companies (WaSCs) in England over the period 2019–2023 for evidence of any of the 28 tactics identified by Goldberg and Vandenberg[6]. While the framework of 'strategies' by Legg et al.[5] and 'tactics' by Goldberg and Vandenberg[6] have much in common, Goldberg and Vandenberg[6] place greater focus on communication strategies directed at the public (Supplementary Table 1), which more closely aligns with the aims of this analysis. For example, tactics such as 'appeal to emotion', 'taking advantage of scientific illiteracy' or 'taking advantage of the victim's [public] lack of money or influence' were more applicable to the communications observed by the water industry in England. Examples in this study come from multiple sources, including government agency reports, transcribed committee minutes/reports, water company websites, information leaflets and social media communications, and some of our own research experiences. Tactics, as defined by Goldberg and Vandenberg[6], are highlighted in the text by the respective number from Table 1 preceded by a 'T' (that is, T1–28). To our knowledge, this is the first such examination of English WaSCs. This analysis aims to provide evidence to test the hypothesis that the WaSC industry may have actively maintained the status quo through 'manufactured doubt' and 'spin'.

Background to the UK water industry

Most of the water industry in the UK was privatized in 1989, with England split into 9 privatized WaSCs and a further 13 smaller companies providing potable drinking water services only. The water company in Wales was left as a not-for-profit company, leaving only Scotland and Northern Ireland under public ownership. Complications in water quality governance arise from river basins crossing boundaries between Wales and England as well as Scotland and England, but the ownership model in England is dominant (84% of UK customers) since England has a population of approximately 57 million people out of a total of 68 million in the UK. The English water industry is now 70% foreign-owned by investment firms, private equity, pension funds and businesses lodged in tax havens spread across 17 countries[12]. The financial exploitation of national water resources in England, alongside long-term degradation of wastewater treatment infrastructure and ineffective regulation, raises globally important issues around water security, business ethics and environmental stewardship. Privatized WaSCs in England are thought to be carrying £56 billion (~US$68 billion) in debt (in 2023) while reportedly having paid out £76 billion (US$92 billion) in shareholder dividends since privatization[12]. In June 2023, an emergency meeting took place between the financial regulator (the Water Services Regulation Authority, Ofwat), the Department for Environment, Food and Rural Affairs (DEFRA) and the UK Treasury owing to concerns that Thames Water (with 15 million customers, the largest WaSC in England) was on the brink of financial collapse and unable to service its debts[13]. This raises the question of whether global investors can be held to account for a WaSC's environmental performance, which includes discharges of untreated sewage, clean water leaks and predicted water resource shortages exacerbated by climate change[14].

The Environment Agency (EA) in England polices the environmental performance of English WaSCs. Between 2019 and 2023, there were approximately 1.7 million monitored discharges of untreated sewage and wastewater to English rivers and coasts, totalling 12.7 million hours, equivalent to almost 1,500 years (Table 2, Fig. 1 and ref. 15). Many of these discharges breached the permits issued by the EA and are suggested as illegal (that is, non-permitted discharges)[16,17].

Multiple investigations into the water industry and its regulators are ongoing[17]. The environmental regulator, the EA of England, and the financial regulator, Ofwat, are investigating the WaSCs[17]. The UK House of Lords is also investigating Ofwat[18]. The Office for Environmental Protection (OEP), which was formed in 2021, is investigating the performance of the EA, Ofwat and DEFRA. An interim report by the OEP (2023) concluded that "we believe that there may have been failures to comply with environmental law by all three of the public authorities"[19]. In addition, two high court actions were granted in 2023 for judicial reviews into Ofwat and DEFRA's stormwater overflow reduction plans[20]. In August 2024, Ofwat recommended that three water companies in England (Thames Water, Yorkshire Water and Northumbrian Water) be fined a total of £168 million for "failing to manage their wastewater treatment works and networks, as part of the first batch of outcomes from its biggest ever investigation"[21]. In October 2024, the new UK government ordered an independent commission to review the water industry in England and Wales with the vision to "create a climate resilient and secure water sector that continues to have world-leading drinking water quality and delivers on our government's priorities for public health, enjoyment of our waters, the natural environment, economic growth and food security"[22].

WaSCs have two principal responsibilities that involve the provision and maintenance of suitable infrastructure: (1) to supply clean drinking water to customers and (2) to collect domestic/commercial/industrial wastewater and discharge it back to the environment once it has been treated to agreed national/international standards. In its 2022 annual performance report, the EA said: "In 2021, the environmental performance of England's 9 water and sewerage companies was the worst we have seen for years"[15]. Water companies are given a star rating for their environmental performance. The report's summary states that, when measured against a top 4* rating, most had gone in the wrong direction, with four companies on 2* (requiring improvement) and a further two companies given a 1* rating (poor). The report describes the sector's performance on pollution as "shocking, much worse than previous years". In a damning critique of the way the water industry is managed, the 2022 EA report states:

> "Company directors let this occur and it is simply unacceptable. Over the years the public have seen water company executives and investors rewarded handsomely while the environment pays the price. The water companies are behaving like this for a simple reason: because they can."

The EA reported 1,677 pollution incidents and 62 serious pollution incidents during 2021[15]. The terminology for describing the amount of precipitation (exceptional, extreme and heavy) varies between water company websites. The evidence that has recently emerged shows that light/average precipitation is sufficient to activate many storm overflows[16,17]. Activation may also be initiated by groundwater infiltration into leaky sewerage pipes. The EA does not accept such infiltration as an excuse for storm overflow use, considering it to be in breach of the discharge permit[23]. The European Court of Justice ruled in 2012 that the UK Government's attempt to allow discharges during 'heavy rainfall' was not acceptable and insisted that such discharges

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Table 1 | The 28 tactics identified by Goldberg and Vandenberg (2021) that industries use to manufacture doubt and examples we identified from England's water industry for 22 of these

Number	Tactic	Examples used by the water industry
T1	Attack study design	• Water companies have attempted to discredit work by scientists that has highlighted breaches of sewage discharge permits
T2	Gain support from reputable individuals	N/a
T3	Misrepresent data	• Spill data—accepted by the EA and water companies as not accurate and in breach of permits (Environmental Audit Committee (EAC) minutes) • Both Thames Water and Southern Water fined for misrepresenting data • Impact of spill data (Reasons for Not Achieving Good Status; RNAGS) are misrepresented as the frequency and quantify of discharges are unknown • Declare that sewage discharges are not raw sewage or play down the severity of spill concentrations
T4	Suppress incriminating information	• Prevented and hid data from EA inspectors (judges summing up) • Refuse/delay Freedom of informtion (FOI) requests—widely reported in the media and more recently during BBC investigation into dry spills
T5	Contribute misleading literature;	• Evidence of using outdated literature • Misleading public communications over the dilution of stormwater overflows and the impact of CSOs on overall water quality • Deny discharges are raw sewage
T6	Host conferences or seminars	• Host conference whereby the main focus is influencing customer behaviours as a means of deflecting blame
T7	Avoid/abuse peer review	N/a
T8	Employ hyperbolic or absolutist language	• Play down environmental impacts of discharges and play up the impact on people's homes if they were not to discharge sewage, for example, flood schools and hospitals with sewage • Alarmist figures to fix the problem (£660 billion), which focuses on increased bills for customers
T9	Blame other causes	• Blame the farming industry, road run-off and people putting incorrect things such as fats and sanitary products down the toilet. These are all valid, but when is this education versus deflection? • Considerable investment in the education of other causes and playing down their own impacts
T10	Invoke liberties/censorship/overregulation	N/a
T11	Define how to measure outcome/exposure	• They have been policing themselves, so they have been defining how to measure outcomes. Self-reporting of discharges has been shown to be inaccurate • Defining discharges as impacting and non-impacting • Been allowed to define what exceptional circumstances are, with large variation in terms used by different water companies
T12	Take advantage of scientific illiteracy	• 95% of rainwater claims and just grey water rely on the perception that the public will not know it might be inaccurate or how bad the other 5% might be. • Inaccurate claims relating to the age of the England sewage and wastewater infrastructure
T13	Pose as a defender of health or truth	• UK Water argues that the water industry has been responsible for cleaning up the rivers since the 1950s as opposed to being responsible for their poor water quality • 'The truth about English rivers', written by the body representing the UK water industry, uses inaccurate and outdated data to support claims that critics are wrong • Uses language such as 'sound science' adopted by other polluting industries to seed doubt
T14	Obscure involvement	N/a
T15	Develop a public relations strategy	• Evidence of similar and coordinated messaging by the industry representative lobby UK Water.
T16	Appeal to mass media	• Develop a coordinated strategy that makes business appear greener and more healthy
T17	Take advantage of victim's lack of money/influence	• 'This will cost the consumer more if we have to improve the infrastructure'
T18	Normalize negative outcomes	• Highlight that the discharges are normal and heavily regulated when in fact they should only be used in 'exceptional circumstances' • Develop approaches whereby water users find it normal to check for sewage discharges before taking part in water-based recreation
T19	Impede government regulation	• EA officers were impeded from getting data from water companies
T20	Alter product to seem healthier	• The Storm Water Task Force Group has been changed to the Clean Rivers and Seas Task Force by Southern Water • Wastewater treatment plants have been rebranded as water recycling sites • The industry has recently put themselves forward as the group that helped clean up the rivers • We are protecting homes and businesses
T21	Influence government/laws	• The scaremongering about the bill to stop stormwater overflows focussed on costs that may have influenced the more nuanced language used by the government to 'progressively reduce' • The government uses terminology and figures generated by the water industry
T22	Attack opponents	• Discredit work of scientists that highlights that sewage discharges may have been in breach of permits
T23	Appeal to emotion	• Emotive language used in communications with the general public, for example, without stormwater releases, schools and hospitals would be flooded
T24	Inappropriately question causality	• Just 4% impact on water quality, when the real impact is unknown without more accurate data on sewage discharge volumes
T25	Make straw-man arguments	• Not 'raw sewage' • Focus on agriculture doing harm
T26	Abuse credentials	N/a
T27	Abuse data access requests	• BBC: "The remaining six water companies in England said they couldn't provide information because they were already being investigated for potential illegal spilling by industry regulator Ofwat and the EA"
T28	Claim slippery slope	N/a

N/a, tactics for which we did not find evidence; EA, Environment Agency; CSO, combined sewer overflow.

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Table 2 | Numbers and duration (hours) of known sewage and wastewater discharges into rivers and coastal waters in England between 2019 and 2023 (ref. 15)

Year	Number of known sewage and wastewater discharges	Hours of discharges (millions)
2019	292,864	1.5
2020	403,171	3.1
2021	372,533	2.7
2022	301,091	1.8
2023	464,056	3.6

are permitted only in 'exceptional situations'. A damning 2022 report by the House of Commons Environmental Audit Committee (EAC) in its 4th Water Quality in Rivers Report[24] highlighted

> "…that rivers in England are in a mess. A 'chemical cocktail' of sewage, agricultural waste, and plastic is polluting the waters of many of the country's rivers. Water companies appear to be dumping untreated or partially treated sewage in rivers on a regular basis, often breaching the terms of permits that on paper only allow them to do this in exceptional circumstances."

The looseness of the term 'exceptional' has provided a loophole in the legislation that has been widely exploited (T11). For example, Hammond et al.[16] demonstrated that water companies were substantially underreporting the number of untreated sewage discharges, many of which occurred under 'normal' rainfall or even during dry weather. At two wastewater treatment plants (WWTPs), almost 1,000 previously unreported spills were identified, with many violating permits to discharge water via storm overflows[16]. Further reports by the same authors, analysing monitored discharge data obtained by freedom of information request from WaSCs, have identified more than 5,000 days of sewage spills in breach of permits[16]. The accumulation of hotspots of microplastic contamination on riverbeds also points to the routine discharge of inadequately treated wastewater into rivers under conditions of low river flow[25]. An earlier study showed that even moderate rainfall and flooding were sufficient to wash microplastics downstream and prevent them from accumulating on riverbeds[25,26]. As such, the recovery of large quantities of riverbed microplastics downstream of a WWTP or combined sewer overflow (CSO) is clear evidence of untreated wastewater discharges during light/normal rainfall (or during an otherwise dry period). These two studies were published just 2 months apart in 2021; both provided compelling evidence of untreated wastewater discharges into low river flows or dry weather spills in English river catchments. The water industry immediately refuted each of the study's designs (T1) and assumptions (T22)[27–29], yet when challenged to provide evidence to the contrary, to our knowledge, none has been forthcoming. The data from some of these reports (for example, refs. 30 and 31) was subsequently used as evidence by Ofwat in fines totalling £158 million to Thames Water, Yorkshire Water and Northumbrian Water[32].

Academics and campaign groups wanting to understand the frequency, severity and risks posed by sewage discharges in England have routinely had environmental information requests (EIRs) denied (T27 and T4) or incomplete datasets provided after prolonged delay (T4)[33,34]. It is a legal requirement that all EIRs be satisfied within 20 working days unless extensions are requested. A 2023 analysis of spill data from three water companies by BBC researchers in collaboration with several UK universities has confirmed that dry weather spills are widespread and routine[35]. In response to the BBC investigation, representatives from both Southern Water and Wessex Water reported that "dry spills are a complex issue" and "often arise due to infiltration of groundwater"[35]. Six water companies said they were unable to provide information to the BBC investigation because they were already being investigated for potential illegal spilling by industry regulators Ofwat and the EA[36]. Subsequently, the UK Information Commissioner's Office reported in July 2024 that a recent tribunal case[36] found that a legal exception, which says releasing information will prejudice an investigation, did not apply to the information requested through EIRs[34].



Fig. 1 | CSO duration for 2023. Circles show individual overflows, with the size being proportional to duration (0–8,700 h) and the colour differentiating the WaSCs. Figure adapted from Top of the Poops under a Creative Commons license CC BY-SA 4.0.

The use of fines by regulators has not improved standards. Summing up in the *EA* v. *Southern Water Services Limited* case[37] in July 2021, which was the largest investigation in the EA's 25 year history, the judge stated

> "Southern Water Services Limited has pleaded guilty to 51 counts of discharging untreated sewage into controlled coastal waters… It has been estimated that the total volume of untreated sewage across all of the sites was in the region of 16–21 billion litres."

> The court heard that, when "…the Environment Agency sought to investigate these offences it met a level of obstruction that it says was unprecedented in its experience of a company of this size. On multiple occasions, employees refused to permit Environment Agency officers to take away documentation that it wished to seize under its statutory powers, refused to allow them to walk around sites unaccompanied, citing 'health and safety', and refused to answer questions, despite the Agency's powers to require answers."

These offences (T4) led to the largest ever fine of a UK water company of £90 million[37]. In July 2023, Thames Water was fined £3.33 million for discharges of raw sewage into rivers[38]. In summing up, the judge[39] said she believed Thames Water had shown a "deliberate attempt" to mislead the EA by omitting water quality data readings and submitting a report to the regulator denying responsibility (T3, T4 and T19).

How the water industry in England spins science

The water industry and its representatives proactively and reactively communicate with the public. The following are typical examples of such communication, which have been evaluated for evidence of the manufacture of doubt and spin[6]. Biocide industries have been documented rebranding their insecticides, pesticides, fungicides and herbicides as 'plant protection products', which softens their image from an industry that aspires to kill target organisms to one that offers protection. Similarly, WaSCs have introduced alternative terms as part of a public relations strategy to make their operations appear greener. For example, the term 'water recycling centre' began to replace 'sewage treatment works'[39]; Southern Water's Storm Water Task Force was rebranded as the Clean Rivers and Seas Task Force. These could be viewed as a coordinated strategy to make the business appear greener (T15, T16 and T20).

When challenged about stormwater releases, the water industry has repeatedly refuted that they discharge 'raw sewage'. This term is graphic and emotive and can be visualized by the public very easily (T23). Examples of how the industry has softened this terminology can be found in Box 1. The industry routinely utilizes alternative phrases that metaphorically dilute the offensive connotation to the point that they are no longer accurate (see rebuttal in Box 1) (T3, T5, T8 and T12). The water industry has also tried to play down the severity of storm overflow discharges by describing their impact as 'insignificant', 'minimal' or 'temporary'. These are disingenuous because, of the 370,000+ CSO spills in 2020, only 11% were investigated by the EA for impact[40–43], ensuring that no one can truly know that the impact is minimal. Moreover, sewage is just one component of untreated wastewater, which contains millions of putative human pathogens and antimicrobial resistance genes per litre[44] as well as high levels of nutrients and microplastics, and hundreds of micropollutants, making it likely that its discharge to a river would be impactful from the perspective of several metrics, especially if such discharges are chronic, as many are[45,46]. Wessex Water state on their website that stormwater overflows have 'minimal or no ecological impact', yet they also claim the following:

> "We don't currently know the real time impact storm overflows have on water quality. It is possible that a storm overflow that discharges for a couple of hours could discharge less harmful sewage than a storm overflow that discharges for just 30 minutes. It all depends on the contents of the combined sewer at the time of heavy rainfall."[47]

Therefore, we conclude that the messaging from the water companies is mixed and errs on the side of minimizing the impact of the sewage spills (T5 and T12).

The water industry and regulatory authorities refer to the use of screening to remove solid objects (not faeces), as 'preliminary treatment', suggesting that the components of sewage that people are most concerned with (that is, the faeces and pathogens) have started to be removed, which they are not (T11, T12, T20 and T23)[48]. Referring to sewage discharges as 'preliminarily treated' gives the public a false impression of the risk, playing down the potential harm to human health and the environment that can be caused from its discharge to the environment. For example, Thames Water highlights that "we've needed to significantly extend Mogden [WWTP] to reduce the number of times partially-treated sewage overflows into the River Thames when the works become overloaded after heavy rain"[49]. Screened sewage is identical to the pre-screened sewage in every significant biological and chemical metric (T3, T5, T8, T11, T12, T13 and T20), thereby posing nearly identical environmental and human health threats as unscreened sewage[50].

Another variation on these rebuttals from the water industry is to claim that it is not raw sewage but 'grey water' as it has been diluted by other household discharges (Box 1), which often gives the domestic waste a grey appearance. Thames Water recently described grey water as "relatively clean waste water from baths, sinks, washing machines and other kitchen appliances"[51]. Discharges still contain raw sewage, regardless of the colour, not to mention that the 'sewage fungus', which gives rivers a grey appearance, is actually a complex of bacteria, not fungi (T5). Furthermore, they still contain harmful faecal pathogens at concentrations that exceed legal limits[22,52], as well as discharges from dishwashers, showers and washing machines that contain chemicals known to be toxic to aquatic wildlife[53,54].

Industry communications will often alter the meaning of key terms by softening terms, such as 'heavily diluted wastewater', 'mainly rainwater' and 'pretty much rainwater' (T5 and T8; Box 1). A common industry phrase communicated across social media is that wastewater discharges "were 95% rainwater" (Southern Water) or, in some cases, "99% rainwater" (South West Water) (Supplementary Fig. 1). Given that all pipes have a different degree to which they are already filled with sewage (that is, dry weather flow), it is not possible to say what proportion of the CSO discharge is composed of sewage and rainwater—it will depend on many factors, including time of day, as the pipes have the greatest free capacity in the early morning and the least free capacity at approximately 10:00[55].

Following questions by the EAC in October 2021, the chief executive officer (CEO) of Severn Trent Water repeatedly claimed that "the company did not discharge raw sewage" and "because storm overflows discharged a mix of sewage 'heavily diluted' with rainwater, the contents of any discharge were 'pretty much already rainwater'"[56]. The committee suggested that, while this claim may have been intended to reassure them about the discharges from overflows on the Severn Trent network, they did not find it convincing[56]. At the same committee, the then CEO of Southern Water, in response to questioning, stated that "In most of our spills, the constituents are 97.1%—it sounds terribly precise, but roundabout 97% is rainwater, surface water, so not sewage"[56]. The claim that spills are so diluted that it is not sewage we believe to be factually incorrect. The attempt at precision for a highly variable measurement further reinforces the impression that the industry wishes to be seen as an authority with a rich understanding of its business and its possible impacts (T5). Following recent challenges by a citizen science group, SOS Whitstable, the industry has since changed these claims from "95% rainwater" to "up to 95% rainwater" (T3) (for example, refs. 57 and 58). Recent studies of stormwater discharges have found that, while they may only contribute 8% in terms of volume compared with treated effluent, they contribute more than 90% in terms of total faecal coliforms[52]. Similar studies have highlighted that CSO discharges contain very high numbers of faecal indicator species that fail approved water quality criteria[40–42]. There is considerable evidence that treatment cannot remove all pollutants, such as endocrine-disrupting chemicals, which can impact the health of ecosystems, from wastewater effluents[53,54]. These statements made by the water industry, as well as not always being factually correct (that

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BOX 1

Examples of water industry communications playing down the impact of raw sewage, taken from company web pages, 20 March 2023

Water industry quotes	Rebuttal to quotes
Quote 1: "Contrary to popular belief, stormwater releases are not 'raw sewage'. They're made up of wastewater – primarily from washing machines, showers and dishwashers – and are heavily diluted by rain." (Southern Water)	Quote 1: Stormwater releases contain raw sewage as they are derived from the sewerage system.
Quote 2: "Because of the job they do, and despite calling them 'sewer overflows', most of the water they release is rainwater, not raw sewage." (Anglian Water)	Quote 2: The sewerage system will always have sewage in it. The ability of sewers to accommodate rainfall in addition to the sewage is highly variable. Some sewers are already running near capacity and, therefore, have little to no capacity to accommodate any rainfall.[66]
Quote 3: "Although it does contain some untreated sewage, storm discharge is heavily diluted because it's mostly rainwater" and "In most cases, storm discharges are heavily diluted by rainwater. So, although they're unpleasant, their impact to river water is likely to be minimal." (United Utilities)	Quote 3: There is no room for rainwater where the sewer system is running near capacity. CSOs discharging from such a system will not experience very much dilution at all. Moreover, there is ample evidence in the academic literature to show that the level of microbial contamination in stormwater will increase in a 'first flush'[85].
Quote 4: "When we do need to use them, they can sometimes affect river and bathing water quality, albeit temporarily." (Thames Water)	Quote 4: There is no evidence that rivers are only temporarily impacted by CSOs. Many CSOs discharge nearly every day of the year, as the data show[86].
Quote 5: "Storm overflows have minimal or no ecological impact because what is released is diluted wastewater" and "Generally, storm overflows have an extremely small impact on rivers." (Wessex Water)	Moreover, there is no systematic data being collected to assess whether the downstream environment is impacted, only temporarily, from CSOs. Quote 5: Same response as to quotes 2, 3 and 4.
Quote 6: "Storm overflows have not been routinely built into sewers since the 1960s, but over 14,000 of them remain in England. They are much less environmentally harmful than many other sources of pollution like sewage from treatment works or agricultural runoff..." (Water UK)	Quote 6: It should always be the case that CSO discharges will be more impactful to the environment than treated sewage discharges of the same volume. If this was not the case there would be no point in treating sewage. The fact that some agricultural runoff can be more impactful than CSO discharges is irrelevant.

is, misinformation; T5), could be considered as underplaying the risks to human health and the environment that arise from such discharges (that is, greenwashing and spin; T12).

'Nature-based solutions' (NBS) is the latest term used among those in restoration ecology and climate change and flood mitigation. Several industries have been accused of greenwashing through their 'net zero' and 'carbon neutral/positive' initiatives to make the public think businesses are environmentally friendly[59–61]. NBS have recently entered the UK water industry lexicon to foster environmental positivity in strategic approaches to the problem of stormwater discharges. Large-scale infrastructure changes are costly for the water industry and are communicated as impacting consumer bills (T17). In contrast, NBS approaches are seen as environmentally friendly as they do not consume as much carbon as engineering solutions such as the storm tanks used to relieve pressure on a combined sewerage system receiving rainfall. They also provide a means for the water industry to support nature conservation efforts, which helps rebuild trust and positive public relations (T13). For example, the restoration of oyster beds is considered a green alternative to dealing with high nutrient loads owing to their capacity to filter large volumes of water and form large biogenic reefs that increase biodiversity. NBS in pollution control are not new—reed beds, for example, have been used successfully for many decades to trap contaminants in the mining and water industries[62]. All conventional sewage treatment has a biological component, which is a NBS to break down organic material. The general public should be more clearly informed that nature can only provide solutions if the human pressures are sufficiently alleviated; in the case of the water industry, the need is for repairing and replacing infrastructure (T5) and, ultimately, reducing the load on sewerage networks (T12).

Following the 2021 court case of *EA* v. *Southern Water Services Limited*, the judge highlighted that oyster populations along the south coast of England had declined in areas receiving illegal sewage discharges[37]. While the declines in oyster populations could not be fully attributed to sewage, the judge stated:

> "... there is a degree of scientific consensus that a reason for a substantial part of the reduction in the oyster population is water pollution or water quality. The demise of the oyster fishery in the Solent has had a huge impact on the local economy. Conversely, modelling shows that if water quality improves there is likely to be a significant increase in the value of bivalve shellfish harvesting in the Solent"[37]

Restoration projects under the guise of NBS (for example, refs. 63 and 64) without the concomitant declines in sewage discharges mean habitats may continue to be at risk, and this whole approach could

BOX 2

Examples of water industry communications highlighting that what they do is permitted and normal, taken from company web pages, 20 March 2023

"Storm discharges are legally allowed, under the conditions of the Environment Agency permit." (Thames Water)

"This is permitted by the Environment Agency and closely monitored by them and us." (Yorkshire Water)

"Storm overflows have minimal or no ecological impact because what is released is diluted wastewater... Although overflows are permitted by the Environment Agency, we agree that they have no place in the 21st century, but it will take time and significant investment to progressively eliminate them." (Wessex Water)

"Each of our CSOs has been assessed for the environmental risk potential, as defined by the Environment Agency, and they have each been permitted to act as a 'storm overflow'. But despite being consented by our regulator, it is understandable that no one finds it acceptable that even extremely diluted sewage reaches our rivers." (Anglian Water)

"Our storm overflows operate within the permits set by the Environment Agency and have been designed to provide excellent bathing water quality." (Northumbria Water)

be considered further greenwashing and a cynical public relations exercise (T12).

Online 'near real-time' data showing CSO discharges, now available from seven WaSCs, inform the public of sewage discharges to enable them to better control their exposure to the pollution hazard by choosing where to participate in swimming and other recreational water activities. This approach normalizes the negative outcomes of sewage discharges, making it routine for the bather to view a website before planning a day at the river or beach (T18). Southern Water has event duration monitors fitted to 100% of its wastewater and CSO outfalls, and launched in 2018 a public-facing website (Beachbuoy, https://www.southernwater.co.uk/water-for-life/beachbuoy) to inform water users of discharges in real time. Hailed as leading in the sector for the UK, the company was criticised in September 2021 for the way data are displayed to the public[65]. The real-time maps offer a traffic light system: red indicates a discharge at a specific location in the last 24 h, and yellow in the previous 72 h, with green indicating no recent spills. During heavy and/or prolonged rainfall, England's south and southeast coasts are often red with spill alerts across 700 miles (1,126 km) of coastline. The traffic light system was recently replaced with white markers instead of red, until the spill was verified as genuine. Responding to criticism, the water company told the BBC that "Beachbuoy previously did not account for tidal conditions and duration, and cautiously took the worst-case scenario, leading to flagging bathing waters, which causes unnecessary worrying for the public and the tourism industry alike. Now if the outfall is a long way out to sea, the release was short and the tidal conditions meant there could be no impact on a bathing water, the tool will no longer trigger a warning to users." Where spills are not identified as genuine within the 24 h window, markers will remain white on the map. The absence of a yellow marker means that bathers will only find out about sewage discharges retrospectively, defeating the purpose of a spill warning system[65]. Because of the uncertain quality of the discharge data, the industry has introduced a delay in its own reporting, which reduces the protective intention of the alerts and decreases the public's understanding of the true nature of spills in a region (T3).

Other modifications include dynamic outfall mapping, whereby spills are categorized as 'false alarms', plus 'genuine impacting' and 'genuine non-impacting', depending on the state of the tide. Non-impacting spills have been classified as unlikely to cause designated bathing water sites to fail faecal bacterial measurements. In July 2023, on the basis of this criticism, Southern Water changed the terminology from 'non-impacting' to 'non-impacted' to better represent the condition of the bathing water location. Critics have pointed out that these 'non-impacting' discharges or indeed 'non-impacted' discharges could well be 'impacting' water quality for human and ecological health in areas close to designated bathing water locations (T3, T11 and T12). Therefore, collectively, these are seen as a tactic to play down the impacts of sewage discharges on coastal environments.

WaSCs observed during this study highlight that the use of CSOs is normal (T18; normalizing negative outcomes) and heavily regulated by the EA (Box 2) and that, without their use, homes and businesses would be flooded with sewage (Box 3). Some companies highlight that schools and hospitals could be flooded without the discharge from CSOs. Presenting the public with a binary choice of spilling sewage into schools and hospitals or the river appeals to a simplistic argument that will allow the public to tolerate CSOs (T23; appeal to emotion). Recent studies have revealed the chronic under-capacity of the English wastewater systems as a fundamental cause behind the increased frequency and duration of CSO spills[66]. In December 2022, the CEO of the financial regulator Ofwat wrote

> "We expect companies to deliver the service improvements they were funded to deliver. No ifs, no buts" and "The lack of investment from companies we're seeing at the moment is extremely disappointing, especially in light of the poor performance for customers and the environment. Failure to invest or delays to investments means that vital improvements are not being made or are late. I am expecting these companies to get a grip on their investment programme and make up for the shortfalls to deliver the associated improvements in service."[67]

Both water companies and politicians have often blamed the current situation on historical Victorian drainage (Queen Victoria's reign ended in 1901; T3 and T12). However, while it is true that a good deal of drainage infrastructure was built during Victorian times in London (mid-1870s), much of the network in London and other cities and towns was built much later during the urban expansion of the twentieth century[68]. Furthermore, several reports have emphasized that infrastructure investment does not match population growth, climate change and increased use of impermeable surfaces[66,67].

One tactic used by those looking to manufacture doubt is to deflect causality and obscure involvement (T9 and T14) by suggesting 'it's not us, it's them' or developing a straw-man strategy (T25). There are many pressures on aquatic systems, including agriculture, industrial and domestic discharges and urban runoff. The EA has introduced metrics for rivers to determine the dominant sources behind the 'reasons for not achieving good status' (RNAGs). RNAGs attempt to apportion contributions to poor water quality to a source[69]. WaSCs frequently highlight that their contribution to rivers not achieving good status

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BOX 3

Examples of water industry communications that use emotive language, taken from company web pages, 20 March 2023

"If we didn't have storm overflows then the diluted storm water wouldn't have anywhere to go, which would result in homes, businesses and streets being flooded." (South West Water)

"When this occurs, excess water is released into rivers and the sea, to prevent homes, businesses, schools, and roads flooding.' "Storm releases are made for one reason – to protect homes and businesses from flooding. For instance, in heavy rain storm releases from our Budds Farm wastewater treatment works (and the related storm overflows) protect more than 2000 properties including schools and hospitals." (Southern Water)

"At times of heavy or continued rainfall, the sewer system can't always cope with the extra volume. Storm overflows are used to prevent sewers flooding our homes, gardens and streets. They act as a safety valve, diverting some of the rainwater and foul water into watercourses. We call this mixture 'storm discharge'." (Thames Water)

"As rainwater can be unpredictable, we have permitted storm overflows on our sewer network to act as a relief valve. They help to reduce the pressure on sewers during heavy rainfall events and stop the system from backing up and flooding homes and gardens by allowing heavily diluted wastewater to be discharged into watercourses." (Yorkshire Water)

"Sewers operate this way to help prevent the flooding of streets, homes and businesses." (United Utilities)

"Storm Overflows (SOs) help protect homes, buildings, and land from flooding. They're 'backup sewers', and collect overflowing stormwater when our normal sewers are overloaded due to heavy rainfall. Spills to environment contain mainly rainwater along with dilute sewage and are screened to remove debris." (Northumbria Water)

BOX 4

Examples of water industry communications deflecting or playing down issues of poor water quality, taken from company web pages, 20 March 2023

"Did you know agricultural runoff is the biggest polluter of rivers and waterways as it often carries along pesticides, fertilisers and animal slurry?" and "Agricultural impacts alone account for 46% of the reasons for poor river health in our region." (South West Water)

"The Environment Agency estimates that storm overflows lead to around 5% of river and sea pollution in the North West, with water quality in the natural environment affected by rain running off highways and farm land and private drainage being incorrectly connected." (United Utilities)

"In most cases, storm discharges are heavily diluted by rainwater. So, although they're unpleasant, their impact to river water is likely to be minimal. The Environment Agency reports that, overall, they do less damage to the environment than other sources of pollution." (Thames Water)

"In truth, storm overflows account for just 4% of all the reasons for rivers and waterways not achieving the best water quality. As is clear from Environment Agency river quality data, there are many contributing factors outside of water company control that cumulatively have a more significant impact on the health of our waterways." (Anglian Water)

"Combined Sewage Overflows are vital in protecting people from flooding and are widely used across the UK. CSOs are only responsible for 3% of reasons for rivers in England not achieving good ecological status. The water industry accounts for 24% of the remaining quality issues in England's rivers and other sectors, including agriculture, housing and transport, accounting for 76% of the reasons for rivers failing to achieve good ecological status." (Severn Trent)

"Government should work with the water industry, NGOs and others to agree a plan in 2022 to progressively eliminate the four per cent of harm caused by storm overflows to English rivers, starting with the most sensitive catchments." (UK Water)

is less than that of agriculture and that CSOs only make up a small proportion of this contribution to poor status (T9 and T25; see Box 4 for examples). RNAGs were developed before the water industry was required to fit CSO monitors and thus before the full extent of sewage discharges was known. Monitoring has revealed that such discharges are common during periods of no or moderate rainfall in all regions. The lack of accurate data on the extent to which CSOs are discharging, especially the volumes being discharged, undermines confidence in the RNAG system. The water industry continues to take advantage of RNAGs' flaws to play down the impacts of untreated wastewater discharges (T3 and T5; Box 4). Albini et al.[70], in a modelling study of four different rivers, recently concluded that sewage discharge significantly impacts water quality and benthic riverine communities, regardless of the surrounding land uses. Deflecting blame without comprehensive monitoring has delayed regulation and policy change (T19).

The industry has been keen to portray CSO discharges as the result of the public flushing wet wipes down toilets (T9). For example, UK Water in November 2021 stated that "Wet wipe-caused blockages in the sewer are a leading cause of sewage spills from storm overflows"[71]. Southern Water sponsored and organized a 1 day conference entitled the Zero Pollutions Conference in July 2019, in which half the day was dedicated to 'customer engagement influence behaviour' with speakers from South West Water, Anglian Water and Southern Water[72].

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Fig. 2 | An outfall from a CSO on the River Tame in Greater Manchester, UK. Figure reproduced with permission from ref. 25, Springer Nature Limited.

A variety of companies have subsequently run targeted public education initiatives such as 'The Unflushables' (Southern Water), 'Unblocktober' (Thames Water), 'Waging War on Flushable Wipes' (Severn Trent Water), 'Stop the Block' (South West Water) and 'Wipe out Wipes' (Wessex Water). Although wet wipes do combine with fats to create fatbergs, they are a relatively small component of the wider CSO problem (T3, T5, T12 and T25). Others have argued that the main reasons for stormwater overflows are the lack of capacity from increased population and poorly maintained infrastructure such as cracking pipes[66]. The industry arguably encourages such a narrative to mask its complacency and lack of investment (T24). Wet wipes are found in huge numbers in urban river channels and are a particular problem on the bed of the Thames in London. They provide a very graphic illustration of the amount of untreated sewage discharged into England's rivers (Fig. 2).

Event duration monitors have confirmed that such discharges often occur during periods of no or moderate rainfall. A recent chair of England's EA suggested that the threat of closer monitoring caused some WaSCs to acknowledge significant breaches of regulatory permits[73].

UK Water, the body representing the water industry, published an article in 2021 called "The Truth about English Rivers", which was a rebuttal[74] of the criticisms directed at the water industry's performance by campaigners and the media (T22). The article mentioned inaccurate reporting, a distorted and misleading picture and that "commentators are working from the same, shared set of facts about reality on the ground". The article claimed to be the 'truth' and contained many issues addressed in this manuscript relating to misrepresented data. Two years later, in May 2023, Water UK publicly apologised on behalf of the industry, accepting it had not moved fast enough on sewage discharges or infrastructure investment and committed to increased funding plus greater transparency and accountability[75].

One of the inaccuracies is the EA's oft-cited statement that "Water quality in our rivers is now better than at any time since the start of the Industrial Revolution". The water industry and several politicians have widely promoted this statement (for example, Severn Trent, 23 November 2021). This statement was recently reviewed by a team of academics[76], who concluded that the conclusions were, at best, mixed, with pressures from nutrients and pesticides having increased in many areas since the 1940s and diffuse-source pollution and novel pollutants, including microplastics and forever chemicals, increasing in recent decades to pose substantial water quality threats. A recent study of nitrogen isotopes in macroalgae in the Mersey Estuary has shown that sewage pollution is now higher than at any point since privatization[77].

BOX 5

Examples of water industry communications highlighting the cost implications for the public should they try to fix stormwater overflows (CSOs), taken from company web pages, 20 March 2023

"Many storm overflows are decades, if not hundreds of years old, and this legacy infrastructure is an antiquated system for the modern world. Removing them completely would be complicated, disruptive and expensive, with an estimated price tag of £600 billion to effectively re-plumb major towns and cities across the UK." (Anglian Water)

"The cost, both financially and environmentally, to remove all storm overflows would be substantial. It would cost billions and cause major disruption digging up urban and rural areas." (South West Water)

"At Anglian Water, we've been working through them for years, and fixing them where they cause problems. This isn't new and it isn't something that is driven by the recent media or political coverage. It's because we don't think they are fit for purpose and we want any issues with them resolved. We must do this in a prioritised way however, as the engineering solutions are not straightforward and the cost to customers is significant." (Anglian Water)

"The cost of replacing assets like CSOs with new systems and flood mitigation (as properties still have to be protected from flooding) would run into many billions. Customer bills would rise dramatically, not to mention the disruption caused by completely replumbing the major cities in our region." (Anglian Water)

"Building new storage tanks would have significant carbon consequences, cost billions of pounds and would be extremely disruptive – so the best option is to separate surface water from combined sewers. ...It is estimated it would cost around £10 billion to eliminate overflows in our region. To help fund this, customers would see around a 50% increase in bills over 10 years." (Wessex Water)

Northumbrian Water, in their draft drainage management plans, propose four potential options to deal with storm water overflows for customers and stakeholders to consult upon. These four options come with either a predicted 13%, 17%, 34% or 38% rise in bills by 2045, which do not include inflation.

No. 17 in the Goldberg and Vandenberg[6] tactic list is "take advantage of the consumers' lack of money/influence". This tactic is employed in the industry's argument that solving the UK's sewage crisis would cost consumers more than they can afford, even before the problem has been holistically assessed for all interrelated solutions (T17; Box 5). The upper end of the costs predicted to resolve the issue of storm overflows was stated as £660 billion in a report published in 2021 and co-written

by the water industry, the Water Services Regulation Authority, DEFRA and the EA[78]. This value was based on an estimation of digging up every road in the UK with a combined foul and surface water sewer. The water industry and many members of parliament in the serving government also mentioned the excessive cost to the billpayer in response to opposition pressures to reform the industry (T21). Attempts to maintain the status quo through scaring stakeholders over the 'cost of action' have stymied the discussion, investment and legislative reform (Box 5).

Conclusions

In our analysis of England's water industry, we identified 22 (Table 1) of the 28 tactics reported by Goldberg and Vandenberg[6] to deflect blame and diminish responsibility. These 22 tactics broadly map onto the 5 macro-strategies and the 19 meso-strategies proposed under the alternative framework of Legg et al.[5] to compare industry approaches to manufacture doubt and manipulate the use of science in policy and practice. Many of these strategies[5] and/or tactics[6] have been used to control the narrative and reduce, weaken and delay legislation that would place tighter controls on the very water industry activities that keep profitability high. Collectively, they provide evidence of industry approaches to manufacturing doubt that have detrimental effects on both environmental and public health. No river in England has a good overall health status, and only 15% have good ecological health status[24]. The UK government and EU parliament, under Directives 2005/29/EC and 2011/83/EU, have recently put forwards plans to prevent greenwashing by the commercial sector. Buse and Bayliss[12] commented that water companies "…urge consumers to use less water—yet given their own record, this amounts to little more than using the standard corporate playbook of shifting responsibility to users." Giakoumis and Voulvoulis[79] stress that, while the general public can play their part to flush "pee, poo and paper" only, the most important issue to address is the investment in the sewerage infrastructure and that "Consumers are called upon to act in the public interest, while the private equity owners operate in the interests of shareholders." Water companies routinely cite volumes of drinking water consumed, yet they are extremely reluctant to give volume data on the untreated wastewater they discharge to the environment, despite clear demonstrations of the ease of estimating volumes from existing datasets[16]. During cross-examination by the parliamentary EAC in October 2021, several water company bosses conceded that they were unaware of any optimized technologies that could measure volumes and were seeking options globally[78,80]. On the other hand, Thames Water has published discharge volumes for Mogden WWTW (serving 2 million customers) for a decade. Scottish Water estimated that, during 2022, their assets discharged an estimated 47 million cubic meters of wastewater into river and coastal waters[81], suggesting that estimations of volumes were feasible in many UK water company contexts.

Climate change will make water an increasingly valuable commodity globally[82]. The communications and misinformation strategies highlighted in our analysis are important industry tools to deflect criticism and perpetuate the status quo. We propose that industry communications should also be regulated so that any signs of manufactured doubt, misinformation and spin can be challenged. A closing example is provided from an advertising campaign by Anglian Water, which extolled the virtues of the company's wildlife-friendly wetlands. In June 2023, the advertisement was banned and sanctioned by the Advertising Standards Authority[83], who concluded that Anglian Water "…carried out activities that caused harm to the environment, which contradicted the overall impression of the advertisement." We are hopeful that such criticisms in the future will come from the industry regulators themselves and that future pressure will not be reliant on the complaints of 'nine citizens' to correct the messaging of the water industry.

Methods

Several approaches have been employed to examine the tactics/strategies employed by industries to influence science and policy[5,6]. A review of individual water company communications in England from their webpages (Supplementary Table 2) and those of their trade association (UK Water) and ad hoc social media messages was conducted and mapped against the 28 tactics used by the industry to manufacture doubt by Goldberg and Vandenberg[6]. We also conducted a mapping exercise to compare commonalities in the frameworks for strategies proposed by Legg et al.[5] and the tactics proposed by Goldberg and Vandenberg (ref. 6 and Supplementary Table 1). This process highlighted that, at least for the higher (macro- and meso-strategy) strategies proposed by Legg et al.[5], there were considerable overlaps between the two frameworks. This enabled us to discuss, at least in broad terms, the substantive evidence for water industry strategies/tactics for manufacturing doubt. There are many definitions of greenwashing[84], and the one adopted here from Webster's *New Millennium Dictionary of English* (https://www.merriam-webster.com/) is the "practice of promoting environmentally friendly programs to deflect attention from an organisation's environmentally unfriendly or less savoury activities". In addition, the analysis is supported by evidence from sources such as formal responses of leaders within the EA, Ofwat, the OEP, DEFRA, judiciary summary reports from the Department of Justice, transcripts of minutes of cross-party parliamentary committees in England, the EAC, the Environment, Food and Rural Affairs Committee, the water industry, CEOs of water companies and Water UK (the mouthpiece for the water industry).

Each tactic we have mapped is highlighted in the text by the number of the tactic in Table 1 preceded by a 'T' (that is, T1–28). Where it is not mentioned, to the best of our knowledge the water companies have not issued a response to criticisms and accusations.

Data availability

The authors confirm that all data generated or analysed during this study are included in this published article.

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Author contributions

A.T.F. conceived the manuscript. A.T.F., A.C.S., P.H. and J.W. contributed to the researching, writing and editing of the manuscript.

Competing interests

A.T.F. is an academic who has received funding from research councils, the European Union, government agencies, philanthropic organizations and industry, which has included water companies. A.C.S. is an academic who has received funding from research councils, government agencies and industry, which has included the UKWIR. P.H. is a retired mathematician and computer scientist, previously funded as an academic at University College London and the University of Oxford for research unrelated to the water industry. He currently supports a citizen science campaign group called Windrush Against Sewage Pollution. J.W. is a physical geographer who has received funding from the Natural Environment Research Council. He has not received funding from the water industry.

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