PFAS sources, pathways, and environmental data (2026)

Chief Scientist's Group research report

Environment Agency

Date: November 2026

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Published by:

Environment Agency

Horizon House, Deanery Road, Bristol BS1 5AH

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Keywords: pfas, sources, pathways, receptors, monitoring, water, leachate, regulation, sectors, risk

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Citation: Robertson, R., Mackeown, G., Carmichael, H., Holmes, L. Dunbar, M. (2026) PFAS sources, pathways and environmental data. Environment Agency, Bristol.

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This report has been produced through research commissioned by the Environment Agency's Chief Scientist's Group, in partnership with the Environment Agency PFAS Working Group.

This report has undergone the Environment Agency's standard scientific, technical and governance quality assurance processes. Given the significant public, policy and stakeholder interest in PFAS, the report was also subject to independent external peer review by three recognised experts in PFAS science, environmental monitoring and statistical analysis. Reviewer comments were considered during preparation of the final report and incorporated where appropriate.

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If you have any comments or questions about this report or the Environment Agency's other scientific work, please contact research@environment-agency.gov.uk.

Dr Robert Bradburne
Chief Scientist

Acknowledgements

We thank the technical specialists of the Environment Agency PFAS Working Group - James Lymer, Dr Alun James, Martin Flack, Dr Nicola Ingrey, Andy Oliver, Helen Pickering, Josh Wright, Lauren Morton, Alex Taylor, Dr Claire Massey, Angela Haslam, Elly Hodkinson, Kerry Jones, Michael Hughes, Kirsty Darby, and Petroula Mantzou - for their valuable contributions and comments. We would especially like to acknowledge the members of the PFAS Project Board - Dr Emma Pemberton, Ruth Jones, Steve Dungey, and Matt Womersley for their thorough strategic and technical review of the report prior to publication. We thank the large number of Environment Agency field, laboratory and technical staff who collected the samples, undertook laboratory analysis and ensured the data were available for us to use.

We also thank the three independent external reviewers who provided scientific and technical review of the report prior to publication. Their comments and challenge helped strengthen the final report.

Executive Summary

This report presents an updated national assessment of the occurrence of PFAS in English waters, sources and pathways to the environment. It builds on the Environment Agency's 2021 report, reflecting significant advances in monitoring capability, analytical methods, and operational and regulatory activities over the past five years. The analysis presented in this report strengthens the national evidence base on PFAS. It also informs proportionate regulation and future policy and operational decision-making across government, regulators and industry.

Since 2021, the Environment Agency has significantly expanded its PFAS monitoring and analytical capability. Building on fully quantitative monitoring of PFOS and PFOA since 2016, we can now measure more than 60 PFAS in freshwater and 45 in fish and mussel tissues. We have developed a method for the ultra-short-chain PFAS trifluoroacetic acid (TFA), which is not captured through our target analytical suite. We have also broadened the range of samples investigated, extending our analysis to more complex media such as landfill leachate.

Together, these developments represent one of the most extensive PFAS environmental monitoring programmes globally, providing a substantially stronger evidence base for environmental assessment and regulatory decision-making. Additionally, we have applied modern statistical approaches in our analysis of the data to improve our understanding of long-term trends, spatial variability and seasonal effects, increasing confidence in our findings.

The expanded evidence base delivers several important findings. Most notably, our analyses show clear and reliable declines in concentrations of key legacy PFAS in English rivers between 2016 and 2025. Our long-term monitoring datasets demonstrate statistically robust declines in PFOA and PFOS concentrations, providing evidence that regulatory restrictions, changes in use and targeted interventions can deliver measurable reductions in environmental concentrations. Between 2016 and 2025, PFOS and PFOA concentrations in English rivers declined by approximately 30% and 29%, respectively. PFOS concentrations in freshwater fish declined by approximately 49% between 2016 and 2024, with more modest declines observed in coastal fish populations.

Trends in concentrations over time are less clear in other waterbody types. Lakes generally have higher PFAS concentrations than rivers, while shorter monitoring records and more complex environmental processes increase uncertainty in trends in occurrence over time in estuarine and coastal waters. PFOA concentrations show evidence of decline in lakes, groundwater, estuarine and coastal waters, although with less confidence than in rivers. In contrast, PFOS concentrations in lakes, estuarine and coastal waters have generally remained stable over the monitoring period.

Our monitoring demonstrates that PFAS are widespread across all environmental media investigated. Concentrations of PFAS are greater in rivers and lakes compared to groundwater. In freshwater, groundwater, and landfill leachate samples, short-chain carbonyl PFAS dominate the PFAS measured due to their mobility, persistence and

formation from precursor compounds. A pilot monitoring study found TFA at higher concentrations than any other PFAS analysed.

In contrast, long-chain PFAS, particularly PFOS, continue to dominate the PFAS body burden in fish and shellfish sampled. Our findings indicate a changing PFAS profile in England over time. While concentrations of some legacy PFAS are declining, the widespread presence of short-chain PFAS, precursor compounds and other replacement PFAS is evident.

The report also highlights the value of complementary analytical approaches. Preliminary results using the Total Oxidisable Precursor (TOP) assay to English water samples indicate that the Environment Agency's targeted PFAS suite captures approximately 46% of oxidisable PFAS. These findings are helping to improve our understanding of the occurrence of PFAS in the environment that we are unable to fully characterise through our routine fully quantitative targeted analytical methods. This report also describes the Environment Agency’s role in managing PFAS in England and the regulatory frameworks used to assess and manage risks.

Our understanding of PFAS sources, the nature of emissions and pathways through the environment has been strengthened through engagement with regulated industry, targeted investigations, and evidence gathering across key sectors and waste management activities. Supporting this work, the Environment Agency’s PFAS Risk Screening Project developed a national GIS-based Risk Explorer that integrates information on potential PFAS sources, environmental sensitivity and monitoring data to identify and prioritise potential PFAS source areas.

The Risk Explorer has been used to screen and prioritise more than 40,000 potential PFAS source sites across England. This has helped to target further investigation, inform regulatory action and support proportionate risk-based management of PFAS. In parallel, we are developing consistent guidance for PFAS remediation in soil and water, including approaches for evaluating remediation options and assessing the effectiveness of different technologies for managing contamination.

Important uncertainties remain. For example, the distribution and occurrence of PFAS in soils, sediments, air and marine environments is less well understood and characterised. Working with others, the Environment Agency is delivering an extensive programme of research, monitoring and regulatory action to strengthen the evidence base, support delivery of the PFAS Plan and manage risks from PFAS.

Overall, this report reflects the significant progress made in our understanding of PFAS sources, occurrence and environmental behaviour in England over the past decade. Our evidence shows that regulatory restrictions and changes in use of some legacy PFAS are associated with declining environmental concentrations, with clear reductions observed in English rivers and biota. Together, this strengthened evidence base supports proportionate, precautionary and effective management of PFAS. It provides a strong foundation for future monitoring, decision-making and regulation to reduce emissions, manage environmental risks and protect the environment over the long term.

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1. Introduction

This report examines the sources, pathways, occurrence and environmental behaviour of PFAS in England. It draws primarily on Environment Agency monitoring data, together with evidence from research, industry engagement and regulatory investigations. The report supports environmental management, risk assessment and regulatory decision-making. Building on the Environment Agency's 2021 PFAS report, it reflects substantial advances in monitoring, analysis and scientific understanding over the past five years and provides a more comprehensive assessment of PFAS occurrence and exposure across England.

While environmental thresholds and regulatory standards help indicate the potential significance of PFAS contamination, their application remains limited because such values are available for only a small number of PFAS and are often developed for different environmental receptors and assessment purposes. Therefore, this report focuses on the distribution, occurrence and trends of PFAS within the environment rather than formal environmental risk or regulatory compliance assessment. Where environmental thresholds are referenced, they are provided for contextual interpretation only and should not be interpreted as compliance assessments against statutory standards.

1.1 Background

PFAS are a broad and diverse class of synthetic organofluorine chemicals. Since the 1940s, they have been used across an array of industrial applications and consumer products for their surfactant, lubricating and repellent properties. Common use sectors include firefighting, aviation, food contact materials, transport, electronics, textiles, cosmetics and personal care products, construction, medical devices, pesticides and pharmaceuticals (Gaines, 2023; Glüge et al., 2020).

Given their widespread use, environmental releases may occur throughout their lifecycle, including manufacture, storage, use, and disposal (Cordner et al., 2024; Evich et al., 2022). Key sources include PFAS manufacturing and industrial facilities, commercial and domestic product use, and sites where aqueous film-forming foams (AFFFs) have been used. Waste management sites, such as landfills and wastewater treatment works (WwTWs), can also receive PFAS and act as pathways where they are released back into the environment (Dasu et al., 2022). However, for many PFAS, the potential for environmental release throughout their lifecycle remains poorly understood.

There is no single globally adopted definition of PFAS. In Europe, a widely accepted scientific and regulatory definition is that proposed by the Organisation for Economic Co-operation and Development (OECD, 2021). It defines PFAS with a few noted exceptions as any chemical containing at least one perfluorinated methyl group ($–CF_3$) or perfluorinated methylene group ($–CF_2–$) (without any hydrogen, chlorine, bromine, or iodine atom attached to it). This structural definition provides a practical basis for

identifying and regulating PFAS, categorising them into non-polymeric and polymeric groups. Appendix 1 outlines differing definitions and substance groupings.

Under this definition, PFAS include thousands of substances that vary widely in structure, properties and potential risks (ECHA, 2023). More than 200 use categories have been identified, reflecting their ubiquity and functional versatility in modern life (Glüge et al., 2020). However, only a small subset, likely in the low hundreds, are thought to be used commercially in significant volumes across the UK (Buck et al., 2021; HSE, 2023).

The carbon–fluorine (C–F) bond is the strongest single bond in organic chemistry (O'Hagan, 2008), making PFAS highly resistant to chemical, thermal and physical degradation. This stability, combined with low surface tension and water- and oil-repellent properties, has made PFAS integral in products requiring durability and high performance.

Despite their utility, many PFAS share characteristics that together pose a hazard to the environment and human health (HSE, 2023):

- **Extreme persistence:** PFAS and their breakdown products resist degradation and can remain in the environment for hundreds of years.
- **Mobility:** many PFAS are mobile in water, and their potential for long-range transport is high, leading to widespread dispersal and contamination.
- **(Eco)toxicity:** exposure to certain PFAS has been associated with adverse health effects in humans and wildlife, raising concerns about potential long-term impacts.
- **Bioaccumulation and biomagnification:** some PFAS can bioaccumulate in biota, with concentrations increasing up the food web.

These properties, combined with decades of widespread use and past management practices, have resulted in many PFAS becoming globally distributed pollutants. They have been detected across a range of environmental and biological media, including biota, soils, water, air, food and human serum (Ackerman Grunfeld et al., 2024; Cousins et al., 2022), including in remote and otherwise pristine regions such as the Arctic and Antarctica (Joerss et al., 2020; Xie and Kallenborn, 2023).

Production and use of PFAS have continued to increase since their introduction (Evich et al., 2022), contributing to rising environmental concentrations and ongoing exposure for current and future generations (Cousins et al., 2020; De Silva et al., 2021). This has resulted in a widespread legacy of contamination that is technically challenging and costly to remediate (Ling, 2024; Ling et al., 2026). In England, national-scale screening suggests that approximately 2,900 to 10,200 sites may present a higher potential for PFAS occurrence, depending on the screening threshold applied (Jacobs, 2023); however, these results are intended to support prioritisation and do not constitute site-specific assessments of contamination or risk. The widespread and costly legacy of PFAS contamination has increased interest in applying the polluter pays principle to help recover remediation costs and encourage future emission reductions.

A range of remediation technologies are currently being developed. Many remain technically complex, energy intensive and costly, and achieve only partial removal or

containment rather than complete destruction of PFAS (Environment Agency, 2025a). This highlights the importance of preventing releases at source before contamination occurs. Even when sources are removed, PFAS concentrations often decline only slowly because many PFAS persist in the environment and act as long-term reservoirs of contamination. In addition, some types of volatile PFAS, including certain fluorinated gases (F-gases), can also contribute to climate change (Sovacool et al., 2021).

To date, PFAS regulation has largely focused on a relatively small proportion of the PFAS group, particularly legacy long-chain PFAS. Declining trends observed in select surface waters and biota in England and globally indicate the effectiveness of these measures (Environment Agency, 2025b, 2024; Muir and Miaz, 2021). However, trends can vary between environmental media and locations, reflecting the complex and spatially variable nature of PFAS. In addition, reductions in emissions may take many years to translate into measurable reductions in environmental concentrations, making it difficult to assess the effectiveness of interventions.

Furthermore, PFAS regulation has largely followed a substance-by-substance approach. In some cases, this has led to the replacement of regulated PFAS with structurally similar shorter-chain alternatives that may pose comparable risks, raising concerns around “regrettable substitution” (Manojkumar et al., 2023). Many of these replacement PFAS can themselves transform into highly persistent terminal compounds under environmental conditions, and their increasing environmental detection reflects both growing use and persistence (Megson et al., 2025; Miaz et al., 2020). Therefore, for many PFAS, there remain limited data on hazards, exposure, uses and volumes, making it difficult to manage long-term risks (Ng et al., 2021).

In response, there is growing international interest in broader, class-based restrictions and mixture-based risk assessment approaches (Cousins et al., 2020; ITRC, 2023). Reflecting this shift, the EU is considering restrictions on PFAS as a group by limiting their manufacture, use, and placing on the market, while allowing continued use only where considered essential and where suitable alternatives are not available. This approach is currently being evaluated by the European Chemicals Agency’s (ECHA) scientific committees (ECHA, 2024).

A similar approach is being considered in the UK. The Health and Safety Executive (HSE) and the Environment Agency are progressing measures under UK REACH to restrict the sale and use of PFAS in firefighting foams (FFF) (HSE, 2025).

Complete elimination remains challenging, as PFAS contribute to the performance and functionality of many products and suitable alternatives are not yet available for all uses. However, the development of PFAS-free alternatives is ongoing, and some manufacturers and suppliers have begun to phase out the use of specific PFAS in products and production processes (PFAS Central, 2026).

1.2 Perfluoroalkyl acids and their precursors

To date, regulatory and scientific efforts have primarily focused on non-polymeric PFAS, particularly perfluoroalkyl acids (PFAAs) such as perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS). This reflects their historic widespread use, persistence, mobility, bioaccumulative behaviour and demonstrated toxicity in humans and wildlife. PFAAs are characterised by a fully fluorinated hydrophobic carbon chain attached to an acidic functional group (Figure 1).

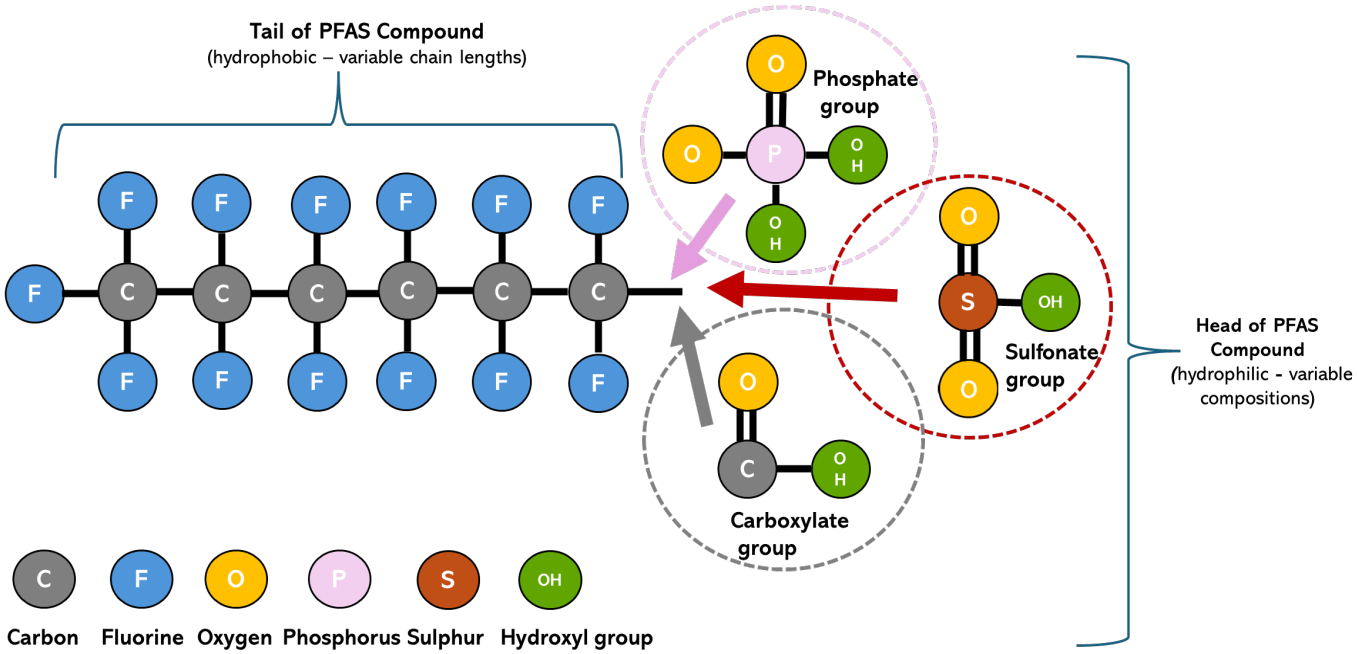


Figure 1: Simplified representation of a non-polymeric perfluoroalkyl substance. Variations in chain length, degree of fluorination, branching, charge and type of functional group create a wide diversity of PFAS with differing environmental fate and behaviour.

PFAAs are often described as terminal or “arrowhead” substances due to their extreme stability and resistance to further degradation, allowing them to persist in the environment for many decades or longer. Many other PFAS, known as “precursors”, can eventually degrade into these terminal PFAAs over time in the environment or biota. This transformation may occur slowly and under specific environmental or biological conditions, complicating monitoring and management (Pickard et al., 2024, 2022). This has raised concerns that continued releases of precursor PFAS could delay environmental recovery and continue to contribute to future PFAA burdens, even where direct emissions of legacy PFAAs have been reduced. The extent of their contribution and which precursor substances are the most relevant remain areas of ongoing research.

As a result, PFAAs are a key focus for regulation, as they represent stable terminal products of the PFAS group and are central to risk assessment and management. This underpins the “arrowhead” concept, where precursor substances are grouped with the stable terminal PFAAs they can ultimately form. For example, the United Nations Stockholm Convention on Persistent Organic Pollutants (POPs) applies risk management

restrictions to PFOS as 'PFOS, its salts and PFOS-related substances' (Appendix 2). Figure 2 illustrates the main PFAS subgroups as defined by OECD's 2021 definition, including the classification of PFAAs and their precursor substances.

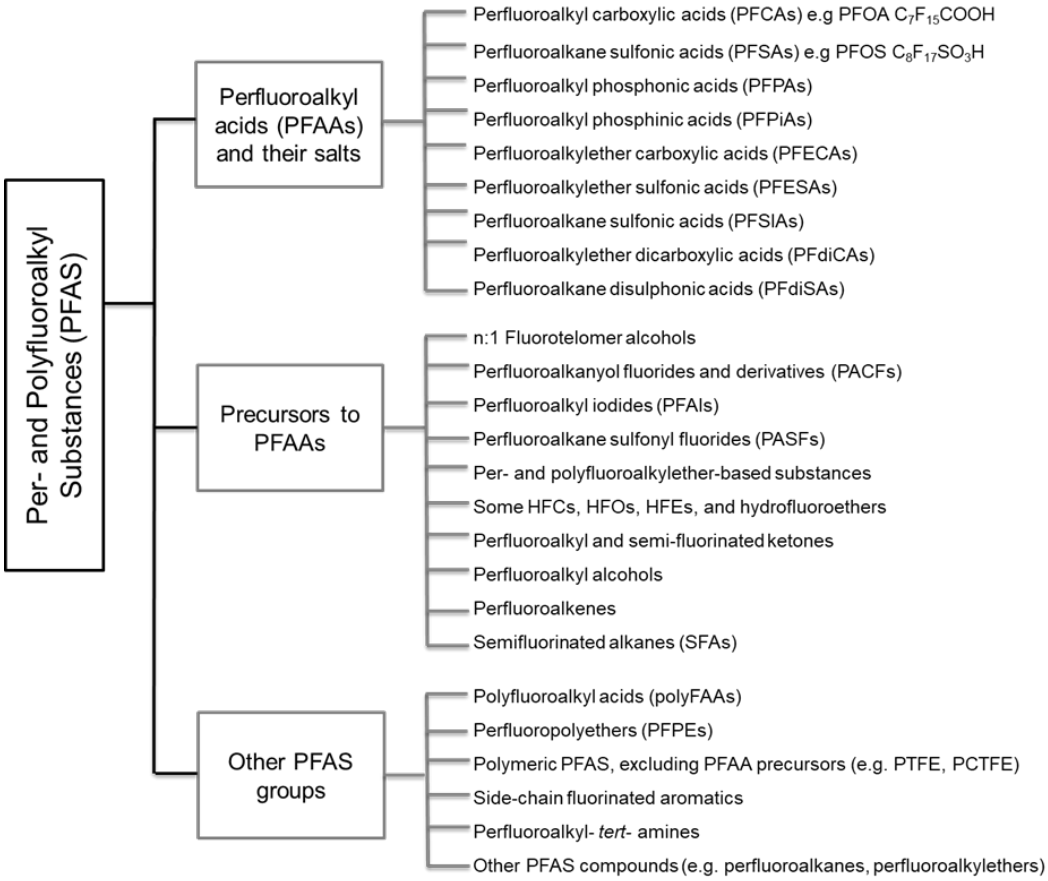


Figure 2: Main PFAS subgroups as identified by the OECD with a focus on PFAAs or 'arrowheads' and their precursors. The figure presented is adapted from Figure 9 in the OECD (2021) report, which provides further detail on PFAS grouping and terminology.

PFAAs are generally classified as either short-chain or long-chain based on the length of their perfluoroalkyl chain. Long-chain perfluoroalkyl carboxylic acids (PFCAs) contain 7 or more carbon atoms, while long-chain perfluoroalkyl sulfonic acids (PFSAs) have 6 or more (Buck et al., 2011; OECD, 2021). Structural features such as chain length, functional group, and molecular charge strongly influence how PFAS behave in the environment, including their fate, bioaccumulation and (eco)toxicity (Brendel et al., 2018; Brunn et al., 2023; Leung et al., 2023; Sun et al., 2022).

Long-chain PFAAs are generally less mobile but more bioaccumulative and toxic than their short-chain counterparts. However, this does not indicate a lack of hazard for some short-chain PFAS (Brendel et al., 2018; Sunderland et al., 2019). Long-chain PFAAs also bind more readily to sediments and clay-rich or organic soils (Ehsan et al., 2024; Lyu et al., 2022). This makes them less likely to leach into groundwater and more likely to bioaccumulate in the tissues of organisms, leading to biomagnification through the food

web. Among compounds of the same chain length, PFSAs tend to bioaccumulate more than PFCAs, largely due to the sulfonate group (Brunn et al., 2023).

In contrast, short-chain PFAAs are typically more water-soluble and mobile, increasing their ability to migrate through soils and water and increasing the risk of groundwater and surface water contamination. They may also be more readily taken up and translocated within plants, including crops (Costello and Lee, 2020; Fawad et al., 2026; Felizeter et al., 2014; Li et al., 2022). Together, these characteristics reduce the effectiveness of many conventional remediation techniques for short-chain PFAS (Neuwald et al., 2022).

Molecular charge also affects PFAS mobility and adsorption behaviour. Most PFAAs exist predominantly as anions under typical environmental conditions, reducing their interaction with negatively charged soil surfaces and increasing their potential to move through water (ITRC, 2023). Figure 3 summarises how PFAS chain length influences adsorption, mobility, bioaccumulation, and toxicity.

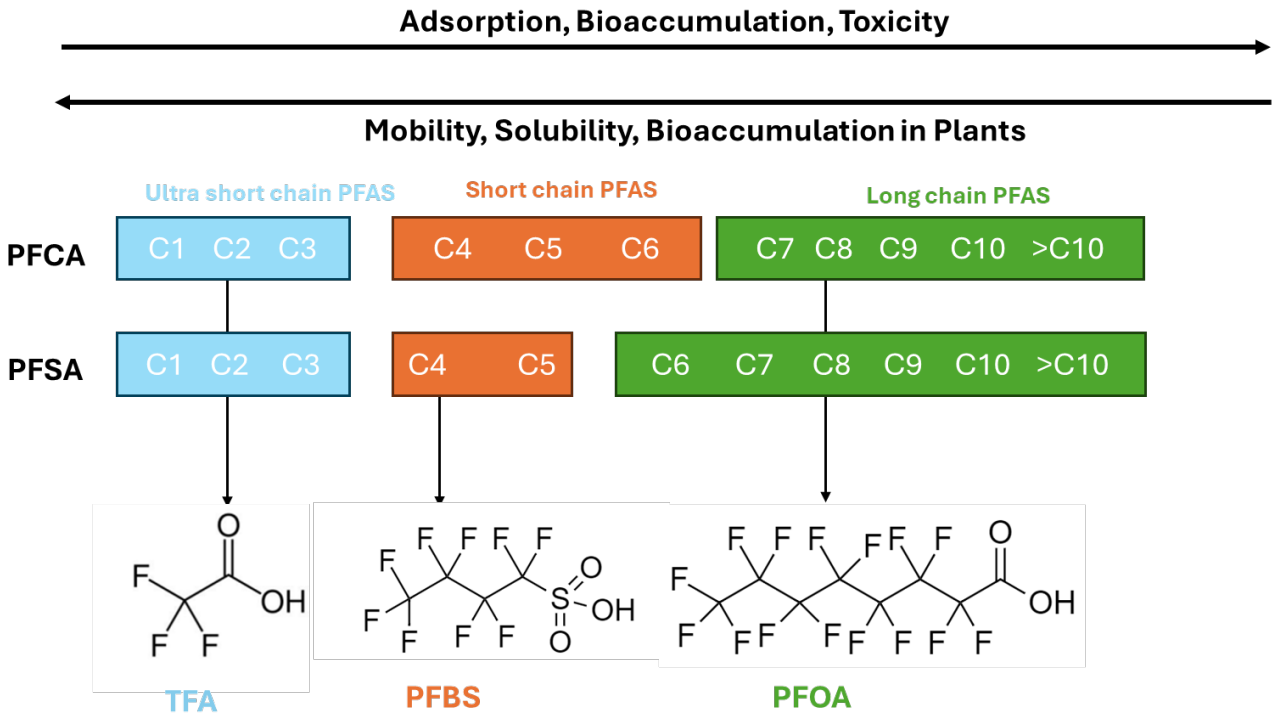


Figure 3: Classification of PFAS properties by chain length, illustrated using TFA (trifluoroacetic acid), PFBS (perfluorobutane sulfonic acid) and PFOA (perfluorooctanoic acid).

Increasing attention is being given to ultra-short-chain PFAS, such as trifluoroacetic acid (TFA), due to their environmental persistence, mobility and potential health implications (Arp et al., 2024; Cahill, 2024; Garavagno et al., 2024). Globally, TFA has been detected across a wide range of environmental media, including surface waters, drinking water, groundwater, wastewater effluent, oceans, crops, vegetation, soils, sediments, precipitation and ice cores from remote regions (Zhi et al., 2024). Reviews of global monitoring data indicate that its occurrence is widespread and increasing, and that in some aquatic systems TFA can represent a substantial proportion of total PFAS (Arp et

al., 2024; Freeling and Björnsdotter, 2023). Its widespread presence is largely linked to the degradation of precursor substances used across multiple sectors, including fluorinated gases, pesticides, and pharmaceuticals (Diehle et al., 2025; Joerss et al., 2024; Joudan et al., 2021). Due to its small size, high mobility and negative charge, TFA is one of the hardest PFAS to remove from the environment (Scheurer et al., 2017).

Overall, the intrinsic physical and chemical properties of PFAS, combined with the large number of substances and their occurrence at very low concentrations, make them difficult to detect, quantify, and monitor in environmental media. Monitoring capabilities are currently most advanced for water, while analysis in soil, sediment, biota, and air remains less developed globally. In addition, most routine analytical chemistry methods focus on a relatively small subset of well-studied non-polymeric PFAS, despite regulatory standards increasingly considering a broader range of PFAS compounds.

Interpreting PFAS concentration data can also be challenging, as concentrations may vary spatially, seasonally, and over time, and often include results below detection limits. Screening approaches, including non-targeted analysis (NTA) and fluorine mass balance methods, can be used to identify a wider range of PFAS and assess contamination beyond the scope of routine targeted monitoring programmes (Idowu et al., 2025; Megson et al., 2025). However, targeted compound-specific analysis remains important for routine monitoring, regulatory compliance, and remediation decision-making.

1.3 Human and wildlife health effects

While recognising the importance of human health impacts, this section focuses primarily on wildlife and environmental effects, consistent with the Environment Agency's statutory remit. Human health effects are briefly summarised for context, but detailed assessment is led by other government health bodies (HSE, 2023, 2025a; UKHSA, 2026). A database of academic studies on the health effects of different PFAS is available through the PFASTOX database.

PFAS exhibit a wide range of biological effects in humans and wildlife, reflecting their structural diversity, persistence and ability to accumulate in living organisms. Many PFAS bind preferentially to proteins rather than lipids, leading to accumulation in protein-rich tissues and affecting multiple organ systems, with some compounds persisting in the body for years (Fenton et al., 2020; Roth et al., 2023; Sunderland et al., 2019). Human biomonitoring shows the widespread presence of PFAS in human populations (Biggeri et al., 2024).

Human exposure to PFAS occurs through multiple pathways, with diet considered the main non-occupational exposure route. Higher internal concentrations are often reported in communities affected by contaminated sources and in occupationally exposed populations (Christensen and Calkins, 2023). Evidence increasingly links exposure to certain PFAS, particularly legacy long-chain PFAAs, with effects on immune suppression, certain cancers, fertility, development and metabolism (Habib et al., 2024; Obsekov et al., 2023; Schillemans et al., 2023; Souza et al., 2026; Xing et al., 2024; Yeoh et al., 2025).

Furthermore, recent research suggests that certain PFAS may act synergistically with microplastics (Soltanighias et al., 2024).

Fish are particularly relevant because PFAS are readily transported in water and can enter food webs at multiple points. Fish are commonly used as sentinel species because they are exposed through both water and diet, helping to show how PFAS accumulate and transfer through aquatic food webs (Lee et al., 2020). As such, the ecotoxicological evidence base is more developed for aquatic species.

Similarly, evidence for adverse effects in wildlife is strongest for PFOS and PFOA. Reported effects in fish include impacts on growth, reproduction, development, metabolism, hormones, immune function and behaviour (Hong et al., 2025; Lee et al., 2020; Zhu et al., 2026). PFAS have also been detected across a wide range of organisms, including crustaceans, amphibians, terrestrial invertebrates, birds, mammals, algae and plants, with studies reporting similar effects (Ankley et al., 2020).

Despite a rapidly expanding (eco)toxicological evidence base, uncertainty remains regarding the toxicity of many PFAS across different species, particularly given the limited availability of long-term toxicological and epidemiological studies for most compounds. Uncertainty also remains around their mechanisms of action, mixture effects, and the consequences of long-term, low-level exposure in ecosystems at a community level. These uncertainties are compounded by differences between controlled laboratory studies and real-world environments, as well as limitations in field studies, including small sample sizes and sometimes inconclusive results.

Effects also vary with species-specific differences in uptake, distribution, metabolism and elimination, as well as age, exposure concentration and time (Bangma et al., 2022; Roth et al., 2023). For example, findings from the H4 Chemicals Indicator for England show declining PFOS concentrations in freshwater fish and harbour porpoise, no observed change in otters, and uncertain trends in buzzards, red foxes and estuarine/coastal fish (Environment Agency, 2024).

Interpretation is further complicated by the complex environmental behaviour of PFAS. This depends on substance properties, precursor transformation and environmental conditions such as sediment composition, temperature and salinity. Therefore, for many PFAS and species, the evidence base continues to evolve, making broader environmental trends sometimes difficult to interpret.

Nonetheless, available scientific evidence is clear that some PFAS can cause harm to aquatic and terrestrial animals and plants at environmentally relevant concentrations due to their (eco)toxicological properties, persistence, mobility and ability to bioaccumulate in organisms. Together, these characteristics justify measures to minimise emissions and reinforce the need for grouping approaches, ecological risk assessment, and proportionate precautionary regulation (Anderson et al., 2022; Cousins et al., 2020; Roth et al., 2023). These approaches, including actions being taken by the Environment Agency and others, are discussed in the following chapters.

2. Policy and regulatory landscape

2.1 Our statutory and regulatory role

The Environment Agency regulates chemicals in the environment for England. We work to protect the environment and human health by assessing chemical risks to the environment, monitoring emissions, implementing controls on industry and enforcing regulations. We regulate a wide range of business activities, including chemical manufacturing, waste management and water discharges, and enforce regulatory compliance.

Since the early 2000s, the Environment Agency has played a central role in building the evidence base on PFAS, improving understanding of their sources, behaviour, and impacts in the environment. This approach is evidence-based and risk-led, reflecting both the complexity of PFAS as a class and the evolving nature of scientific understanding. Environmental monitoring is a core part of our work, helping to identify where the risks are greatest and to prioritise regulatory action and further investigation.

Findings are shared with the Government and other regulators to support their regulatory and advisory roles and to ensure that policies and controls on PFAS are informed by robust and up-to-date evidence. The Environment Agency also works closely with academia, industry, and international partners, recognising that effective management of PFAS requires collaboration and shared understanding across sectors and borders.

The Environment Agency's work on PFAS includes:

- Ensuring compliance with legislative requirements where the Environment Agency is the competent authority
- Providing technical advice to the Government to support the development of environmental policy
- Supporting and undertaking scientific research to improve understanding of PFAS
- Operating an advanced PFAS surveillance programme, using environmental monitoring data to assess the state of the environment, identify trends, assess risks and support regulatory decision-making
- Identifying current and legacy uses of PFAS and working with stakeholders and industry to map potential sources and manage associated risks
- Investigating PFAS in river catchments and working with water companies to understand sources entering sewers and explore wastewater treatment options
- Collaborating with national and international regulators and partners to share data, align regulatory approaches and guide risk management strategies

The Environment Agency supports the application of the polluter pays principle through its regulatory, permitting, enforcement and contaminated land functions, seeking, where appropriate, to ensure that those responsible for pollution bear the costs of its management, investigation and remediation. However, regulating such a large and chemically diverse group of substances is inherently complex.

PFAS regulation cuts across several regulatory frameworks. Some are specific to chemicals, while others provide broader controls on pollution and environmental protection.

Key frameworks currently used to regulate and manage PFAS in England include:

- UK Registration, Evaluation, Authorisation and Restriction of Chemicals (UK REACH) (assimilated Regulation (EC) 1907/2006)
- The POPs Regulations (assimilated Regulation (EU) 2019/1021 as amended)
- The Environmental Permitting (England and Wales) Regulations 2016
- The Water Framework (Water Framework Directive) (England and Wales) Regulations 2017
- The Water Supply (Water Quality) Regulations 2016
- Environmental Protection Act 1990, Part 2A (Contaminated Land Regime)
- The Fluorinated Greenhouse Gases Regulations 2015

PFAS may also be indirectly regulated under other legal frameworks, including those governing:

- Classification, Labelling and Packaging
- Plant protection products and biocides
- Veterinary medicines and human pharmaceuticals
- Waste management
- Health and safety
- General product safety
- The Control of Major Accident Hazards COMAH
- Control of Substances Hazardous to Health

Each framework has distinct legislative requirements, spanning multiple statutes and regulatory agencies. As a result, responsibilities for chemicals, water, waste, products, land contamination, food and public health are distributed across several regulators, each with different duties, powers and thresholds for action. Further detail on these frameworks, the Environment Agency's role within them, and the responsibilities of other regulators is provided in Appendix 2.

2.2 Regulatory direction and the PFAS Plan

The UK Government's PFAS Plan, led by Defra, recognises that the persistence and widespread presence of PFAS pose risks to the environment and public health, requiring action to prevent irreversible harm. It makes clear that a lack of full scientific certainty should not delay proportionate and preventative action. This establishes a clear policy basis for earlier and more proactive regulatory intervention where justified by the available evidence to manage PFAS environmental risks. It sets out a coordinated approach to improve understanding of PFAS sources, tackle environmental pathways and act to reduce exposure. Alongside actions to reduce risks to the environment and public health across the full life cycle of PFAS, it supports innovation and economic growth.

The PFAS Plan includes 49 actions to strengthen regulation and risk management. These include expanding environmental monitoring, improving understanding of PFAS sources and pathways, using existing regulatory controls and developing additional controls where needed, and developing environmental standards and statutory limits as the evidence base, analytical capability and assessment methods continue to evolve.

It provides a platform for continued engagement with industry, regulators and other stakeholders to reduce PFAS emissions, manage risks and support the transition to suitable alternatives where these are available. The PFAS Plan also supports continued engagement with international partners and reflects the need for coordinated action on the transboundary challenge presented by PFAS.

The Environment Agency continues to work with Defra, the devolved administrations and other partners to manage PFAS risks in a coordinated, risk-based and proportionate way, supported by wider policy frameworks such as the Environmental Improvement Plan and the New vision for water white paper.

Overall, the PFAS Plan provides a science-based and evidence-led strategic framework for managing PFAS risks in the UK. It supports coordinated action across government, regulators, industry and other stakeholders to improve understanding of PFAS, strengthen regulatory controls, reduce emissions and protect the environment and public health. The Environment Agency will play an important role in delivering the actions in the PFAS Plan through its monitoring, research, regulatory and advisory functions, building on the evidence and expertise developed in recent years.

3. Uses, sources and pathways of PFAS

This chapter explains how uses of PFAS translate into environmental releases, how PFAS move through the environment and how the Environment Agency is developing an evidence base to prioritise proportionate, risk-based action in England. Understanding the sources and pathways of PFAS in the environment is complex. This is because of the large number of individual substances and their widespread use over many decades. Some PFAS can accumulate in the food chain, and others are very mobile in soil and water. Many will eventually degrade to highly persistent terminal products, PFAAs or 'arrowheads', as described in Section 1.2. To help prioritise risk management, we have gathered information on the uses and sources of PFAS used, or potentially used, in England.

3.1 Uses of PFAS

More than **200 uses of PFAS** have been identified, spanning 64 use categories and over 20 industry sectors for 1,400 PFAS (Glüge et al., 2020). These include use in or as:

- chemical intermediates and processing aids for fluoropolymer production
- surface coatings for textiles, food contact materials and packaging
- cleaning agents, varnishes, polishes and waxes
- paints
- printing inks
- lubricants
- cosmetics and shampoos
- pharmaceutical active substances
- plant protection products
- medical devices and products
- feed additives
- the electronics and energy production sectors
- construction materials, including as polymers
- heat exchange systems
- fire suppression systems and firefighting foams
- mist suppressants for metal plating

Using information in the UK REACH database, the PFAS regulatory management option analysis (RMOA) estimated that between 2,600 and 36,700 tonnes of PFAS across all groups, were present on the Great Britain (GB) market in 2023. However, this does not include substances manufactured or imported at less than one tonne per year, substances still subject to transitional provisions following the UK's withdrawal from the EU, polymers and substances incorporated into articles. Further information on UK REACH is provided in Appendix 2.

The majority of PFAS potentially available on the GB market are thought to fall within the following broad chemical groupings, which are used across a wide range of industrial and commercial applications:

- **Polyfluoroalkyl substances**, used in fire suppression systems and FFF, foaming agents, refrigerant gases, electronic equipment manufacture, cleaning agents and certain medical devices.

- **Perfluoroalkenes**, used in insulation foams and in the manufacture of polymers, petrochemicals and fine chemicals.

- **Perfluoroalkanes and perfluorocycloalkanes**, used in cleaning agents, surface treatment agents, processing aids, refrigerant gases, heat-transfer fluids and semiconductor manufacture.

Estimates of emissions by industry sector indicate that the largest emissions arise from fluorinated gases (F-gases), textiles, upholstery, leather, apparel and carpets (TULAC), and medical devices and medicinal products (HSE, 2023). This is followed by food contact materials, electronics (including batteries), construction, and FFF, in that order from highest to lowest estimated emissions. Emissions from these sectors occur to air, water and land, except for F-gases, which are emitted primarily to air. These sectors are discussed further in Chapter 4. Similar patterns in the relative importance of different sources have been identified in the ECHA PFAS restriction proposal (ECHA, 2025). However, estimates of PFAS uses, production volumes and emissions remain uncertain and continue to evolve as new information becomes available.

Researchers and industry have been exploring alternatives that provide similar functionality without the persistence and, where relevant, toxicity. Alternatives already exist for many applications. For example, a recent database developed by Figuière et al. (2025) documents known uses of PFAS, describes the functions they provide, identifies potential alternatives capable of delivering equivalent or similar performance and evaluates their suitability as replacements. The database covers 325 PFAS applications across 18 use categories and identifies more than 530 PFAS-free alternatives.

Further information on the availability and use of alternatives across many sectors is also provided in the PFAS RMOA, while the OECD has also explored alternatives in sector-specific papers. Nevertheless, given the widespread use of PFAS across multiple sectors, transitioning to alternatives will require time, investment, and strong evidence on their performance, safety and sustainability to avoid regrettable substitution.

3.2 Sources of PFAS

PFAS can enter the environment during manufacture, use, and end-of-life processes such as disposal, recycling and legacy releases. Drawing on both international literature and UK-specific evidence, the major sources of PFAS entering the environment are:

- Industrial facilities that produce or process PFAS or use PFAS in manufacturing or other activities
- Firefighting activities and areas where fluorine-containing foams have been stored, used or released, such as fire stations, training facilities, military bases, airports, refineries and petrochemical facilities
- Waste management facilities, such as landfills, which can act as secondary sources
- Wastewater treatment works (WwTWs), including treated effluent and the production and land application of biosolids

Diffuse sources will also contribute to the overall environmental load. Examples include washing and general wear and tear of treated textiles; use and disposal of personal care products, surface coatings, cookware, paints, lubricants, building materials and PFAS-containing machinery. These activities can result in direct releases to the environment, or indirectly via run-off and subsequent discharges to water from WwTWs.

Historic use has also resulted in legacy contamination of soils and groundwater. Legacy contamination continues to contribute to the environmental burden because of the long lifespan of some PFAS products and the intrinsic persistence of PFAS. Understanding PFAS sources, the pathways by which they move through the environment, and the receptors they may affect is essential for identifying appropriate risk management actions.

3.3 Pathways and receptors of PFAS

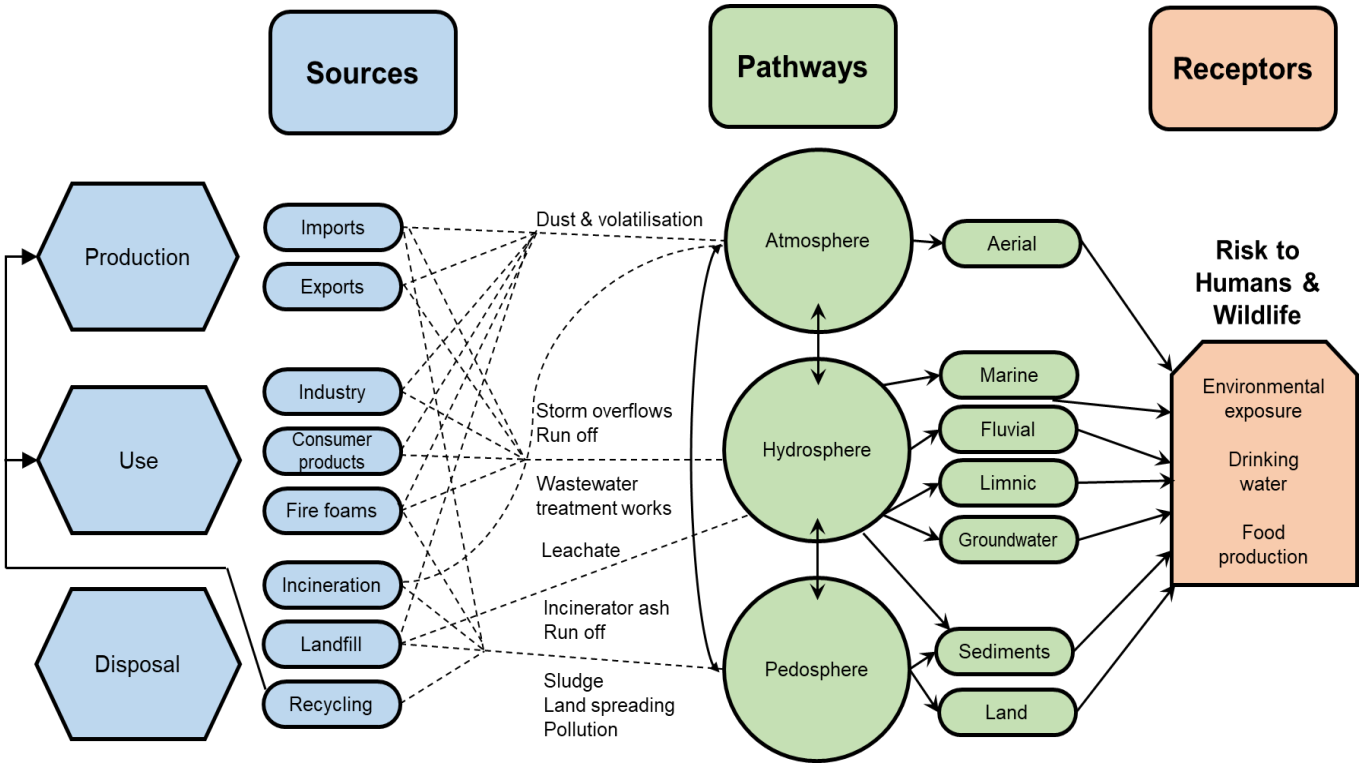


Figure 4. Overview of the relationships between PFAS sources, environmental pathways, and receiving environmental compartments and receptors.

Sources include activities associated with production, use, and disposal, such as industrial operations, product use, and waste management. Areas close to point sources typically show higher levels of PFAS. Pathways include air, water and land. Figure 4 highlights the interconnectivity between environmental compartments and the multiple pathways by which PFAS migrate, leading to contamination of environmental media and exposure of ecosystems and wildlife, with secondary implications for food production and drinking water.

Air: PFAS can be released directly to the atmosphere and transported by binding to airborne particles, as well as through partitioning and migration between air, water and surfaces (Environment Agency, 2025c). Volatile and semi-volatile precursor PFAS such as fluorotelomer alcohols (FTOHs) and perfluoroalkane sulfonamides (FOSAs) can travel thousands of kilometres via atmospheric currents and degrade into PFAAs, contributing to their widespread global detection and complicating differentiation between local and distant sources (Faust, 2023; Liu et al., 2023). Atmospheric emissions associated with fluorochemical manufacture and fluorinated gases may also contribute to environmental contamination through direct emissions of some PFAS and through the formation of persistent transformation products such as TFA (Miller et al., 2026). Atmospheric PFAS are returned to terrestrial and aquatic environments through wet and dry deposition (De Silva et al., 2021; DeLuca et al., 2022).

Water: For many PFAS, water plays a central role in environmental cycling, acting as both a transport pathway and a sink, with substances moving through rainwater, rivers, groundwater, and marine systems. Rivers are particularly important, acting as the main route by which PFAS are transported from source areas through the terrestrial environment and into coastal and marine waters. Oceanic currents may also play a key role in the long-range transport of PFAS in marine environments. In addition, at the air–water interface, PFAS can be resuspended into the atmosphere via sea-spray aerosols or deposited along coastlines through the formation of PFAS-enriched sea foam (Sha et al., 2024). PFAS can also accumulate in the cryosphere, including snow and ice (Arulananthan et al., 2025).

Soil and sediments: PFAS may sorb to soils and sediments, with the extent of sorption varying between substances but generally increasing with chain length (Barber, 2021; Beriro et al., 2025). The extent of sorption is also influenced by soil composition, pH, temperature and moisture content. Soils with high organic carbon content can act as temporary reservoirs for PFAS. At highly contaminated sites, particularly those affected by AFFF, some PFAS may also occur as self-associated aggregates rather than solely being sorbed to soil particles, which can influence their mobility and persistence in the subsurface (Tsianou et al., 2023).

Changes in environmental conditions may subsequently remobilise PFAS into groundwater or surface waters (Liu et al., 2023; Wang et al., 2023). Soils and sediments can therefore act both as sinks and as long-term secondary sources (Brusseau et al., 2020). Uptake by plants and soil organisms provides an additional pathway into terrestrial food webs.

Biota: Certain PFAS can bioaccumulate in wildlife and biomagnify through food webs, resulting in elevated concentrations in species at higher trophic levels. Consumption of plant material and protein-rich foods such as fish may therefore represent important pathways for human exposure to PFAS. Consequently, PFAS discharges to the water environment may contribute to human exposure through the accumulation of PFAS in fish and other aquatic organisms (Langberg et al., 2024). Studies have reported the presence of PFAS in human serum, plasma, and whole blood across populations worldwide (Labay and Blum, 2026).

The environmental fate and behaviour of individual PFAS are influenced by both their molecular structure, physicochemical properties and the environmental conditions they are exposed to. For example, chain length, ionic state, functional groups, solubility, and partition coefficients influence how PFAS move, persist and partition in the environment. Factors such as soil type, groundwater depth and flow, surface water connectivity, co-contaminants and climate also affect behaviour. As a result, residence times vary between environmental compartments, with precursor transformation adding an additional layer of complexity.

3.4 Source apportionment

Understanding the relative contribution of different PFAS sources is important for supporting targeted interventions and improving water quality. Source apportionment combines environmental monitoring data with information on land use, discharges and river flows to estimate the contribution of different sources within a catchment. These assessments can be based on either concentrations or loads. For PFAS, load-based approaches are particularly useful for understanding the relative importance of different sources, assessing environmental exposure, and evaluating whether regulatory and remediation interventions are reducing emissions over time.

The Environment Agency is exploring opportunities to apply source apportionment approaches to PFAS, building on experience gained through the Source Apportionment Geographic Information System (SAGIS), which is used to support nutrient permitting at WwTWs and assess the impacts of population growth on receiving waters. However, it can be particularly challenging due to the wide range of contemporary and legacy sources. PFAS can be released to the environment over long periods, both from identifiable sources and through diffuse pathways.

Although national-scale PFAS source apportionment is currently limited by data availability, recent work in the River Mersey catchment demonstrates the potential of load-based approaches. Byrne et al. (2024) estimated an annual export of approximately 68.1 kg of eight monitored PFAS from the River Mersey catchment and provided the first UK estimates of contributions of WwTWs to total river PFAS. Findings suggest that around one third of WwTW-derived PFOA may be retained within the catchment. The study also indicated that approximately half of PFOS exported from the catchment may originate from sources other than WwTWs. Further analysis by Byrne et al. (2025) identified the Upper Mersey as a major source area, with several smaller sub-catchments contributing

disproportionately high PFAS loads. The spatial distribution of these loads was consistent with catchments containing a greater density of industrial land uses, including waste management and landfills.

Despite challenges associated with data sparsity, overlapping sources and precursor transformation, source apportionment remains an important tool for supporting proportionate regulation, monitoring, remediation and enforcement. To further develop these approaches, we are undertaking further research to quantify PFAS loads from river catchments with distinct source profiles and improve understanding of the relative contribution of different sources.

Source apportionment approaches may also be strengthened through the application of non-targeted analysis and emerging environmental forensic techniques, including chemical fingerprinting. These approaches have the potential to improve identification of source-specific PFAS profiles and help distinguish between co-mingled sources where conventional load-based assessments alone may be insufficient. However, application of these techniques to PFAS remains an area of active research and development.

3.5 The PFAS Risk Screening Project

The PFAS Risk Screening Project was launched in 2020 to improve understanding of PFAS in the environment and support regulatory action. The project responded to evidence of widespread PFAS contamination, the need to better understand likely sources and pathways, and the need to prioritise sites for further investigation and regulatory activity.

Central to the project is the PFAS Risk Explorer, a Geographic Information System (GIS)-based risk screening tool designed to improve understanding of potential PFAS sources, pathways and risks across England. The tool supports the identification, prioritisation and management of sites that may warrant further investigation, helping to target regulatory effort, monitoring and evidence gathering in a proportionate way.

The PFAS Risk Explorer brings together a wide range of spatial datasets to provide comparative risk scores for more than 40,000 sites across England. These scores are based on factors such as site type, land use, environmental setting, soil characteristics, collected PFAS monitoring data, and connectivity to surface water and groundwater. The tool enables a consistent national screening approach, supports prioritisation of higher-risk sites, and helps identify sectors and site types that may be contributing to PFAS occurrence in the environment. Depending on the risk threshold applied, around 2,900 sites fall within a higher screening risk category, increasing to approximately 10,200 sites under a more precautionary approach.

Priority source areas in England include air transport and military sites, fire stations and firefighting training facilities, WwTWs, landfills, some Control of Major Accident Hazards (COMAH) sites and industrial sectors such as textile, paper, chemical manufacturing. These outputs have informed targeted desk-based studies of highest-risk sites, improving

understanding of sector-specific risks and helping to identify potential hotspots. These sites and sectors are discussed further in Chapter 4.

The tool does not quantify PFAS concentrations or loads, and a higher risk score does not necessarily mean that a site contributes the greatest PFAS burden. Instead, the scores provide a screening-level indication of the likelihood that a site may contribute to PFAS occurrence, helping to prioritise further investigation. Sites identified through the PFAS Risk Explorer still require site-specific assessment, including targeted monitoring and the development of conceptual site models, to confirm the presence of PFAS and assess potential risks to human health and the environment.

In addition to the Risk Explorer, the PFAS Risk Screening Project has delivered a range of supporting studies and tools that have strengthened the evidence base. These include assessments of ecological receptors, PFAS treatment technologies, the costs of remediating contaminated sites, and approaches for estimating background PFAS concentrations in soils (Beriro et al., 2025).

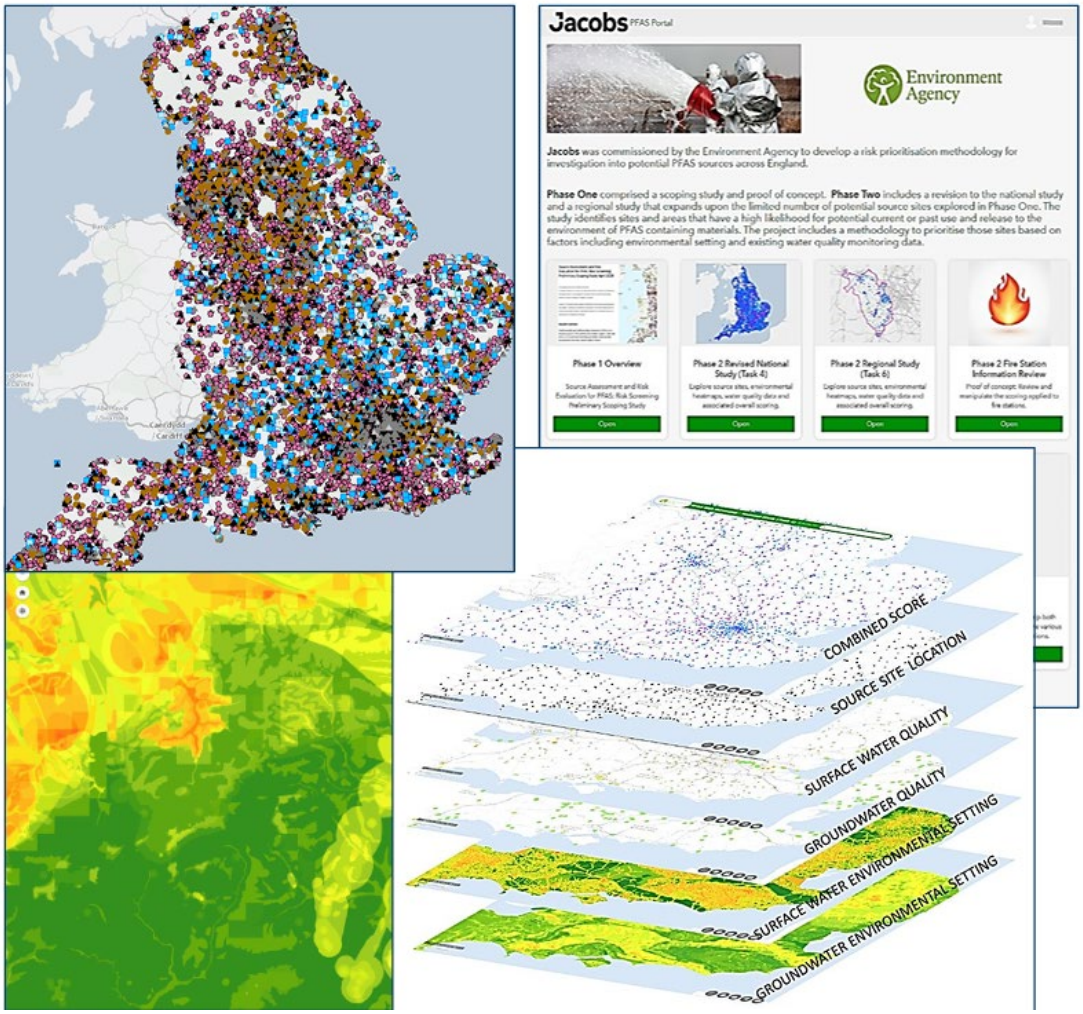


Figure 4: A screenshot of the PFAS risk explorer

4. Key PFAS uses, sectors and site profiles

This chapter provides an overview of the key uses, sectors and site profiles associated with PFAS use, release and historic contamination in England. It also explains how the Environment Agency identifies and prioritises potential PFAS sources for further investigation and regulatory action, drawing on evidence from industry engagement, environmental monitoring and regulatory information gathering.

4.1 PFAS sources: how we identify and prioritise risks

As part of the PFAS Risk Screening Project, site profiles were developed between 2020 and 2024 to improve understanding of PFAS use and occurrence across different site types and to support the identification and prioritisation of potential PFAS source areas. These profiles support site assessment and investigation, particularly in relation to soil and groundwater contamination, and extracts are incorporated into some of the summaries below. Each profile describes how PFAS are used within a particular activity, how they may be released to the environment, the substances most likely to be present, and the areas where releases are most likely to occur. Where available, this information is supported by published research and case studies. Site-specific PFAS profiles are published on the CL:AIRE website. Given the number of profiles available, they are referenced rather than reproduced in this report.

A semi-quantitative risk framework supports prioritisation of sites. This framework evaluates two main factors:

- Magnitude of PFAS use (expected quantities)
- Likelihood of environmental release, considering how PFAS are used, for example as raw material ingredients, process intermediates, or formed during industrial processes, the frequency of use, and the effectiveness of containment measures.

Multiplying these scores produces an overall source potential rating, helping to focus attention on areas of greatest concern. While these scores are generic and based on professional judgement, they provide a useful starting point for prioritisation and can be refined as new information becomes available. Separate standalone profiles were also developed for firefighting foams and fire suppression systems, recognising the widespread use of these products and their high potential to release PFAS to the environment across a range of industries and activities.

Building on this work, we undertook structured surveys of a number of regulated industrial sectors during 2024 and 2025, including chemicals, paper and pulp, textiles, refineries and fuel, and metals manufacturing and finishing, to improve understanding of the range and amounts of PFAS used and their potential pathways to the environment. Where further information is required, Regulation 61 notices can be issued under the Environmental Permitting (England and Wales) Regulations 2016 (EPR). These can be issued to obtain more detailed information on current and historic PFAS use, potential emissions, and the

presence of PFAS in surface water discharges, sewer discharges and waste streams. The information gathered to date as a result of issuing Regulation 61 notices has strengthened the evidence base and is helping to inform future regulatory decisions at both site and sector level. Alongside formal information gathering, we continue to engage with sectors to address evidence gaps and improve understanding of PFAS releases to the environment.

The Pollution Inventory (PI) complements this work by providing information about the releases and transfers of substances from industrial activities we regulate. This includes annual emissions of certain substances to air, controlled waters and land, and off-site transfer in wastewater and waste. While PFAS do not have a dedicated reporting category, some substances reported under existing categories are likely to fall within the OECD (2021) definition of PFAS (OECD, 2021), including:

- Perfluorocarbons (PFCs)
- Hydrofluorocarbons (HFCs)

Estimates of PFAS emissions by industry sector in England, including a breakdown of air emissions of PFCs and HFCs from each regulated industry between 2022 and 2024, are provided in Appendix 3. However, the PI data captures only emissions reported by larger regulated facilities that exceed the relevant reporting thresholds and therefore does not provide a complete picture of PFAS emissions across a sector. Reported emissions may also be subject to some degree of uncertainty.

4.1.1 Textiles, upholstery, leather, apparel and carpets

The textiles, upholstery, leather, apparel, and carpets (TULAC) industries are among the largest global users of PFAS, using them to provide water, oil and stain resistance. Despite this very high global use, very little information is available on actual PFAS use and potential environmental impacts related to textile manufacturing in England.

The global nature of textile supply chains also makes it difficult to fully understand the types and quantities of PFAS present in imported products. Additionally, it is often difficult to determine whether individual PFAS have been intentionally used, are degradation products, or are present as impurities.

PFAS may be applied during fibre production, fabric finishing and, in some cases, at the point of sale. Over 120 PFAS substances are known to be used in the TULAC sector (ECHA, 2025; Kotthoff et al., 2015; Wood, 2020).

PFAS are incorporated into impregnation agents and waterproof membranes, with fluorotelomer-based substances, side-chain fluorinated polymers (SCFPs) and fluoropolymers, particularly polytetrafluoroethylene (PTFE), are commonly used.

While the sector has shifted away from historical reliance on PFOS and PFOA towards short-chain alternatives, legacy contamination remains an ongoing issue. Wastewater from textile facilities often contains a wide range of PFAS, including fluorotelomer alcohols and sulfonates. Environmental releases may occur through process water, effluent discharges and the disposal of treated offcuts (Cantoni et al., 2024; Heydebreck et al., 2016; Jia et al.,

2023). Growing awareness of PFAS in textiles is creating an incentive for companies to transition towards PFAS-free alternatives (Wood, 2020). However, eliminating and replacing PFAS remains challenging in some applications, particularly personal protective equipment such as firefighter uniforms.

In England, textile facilities are concentrated predominantly in Yorkshire and Lancashire. Information provided in site surveys indicates that PFAS that currently or historically held and used at these facilities include, SCFPs, which may degrade and form non-polymeric PFAS (Titaley, 2024). Although PFOS and PFOA are banned in textile production, these PFAS may still be released from these sites through historical contamination. The PFAS RMOA estimates PFAS use in the TULAC sector in GB at 4,100–14,300 tonnes per year (HSE, 2023). Emissions of polymeric PFAS (380–1,420 tonnes per year) exceed those of non-polymeric PFAS (110–390 tonnes per year) across air, water and land.

Draft UK Best Available Techniques (BAT) conclusions for the textiles sector make explicit reference to PFAS (Environment Agency, 2025d), emphasising the importance of addressing their use and emissions at the sector level.

4.1.2 Paper and pulp manufacturing (including food packaging)

PFAS have been used in paper and cardboard products since the 1950s, primarily to provide grease and moisture resistance in food packaging and paper products used for medical applications (Langberg et al., 2024). Historically, PFOS was used in coatings and additives before being replaced by C8 fluorotelomers, which can degrade to PFOA, and subsequently by C6 fluorotelomers. Environmental contamination can occur through wastewater discharges, the land spreading of PFAS-contaminated sludge and potentially through emissions in the drying process (Lendewig et al., 2025). The use of recycled fibre, imported materials and complex supply chains can make it difficult to identify the sources of PFAS and how they move through the manufacturing process.

International case studies have highlighted PFAS contamination in the environment surrounding paper mills, with PFOS and PFOA still detected despite regulatory restrictions (Bauer et al., 2025). To better understand the situation, in 2023 the Environment Agency commissioned a targeted testing programme to investigate PFAS presence and potential releases at five UK paper mills. The mills process different types of recycled paper and produce a range of paper products. Samples were collected from key stages of production, including input paper, input water, effluent, sludge and output products, and analysed for 23 PFAS (Environment Agency, 2023a).

PFOS, PFOA and PFHxS were not detected above the limit of detection in any of the 15 input paper samples, suggesting these substances are not typically present in recycled paper. However, these substances were found at very low levels in some input water, effluent and sludge. The results suggest that PFOS and PFOA may be introduced via input water rather than recycled paper and may become concentrated in effluent and sludge during the paper-making process. Polyfluoroalkyl phosphate esters (PAPs) were also detected, although reported values are indicative due to limitations in available testing standards. PAPs, particularly 6:2 mono/diPAP and 8:2 mono/diPAP, have been used in

food contact paper as surface treatment agents that bind fluorinated chains to cellulose fibres (Begley et al., 2005; Dueñas-Mas et al., 2023; Håkon A. Langberg et al., 2024).

Food contact materials (FCMs) have historically represented one of the most significant uses of PFAS within paper, cardboard and moulded-fibre packaging. The dominant PFAS used in food packaging manufacture have been side-chain fluorinated polymers, most commonly based on fluorotelomer alcohol (FTOH) chemistry (Glüge et al., 2020; Trier, 2017). Perfluoropolyethers (PFPEs) and fluoroplastics have also been reported to play an important role in food packaging production (ECHA, 2025).

PFCAs, including perfluorobutanoic acid (PFBA), perfluorohexanoic acid (PFHxA) and PFOA, have been detected in FCMs, likely arising as impurities or oxidation products of FTOHs (Dueñas-Mas et al., 2023). As a result, FCMs are recognised as important pathways of direct human exposure to PFAS (Phelps et al., 2024). Fidra, an environmental charity, carried out a study to explore the extent of PFAS use in food packaging in the UK. They noted widespread use of PFAS across the UK food sector in FCMs. Supermarket bakery bags, microwave popcorn packaging, pizza boxes and takeaway packaging were highlighted as items very likely to contain PFAS (Fidra, 2020). These materials may also release PFAS to the environment during disposal, including through landfilling, composting and littering (Goossen et al., 2023). The PFAS RMOA estimates emissions from FCMs at approximately 180–560 tonnes per year (HSE, 2023).

When surveyed by the Environment Agency in 2020, the Confederation of Paper Industries stated that the vast majority of its members' product range is PFAS-free, but two companies were known to use PFAS in small quantities in food packaging and medical non-packaging products as grease and detergent proof barriers.

More recent surveys issued to Environment Agency-regulated paper and pulp facilities indicate that PFAS use has continued to decline, with most sites reporting no current use. Although the sector has largely transitioned towards PFAS-free alternatives, legacy contamination associated with historical use remains a concern.

4.1.3 Metal manufacturing and finishing

The regulation of the metals sector in England is complex, and some facilities that use PFAS may not require an environmental permit from the Environment Agency, while others may be regulated under different activities. As a result, PFAS use and emissions may not always be captured consistently across the sector.

PFAS are used in the metal sector primarily for their surfactant properties and chemical stability, particularly in metal plating and surface treatment processes where they act as mist suppressants, wetting agents, corrosion inhibitors and coating aids. Uses are most associated with chromium electroplating, but may also occur in nickel, zinc and other metal finishing applications.

Regulatory assessments and peer-reviewed studies consistently identify hard chromium electroplating as one of the most significant and higher-risk PFAS uses in the metal sector, owing to reliance on PFAS-based mist suppressants to control toxic chromium (VI)

aerosols (Cuthbertson, 2020; ITRC, 2023; UBA, 2022). PFAS releases may occur through process water, spills, and adsorption to concrete and equipment. The PFAS RMOA estimates annual PFAS emissions from metals and chromium plating at >0.6 tonnes across air, water and land.

Historically, PFOS-based mist suppressants were common, but have largely been replaced by 6:2 fluorotelomer sulfonate (6:2 FTS) and other alternatives as restrictions on PFOS have been introduced (Posner, 2020). In some regions, other alternatives such as F-53B, 6:4 FTS and, to a lesser extent, PFBS have also been reported (Bao et al., 2019; Pan et al., 2018; Zhang et al., 2025). International case studies have shown that, despite substitution, PFOS remains detectable in wastewater from some facilities, possibly due to desorption from infrastructure.

F-53B has not been registered under UK REACH; however, it has been detected at low concentrations in the River Thames and has also been identified in UK otters (O'Rourke et al., 2020). It was not known to be used in the UK at the time, although low-level use, potentially associated with mist-suppressant applications, cannot be ruled out given its international use as a replacement for PFOS. Further information on the use of F-53B is available in an Environment Agency risk evaluation report (Environment Agency, 2023b).

PFOS use as a mist suppressant in non-decorative hard chromium (VI) plating in closed-loop systems was permitted under Commission Regulation (EU) 2019/1021 as a specific exemption until September 2025. In 2018, the Environment Agency was notified of a stockpile of 56.8 kg of neat PFOS for use as a mist suppressant. A further notification for the same use was received in 2020. No active notifications were received prior to the expiry of the specific exemption in September 2025, indicating that continued use of PFOS for this purpose is unlikely in England.

Advances in decorative chrome plating have enabled chromium (III) to be used in place of chromium (VI), largely removing the need for PFOS or alternative mist suppressants in these applications. However, mist suppressants are still used in some hard chrome plating processes.

Survey responses confirmed that PFAS-based alternatives continue to be used as mist suppressants in chrome plating, with 6:2 FTS identified as the predominant substance. Where it has been included in the testing suite, 6:2 FTS is frequently the most abundant PFAS detected in groundwater and surface water in England (Section 7.2).

In parallel, a UK BAT review for the Surface Treatment of Metals (STM) sector is currently underway (Environment Agency, 2025d). This review is considering the inclusion of low-volume, but high-impact chemicals used within the sector, including PFAS.

PI data for 2022–2024 indicates that the metals sector reported no PFC emissions above the reporting threshold. Releases of HFCs were recorded in the annual air-emissions data; as noted in Appendix 3, these arise from activities within the non-ferrous sub-sector.

4.1.4 Chemical manufacturing industry

Certain parts of the chemical manufacturing sector, particularly facilities involved in the production or use of PFAS and other fluorinated chemicals, have been identified as areas of elevated PFAS concern due to the potential for environmental releases. Survey returns indicate substantial variation across the sector in the use, application and manufacture of PFAS, presenting a diverse set of regulatory challenges. While some facilities use PFAS in relatively small quantities, others rely on them for core manufacturing processes.

Particular attention has been given to fluoropolymer and fluorocarbon manufacturing because these activities involve the manufacture, processing and use of large quantities of fluorinated substances. Although this represents a relatively small industrial sector in England with two known UK production facilities, it is of regulatory significance because of the volumes of fluorinated substances handled.

Fluoropolymers, including PTFE, polyvinylidene fluoride (PVDF), fluorinated ethylene propylene (FEP), perfluoroalkoxy alkane (PFA), fluoroelastomers and ethylene tetrafluoroethylene (ETFE), are widely used in industrial, construction, electronic, medical and consumer materials because of their thermal stability, chemical resistance and low-friction properties (Glüge et al., 2020; OECD, 2021). Further information on the use of PFAS in construction and lubricants is provided in the PFAS RMOA.

In addition to the fluoropolymers themselves being PFAS, many fluoropolymer manufacturing processes have historically relied on non-polymeric PFAS, particularly fluorosurfactants, as polymerisation aids (Lohmann et al., 2020). PFOA and related substances were the most widely used fluorosurfactants for many years. As a result, PFOA and other PFCAs have been detected in air and soil around fluoropolymer manufacturing facilities in the UK, Europe and the USA (Dalmijn et al., 2025, 2024; D'Ambro et al., 2021; Dauchy, 2023).

Regulatory restrictions and voluntary phase-outs of PFOA have led manufacturers to adopt alternative fluorosurfactants, largely based on proprietary formulations of ammonium salts of perfluoroether carboxylic acids (PFECAs), including HFPO-DA, ADONA and EEA-NH4 (Environment Agency, 2023c; Wang et al., 2015). While these changes have substantially reduced emissions of legacy substances, concerns remain regarding historic contamination, the behaviour of replacement PFAS and their potential long-term impacts.

Although some non-polymeric PFAS emissions continue to occur from fluoropolymer manufacturing, the European fluoropolymer industry has reduced emissions of fluorinated polymerisation aids significantly since the 1990s (Ameduri, 2023; Environment Agency, 2026a).

The PFAS RMOA estimated annual PFAS emissions from PFAS manufacturing at <50 tonnes across air and water (polymeric and non-polymeric PFAS). Understanding both historic and current emissions at these sites remains a key priority for the Environment Agency, helping to inform regulatory approaches, permitting decisions, and environmental management activities.

Beyond fluoropolymer and fluorochemical manufacturing, sector survey responses identified a range of activities with the potential to generate PFAS-related risks through use and emissions. These include the production of organic and inorganic chemicals, such as pharmaceuticals (Section 4.1.7), plant protection products and biocides (Section 4.1.8), as well as chemical manufacturing processes involving electrolytic processes.

Other relevant sectors include semiconductor fabrication, including photolithography and etching, lithium-ion battery manufacturing and recycling, and the manufacture of AFFF (Guelfo et al., 2024; Rensmo et al., 2023). The chemical manufacturing industry also stores significant quantities of PFAS-containing AFFF for fire suppression, representing an additional potential source of environmental contamination. The PFAS RMOA estimates that the electronics sector (including semiconductors) and energy (including batteries) emissions are ~115 tonnes per year for polymeric PFAS and 55 tonnes per year for non-polymeric PFAS.

Given the diversity of PFAS use and the range of activities across this sector, a single uniform approach is unlikely to be effective. Instead, a tailored regulatory strategy is required to adequately address the unique PFAS risks posed by individual facilities and subsectors.

Regarding known PFAS emissions to air, PI data for 2024 show that the chemicals manufacturing sector accounted for 100% of industrial sites reporting PFC emissions above the reporting threshold. PI data further indicate that in 2024 the chemicals manufacturing sector accounted for approximately 17.8% of industrial sites reporting HFC releases. However, evidence on emissions from many subsectors remains limited (Appendix 3).

4.1.5 Food and drink products

Within the food and drink manufacturing sector, PFAS may arise from several potential sources. Raw ingredients can represent a pathway for PFAS to enter food products where crops or livestock have been exposed to PFAS through pesticides, veterinary pharmaceuticals (Section 4.1.7) or contaminated irrigation water, including where PFAS contamination may be associated with the application of biosolids (Section 4.2.4).

Fish, seafood, meat and eggs generally make the greatest contribution to dietary exposure (EFSA et al., 2020; Håkon Austad Langberg et al., 2024). PFAS, including TFA, are also increasingly being detected in beverages such as wine and beer (Burtscher-Schaden, 2025; Hoponick Redmon et al., 2025).

Additional sources may also include PFAS-containing components in processing equipment, such as oil-repellent conveyor belts, gaskets and seals. Certain cooking utensils and non-stick equipment used during food preparation may also contain PFAS (Schaider et al., 2017).

Refrigeration systems may also be relevant where fluorinated gases (F-gases) are used as refrigerants (Section 4.1.6). Leaks from these systems can result in fugitive emissions to the atmosphere (McLinden and Huber, 2020; Sovacool et al., 2021). PI data associated

with this sector indicate reported HFC emissions of approximately 13.7 tonnes in 2022, 14.0 tonnes in 2023 and 16.1 tonnes in 2024, which are likely to be associated with their use in refrigeration and cooling applications.

The Food Standards Agency (FSA) is the competent authority for food safety and consumer protection in England and Wales. While there are currently no statutory maximum levels for PFAS in food, and no PFAS-specific restrictions applicable across all FCMs in GB, the FSA continues to assess PFAS risks through monitoring and exposure assessment. The FSA has sought independent scientific advice from the Committee on Toxicity of Chemicals in Food, Consumer Products and the Environment (COT) on the development of appropriate standards. The FSA is also working to develop and validate analytical methods for PFAS in food. This work aligns with the UK Government's PFAS Plan, which sets out a cross-government, science-led approach to improving understanding of PFAS sources, reducing exposure, and supporting evidence-based risk management to protect public health and the environment.

In 2020, the European Food Safety Authority (EFSA) assessed risks to human health from the consumption of PFAS in food and derived a Tolerable Weekly Intake (TWI) of 4.4 ng/kg body weight (EFSA et al., 2020). This TWI relates to the sum of four PFAS (PFOS, PFOA, PFHxS and perfluorononanoic acid (PFNA)) considered to contribute most to the levels of PFAS found in human serum. The TWI has informed regulatory thresholds and risk assessments in several European countries, including Denmark, Sweden, Germany and Belgium. It was also used in the derivation of soil screening values in England and Wales (CL:AIRE, 2024; Environment Agency, 2026b). In addition, the EU EQS proposal for a group of 25 PFAS has used this TWI as its basis.

Some concerns over the basis of the EFSA TWI have been raised previously, including by the UK Committee on Toxicity (CoT) in a statement published in 2022 (COT, 2022). The CoT questioned the biological significance of the response (immunological) on which the TWI is based. They were, however, unable to propose an alternative value. CoT are currently reviewing available toxicological data to determine either an oral health-based guidance value (HBGV) for PFAS or several HBGVs, according to the data available. The outcome of the CoT review will inform future PFAS threshold derivation, but the timeline for completion is unknown.

Although legal limits have not yet been established, the UK Government's PFAS Plan signals an intention to strengthen the evidence base and consider further regulatory measures to reduce PFAS exposure and environmental contamination. In the meantime, regulation is based on general food safety legislation, supported by coordinated action across Defra, the Environment Agency and industry guidance aimed at minimising contamination.

4.1.6 Fluorinated Gases (F-gases)

F-gases are highly volatile hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulphur hexafluoride and other gases that contain fluorine. They are used mainly in refrigeration, air-conditioning, heat pumps, aerosols, solvents, foam-blowing agents and specialist

industrial applications. HFCs account for the majority of current F-gas use. Information on GB regulation of F-gases is provided in Appendix 2.

Many were introduced as replacements for ozone-depleting substances (ODS) because they do not damage the ozone layer. While ODS have now been largely phased out in the UK, the use of some F-gases has increased. However, many are potent greenhouse gases and some also meet the OECD 2021 definition of PFAS as they contain multiple $–CF_3–$ or $–CF_2–$ groups. Examples include 1,1,1,2,2-pentafluoroethane (HFC-125) and 1,1,1,2-tetrafluoroethane (HFC-134a).

Under the Kigali Amendment to the Montreal Protocol, the UK is required to phase down HFC consumption by 85% between 2019 and 2036. As HFCs are phased down, manufacturers and users must transition to alternatives, many of which are already commercially available across a range of sectors (Glüge et al., 2020). However, some of the most technically viable replacements are also fluorinated compounds that meet the definition of PFAS. For example, hydrofluoroolefins (HFOs) generally have much lower GWPs than HFCs and are increasingly used in refrigeration and air conditioning.

Unlike many of the PFAS sources discussed elsewhere in this chapter, F-gases are released primarily to the atmosphere rather than directly to land or water. The PFAS RMOA estimates that F-gases contribute around 4,050 tonnes of PFAS emissions to the atmosphere annually in GB. Their environmental fate is therefore strongly influenced by atmospheric transport and transformation.

While fluorinated gases are not routinely monitored within the Environment Agency's PFAS monitoring programmes, they are increasingly recognised as important indirect contributors to PFAS contamination through the formation of persistent transformation products (Arp et al., 2024; OECD, 2021). The dominant quantifiable source of TFA is considered to be atmospheric release of certain F-gases from refrigerants and blowing agents (Freeling et al., 2020; Holland et al., 2021; Adlunger et al., 2021). These may then enter terrestrial and aquatic environments through rainfall and other atmospheric deposition processes (Freeling and Björnsdotter, 2023; Joerss et al., 2024; Scheurer et al., 2017).

Some HFOs such as HFO-1234yf can generate TFA more readily and at higher yields than HFCs (Khan et al., 2026). A modelling study estimated that 17 (15–19) t/yr of TFA was deposited over the UK in 2024 from global emissions of hydrochlorofluorocarbons (HCFCs) and HFCs, and a further 7.3 (3.1-13) t/yr from HFO-1234yf used in mobile air conditioning (Hart et al., 2026a, 2026b).

F-gases are therefore an important PFAS use across many sectors because of their role as precursor sources of TFA and their potential influence on future PFAS concentrations in the environment. Understanding the occurrence and environmental behaviour of TFA is becoming increasingly important in the assessment of PFAS within the water environment (Section 8.3).

4.1.7 PFAS in pharmaceuticals and medical applications

Pharmaceuticals and medical devices have received increasing attention because of their widespread use, growing evidence base and potential to release PFAS to the environment. PFAS may be present as active pharmaceutical ingredients, manufacturing intermediates, processing aids and fluoropolymer components of medical devices and other healthcare applications. Examples include certain inhalers, specialised medical equipment and associated packaging materials.

Evidence indicates increasing use of the $–CF_3$ group in bioactive substances, including pesticides and human and veterinary medicines. Incorporation of this group is a common strategy to modify pharmacokinetic properties, including drug stability, solubility, and bioavailability (Johnson, 2022; Zhou et al., 2016). Any compound containing the $C–CF_3$ moiety has the potential to degrade to TFA (Greinke et al., 2026; Hart et al., 2026a; Lindley, 2025). However, the extent to which pharmaceutical compounds are transformed into TFA remains poorly characterised.

The European Federation of Pharmaceutical Industries and Associations (EFPIA, 2023) estimates that approximately 30% of commercially available active pharmaceutical ingredients (APIs) contain fluorine. Many of these would be considered PFAS due to the presence of a single $-CF_3$ group. Several of these are essential medicines, for example efavirenz (AIDS), mefloquine (malaria), fluoxetine (depression) and gemcitabine (cancer). Hammel et al. (2022) identified approximately 360 approved fluorinated pharmaceuticals globally, of which 107 meet the OECD 2021 PFAS definition (OECD, 2021).

PFAS may also be used as starting materials, intermediates, reagents and catalysts during the manufacture of both PFAS and non-PFAS active pharmaceutical ingredients (EFPIA, 2023). Fluoropolymers such as PTFE and ETFE are also used in equipment associated with the manufacture of pharmaceuticals, plant protection products and biocidal products.

The PFAS RMOA estimates annual PFAS emissions from the medical devices and medicinal products sector at approximately 560 tonnes, of which more than 70% are emissions to air (HSE, 2023). However, this estimate combines manufacturing and use-stage emissions and therefore does not provide separate estimates for each life cycle stage. A proportion of these emissions is expected to arise from the use stage, particularly from pressurised metered-dose inhalers and volatile inhaled anaesthetics such as isoflurane, sevoflurane and desflurane (Environment Agency, 2026a; Kalmar et al., 2024).

There is increasing recognition that PFAS associated with pharmaceutical manufacturing, formulation and medical products require targeted management and control measures, particularly where releases to the environment may occur.

4.1.8 PFAS in pesticides

Ogawa et al. (2020) identified 424 organofluorine plant protection products (PPPs), representing approximately 16% of all PPPs in global production at the time. At least some of these are likely to meet the 2021 OECD PFAS definition. Joerss et al. (2024) reported that 143 PPP active substances contained a trifluoromethyl ($–CF_3$) group, of which 39

were approved for use in the European Union. PFAS are also present in certain biocidal products.

In the UK, there are at least 31 active pesticide substances approved for use that currently meet the 2021 OECD definition of PFAS, of which 25 are in active use. It is estimated that PFAS-containing pesticides were applied to approximately 7.76 million hectares of arable crops in the UK in 2024 and represented 16% of the most widely used pesticides in the arable sector (Donley et al., 2024; Fidra, 2026; Lasee et al., 2022). The use of PFAS containing pesticides provides a direct route for PFAS to enter the environment and is considered a major driver of TFA contamination in the UK. However, quantifying these emissions remains challenging due to limited data on degradation rates and TFA yields (Johnsen et al., 2026).

The five most widely used PFAS active substances, diflufenican, flufenacet, lambda-cyhalothrin, trifloxystrobin and mefentrifluconazole, were collectively applied to more than 41 million hectares of arable crops in the UK between 2014 and 2024 (Fidra, 2026).

4.1.9 Firefighting and fire suppression activities

Firefighting foams (FFF), particularly aqueous film-forming foams (AFFF), represent one of the most significant use-related sources of PFAS contamination in England. Unlike many other applications discussed in this chapter, FFF have historically involved the deliberate and repeated application of PFAS-containing formulations directly to land, often at high volumes and with limited containment. This has resulted in highly localised but persistent contamination, particularly affecting soils and groundwater, with impacts that commonly extend decades beyond the period of active use (ITRC, 2023; Mattila et al., 2024).

The PFAS associated with FFF comprise complex mixtures of terminal substances and precursors. Older formulations were dominated by long-chain sulfonates, notably PFOS, while more recent products have relied primarily on short-chain C6 fluorotelomer-based PFAS. A wide range of precursor substances have been used across different generations of foam, many of which can degrade over time to PFAAs (Harding-Marjanovic et al., 2015). As a result, PFAS profiles at foam-impacted sites often differ from those associated with wastewater discharges, landfill leachate or industrial manufacturing, and may evolve as precursor transformation proceeds and historical residues continue to leach from soils and infrastructure (HSE, 2025a; ITRC, 2023). The long service life of AFFF, combined with incomplete historical records of product composition and usage, can make source identification and site characterisation challenging.

Releases associated with FFF have arisen through multiple activities, including routine fire-training exercises, emergency incident response, accidental discharges, testing and failure of fixed fire-suppression systems, and subsequent equipment washing and waste handling. These pathways affect a wide range of sites, including airports, fire stations, COMAH sites, industrial facilities and transport hubs. Fixed foam systems installed in hangars, warehouses and high-hazard industrial settings have been a particular source of discrete contamination incidents (Anderson et al., 2016), with large volumes of foam released rapidly during system activation or malfunction. Residual PFAS retained within

tanks, pipework and pumps can continue to leach long after fluorinated foams have been replaced, as effective decontamination of PFAS from AFFF-impacted fire suppression systems is extremely difficult (Nguyen et al., 2025).

Dedicated fire-training grounds are consistently identified as having the highest source potential. Historic practices often involved repeated application of large quantities of AFFF directly to unsealed surfaces, typically without capture or treatment of runoff. Numerous investigations have demonstrated extensive contamination of soil, groundwater and nearby surface waters at such sites, with PFAS adsorbed into concrete pads and underlying substrates acting as long-term diffuse sources (Reinikainen et al., 2022). Although modern training practices increasingly emphasise containment, foam capture and the use of fluorine-free alternatives, legacy contamination frequently remains a dominant driver of risk and requires site-specific investigation rather than reliance on generic assumptions.

FFF are a regulatory priority under UK REACH. The HSE, in collaboration with the Environment Agency, has developed proposals to restrict the placing on the market and use of PFAS in FFF, reflecting both their environmental persistence and the availability of non-fluorinated alternatives for many applications (HSE, 2025a, 2025b). In addition, several foam-related PFAS, including PFOS, PFOA and PFHxS, are regulated under the POPs Regulations. Where concentration thresholds for these substances are exceeded, PFAS-containing foams and contaminated residues must be managed as POPs waste and destroyed or irreversibly transformed, most commonly through high-temperature incineration (Section 4.2.3). Replacement of foams and decontamination of systems can be complicated by residual PFAS retained within infrastructure, and inappropriate handling of legacy stocks remains a recognised risk.

Based on the best available national evidence, PFAS emissions associated with FFF use in GB have been estimated at approximately 48 tonnes per year, although significant uncertainty remains regarding the relative contribution of legacy versus current use (HSE, 2025a, 2025b). Monitoring data consistently identify elevated PFAS concentrations in groundwater and surface waters in the vicinity of known foam-use locations, but quantitative source apportionment is often constrained by incomplete historical records and the complexity of precursor transformation. As with other legacy sources discussed in this chapter, targeted site investigation and monitoring are therefore required to establish the extent of contamination and inform proportionate regulatory or remedial responses.

4.1.10 Refineries and fuel storage

Refineries and fuel storage sites have historically used PFAS primarily in fire suppression systems and emergency response foams, rather than in routine process applications. International case studies show that the scale of operations and fuel inventories at these sites mean that major fire incidents can involve the deployment of thousands of litres of AFFF (Reinikainen et al., 2022). PFAS have been found in drainage systems, sediments, and groundwater, with legacy PFOS dominating older contamination profiles. Although modern containment aims to minimise releases, historical practices and residual PFAS within infrastructure remain ongoing concerns.

A well-documented UK example is the 2005 Buncefield oil storage depot fire. A series of explosions at the site triggered a catastrophic fire that burned for three days. During the operation to extinguish the fire, approximately 786,000 litres of firefighting foam were used, of which some contained PFOS. Joint investigations by the HSE and the Environment Agency identified the event as a major pollution incident, with releases of foam-laden firewater and fuel to drainage systems, soils, groundwater and nearby surface waters. Due to the use of PFAS-containing AFFF, the contamination is particularly long-lasting (HSE, 2025).

Information provided through surveys confirms that PFAS use at refineries and fuel storage sites is predominantly associated with AFFF. While operators generally demonstrate good awareness of the environmental risks associated with foam use, the persistence of PFAS means legacy contamination is likely to remain at some sites. Regarding emissions to air, no HFC or PFC emissions above the relevant reporting thresholds were recorded for this regulated sector in the 2024 PI data.

4.1.11 Oil and gas (onshore exploration and extraction operations)

PFAS use within the oil and gas industry includes applications in drilling fluids, surfactants and fire suppression systems. PFAS may be used to reduce fluid loss during drilling, improve the stability of drilling muds and enhance reservoir permeability (Hussain et al., 2022). Potential release pathways may include flowback fluids, wastewater, spills and leaks from infrastructure, while the historic use of AFFF at exploration, production and storage facilities may also contribute to legacy contamination. Information provided by operators through surveys confirms these uses.

Although data on current UK practices remain limited, international studies have documented PFAS contamination in groundwater and surface water near oil and gas facilities.

4.1.12 Control of Major Accident Hazards Regulations (COMAH) Sites

Sites regulated under the Control of Major Accident Hazards Regulations (and predecessor legislation) store large volumes of hazardous substances, which may include flammable substances. They are required to have extensive fire prevention systems, which need to be demonstrated through emergency exercises. Historically, these systems relied on fluorinated foams, with maintenance, training and accidental discharges contributing to PFAS releases. As PFAS are not classified as dangerous substances under current regulations, their consideration in risk assessments has been limited. PFAS from past activities may be absorbed to concrete and released to the environment even after use of fluorinated foams has ceased. Source potential is highest in emergency exercise areas and wastewater systems, where containment failures can result in significant environmental impact.

4.1.13 Civil and military airfields

PFAS have been widely used at civil and military airfields, primarily through the use of AFFF for fire suppression and training, as well as other sources such as hydraulic fluids, specialist coatings, lubricants and maintenance activities. Historically, live fire-training practices involved direct application of foam to ground surfaces, often with minimal containment of runoff. Hangar fire-suppression systems have also contributed to contamination, with several documented accidental discharges over past decades. PFAS have been detected in groundwater and surface water near airfields, with legacy contamination persisting due to adsorption into concrete, asphalt and drainage infrastructure (Srivastava et al., 2024). Common PFAS identified at airfield sites include PFOS, PFOA, 6:2 FTS, 6:2 FTAB and cyclic PFAS such as perfluoroethylcyclohexane sulfonate (PFECHS) (Liu et al., 2022; Yan et al., 2024). Source potential is highest at training grounds and hangars with a history of foam use, where the magnitude and likelihood of release were historically high.

The Environment Agency is currently undertaking a review of environmental permits for water-discharge activities at English airports and military airfields, recognising that PFAS may still be entering the water environment as a result of both historic and ongoing use of PFAS-containing firefighting foams.

4.1.14 Other military installations

Military installations, excluding air transport sites, have historically used PFAS-containing AFFF for both training and emergency response. Additional potential sources of PFAS include vehicle maintenance areas, hydraulic fluids, bilge water systems and certain pesticide applications. International investigations have consistently found widespread PFAS contamination in groundwater at military sites, typically linked to historic foam use. Fire training grounds present the highest source potential, reflecting repeated, high-volume foam applications over potentially long periods.

The Environment Agency is working closely with the Ministry of Defence (MOD) to assess current and historical PFAS use across military estates and to evaluate potential contamination at these sites. This includes identifying source areas, assessing pathways to groundwater and surface water, and understanding the extent of any legacy contamination.

Emerging research indicates that munitions manufacturing, storage, disposal and firing activities may represent an additional source of PFAS to the environment (Koban and Pfluger, 2023; Olsavsky et al., 2020). Although current evidence remains limited and not all of these activities are regulated by the Environment Agency under the Environmental Permitting Regulations 2016, they represent a potentially relevant pathway for environmental release that warrants further investigation.

4.1.15 Paints and coatings

Paint and coating production is related to chemical manufacturing but is primarily a formulation process in which pre-manufactured raw materials, including pigments, resins, binders, solvents, fillers and additives, are received and physically blended, dispersed and mixed to produce finished coating products. Typically, formulation does not involve intentional chemical reactions to create new chemical substances.

PFAS may be added to paints and coatings to improve flow, levelling, gloss and durability, as well as to reduce bubbling and peeling (Environment Agency, 2026a). They are also used in speciality coatings to provide stain-resistant, graffiti-resistant and water-repellent properties (Cahuas et al., 2022; Fernandez et al., 2021; Liu et al., 2025). PFAS in these applications may include fluoropolymers, side-chain fluorinated polymers, perfluoropolyethers (PFPEs), and PFAS-based surfactants, including some short-chain fluorinated surfactants used in paints and coatings (OECD, 2022a).

A 2020 survey conducted by the Environment Agency with the British Coatings Confederation, representing around 95% of UK sales of coatings, inks and wall coverings, indicated that at least one member receives PFAS-containing raw materials. However, no information was provided on quantities used or potential emissions (Environment Agency, 2021).

4.1.16 Cosmetics

The production of cosmetics is primarily a formulation process involving the physical blending and mixing of pre-manufactured raw materials, such as oils, surfactants, emulsifiers, waxes, fragrances, preservatives, colourants and active ingredients, to produce finished cosmetic products. PFAS are intentionally used in some cosmetics to improve product performance. The PFAS RMOA estimated annual PFAS emissions from cosmetics to water and land at approximately 0.9–1.8 tonnes. Although PFAS with known uses in cosmetics are not classified as hazardous to human health, products containing polymeric PFAS, FTOHs and PAPs may act as sources of PFOA, PFOS and other PFAAs through degradation (Bălan et al., 2024; Fujii et al., 2013; Harris et al., 2022; Whitehead et al., 2021).

Identifying and quantifying PFAS in cosmetics remains challenging because of the wide range of fluorinated ingredients, limited analytical standards and emerging analytical methods (ECCC and Health Canada, 2024). In a 2020 survey, the Environment Agency received a consolidated response from the Cosmetic, Toiletry and Perfumery Association (CTPA), which reported the use of nine individual PFAS within the UK cosmetics sector.

More recently, the CTPA has reported that many manufacturers are voluntarily moving away from the use of PFAS in cosmetics. This suggests that cosmetics may become an increasingly minor source of PFAS. However, the CTPA does not represent the entire cosmetics sector, and there remains uncertainty regarding PFAS use in products supplied by non-member companies.

4.2 PFAS in waste management sectors

4.2.1 Landfills

Landfill represents a final disposal route for a wide range of PFAS-containing waste streams, including domestic, commercial and industrial waste. As waste degrades through microbial, biochemical and physico-chemical processes, landfill leachate and gas are generated, providing potential pathways for PFAS releases to water, air and surrounding environments.

International studies consistently identify landfills as important sources of PFAS to the environment, particularly via leachate (Chen et al., 2023; Hamid et al., 2018; Tolaymat et al., 2023). Short-chain PFAS (C4 – C7) are commonly dominant in leachate, reflecting both their mobility and the transformation of precursor compounds within landfill matrices (Section 8.1). The behaviour of PFAS in landfills is complex and influenced by waste composition, landfill age, climate, and internal processes such as sorption and biodegradation.

Modern permitted landfills in England are designed with engineered containment systems, including geological barriers, synthetic liners, leachate collection and treatment infrastructure, gas management systems and engineered caps, to minimise pollutant migration. In contrast, many historic landfills, particularly those constructed before 2001 and before the introduction of modern environmental permitting and engineering standards, were developed to much lower standards. These sites may pose a higher risk of contaminant discharge through the base, sidewalls and surface, increasing the potential for environmental contamination.

England has approximately 19,000 historic landfill sites and 1,800 permitted operational and closed sites (Environment Agency, 2026c, 2026d). Given the widespread historic use of PFAS, the difficulty of segregating PFAS-containing wastes from general waste streams at source, and the fact that many PFAS-containing wastes would not necessarily have been classified as hazardous waste, many landfills are likely to have accepted PFAS-containing waste during periods when landfill pollution controls and containment standards were generally less stringent than those applied today.

In response, Defra and the Environment Agency have undertaken a phased programme of work to better understand the presence, behaviour and risks of PFAS in landfill leachate. This has included literature reviews, analytical method development, site investigations and targeted sampling at a range of operational, closed and historic landfill sites (Defra, 2024; Jacobs, 2021). These studies provide evidence that PFAS are commonly present in both raw and treated leachate, that treatment effectiveness varies by compound and process, and that landfill discharges may contribute to PFAS burdens in receiving environments. Detailed results are reported in the respective study outputs.

Ongoing work aims to further characterise PFAS sources, compound profiles and treatment performance and assess the effectiveness of current management practices, including on-site treatment and the transfer of leachate to wastewater treatment works.

Emerging evidence also indicates that volatile precursor PFAS, including FTOHs, fluorotelomer olefins (FTOs), FOSAs and perfluorooctane sulfonamidoethanols (FOSEs), may be present in landfill gas, representing a potentially important atmospheric pathway (Goukeh et al., 2025; Smallwood et al., 2023; Titaley et al., 2023). Following release, these compounds can undergo oxidation in the atmosphere to form more polar ionic PFAS (Ahrens, 2011; Lin et al., 2024).

To improve understanding, the Environment Agency is planning further studies on PFAS releases via landfill gas and the potential removal of PFAS through gas management systems. Additional work will include estimating the annual volume and tonnage of PFAS released in treated leachate discharged to surface waters from permitted landfills, alongside the development of best practice guidance for leachate sampling methodologies at landfill sites.

4.2.2 Recycling facilities

UK industrial and commercial recycling facilities operate under defined waste acceptance criteria and the Environmental Protection (Duty of Care) Regulations 1991. All waste must be appropriately classified and managed. Where waste contains PFAS listed under the POPs Regulations, currently PFOS, PFHxS and PFOA, above the relevant concentration thresholds set out in Annex IV of the POPs Regulations, it must be classified as POPs waste. It must then be managed through disposal or recovery in a way that destroys or irreversibly transforms the POP content, which is typically via high temperature incineration.

Waste containing other PFAS that are not listed under the POPs Regulations is not subject to these specific requirements. Furthermore, the majority of PFAS are not currently classified as hazardous and therefore do not trigger hazardous waste obligations. As a result, there is limited legal obligation or incentive to identify, separate, or fully destroy PFAS-containing waste. This means other PFAS, including fluoropolymers may persist through certain thermal and mechanical recycling processes (Cui et al., 2019; Lohmann et al., 2020), leading to contamination of recycled materials. These materials may then be used in new products, where additional PFAS could also be applied.

Household recycling presents a recognised challenge. PFAS-containing items such as food-contact packaging, textiles, non-stick cookware, stain-resistant treatments, and personal-care products may be placed into recycling streams. This creates both:

- a barrier to achieving a more circular economy, as PFAS-containing materials can contaminate recycled feedstocks and limit opportunities for safe material recovery; and
- a potential source of environmental release, particularly where PFAS are mobilised during waste-sorting, shredding, baling, or subsequent processing activities.

Current understanding of how PFAS enter and move through supply chains remains limited, particularly at end of life. As a result, information on the extent to which PFAS may be present in recycled materials and subsequently incorporated into new products is still

developing. Improved supply chain transparency, clearer product information, better segregation of PFAS-containing wastes and more defined waste management routes will be important to reduce PFAS entering recycling infrastructure, support diversion to appropriate treatment or destruction routes, and underpin future circular economy goals.

4.2.3 High temperature incineration

PFAS were originally designed to be highly stable and resistant to extreme environmental conditions, including high temperatures, which is why they have been extensively used in products such as AFFF (Mattila et al., 2024; Padhye et al., 2026). As a result of this stability, their destruction is technically challenging.

In England, PFAS-containing wastes are managed through a range of treatment, recovery and disposal routes. At present, the only proven large-scale method capable of achieving near-complete mineralisation (defluorination) of almost all PFAS is high temperature incineration (HTI) at hazardous waste incinerators (Environment Agency, 2025a).

In the UK, PFAS-containing wastes currently considered most suitable for high-temperature incineration include AFFF and AFFF-contaminated liquids and solids arising from training and emergency response activities. Additional waste streams may include concentrated process residues, sludges and absorbent materials generated during remediation or clean-up of PFAS-contaminated sites, where source-segregated treatment is feasible. To date, AFFF containing PFAS regulated as POPs has been the primary focus of regulatory assessment and permit controls (Environment Agency, 2025a; Hofman et al., 2025).

In GB, HTI capacity for PFAS-containing wastes remains limited (Office for Environmental Protection, 2026). While specialist treatment capacity is available, overall capacity for hazardous waste incineration is finite. As more PFAS material is directed toward incineration as the optimal disposal route, facilities may experience increasing demand on available throughput. There are regulatory actions available that would allow the export of PFAS wastes for incineration, or for treatment using any new equivalent technologies, pending increased UK capacity (HSE, 2025b).

The Environment Agency recently assessed the effectiveness, feasibility and environmental suitability of current and emerging PFAS-remediation technologies for treating PFAS-containing wastes, with particular focus on HTI and firefighting foams (Environment Agency, 2025a). This work examined the operating conditions required for effective PFAS destruction, the risks associated with products of incomplete combustion, and the potential role of alternative treatment methods. It found that HTI can achieve destruction efficiencies greater than 99.99% for PFAS-containing FFF when operated under the following conditions: temperatures ≥1,100°C, a 2–3 second residence time in the secondary combustion chamber, adequate turbulence to ensure complete mixing, and properly balanced stoichiometry (waste, fuel, oxygen and gas-phase components).

Suboptimal conditions, such as lower temperatures, may result in the formation of products of incomplete combustion (PICs), including short-chain PFAS with high global-warming potential, such as CF_4 and C_2F_6. Optimising the incineration process to

avoid CF_4 formation should therefore be a primary goal. While analytical methods for identifying and quantifying the full range of PICs are still evolving, low concentrations of C_2F_6 are generally considered a reasonable indicator that a high degree of PFAS mineralisation has been achieved.

To address these risks, the Environment Agency has updated the permits for HTI accepting PFAS-containing AFFF, requiring them to operate under conditions that ensure effective PFAS destruction. The Agency is also working with operators to carry out destruction-efficiency trials and improve monitoring using Other Test Method (OTM) 45 and OTM 50 to verify how effectively PFAS in AFFF is being mineralised in practice (Hofman et al., 2025; Environment Agency, 2026a).

As discussed in Section 4.2.2, most PFAS waste streams are not currently classified as POPs waste, as only a limited number of PFAS are regulated as POPs and these substances are typically present below the applicable POPs waste thresholds (Office for Environmental Protection, 2026). As a result, they may be disposed of through municipal waste incinerators. This is of possible concern because municipal waste incinerators typically operate at lower temperatures of around 850°C and may unknowingly treat PFAS-containing wastes without ensuring complete mineralisation.

Therefore, further work is needed to test for PFAS in municipal waste incinerator outputs, including incinerator bottom ash, and assess PFAS destruction efficiency across different waste streams (Björklund et al., 2024, 2021). However, current evidence suggests that municipal waste incinerators in England are not a significant source of PFAS emissions. They may also be effective for the destruction of fluoropolymers, although further work is required to improve understanding of the formation, identification and fate of potential PICs (Gehrmann et al., 2024).

Cement kilns may offer a potential alternative solution for the destruction of PFAS-containing wastes (Environment Agency, 2025a). Cement kilns operate at very high temperatures, often exceeding 1,100 °C, which are sufficient to promote the thermal destruction of many organic contaminants. During operations, the co-incineration of calcium-rich mineral feedstocks can favour the formation of stable inorganic fluoride products such as calcium fluoride (CaF_2). Studies also suggest that calcium species can promote mineralisation of PFAS at comparatively lower temperatures than traditional incineration, increasing the likelihood of PFAS breakdown rather than partial degradation (see references within Environment Agency 2025a). However, evidence on PFAS destruction in full-scale cement kilns remains limited, and the Environment Agency is currently undertaking work to assess PFAS destruction and removal efficiency at UK cement kilns (Environment Agency, 2025e). If demonstrated to be effective and safe, this disposal route could unlock additional incineration capacity, enabling the regulated, safe and controlled destruction of PFAS stockpiles.

Overall, where not already a legal requirement, destruction of PFAS through HTI is preferred over landfilling because it avoids long-term environmental contamination. Although alternative destructive treatment technologies are under development, few are currently commercially available, and HTI remains a key method for PFAS management

globally (Environment Agency, 2025a). Further research is needed to optimise incineration processes and characterise emissions under different operating conditions.

4.2.4 Wastewater treatment works

WwTWs act as major collection points for PFAS from domestic, industrial and diffuse sources and represent an important pathway by which PFAS can enter the wider environment. PFAS are widely detected in influent, effluent and sewage sludge at WwTWs in the UK and internationally, reflecting their extensive use, persistence and mobility within catchments (Aro et al., 2021; Gewurtz et al., 2024; Gobelius et al., 2023; Thompson et al., 2022).

Conventional wastewater treatment processes are not designed to remove or destroy PFAS, and removal efficiencies vary substantially between compounds and treatment configurations. PFAS may be discharged in liquid effluent or transferred to sewage sludge, including biosolids applied to land. Transformation processes within WwTWs can also convert precursor substances to more stable terminal PFAS, which may occur at higher concentrations in effluent than influent. UK studies have demonstrated substantial variability in precursor transformation, with some WwTWs exhibiting marked increases in terminal PFAS following treatment, while others show little change or decreases, depending on site-specific conditions (Neill and Megson, 2024; Pascu et al., 2025). Volatile PFAS may also be emitted to air and have the potential for long-range transport. These complexities make WwTWs a sustained focus for regulatory attention and stakeholder engagement.

In England, the main evidence base on PFAS in WwTWs is provided by the Chemical Investigations Programme (CIP), a phased, water-industry-led programme coordinated with the Environment Agency and other regulators to address emerging chemical risks. Full technical detail, monitoring data and site-specific findings are reported in the CIP publications available online (UKWIR, 2025).

- CIP1 (2010–2015) focused on identifying sources and pathways of chemicals entering rivers via WwTW discharges, providing an initial characterisation of treated wastewater.
- CIP2 (2015–2020) expanded monitoring across more than 600 WwTWs, including PFAS in effluents and in rivers upstream and downstream of discharge points, with a focus on PFOS and PFOA at sites discharging to low-dilution catchments (Comber et al., 2021; UKWIR, 2021). Across more than 9,000 effluent samples, PFOS and PFOA were detected in at least 98% of samples, with mean concentrations of around 0.005 µg/L (Environment Agency, 2025f). Data for PFOA and PFOS show concentrations are highly variable between WwTWs.
- CIP3 (2020–2022) included more detailed investigations to address gaps identified in earlier phases. It included investigations of PFOS in estuarine discharges and of PFOS, PFOA and additional PFAS in groundwater and sludge. Targeted sampling of sewer and river catchments was also undertaken to help identify potential sources of PFOS and PFOA, with sectors identified including landfill leachate,

petrol stations, car washes, abattoirs, iron phosphating, nursing homes and commercial bakeries.

Together, these studies show that PFAS are widespread within wastewater systems and that WwTWs can act both as receivers of PFAS from catchments and as pathways for onward release to surface water, groundwater and land via biosolids spreading.

From a sector perspective, WwTWs present a particular challenge for PFAS management. Advanced treatment technologies, including granular activated carbon, ion exchange and membrane processes, can achieve substantial PFAS removal from water. However, these technologies can be costly to install and operate, their performance may be reduced for shorter-chain PFAS, and they generate secondary waste streams that require further treatment or disposal. As a result, source control and upstream intervention are widely recognised as more effective long-term strategies for reducing PFAS inputs to wastewater systems (Environment Agency, 2025a).

The current phase, CIP4, is investigating PFOS in sewer catchments with above-average concentrations, alongside studies on the behaviour of PFAS in effluent and biosolids, groundwater risks associated with sludge application, and more comprehensive PFAS sampling in sludge. CIP4 also includes the first national-scale investigation of PFAS and other emerging substances in groundwater associated with treated effluent discharges to ground. Dedicated groundwater monitoring boreholes, conceptual site models and long-term monitoring can be used to characterise contaminant pathways, improve understanding of attenuation processes and assess risks to groundwater receptors.

Outputs from CIP4 are expected to inform future regulatory decisions, including planning for Ofwat's (the Water Services Regulation Authority) Price Review 2029 (PR29), and consideration of whether additional controls are required to manage PFAS risks associated with wastewater treatment and biosolids management. CIP5 (2030-2032) is currently in the planning stages and is expected to include work investigating PFAS transformation within WwTWs.

4.2.5 Spreading of material to land

The spreading of materials, except for sewage sludge, to land is regulated under the Environmental Permitting (England and Wales) Regulations 2016, with the Environment Agency acting as the primary regulator in England. Although not all materials applied to land fall within this framework, at present, the spreading of sewage sludge to agricultural land is governed separately under the Sludge Use in Agriculture Regulations 1989 (SUiAR). These regulations require that sludge application does not impair soil or water quality. While producers have discretion over additional testing, statutory requirements currently cover only a limited range of metals and do not include emerging contaminants such as PFAS, despite evidence that PFAS can be present in sewage sludge.

The Government's January 2026 White Paper, A New Vision for Water, signals a shift towards stronger and more integrated regulation of sewage sludge. In line with recommendations from the Independent Water Commission, government is consulting on reforms to the existing framework, including whether land spreading of sewage sludge

should be brought within the scope of the Environmental Permitting Regulations 2016. Decisions on any future changes will follow consultation and ministerial consideration.

In recent decades, the volume of material applied to agricultural land for soil conditioning and nutrient recycling has increased, reflecting wider policy support for resource recovery and more sustainable waste management. However, there is increasing concern that land spreading may also act as a pathway for PFAS to enter soils and the wider environment, particularly where PFAS-containing biosolids, sludges, composts or other waste-derived materials are applied to land.

Conventionally produced compost from source-segregated food and green waste is likely to present a relatively low PFAS risk compared with some other waste-derived soil amendments. However, PFAS can be introduced through contaminated feedstocks, including certain compostable food packaging (Khair Biek et al., 2025; Kim Lazcano et al., 2020). Compost-like outputs and anaerobic digestion (AD) digestate may present a higher PFAS risk where they are derived from more heterogeneous waste streams, although available evidence suggests that PFAS concentrations are generally low and highly dependent on feedstock composition.

The environmental significance of PFAS associated with materials spread to land depends on factors including the concentrations present in the material, application rates, soil characteristics, and the properties of individual PFAS. Short-chain PFAS are generally more mobile and may present a greater risk of migration to groundwater, whereas some longer-chain PFAS are more likely to remain associated with soils.

Understanding the occurrence, fate and transport of PFAS associated with land spreading remains an important evidence gap. Ongoing work, including investigations undertaken through CIP and wider Environment Agency research programmes, is improving understanding of PFAS in biosolids, sludge management practices and the potential environmental impacts associated with the application of waste-derived materials to land. The findings will help inform future regulatory approaches and risk management decisions in this area.

Further information on sewage sludge, biosolids and the Environment Agency's role in regulating their management, use and disposal can be found here.

5. PFAS monitoring and analytical approach

5.1 Enhancing and prioritising PFAS monitoring

This chapter provides an overview of the PFAS monitoring undertaken by the Environment Agency between 2016 and 2025. This includes surveillance, operational and investigatory monitoring in English groundwaters, rivers, lakes, estuarine and coastal waters, freshwater fish (roach, chub and trout), marine flat fish (dab, plaice and flounder), and shellfish (blue mussels).

Since publishing our previous report on sources, pathways and occurrence of PFAS five years ago (Environment Agency, 2021), we have significantly expanded our PFAS monitoring capability. Prior to 2021, fully quantitative monitoring was limited to PFOS and PFOA in surface waters, while monitoring of additional PFAS relied on semi-quantitative screening methods with higher detection limits, providing indicative rather than precise quantification. We have since developed and now use fully quantitative analytical methods for 68 PFAS in freshwater and groundwater and 45 PFAS in biota, primarily fish and mussels.

Fully quantitative analysis enables PFAS concentrations to be quantified and reported with greater accuracy and precision than semi-quantitative (target screening) approaches. As a result, with the exception of PFOS and PFOA, the fully quantitative data reported here are not directly comparable with the semi-quantitative screening data presented in our previous report (Environment Agency, 2021). We continue to collect semi-quantitative screening data for hundreds of chemicals, and these data are publicly available online. The suite of PFAS included in our fully quantitative analytical methods will continue to be reviewed and updated as new information and analytical capabilities become available.

Monitoring in estuarine and coastal waters remains primarily focused on PFOS and PFOA. We do not routinely monitor for PFAS in soils or air. In addition to the monitoring described above, other organisations also undertake PFAS monitoring, particularly in animal tissue. These include programmes covering offshore waters, offshore fish, cetaceans, otters and predatory birds, which are reported as part of the Defra H4 indicator.

All acronyms used in this report are defined in the main text, and abbreviations for individual PFAS are provided in the metadata tables in Appendix 4.

Our environmental monitoring is one of the most comprehensive PFAS monitoring programmes globally. The Environment Agency's National Laboratory Service contributes to international PFAS method development through collaborations with national and international partners, including PARC and the NORMAN Network. These partnerships support advances in PFAS monitoring and analysis, whilst also helping to improve the interpretation of PFAS data. The NORMAN network, for example, is an international coalition of reference laboratories, research centres, and related organisations that collaborate on the monitoring and assessment of emerging environmental substances.

The Environment Agency's analytical methods and associated quality control procedures for PFAS in water are summarised in Patel et al. (2026). Our current laboratory capability also includes pilot programmes for ultra-short-chain PFAS, including trifluoroacetic acid (TFA), and the Total Oxidisable Precursor (TOP) assay, which helps reveal PFAS precursor compounds that are not identified through routine targeted analysis**.** Results from these pilots are reported in Chapter 8. For clarity, while TFA is classed as a PFAS compound, TFA and other PFAS are referred to separately throughout this section of the report.

Given the thousands of individual PFAS, it is neither practical nor feasible to monitor every substance. Figure 6 illustrates how regulatory priorities and evidence gathered through monitoring programmes have led to focus on a small subset of PFAS compared with the many substances that meet the definition of a PFAS. PFAS are defined based on their chemical structure rather than their harmful properties, and not all substances that meet the definition are likely to be harmful.

The selection of PFAS for monitoring is evidence-based and targeted, focusing on substances considered to be of greatest relevance in England.

Criteria for inclusion in our monitoring programmes include substances that are:

- Highlighted as having hazardous properties and are restricted, proposed for restriction, or subject to monitoring under regulatory regimes.
- Registered for use in the UK under UK REACH at high tonnage (> 10 tonnes per year).
- Identified as a potential contaminant of concern through hazard and risk assessment.
- Potential for significant environmental exposure through source assessment and risk evaluation work.
- Associated with manufacturing sites in England with the potential for environmental exposure.
- Frequently reported in the scientific literature as significant environmental contaminants, particularly in studies with close geographic proximity to England.
- Identified in the environment through earlier semi-quantitative target-screening monitoring, based on presence/absence of detection in PFAS investigations and related programmes.

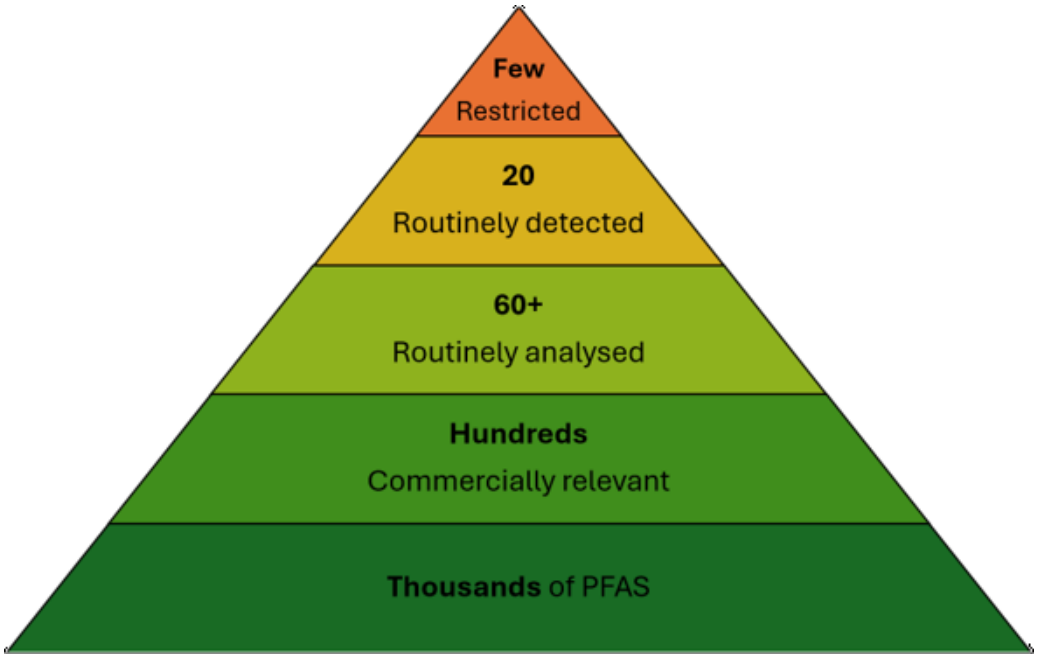


Figure 5: Conceptual illustration of the thousands of PFAS and the smaller subsets that have been the focus of research, monitoring and regulatory activity in the English environment.

5.2 Approach to the analysis of monitoring data

In Chapters 6 and 7 we present analyses of PFAS concentrations in water and biota samples. In Chapter 6 we examine long-term trends, seasonality and spatial variation in PFOS and PFOA concentrations. In Chapter 7 we examine concentration patterns across the full (60+) PFAS analytical suite. Although the data and objectives differ between chapters, both analyses use a consistent statistical analysis approach which we describe below and in more detail in Appendix 5.

Briefly, we fitted statistical models to describe the observed concentration data. Environmental concentration data are often highly variable, and estimates based on only a small number of observations are generally more uncertain than those based on larger datasets. We therefore split the data only where necessary, for example by analytical method and environmental compartment or biota type. We then used statistical models to describe the variation within each dataset. This variation included gradual changes over time, patterns repeated each year (seasonality), and systematic differences among monitoring sites, including uneven numbers of samples.

All reported summary statistics, including mean (average) concentrations, percentage changes and ratios, were derived from the fitted models and are accompanied by estimates of their associated uncertainty. In short, we accounted for the inherent patterns in the data, while maintaining the sample size, allowing the information in each dataset to contribute appropriately to each reported statistic. Compared to further splitting the data

and calculating simple averages, the overall result is an improved description of underlying PFAS concentrations and their trends and patterns.

Many of our PFAS concentration measurements are reported as being below the Minimum Reporting Value (MRV) of the analytical method. In these cases, we know that the concentration is between zero and the MRV, but we do not know its exact value. These observations are sometimes referred to as non-detects; they indicate a failure to detect, not non-occurrence. Our statistical approach specifically accounted for this additional uncertainty, allowing each below-MRV measurement to contribute to the analysis according to the range of values it could plausibly take, rather than treating it as a single assumed value. Being integrated into the data analysis approach described above, this makes best use of the available data. It ensures our reported PFAS concentration statistics are not biased by the presence of non-detects and provides realistic estimates of their uncertainty.

We analysed national-scale time trends in PFOS and PFOA concentrations between 2016 and 2025, while accounting for variation among sampling locations and, for water, seasonal patterns (see Chapter 6). The starting year and availability of data differs across environmental compartments.

Data for the full (60+) PFAS suite are available for a shorter time period (2021-2025) across freshwater, groundwater and biota (see Chapter 7). For the purposes of this report, the term "full PFAS suite" refers to the EA's analytical PFAS suite in use at the time of data analysis and report preparation. Again, the monitoring record begins in different years for different environmental compartments. For each compartment with data, we analysed the data for multiple PFAS together. We accounted for variation in concentration among sites, variation in concentration among the individual PFAS, and for water, seasonal variation in concentration. We excluded very rarely detected PFAS and sparsely sampled sites. The approach takes advantage of the strong co-occurrence among PFAS to optimise the precision of estimated mean concentrations along a spectrum from the more commonly detected to the rarer PFAS. This approach is less suited for the estimation of trends for individual PFAS. Given the time period and numbers of samples for these data, focussing on mean concentrations rather than trends was considered an acceptable compromise. Further details of the methods applied for data analysis are provided in Appendix 5.

We evaluated the fitted models using residual diagnostics that account for the presence of non-detects in the observed concentration data, with particular attention given to residual distributions and temporal patterns. Further details are provided in Appendix 5.

We then used the fitted models to estimate mean concentrations over time (where appropriate), mean concentrations across sites, and mean concentration across the months of the year. In each case, we report 95% confidence intervals for the mean; these indicate the range of mean values that are reasonably consistent with the observed data and fitted model. The width of the confidence intervals is influenced by both the size of the dataset and the amount of unexplained variation in the data: larger datasets and lower residual variation generally result in narrower intervals (more precise estimates).

Our results show that spatial variation between sites is the largest source of overall variability across all environmental compartments. For water data, seasonal effects are the next largest source of variability, followed by long-term trends.

For convenience, we have included our internal reference numbers (WIMS codes) in the metadata table in Appendix 4. You can use these WIMS codes to access our monitoring data online via the Water Quality Explorer tool.

5.3 Water monitoring networks

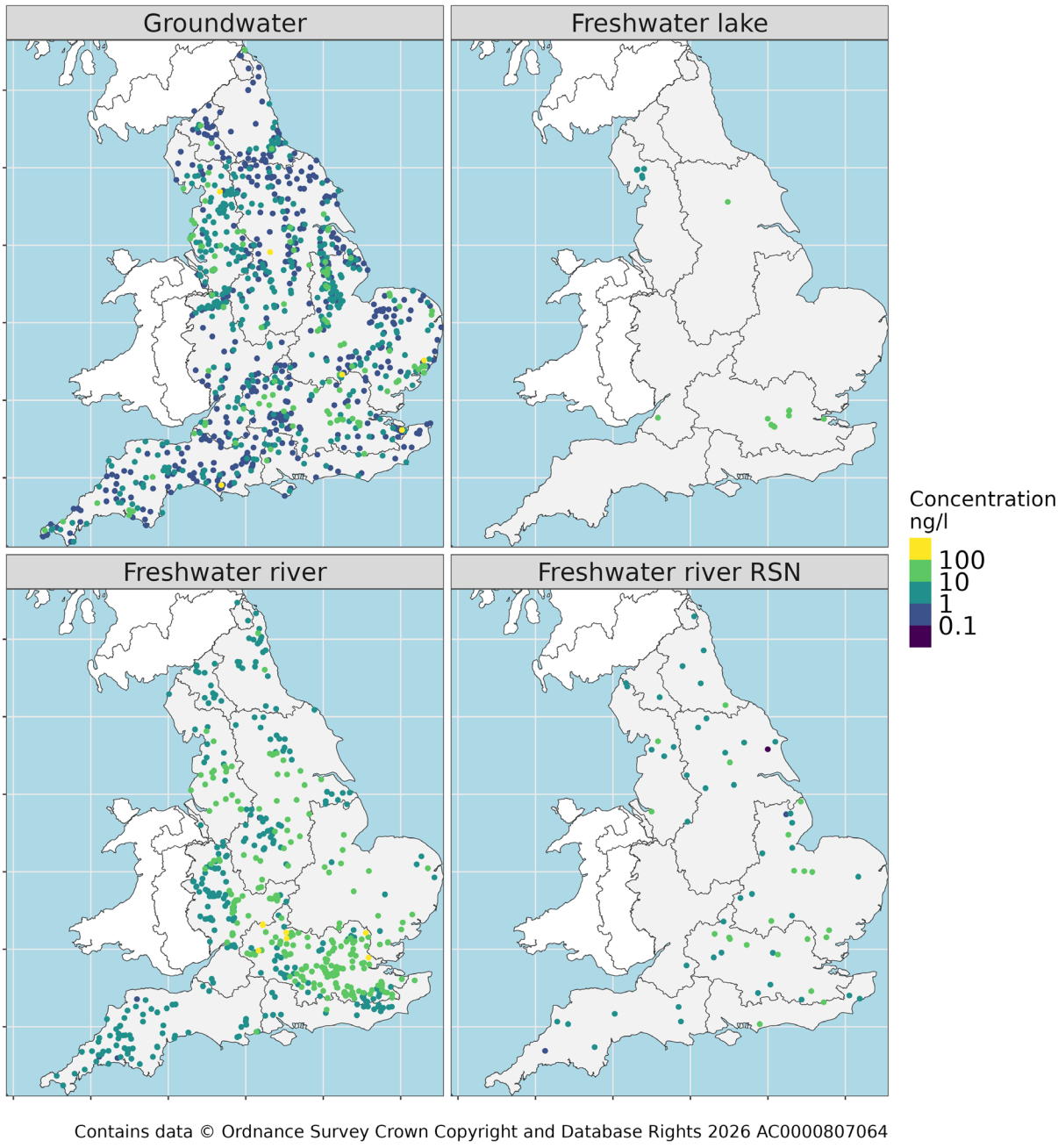
Figure 7 shows sampling locations and total PFAS concentrations at each freshwater and groundwater monitoring site, using the full (60+) PFAS analytical suite. Total concentrations are based on the sum of individual concentrations derived from the model-based reporting framework outlined earlier in Section 5.2. Although total PFAS concentrations are not typically used for environmental risk assessment, they are presented here to allow comparison across all sites for a common year (2025). The maps are provided to show spatial patterns and the broad distribution and density of sampling sites in the monitoring programmes. They illustrate variation in concentrations geographically, with PFAS concentrations typically higher in or near urban areas. Equivalent maps for biota are provided in Appendix 6.

For freshwater, results are presented separately for lakes, the River Surveillance Network (RSN), and a combination of other river monitoring sites. In this report, "freshwater (river) RSN" refers to sites within the RSN, whereas "freshwater (river)" refers to all other river monitoring sites. These sites were sourced from several different monitoring programmes including compliance monitoring for priority substances, designated inland bathing water sites, Harmonised Monitoring / OSPAR Riverine Inputs and Direct Discharges (RID) monitoring and locally tailored programmes. In combination, this combined set of monitoring sites is likely to reflect higher pollutant concentrations than RSN sites for two reasons. First, because most of the sites sampled for investigative work are likely to be those with higher concentrations. Second, because the design of the Harmonised Monitoring / OSPAR programmes focuses more on downstream locations with larger catchments, which integrate the effects of upstream point-source discharges.

The River Surveillance Network (RSN) is a statistically designed monitoring network consisting of 1600 river monitoring sites across England that are used to monitor a wide range of water quality parameters, including PFAS. Because of this design, the RSN can be considered broadly representative in spatial terms of river conditions across England, although it excludes very small headwater streams. Its primary purpose is to provide unbiased estimates of river conditions across the river network.

Groundwater data are collected from the Environment Agency's PFAS surveillance programme, which builds on the existing Groundwater Quality Monitoring Network (GWQMN). The network supports statutory groundwater monitoring requirements, including those under the Water Framework Directive (WFD), with monitoring locations selected using risk-based principles to characterise groundwater bodies and assess

environmental pressures. The surveillance programme is complemented by targeted investigations where local evidence or identified risks warrant additional monitoring.



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Figure 6: Map showing the locations of PFAS monitoring sites across the various EA water sampling networks in England. Additional sites monitored only for PFOS/ PFOA are not shown. *The data presented show total PFAS concentrations (sum of all measured PFAS), expressed in ng/L and shown as model-based estimates for 2025. Regions shown correspond to River Basin Districts.*

5.4 Interpretation of environmental data for PFAS

Environmental quality standards (EQS) are an important tool in water quality legislation in England. They represent the concentration (threshold) of a substance in water above which an adverse effect on aquatic life, and in some cases human health, may occur.

Environmental thresholds such as EQS and predicted no-effect concentrations (PNECs) are useful for contextual interpretation of monitoring data and assessing the potential significance for the environment.

In the UK, at the time of writing, statutory Environmental Quality Standards (EQS) in water are only available for one PFAS - PFOS listed under The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015. For PFOS, both a biota-based EQS (9.1 µg/kg) and annual average water-column EQSs are available, with values of 0.65 ng/L for inland surface waters and 0.13 ng/L for estuarine and coastal waters. Due to its strong bioaccumulative properties, persistence and toxicity, compliance is assessed primarily using the biota-based EQS.

Where EQS values for PFOS are used in this report, they are presented for contextual purposes only, to assist interpretation of broad-scale monitoring trends in water. Our analysis presented here should not be interpreted as a compliance assessment, which is outside the scope of this report.

In the absence of EQSs for other PFAS, thresholds are needed by the Environment Agency for assessing and managing the environmental risks from PFAS in rivers, lakes, estuaries and coastal waters. However, deriving thresholds for PFAS is challenging for several reasons: (i) there are many PFAS, and they are chemically diverse and (ii) sufficient reliable and relevant (eco)toxicity data are only available for a few PFAS. To address this evidence gap, the Environment Agency has undertaken research on threshold derivation for PFAS. This includes an examination of approaches taken to standard derivation internationally by other regulators as well as an assessment of available toxicology data and approaches to standard setting for mixtures and groups of PFAS. Further information is provided in Developing Thresholds for Managing PFAS in the Water Environment (Environment Agency, 2026b) and Review of Approaches for Determining PFAS Thresholds in Water (Environment Agency, 2026e).

6. Temporal trends and spatial variation in PFOS and PFOA concentrations

PFOS and PFOA are the only PFAS in our monitoring datasets with sufficiently long and spatially extensive time series data to support robust assessment of long-term temporal trends in environmental concentration. Several other PFAS, including PFHxS, PFNA and other long-chain PFCAs are included in our current monitoring programmes, and some are of regulatory significance, for example through their inclusion under the Stockholm Convention and associated POPs regulation. However, our monitoring records for these substances are shorter and less spatially extensive, meaning they are not yet suitable for long-term trend analysis.

PFOS and PFOA trends in surface water are explored in Section 6.1, with trends in biota presented in Section 6.2. For surface water, seasonal patterns are also explored where data availability allows. Data for the full (60+) PFAS suite are examined in Chapter 7.

6.1 PFOS & PFOA in surface water

6.1.1 Temporal trends in PFOS and PFOA in surface water

Overview

- Extensive river monitoring provides evidence of sustained long-term reductions in both PFOS and PFOA over the past decade.
- PFOA shows more consistent and greater declines than PFOS across most surface water types.
- In estuaries, PFOA concentrations show a sustained downward trend, while PFOS concentrations have remained broadly stable over the monitoring period, with fluctuations in recent years.
- In coastal waters and lakes, uncertainty is greater, reflecting smaller datasets and shorter monitoring periods.

Figure 8 shows long-term trends in PFOS and PFOA concentrations in surface waters from 2016 to 2025. Table 1 summarises these trends quantitatively, including the estimated overall percentage change since monitoring began and the mean of the annual percentage changes. Box 1 provides guidance on interpretation.

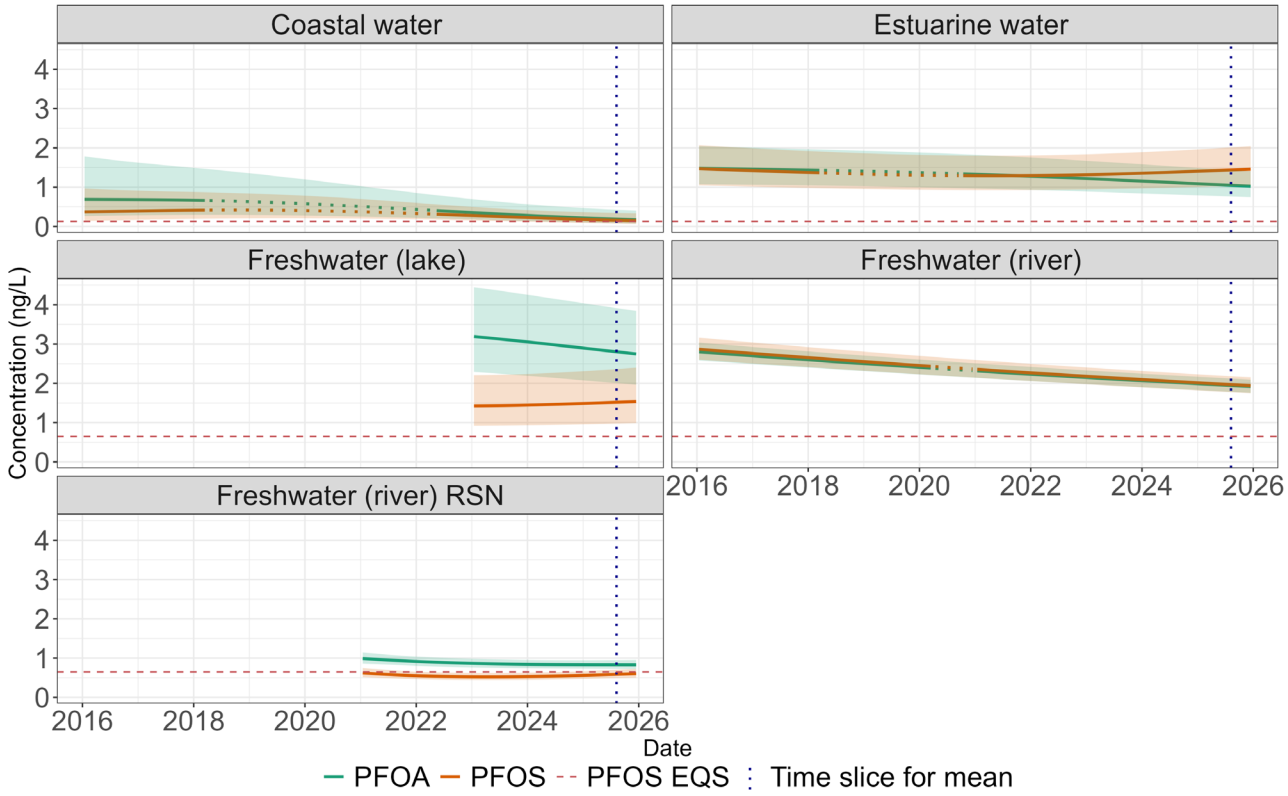


Figure 7: Long-term trends in PFOS (orange) and PFOA (green) concentrations across England at Environment Agency water-monitoring sites. *Coloured lines show model-based estimates of geometric mean concentrations (ng/L), with shaded areas representing 95% confidence intervals for the mean. Dotted sections of the trend indicate periods where very little (fewer than 20 samples per year) or no monitoring data were available. Horizontal dashed lines represent the annual average Environmental Quality Standard (EQS) for PFOS in freshwater (rivers and lakes; 0.65 ng/L) and estuarine and coastal waters (0.13 ng/L). Vertical dotted lines represent the time slice (mid-2025) used to derive mean concentrations for subsequent water analyses in this report. Seasonal patterns and site-specific effects are not displayed in this figure.*

Note: The PFOS EQS is shown for contextual purposes only. The aggregated monitoring data presented in this report are intended to illustrate broad-scale environmental trends and should not be interpreted as a site-specific compliance assessment. Formal water body status assessment using EQS is undertaken separately through WFD classification and reporting processes (Environment Agency, 2026e).

Box 1: How to interpret percentage change values in Table 1

- The percentage shows the estimated change over time.
- The range in brackets shows the likely range of this change.
- If the range is entirely negative or entirely positive, this indicates a clear decrease or increase.
- If the range includes both negative and positive values, there is no consistent decreasing or increasing trend.

Table 1: ***Percentage change (whole period and annual) in PFOS and PFOA concentrations in surface water compartments with 95% confidence intervals in brackets. Whole period changes are from the start year indicated. Values are reported to two significant figures.***

Compartment	Determinand	Start year	Percentage change over whole period (%)	Mean annual percentage change over period (%)
Freshwater (river)	PFOS	2016	**-30** (-34, -26)	**-3.9** (-4.5, -3.2)
Freshwater (river)	PFOA	2016	**-29** (-33, -25)	**-3.7** (-4.3, -3.2)
Freshwater (river) RSN	PFOS	2021	**-9.6** (-18, -1.8)	**-2.3** (-4.3, -0.27)
Freshwater (river) RSN	PFOA	2021	**-16** (-24, -6.8)	**-4.2** (-6.5, -1.7)
Estuarine water	PFOS	2016	**-4.2** (-15, 6.9)	**-0.45** (-1.7, 0.76)
Estuarine water	PFOA	2016	**-27** (-33, -20)	**-3.4** (-4.4, -2.4)
Coastal water	PFOS	2016	**-41** (-85, 66)	**-6.6** (-18, 6.7)
Coastal water	PFOA	2016	**-63** (-89, -8.2)	**-11** (-22, -0.45)
Freshwater (lake)	PFOS	2023	**4.6** (-3.1, 13)	**2.2** (-1.6, 6.4)
Freshwater (lake)	PFOA	2023	**-9.3** (-14, -5.1)	**-4.8** (-7.2, -2.6)

Rivers

In freshwater river sites (excluding the RSN), where the longest and most spatially extensive datasets are available, we can see clear and consistent declines in PFOS and PFOA concentrations over time.

Between 2016 and 2025, mean concentrations of PFOS and PFOA in rivers in England declined by approximately 30% and 29% respectively. Mean annual reductions were 3.9% for PFOS and 3.7% for PFOA. The relatively narrow confidence intervals reflect the large number of sites and long duration of monitoring, providing strong evidence of sustained reductions.

In rivers sampled as part of the RSN, for which monitoring began in 2021, both PFOS and PFOA show declining trends over this period. Between 2021 and 2025, mean concentrations across the RSN declined by around 9.6% for PFOS and 16% for PFOA, and mean annual reductions were 2.5% and 4.2%, respectively. As this analysis is based on a shorter monitoring period and a smaller amount of data, the magnitudes of these trends are more uncertain, as reflected in the wider confidence intervals. Despite this, the direction of change is consistently downward across the monitoring period.

As discussed in Section 5.3, these river trend results should be interpreted in the context of the differing design, objectives and coverage of the underlying monitoring networks.

Estuarine waters

In estuaries, PFOA concentrations show a consistent decline since 2016, decreasing by approximately 27% overall and a mean annual reduction of around 3.4%. Although the monitoring record includes some gaps (Figure 8), introducing some uncertainty, the available data consistently indicate a downward trend in PFOA concentrations.

In contrast, PFOS concentrations in estuaries have remained broadly stable over the monitoring period. Although estimated concentrations were higher in some recent years, there is no evidence of a sustained increase or decrease. The raw data indicate short-term fluctuations, with concentrations in 2024 notably higher than in 2023 and 2025. These fluctuations are largely smoothed by the long-term trend models. One possible explanation is the unusually wet conditions in 2024, which may have increased remobilisation of PFAS from sediments and altered estuarine salinity patterns. Additional data from future years will help determine whether these fluctuations are primarily driven by hydrological conditions or other environmental factors.

Coastal waters

In coastal waters, PFOS concentrations have remained broadly stable over the monitoring period, although confidence in the estimated trend is limited due to the small number of samples and monitoring sites (Table 1). PFOA concentrations show a more consistent downward pattern (Table 1), although confidence in the observed trend is lower than for rivers due to the limited and discontinuous nature of the coastal monitoring dataset, including gaps in sampling (Figure 8) and a smaller number of monitoring sites (Table 2).

Lakes

Monitoring for PFOS and PFOA in lakes began in 2023. Compared to our data on rivers, the limited number of sites and samples, and the short time series, mean that there is less certainty in our estimates. PFOS concentrations in monitored lakes have remained broadly stable over the monitoring period. However, PFOA concentrations decreased by around 9% between 2023-2025 (Table 1). These results should be considered indicative only, given the limited data available.

Wider context

Taken together, the trends observed here are broadly consistent with observations reported in wider literature across surface waters and wildlife, where PFOS concentrations generally increased until the 2000s before stabilising and subsequently declining in response to regulatory controls and changes in use (Badry et al., 2025; Kesic et al., 2023; Muir and Miaz, 2021; Pereira et al., 2021; Rupp et al., 2025; Sun et al., 2026). However, trends can vary between species, environmental media and location. Continued releases from legacy products and legacy contamination, ongoing use of precursor compounds, and remobilisation of contaminated sediments may also influence observed concentrations and contribute to temporal lags between regulatory restrictions and measurable environmental improvements. For example, the H4 Chemicals Indicator for England showed declining PFOS concentrations in harbour porpoise, no observed change in otters, and uncertain trends in buzzards and red foxes (Environment Agency, 2024).

6.1.2 Mean PFOS and PFOA concentrations in surface water

Table 2 presents a snapshot of England-wide mean PFOS and PFOA concentrations in surface waters (mid-2025), derived from our long-term trend analyses. It also reports the number of sites and samples contributing to each estimate across the full analysis period, highlighting the much larger datasets for rivers compared to the more limited data for lakes, estuarine and coastal waters.

A consistent pattern is evident across waterbody types. The highest mean concentrations are observed in rivers and lakes, with concentrations lowest in coastal waters. Across all waterbody types, mean concentrations of PFOS and PFOA are in the low ng/L to low µg/L range.

Lakes

Freshwater lakes show the highest mean concentrations for PFOA (2.8 ng/L; Table 2), with PFOS concentrations also relatively elevated (1.5 ng/L), although lower than in river sites.

Rivers

Rivers (excluding the RSN sites) also show relatively high mean concentrations for both PFOS and PFOA, with similar levels of each (both 2.0 ng/L). In contrast, mean concentrations at RSN sites are approximately two- to three-fold lower. This likely reflects differences in network design. The RSN is a probability-based, spatially balanced survey network, including many lower-order streams, whereas the other river monitoring sites are

predominantly targeted towards statutory monitoring requirements, known pressures, investigations or higher-risk locations.

Estuarine and coastal waters

Estuarine waters generally show lower mean concentrations than rivers and lakes, with PFOS (1.4 ng/L) slightly higher than PFOA (1.1 ng/L). Coastal waters have the lowest concentrations of all water types for both substances (≤ 0.2 ng/L), although these estimates are based on more limited data and are therefore more uncertain.

Table 2: ***Model-based estimates of geometric mean concentration for PFOS and PFOA in English surface water in the middle of 2025 with 95% confidence intervals (shown in brackets), numbers of samples and sites used in each analysis. Values are reported to two significant figures.***

Water category	Determinand	Mean Concentration	Unit	Number of samples	Number of sites
Freshwater (river)	PFOA	**2.0** (1.8, 2.1)	ng/L	9,789	642
Freshwater (river)	PFOS	**2.0** (1.8, 2.2)	ng/L	9,899	661
Freshwater (river) RSN	PFOA	**0.83** (0.73, 0.95)	ng/L	3,817	268
Freshwater (river) RSN	PFOS	**0.58** (0.49, 0.69)	ng/L	3,775	268
Freshwater (lake)	PFOA	**2.8** (2.0, 3.9)	ng/L	1,062	75
Freshwater (lake)	PFOS	**1.5** (0.97, 2.3)	ng/L	882	76
Estuarine water	PFOA	**1.1** (0.77, 1.5)	ng/L	2,808	68
Estuarine water	PFOS	**1.4** (1.0, 2.0)	ng/L	2,812	68
Coastal water	PFOA	**0.19** (0.087, 0.44)	ng/L	132	10
Coastal water	PFOS	**0.16** (0.076, 0.35)	ng/L	127	10

6.1.3 Spatial variability in PFOS and PFOA concentrations in surface water

There is substantial spatial variation in PFOS and PFOA concentrations across all surface water compartments. This variation is summarised in Table 3 using the ratio of concentrations at the 90th and 10th percentile sites (90:10 ratio). Higher ratios indicate greater differences in concentration between monitoring sites. For example, in estuarine waters, a ratio of 16 for PFOS means that concentrations at sites towards the higher end of the concentration range (90th percentile) are approximately sixteen times greater than those at sites towards the lower end of the concentration range (10th percentile). Estimates are based on site-specific modelled concentrations at a common reference point (mid-2025), with uncertainty expressed as 95% confidence intervals (see Appendix 5 for details). Differences in estimated spatial variability should be interpreted in the context of differences in network size, which range from approximately 10 coastal sites to over 660 river sites (Table 2).

Lakes

Variation between sampling locations is greatest in lakes, particularly for PFOS. However, this may in part reflect the smaller monitoring network (76 sites and 882 samples for PFOS) compared to rivers (661 sites and 9,899 samples for PFOS), and therefore greater uncertainty in the modelled concentration estimates.

Rivers

Variation is also high across river monitoring sites, where PFOS concentrations vary more among sites than PFOA. Site to site variation is greater for RSN sites than non-RSN river sites; this is likely due to the higher proportion of RSN sites with very low PFOS and PFOA concentrations.

Estuarine and Coastal Waters

Estuarine waters show similar levels of variation for both substances, while coastal waters show the least variation overall, although estimates are more uncertain due to the relatively small dataset available (68 sites in estuaries compared with 10 coastal sites).

This high spatial variation is a consequence of the spatial variation in PFOS and PFOA sources, as well as the processes by which they are delivered to and transported within water. The underlying distributions of site-specific concentrations are illustrated in Figure A4 in Appendix 6 to provide further visual context for these patterns.

Table 3: ***Mean ratio between 90th and 10th percentile sites for PFOS and PFOA concentrations in water in different environmental compartments. Numbers in brackets indicate 95% confidence intervals.***

Media / Compartment	Determinand	90th:10th percentile site ratio
Coastal water	PFOS	5 (3, 9)
Coastal water	PFOA	8 (5, 13)
Estuarine water	PFOS	16 (13, 21)
Estuarine water	PFOA	18 (15, 21)
Freshwater (lake)	PFOS	85 (65, 110)
Freshwater (lake)	PFOA	42 (34, 50)
Freshwater (river)	PFOS	25 (23, 27)
Freshwater (river)	PFOA	12 (11, 13)
Freshwater (river) RSN	PFOS	46 (41, 52)
Freshwater (river) RSN	PFOA	14 (12, 16)

6.1.4 Seasonality in PFOS and PFOA concentrations – surface water

Figure 9 shows that both PFOS and PFOA exhibit seasonal variation in mean concentrations, with the pattern and magnitude of seasonality differing between rivers, lakes, estuaries and coastal waters.

Rivers

The strongest seasonal patterns were observed in rivers, where concentrations of both PFOS and PFOA were generally highest during summer and lowest during winter. This pattern is consistent with reduced dilution during low-flow conditions and may indicate the influence of relatively continuous sources.

Seasonal patterns appear more pronounced in the non-RSN dataset compared to the RSN sites. As these sites are more frequently associated with targeted monitoring of higher-risk or operationally important locations, this observation may suggest a greater influence of localised sources. However, the monitoring data presented here do not allow attribution of the seasonal signal to specific source types or processes. Other factors, including groundwater inputs, agricultural runoff, fluctuations in industrial discharge, stormwater discharges, diffuse sources, sediment-water interactions and wider catchment processes, may also contribute to the observed patterns.

Estuarine Waters

Estuarine waters also show evidence of seasonal variation, although the magnitude of these changes is smaller than in rivers. PFOS and PFOA exhibit different seasonal patterns. PFOA concentrations decrease from late winter through spring and early summer, before increasing from summer through autumn and into winter. In contrast, PFOS concentrations remain relatively stable, with only a slight decrease into summer. Overall, both substances tend to show slightly higher concentrations in winter. The wide confidence intervals indicate high uncertainty in these patterns (Figure 9), reflecting the smaller number of sites and samples and gaps in the monitoring record.

Lakes and Coastal Waters

Lakes and coastal waters exhibit very limited seasonality, with concentrations remaining relatively stable throughout the year and only slight summer increases observed in lakes.

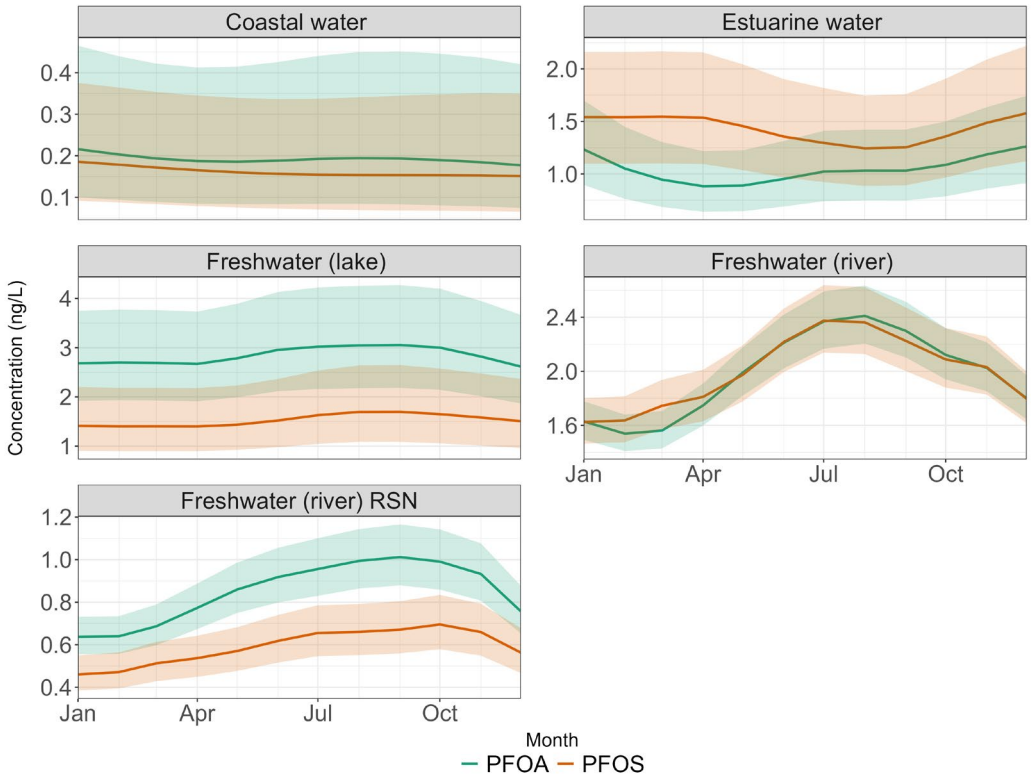


Figure 8: Seasonality patterns in PFOS (orange) and PFOA (green) concentrations across England at Environment Agency water-monitoring sites. *Coloured lines show model-based estimated seasonal cycles (geometric mean concentrations, ng/L), shaded areas represent 95% confidence intervals. Long-term temporal trend is excluded.*

These data are quantified as the peak to trough ratio in Figure A6 of Appendix 6.

Differing seasonal patterns between waterbody types highlight the difficulty of generalising trends, as seasonal drivers are likely to be site-specific and influenced by a combination of physical, chemical, and biological processes.

6.1.5 Summary of observed PFOS and PFOA trends in surface water

The preceding water monitoring data sections demonstrate clear long-term declines in PFOS and PFOA concentrations in rivers, where the longest and most spatially extensive monitoring records are available. Similar declines are less consistently observed in lakes and estuarine and coastal waters, where concentrations are generally stable or trends are more uncertain because of shorter monitoring records and smaller datasets. Alongside these changes, pronounced spatial variability between sites and distinct seasonal patterns are also evident, particularly in rivers.

While these results indicate encouraging downward trends, they also highlight important differences in the behaviour of PFAS between waterbody types and between PFOS and PFOA. The following section synthesises these findings to explore the processes that may underpin the observed patterns.

Across freshwater systems, monitoring data show continued declines in PFOS and PFOA concentrations in rivers, while trends in lakes are less clear, particularly for PFOS. Lakes generally show higher concentrations than rivers. However, the lake dataset is based on fewer sites, fewer samples and a shorter monitoring period than the river datasets, leading to greater uncertainty in the estimates. As a result, the reasons for the observed differences remain uncertain. While factors such as hydrology, sediment interactions and catchment characteristics may contribute, the current analysis does not allow these mechanisms to be evaluated directly. Studies show that PFAS, particularly PFOS, can bind to sediments in lake environments, potentially contributing to their accumulation and long-term retention in lakes (Loken et al., 2025; Oliver et al., 2020; Reif et al., 2022). The extent of this behaviour can vary depending on environmental conditions, including salinity, temperature, pH, and total organic carbon (ITRC, 2023).

Furthermore, the present analysis does not consider site-specific characteristics such as urban influence, wastewater inputs, historical contamination or other local pressures. Additional investigation of the lake monitoring network and its catchment characteristics would be required to determine whether the observed differences reflect broadly applicable lake processes or the characteristics of the specific lakes included within the monitoring programme.

In estuarine waters, there is no clear long-term trend in PFOS concentrations. PFOS is more likely to bind to particles and sediments than PFOA due to differences in its chemical structure (Fatima et al., 2025; Liu and Li, 2024). As salinity increases, PFOS tends to associate more with particles and sediments, while PFOA remains in the water (Navarro et al., 2022). Changes in freshwater input, salinity and sediment movement can therefore influence how PFOS is distributed between water and sediments, with disturbance of sediments or shifts in conditions leading to fluctuations in concentrations. PFOA, being less strongly associated with sediments, is less affected by these processes and tends to follow a smoother downward trend in concentration over time. Short-term tidal variability may also influence observed concentrations, although Environment Agency sampling protocols aim to minimise this effect by targeting ebb tides (Shi et al., 2025).

In coastal waters, trends in concentrations of PFOS and PFOA over time are uncertain due to limited data availability. PFOA shows a more consistent downward pattern in concentrations, while PFOS concentrations have remained broadly stable. These less pronounced patterns likely reflect extensive mixing, dilution and tidal exchange in coastal environments.

Seasonality also influences PFAS concentrations, although interpretation is complicated by multiple interacting factors. The strongest seasonal patterns are observed in rivers, where concentrations are generally highest during summer and lowest during winter. This pattern is consistent with reduced dilution during low-flow conditions and may indicate the influence of relatively continuous sources. Seasonal patterns also appear more pronounced in samples collected from the targeted river monitoring network than the RSN. As the targeted network contains a greater proportion of sites selected for operational, investigative and higher-risk monitoring purposes, this observation may suggest a stronger influence of localised source pressures. However, the current data do not allow attribution of the seasonal signal to specific source types or processes, and factors such as groundwater inputs, diffuse sources, sediment-water interactions and wider catchment processes may also contribute.

Across most surface-water environments, PFOA generally exhibits greater seasonal variability than PFOS. Estuaries show weaker seasonal patterns, consistent with the influence of salinity, tidal mixing and freshwater influences, while data for lakes and coastal waters display little to no consistent seasonality.

Taken together, the analysis highlights several broad patterns in the temporal and spatial behaviour of PFOS and PFOA across the different surface-water environments:

- In rivers, PFOS and PFOA concentrations show seasonal variation, consistent with the influence of hydrological seasonality and catchment processes.
- Lakes generally show higher PFOS and PFOA concentrations than rivers, although the reasons for this difference remain uncertain and require further investigation.
- Estuaries show more variable PFOS behaviour, which may reflect the influence of salinity, sediment interactions, tidal mixing and freshwater inputs.
- Coastal waters show little consistent evidence of seasonality within the available data.
- Across waterbody types, PFOA generally shows smoother downward trends over time than PFOS, while PFOS appears more susceptible to variability associated with local environmental processes.
- Spatial variability between sites indicates that local factors, including site-specific sources and environmental conditions, also influence observed concentrations.

Finally, it is important to note that here we are reporting concentrations (in units of mass/volume) rather than loads (in units of mass/time). To further understand the drivers of seasonal variability in rivers, elevated PFOS concentrations in lakes, and episodic PFOS increases in estuarine waters, additional targeted investigation could be beneficial, including:

- Investigation of concentration-flow relationships and seasonal PFAS loads across the RSN and targeted river monitoring networks to determine whether observed seasonal patterns primarily reflect dilution of continuous inputs, mobilisation of diffuse catchment sources, or other hydrological processes;
- Estimation of PFAS loads within and across catchments and apportionment between sources;
- Sediment-water flux assessments, especially in lakes and estuaries;
- Further breakdown of estuarine trends to understand episodic increases and their underlying drivers.

Together, these approaches would help distinguish remobilisation from genuine changes in emissions and strengthen interpretation of PFOS and PFOA trends across England's surface waters.

6.2 PFOS & PFOA in biota

6.2.1 Temporal trends in PFOS and PFOA in biota

In addition to water column sampling, we collect fish and mussel (biota) samples to analyse substances with the potential to bioaccumulate. Monitoring PFAS in biota presents several challenges compared to water sampling, including animal welfare considerations, species availability and site accessibility. As a result, biota monitoring is carried out at fewer locations than water monitoring, and datasets are more limited. Nevertheless, available long-term data allow assessment of temporal trends for PFOS, and to a more limited extent PFOA, in freshwater, estuarine and coastal fish.

Figure 10 illustrates the observed long-term trends in PFOS and PFOA concentrations in freshwater, estuarine and coastal fish (whole-body), with corresponding percentage changes and mean annual rates of change summarised in Table 4 (Box 1 provides guidance on interpretation of percentage change values).

Freshwater fish

PFOS concentrations in the sampled freshwater fish declined substantially between 2016–2024, decreasing by an estimated 49% overall and by around 7% per year. This pattern is consistent with sustained declines observed in rivers (see Section 6.1.1), suggesting that reductions in environmental PFOS concentrations are also reflected in biota. Although PFOA has been measured in freshwater fish, results are not shown due to a high proportion of values being below the MRV (1 µg/kg prior to 2022 and 0.2 µg/kg thereafter), preventing meaningful trend analysis.

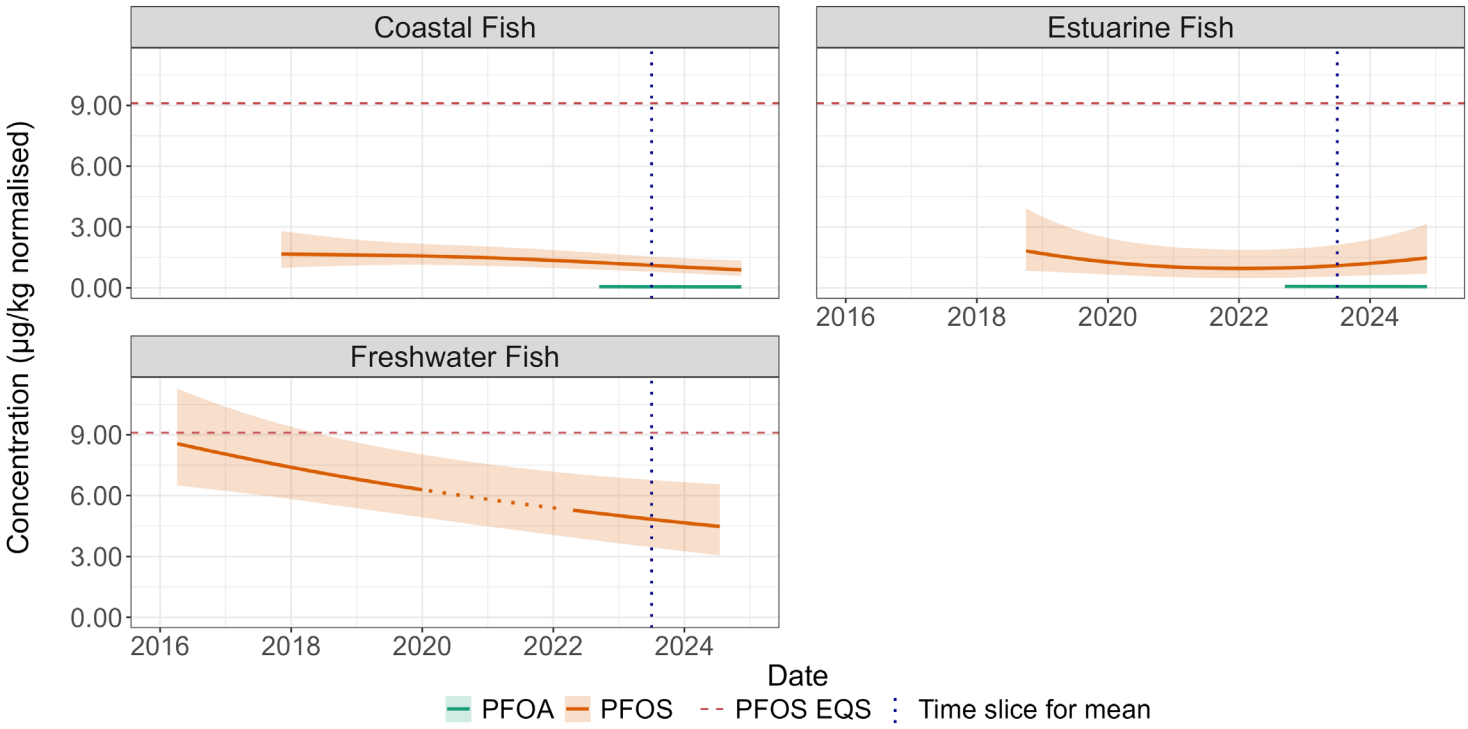


Figure 9: Temporal Trends in PFOS and PFOA concentrations in biota (fish) across England at Environment Agency monitoring sites. ***Coloured lines show model-based estimated geometric mean concentrations (µg/kg) for PFOS (orange) and PFOA (green), with shaded areas representing 95% confidence intervals. Dotted sections of the trend indicate periods where no or insufficient (fewer than 20 samples per year) monitoring data were available. Horizontal dashed lines represent the Environmental Quality Standard (EQS) for PFOS in biota (9.1 µg/kg). Vertical dotted lines represent the time slice used to derive mean concentrations for subsequent biota analyses.***

Note: The PFOS EQS is shown for contextual purposes only. The aggregated monitoring data presented in this report are intended to illustrate broad-scale environmental trends and should not be interpreted as a site-specific compliance assessment Formal water body status assessment using EQS is undertaken separately through WFD classification and reporting processes (Environment Agency, 2026e).

Table 4: ***Percentage change (whole period and annual) in PFOS and PFOA concentrations in biota with 95% confidence intervals in brackets. Whole period changes are from the start year indicated. Values are reported to two significant figures.***

Media	Determinand	Start year	Percentage change over whole period (%)	Mean annual percentage change over period (%)
Coastal Fish	PFOS	2018	**-45** (-71, -3.4)	**-8.0** (-16, 1.0)
Coastal Fish	PFOA	2022	**-19** (-64, 65)	**-8.3** (-29, 18)
Estuarine Fish	PFOS	2018	**-27** (-69, 53)	**-2.8** (-13, 8.9)
Estuarine Fish	PFOA	2022	**-13** (-52, 48)	**-5.5** (-22, 14)
Freshwater Fish	PFOS	2016	**-49** (-69, -23)	**-7.3** (-12, -2.6)

Estuarine & Coastal Fish

In estuarine fish sampled, PFOS concentrations have remained broadly stable since 2018. Although concentrations are higher in some years than others, there is no evidence of a sustained increase or decrease over the monitoring period. Recent increases towards the end of the monitoring record coincide with periods of elevated PFOS concentrations in estuarine waters (Section 6.1.1), suggesting that short-term fluctuations in fish concentrations may reflect changes in environmental exposure.

In contrast, PFOS concentrations in coastal fish sampled show a clearer decline over the monitoring period, with an estimated reduction of around 45% between 2018 and 2024 (approximately 8% per year).

Across estuarine and coastal fish datasets, PFOA concentrations remain low and stable and are consistently lower than PFOS. Wide confidence intervals for percentage change estimates reflect the shorter monitoring period and number of samples.

Summary

Overall, PFOS shows a clear and consistent decline in concentration in freshwater fish, with similar but less certain patterns in coastal fish. No sustained increase or decrease in concentration is evident in estuarine fish. Mean PFOS concentrations in freshwater, estuarine and coastal fish were also below the PFOS biota EQS during the assessment period (Figure 10). These findings provide an additional line of evidence that environmental levels of PFOS are declining, although PFOS concentration in surface waters remain a concern (Figure 8).

PFOA concentrations are generally low. In freshwater fish, concentrations are largely below the MRV and no trend can be determined, while concentrations in estuarine and coastal fish remain low and broadly stable over the monitoring period.

6.2.2 Mean PFOS and PFOA concentrations in biota

Table 5 summarises England-wide mean concentrations of PFOS and PFOA in fish for 2023, derived from the long-term trend analyses. The table also reports the total number of samples and monitoring sites contributing to each estimate across the full analysis period, with the greatest coverage for freshwater fish and more limited datasets for estuarine and coastal fish.

Mean PFOS and PFOA concentrations in fish are markedly higher than concentrations measured in the water column, reflecting the tendency of PFAS to accumulate in fish over time. In freshwater, estuarine and coastal fish species sampled, PFOS occurs at considerably higher concentrations than PFOA, consistent with well-established differences in bioaccumulation behaviour.

The highest concentrations of PFOS are observed in freshwater fish, with a mean concentration of 4.8 µg/kg, around four times higher than in coastal or estuarine fish. This difference likely reflects a combination of factors, including differences in exposure pathways and environmental fate of PFAS between freshwater and marine systems. For long-chain PFAS such as PFOS, localised sediment-associated contamination may contribute to elevated exposure in some freshwater environments. Freshwater fish may also experience greater PFAS exposure than estuarine and coastal fish, consistent with the higher PFOS and PFOA concentrations observed in rivers compared with estuarine and coastal waters (Section 6.1.2). The extent of accumulation is likely influenced by a range of factors, including species, age, trophic level of fish sampled and local environmental conditions.

In marine-influenced systems, PFOS concentrations are similar in estuarine and coastal fish (approximately 1.1 µg/kg). PFOA concentrations are also comparable, at 0.068 and 0.056 µg/kg respectively, indicating little difference between environments.

Table 5: ***Model-based estimates of geometric mean concentration for PFOS and PFOA in biota in the middle of 2023 with 95% confidence intervals (shown in brackets), numbers of samples and sites used in each analysis. Concentrations are reported to two significant figures.***

Media	Water category	Determinand	Mean Concentration	Unit	Number of samples	Number of sites
Biota	Coastal fish	PFOA	**0.056** (0.041, 0.077)	µg/kg	210	19
Biota	Coastal fish	PFOS	**1.1** (0.80, 1.5)	µg/kg	203	19
Biota	Estuarine fish	PFOA	**0.068** (0.035, 0.13)	µg/kg	225	16

Media	Water category	Determinand	Mean Concentration	Unit	Number of samples	Number of sites
Biota	Estuarine fish	PFOS	**1.1** (0.56, 2.1)	µg/kg	222	16
Biota	Freshwater fish	PFOS	**4.8** (3.4, 6.8)	µg/kg	596	69

6.2.3 Spatial variability in PFOS and PFOA concentrations in biota

Concentrations of PFOS and PFOA vary between fish sampling sites across England, as shown in Table 6. We summarise spatial variability using the 90th to 10th percentile site concentration ratios for each fish category. Estimates are based on site-specific estimated concentrations for a common reference point in mid-2023 and are derived using the approach described in Section 6.1.3 for surface waters, which applies equivalently here. Mid-2023 was used as the reference point for biota because, at the time of analysis, biota data were only available up to early 2024 and estimates towards the end of the time series tend to be subject to more uncertainty.

There is substantial spatial variation in concentrations with the degree of variability differing by environment and substance. The greatest site-to-site variation occurs in estuarine fish, where both PFOS and PFOA show differences in concentrations between monitoring locations. However, there is relatively wide uncertainty around the 90:10 ratio estimate, likely reflecting the smaller number of sites and samples available compared with freshwater fish. The pattern is consistent with that observed in estuarine surface waters.

Freshwater fish also exhibit pronounced spatial variation in whole-body concentrations, particularly for PFOS.

In contrast, concentrations of PFOS and PFOA in coastal fish show less variation between monitoring locations as indicated by the lower 90:10 ratios for both PFOS and PFOA, reflecting the pattern observed in coastal surface waters.

The distributions of site-specific concentration estimates underlying these ratios are illustrated in Figure A5 in Appendix 6, which provides visual context to support the quantitative summaries presented in Table 6.

Table 6: ***Mean ratio between 90th and 10th percentile sites for PFOS and PFOA concentrations in biota in different environmental compartments. Numbers in brackets indicate 95% confidence intervals. Values are reported to two significant figures.***

Media / Compartment	Determinand	90th:10th percentile site ratio
Coastal fish	PFOA	2.4 (1.7, 3.8)
Coastal fish	PFOS	3.6 (2.2, 5.9)
Estuarine fish	PFOA	14 (6.1, 28)
Estuarine fish	PFOS	18 (7, 40)
Freshwater fish	PFOS	10 (6.8, 14)

7. PFAS concentrations in the environment

Since 2021, we have expanded our fully quantitative analytical capability to monitor over 60 PFAS in rivers, lakes and groundwater, and 45 PFAS in biota. This represents a significant step forward from earlier semi-quantitative approaches in water only (Environment Agency, 2021), allowing measurement of concentrations of many more individual PFAS in different environmental compartments.

Our data spans between two and five years, depending on the environmental compartment and substance. This data includes PFOS and PFOA concentrations, however we treat this data separately from the longer-term PFOS and PFOA data reported in Chapter 6. We estimated concentrations for individual PFAS by fitting Generalised Additive Models (GAMs) that account for seasonality, site-to-site and concentration differences between individual PFAS. Further detail on the approach is provided in Appendix 5.

Given the large number (60+) and structural diversity of PFAS monitored, we applied a grouped approach to summarise results (see Section 7.1). This provides a practical and consistent framework for assessing multiple substances simultaneously and enables coherent comparison across water body types and biota.

We calculated total concentrations by group from the estimated concentrations of individual PFAS for 2025 in rivers, lakes and groundwater (Section 7.2) and from period-average concentrations for biota (2018-2024) (Section 7.3).

Following assessment of grouped PFAS patterns, we examined concentrations of individual PFAS to identify dominant substances within each environmental compartment and group. This enables comparison of the relative contribution of different PFAS across waterbody types and biota and provides additional insight into differences in environmental behaviour and fate between compartments and substances.

7.1 Approach to PFAS groupings

To support interpretation of results from our PFAS monitoring, we applied a grouping approach. The wide range of PFAS structures and properties means that there are multiple ways in which PFAS can be grouped.

Several published approaches to grouping, based on structural similarity were considered to ensure consistency and scientific robustness in analysis of our PFAS data (Buck et al., 2011; Glüge et al., 2020; HSE, 2023; OECD, 2022b, 2018). The grouping framework proposed by the OECD in 2018 was selected as the primary basis for structural categorisation, as most substances in the EA monitoring datasets align with these group definitions (OECD, 2018). For substances not fully addressed by the 2018 framework, the 2021 update by OECD was used to support group assignment (OECD, 2021). Together, this approach provides a globally recognised, high-level classification system that captures PFAS diversity while remaining practical for interpretation and visualisation.

Within this framework, we further subdivided perfluoroalkyl carbonyl and perfluoroalkane sulfonyl compounds by chain length to reflect differences in environmental behaviour. Long-chain compounds (≥ 7 perfluorinated carbons for carbonyl compounds and ≥ 6 for sulfonyl compounds) generally show greater persistence and bioaccumulation potential and are therefore distinguished from shorter-chain compounds. This additional subdivision provides greater resolution where it most influences environmental fate and hazard evaluation.

While terminology such as PFCA and PFSA are widely used in the scientific literature, these refer to specific subclasses of perfluoroalkyl acids. In this report, we use broader OECD-aligned structural groupings (e.g. “perfluoroalkyl carbonyl compounds” and “perfluoroalkane sulfonyl compounds”) to support a grouping-based approach and maintain consistency with the OECD classification framework.

Table 7 below outlines the 2018 OECD grouping classes used in this analysis, including their subdivision into short- and long-chain groups, their broad environmental characteristics, and the abbreviations applied for labelling and reporting purposes.

Table 7: ***PFAS grouping classes used in this report, their abbreviations, and their general environmental characteristics.***

OECD Grouping (with additional chain-length subdivision)	Abbreviation used	General environmental characteristics
perfluoroalkyl carbonyl compounds (long chain) *	LC-carbonyl PFAS	Moderate mobility, high sorption to soils and sediments, and high bioaccumulation potential.
perfluoroalkyl carbonyl compounds (short chain) *	SC-carbonyl PFAS	High mobility, low sorption to soils and sediments, and low bioaccumulation potential.
perfluoroalkane sulfonyl compounds (long chain) *	LC-sulfonyl PFAS	Low to moderate mobility, high sorption to soils, sediments and biota, and high bioaccumulation potential.
perfluoroalkane sulfonyl compounds (short chain) *	SC-sulfonyl PFAS	Moderate to high mobility, moderate sorption, and low to moderate bioaccumulation potential; generally more sorptive than equivalent-chain length carbonyl PFAS.
per- and polyfluoroalkyl ether-based compounds	Ether-based PFAS	Variable properties; generally high mobility, low sorption, and low to moderate bioaccumulation potential.
perfluoroalkyl phosphate compounds	Phosphate PFAS	Precursor compounds; variable mobility, sorption and bioaccumulation potential, and can transform to terminal PFAAs.
fluorotelomer-related compounds	Fluorotelomer PFAS	Precursor compounds; moderate to high mobility, low to moderate sorption, and can transform to terminal PFCAs.
other PFAA precursors and related compounds - perfluoroalkyl ones	Other PFAS	Variable mobility, sorption and bioaccumulation potential; some substances may transform to terminal PFAS.

OECD Grouping (with additional chain-length subdivision)	Abbreviation used	General environmental characteristics
other PFAA precursors or related compounds – semifluorinated*	Not represented in monitoring suite	Variable mobility, sorption and bioaccumulation potential.
Fluoropolymers*	Not represented in monitoring suite	Typically, low mobility, low bioavailability and low bioaccumulation potential in polymeric form.

**Note: In this report, "perfluoroalkyl carbonyl compounds" are represented primarily by perfluoroalkyl carboxylic acids (PFCAs), while "perfluoroalkane sulfonyl compounds" include perfluoroalkane sulfonic acids (PFSAs) and related precursor compounds. Other PFAA precursors or related compounds – semifluorinated and fluoropolymers are shown for completeness of the OECD PFAS classification system but are not considered further in this report, as no substances from these groupings were present in our current monitoring suite.*

PFAS are also commonly described as either terminal or precursor compounds. Terminal PFAS are highly persistent end-products that do not readily degrade further in the environment and include perfluoroalkyl acids such as PFCAs (e.g. PFOA) and PFSAs (e.g. PFOS). In contrast, precursor PFAS can transform into terminal PFAS in the environment and therefore represent a potential source of these substances. Examples include fluorotelomer-based compounds and some phosphate PFAS.

Our current PFAS analytical suite spans a range of PFAS groups and subgroups. Some groups are well represented while others have limited or no coverage. Specifically, perfluoroalkyl carbonyl compounds consist exclusively of PFCAs, while perfluoroalkane sulfonyl compounds comprise PFSAs and the precursor compounds perfluoroalkyl sulfonamides (FASAs) and perfluoroalkane sulfonyl fluorides (PASFs). The per- and polyfluoroalkyl ether-based compounds are represented by the terminal PFAS perfluoroalkyl ether carboxylic acids (PFECAs) and perfluoroalkyl ether sulfonic acids (PFESAs), and the perfluoroalkyl phosphate compounds consist entirely of perfluoroalkyl phosphinic acids (PFPiAs). The fluorotelomer group includes a range of different fluorotelomer-based substance types. The remaining OECD group categories are either negligibly represented or absent within our current monitoring suite. A detailed breakdown of grouping for the EA's current PFAS monitoring suite is provided in Appendix 4.

The general descriptions of environmental fate and behaviour of substances in each grouping in Table 7 are informed by differences in physicochemical properties, including water solubility, sorption potential (e.g. organic carbon-water partition coefficient; K_{OC}), degree of ionisation and bioaccumulation behaviour. These properties influence the environmental fate and transport of PFAS and determine the differences in mobility and bioaccumulation potential observed between groups. The characteristics presented in Table 7 provide a high-level summary of expected environmental behaviour. Further

information on the physicochemical properties of individual PFAS is available within the Interstate Technology and Regulatory Council (ITRC) PFAS Technical and Regulatory Guidance property tables (ITRC, 2025) and other sources.

7.2 PFAS patterns in freshwater and groundwater

7.2.1 PFAS group patterns in rivers, lakes and groundwater

Although more than 60 PFAS are included in our monitoring programme, many are rarely detected above the MRV. For rivers, lakes and groundwater our analysis included 18 of the 60+ monitored PFAS (Appendix 4), quantified as being above the MRV in at least 1% of samples. We report detection frequencies for individual substances in Appendix 6.

Figure 11 summarises estimated total concentrations of monitored PFAS by group in rivers, lakes and groundwaters. For each group, the point represents the estimated mean concentration at the median (50th percentile) monitoring site, while the horizontal bars indicate the estimated mean concentrations at the 10th and 90th percentile sites. These bars reflect spatial variability in mean concentrations between monitoring sites and should not be interpreted as confidence intervals or uncertainty in the mean estimates.

Concentrations are displayed on a logarithmic scale, where equal distances along the x-axis represent equal proportional rather than absolute changes in concentration. For example, the distance from 10 to 100 ng/L is the same as from 100 to 1,000 ng/L, with both representing a ten-fold increase. Group rankings for the different waterbody types are summarised in Figure 12.

SC-carbonyl PFAS consistently occur at the highest concentrations in rivers, lakes and groundwater, ranking first in every waterbody type (Figure 11; Figure 12).

LC-sulfonyl PFAS generally rank second in rivers (including the RSN) and lakes, indicating continued prevalence despite lower concentrations than SC-carbonyl PFAS. Fluorotelomer PFAS, in contrast, are among the lowest concentrations measured in rivers and lakes but are notably higher in groundwater, suggesting a distinct contamination profile or source pathway within groundwater systems.

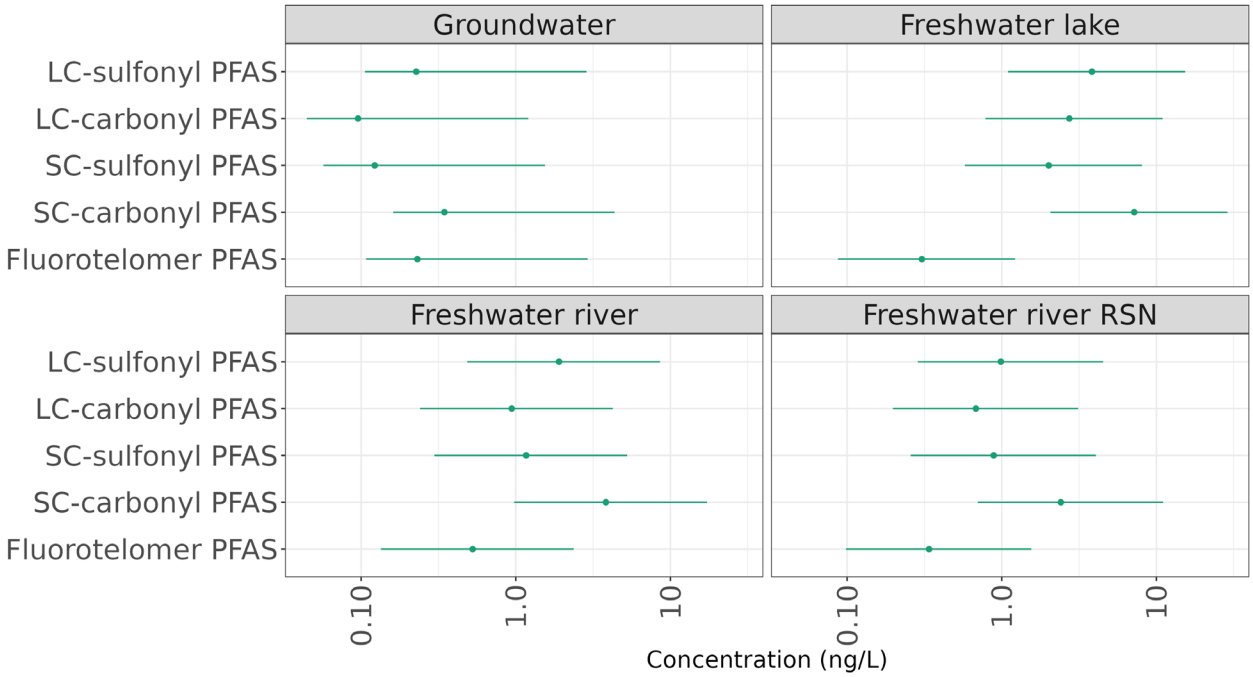


Figure 10: Spatial variability in estimated PFAS group concentrations (ng/L) by waterbody type. ***Points indicate the model-based estimates of geometric mean concentration at the median site, standardised to 2025, while horizontal bars represent concentrations at the 10th and 90th percentile sites. Percentiles are based on ranked site concentrations. The x-axis is presented on a logarithmic scale. Values reflect spatial variation only.***

Groundwater exhibits a distinct pattern of PFAS occurrence compared to rivers and lakes. SC-carbonyl PFAS remain the dominant group, fluorotelomer PFAS are comparatively elevated, and LC-carbonyl PFAS occur at relatively low concentrations. These patterns possibly reflect groundwater-specific conditions, including longer residence times and associated hydrogeochemical processes (Lapworth et al., 2022).

Mean concentrations of PFAS in groundwater also tend to be lower than in lakes and rivers. This is likely due to dilution during infiltration and longer residence times (Lasagna et al., 2013; Sims et al., 2022). In contrast, higher concentrations and detection frequencies of PFAS in rivers and lakes are consistent with shorter residence times and a stronger influence from direct and diffuse inputs, as well as temporal factors, including rainfall, temperature, and seasonal variation.

The widespread dominance of SC-carbonyl PFAS in water is consistent with their high solubility and limited tendency to bind to sediments, resulting in their high environmental mobility allowing efficient transport through surface water and groundwater systems. In addition, many SC-carbonyl PFAS are terminal degradation products of fluorotelomer PFAS. Consequently, fluorotelomer PFAS represent an ongoing source of SC-carbonyl PFAS in the environment, contributing to their widespread occurrence. This pattern may also reflect changes in the types of PFAS in use and thus entering the environment, although the relative contribution of PFAS from current use, precursor degradation and legacy contamination cannot be determined from the monitoring data alone.

In contrast, LC-carbonyl and sulfonyl PFAS, including PFOS, are likely to remain prevalent due to their extensive historical use in industrial applications, continued presence in wastewater discharges and legacy contamination associated with the use of aqueous film-forming foams (AFFFs), combined with extreme environmental persistence and periodic remobilisation from sediments.

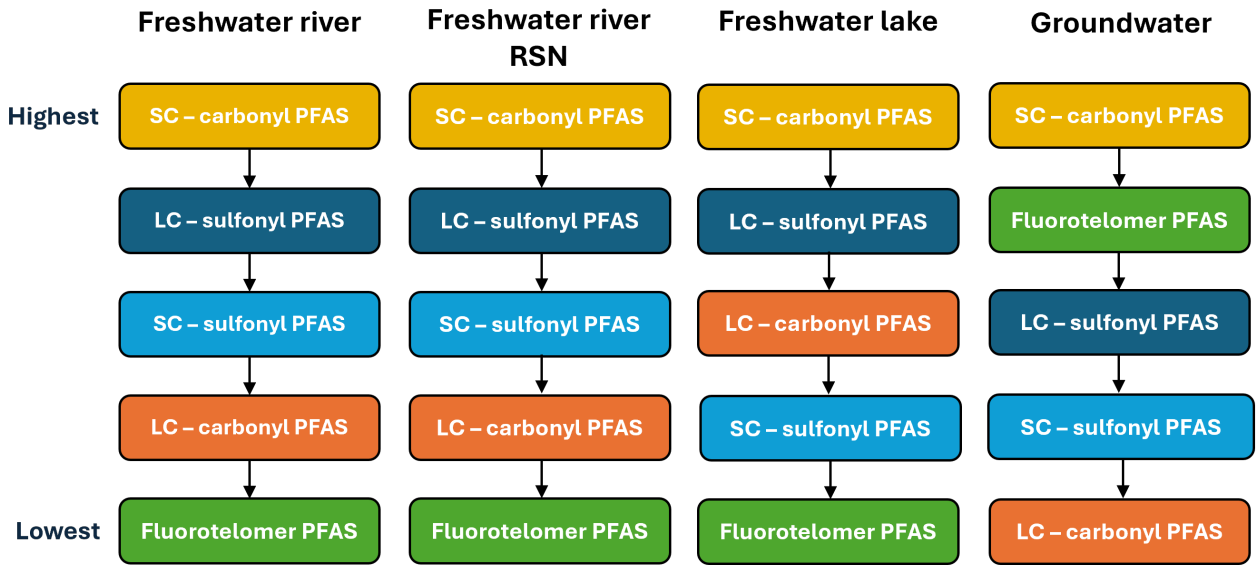


Figure 11: PFAS group rankings (highest to lowest concentration) across waterbody type.

Variability among mid-ranking PFAS groups across rivers, lakes and groundwater likely reflects differences in emission sources, hydrology and environmental fate and behaviour processes. In rivers, contributions from wastewater effluent and urban runoff may influence the presence of SC-sulfonyl PFAS. In lakes however, longer water residence times and reduced flushing are likely to favour the relative persistence of certain long-chain PFAS through greater interaction with sediments and suspended particulates (Muir and Miaz, 2021).

Lower-ranking groups also vary between rivers, lakes and groundwater, with strongly sorbing long-chain PFAS less likely to occur in dissolved form in rivers and groundwater, while fluorotelomer PFAS appear least abundant in lakes, potentially due to sorption and partial degradation. Overall, the patterns summarised in Figure 12 are consistent with known influences of chemical structure, mobility, degradation pathways and hydrological conditions across water environments.

7.2.2 Dominant PFAS in rivers, lakes and groundwater

While the analysis above focuses on PFAS group-level patterns, we can gain additional insight by examining which substances are dominant within each waterbody type (Figure 13). At the substance level, dominance refers to the PFAS with the highest estimated concentrations (ng/L in water), whereas group rankings reflect combined contributions across multiple substances. Numerical values for the individual PFAS concentrations shown in Figure 13 are provided in Table A6 in Appendix 6.

In rivers and lakes, PFBA is the most abundant PFAS, with mean concentrations of around 1.0–1.3 ng/L in rivers and 3.3 ng/L in lakes.

Lakes generally show higher concentrations for several PFAS compared with rivers and groundwater, although this conclusion has greater uncertainty due to fewer monitoring samples and sites.

For most PFAS, concentrations in rivers (excluding the RSN) are higher than those at RSN monitoring sites. The RSN is a probability-based, spatially balanced survey network, whereas the non-RSN river sites are selected primarily for statutory, operational and investigative monitoring purposes. Consequently, observed differences may reflect both differences in environmental pressures and differences in network design. Given the differing objectives and site-selection approaches of the RSN and other river monitoring programmes, this pattern is consistent with a likely greater influence of anthropogenic pressures at non-RSN sites.

Conversely, where concentrations are similar between sampling locations in different networks, this may indicate PFAS that are more widely distributed across the river environment. Our analysis as described here does not provide sufficient evidence to attribute these patterns to specific sources or pathways, and further investigation would be required to determine the relative roles of point sources, diffuse sources, atmospheric deposition and precursor transformation.

An unusual feature of the groundwater dataset is the prominence of the fluorotelomer 6:2 FTSA, which occurs at higher concentrations than any other individual PFAS monitored in groundwater, with a mean of 0.34 ng/L. This may reflect a combination of factors including its widespread use in PFAS-containing products, including fluorotelomer-based firefighting foams introduced to replace restricted PFAS previously used in foams, and environmental behaviour that favours transport through groundwater systems. However, our current analysis does not allow us to determine the relative importance of source strength, precursor transformation, persistence or mobility. Further investigation would be required to understand why 6:2 FTSA is particularly prominent in the groundwater environment.

PFOS and PFHxS are reported separately as linear (L) and branched (B) isomers in Figure 13. Isomers are compounds that share the same chemical formula but differ in their molecular structure. Linear PFHxS generally occurred at higher concentrations than branched PFHxS, while concentrations of linear and branched PFOS were broadly similar across compartments. Variation in the relative proportions of these isomers may reflect differences in manufacturing processes, source characteristics or environmental fate (Agarwal et al., 2025), although the patterns observed in this dataset were generally weak and should be interpreted cautiously.

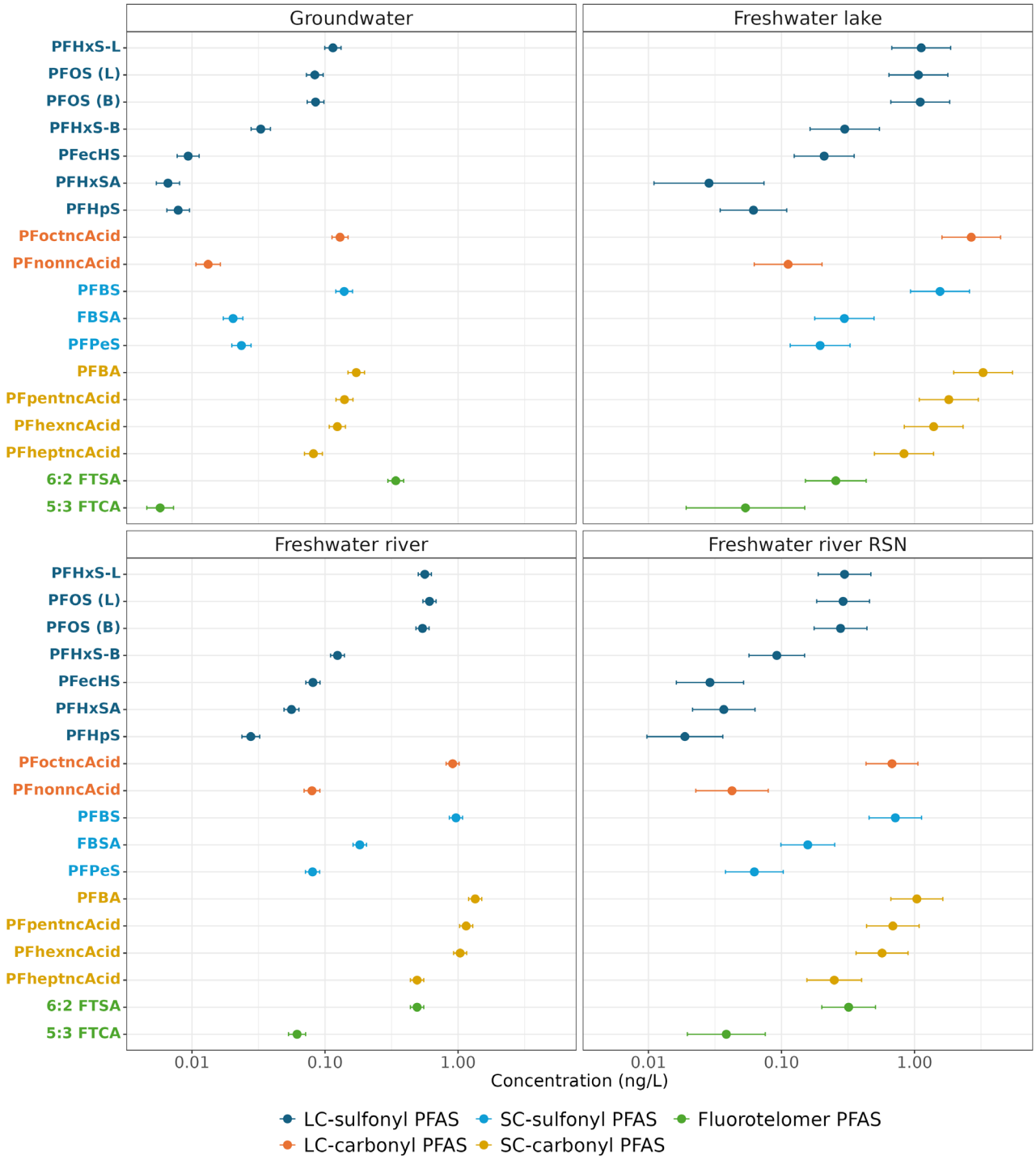


Figure 12: Geometric mean concentrations of individual PFAS in rivers, lakes and groundwater for 2025, grouped by PFAS class. ***Points represent model-based estimates of mean concentrations and horizontal bars represent 95% confidence intervals. Concentrations are displayed on a logarithmic scale. Colours indicate PFAS groupings. Where available, branched and linear isomers are reported separately. Numerical values corresponding to the figure are provided in Table A6 in Appendix 6.***

7.2.3. Comparison of PFAS associated with different sources

Understanding whether higher environmental concentrations of particular PFAS are associated with recognised sources can inform source apportionment, monitoring design and prioritisation of regulatory effort.

The analysis presented in this section is intended as an exploration of PFAS signatures associated with two nationally important PFAS sources: sites linked to AFFF use and permitted landfills. It is not intended to quantify environmental harm or source contribution. Instead, it explores whether particular PFAS occur at elevated concentrations at sites associated with these source types.

Environmental measurements of PFAS will reflect multiple sources, including legacy contamination. Figure 14 explores how PFAS concentrations differ using the monitoring data from rivers and groundwater described in the previous section. We categorised each monitoring site as associated with (and potentially impacted by) an AFFF source, a permitted landfill source, or neither. For each PFAS, we then compared estimated mean concentrations at AFFF-associated and landfill-associated sites (higher-risk sites) with those at sites not associated with either source (lower-risk sites). The results are expressed as concentration ratios relative to the lower-risk sites (for example, AFFF-associated sites: lower risk sites).

Ratios greater than 1 indicate higher relative PFAS concentrations at the sites for that risk category compared to sites where neither AFFFs have been used nor a landfill is present. Ratios less than 1 indicate lower PFAS concentrations. For example, a ratio of 4.3 for PFOA at AFFF risk sites indicates concentrations were 4.3 times as high, or 330% higher than, those at the lower risk sites. A ratio of 1 would indicate no difference in PFAS concentrations between sites associated with different sources. It is important to note that expressing the data as ratios tells us about relative not absolute concentrations.

The figures for absolute concentrations across all data for each compartment are presented in Table A6 in Appendix 6. Further details on the identification of risk sites and the analytical approach are provided in Appendix 5.

The results in Figure 14 show that most PFAS are present at significantly higher concentrations at sites associated with AFFF use or permitted landfill sites (i.e. ratios greater than 1 and confidence intervals not overlapping with 1). There are somewhat unique patterns depending on the PFAS source and whether river or groundwater sites have been sampled. Ratios for some PFAS are occasionally less than 1, but their confidence intervals typically overlap with 1 (no difference) indicating high uncertainty in the effect size.

Permitted landfill sites typically exhibit higher ratios than AFFF sites, but with greater uncertainty. This higher uncertainty may reflect a greater range of potential PFAS contamination sources in landfill, but further data are needed to build an understanding of the differences. For all but two PFAS, AFFF sites exhibit greater elevation in PFAS concentrations (higher ratios) for groundwater than for rivers, but with greater uncertainty. However, the lower figures for rivers may relate to the buffering methodology used to identify potentially impacted sites. In a minority of cases, river monitoring sites classified as

associated with higher-risk PFAS sources may be located upstream of the impacted area, which could reduce the apparent difference between higher-risk and lower-risk sites.

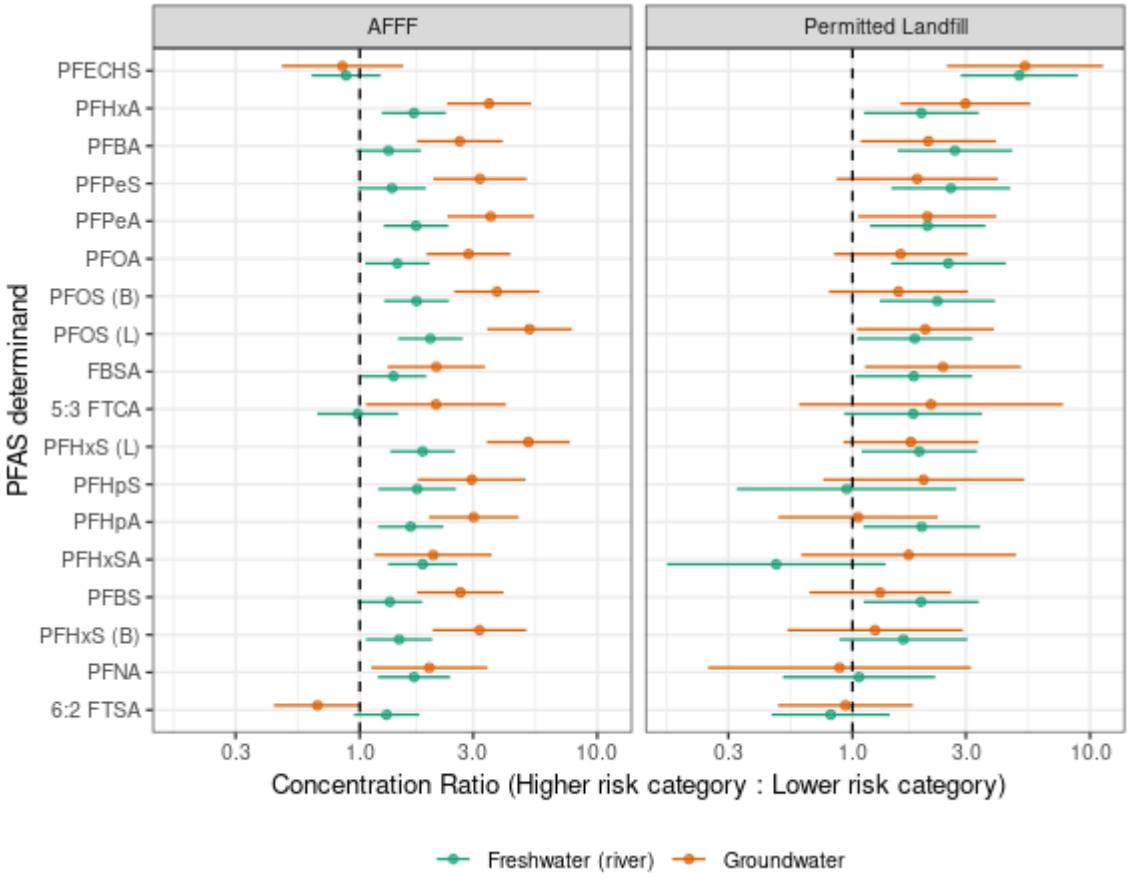


Figure 13: Comparison of estimated PFAS concentrations at sites associated with AFFF use or permitted landfills relative to lower-risk sites, expressed as ratios of mean concentrations derived from model-based contrasts where PFAS profiles vary by site category. Bars indicate 95% confidence intervals for the ratio, dashed vertical lines indicate a ratio of 1 (no difference).

7.3 PFAS patterns in biota

7.3.1 PFAS group patterns in fish and mussels

The data presented here comprise estimated PFAS concentrations in whole-body samples of freshwater fish, estuarine fish, coastal fish, and estuarine and coastal blue mussels. A total of 45 PFAS were analysed in these samples, however only a subset were routinely detected above the MRV. We therefore retained PFAS substances with three or more detected samples above the MRV, resulting in 23 of the 45 PFAS monitored being included in the fish analysis and 14 in the blue mussel analysis (see Appendix 4). Detection frequencies for individual substances are provided in Table A5 in Appendix 6.

As for water samples described in Section 7.2.1, Figure 15 presents PFAS concentrations by group, with points indicating model-based estimated mean concentrations for each

PFAS group and horizontal bars illustrating variability across sites rather than statistical uncertainty.

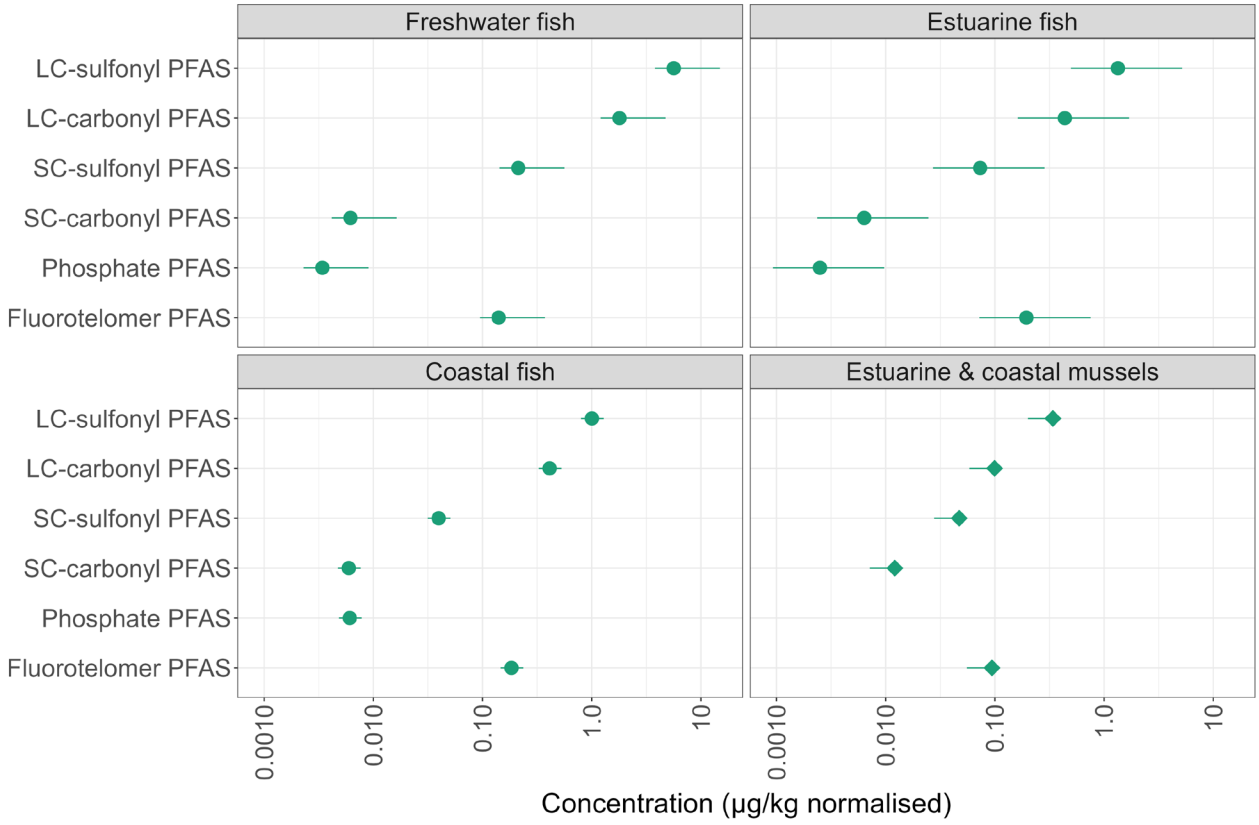


Figure 14: Spatial variability in estimated PFAS group total concentrations (µg/kg) in biota inhabiting freshwater and saline water, averaged across the period 2018-2024. ***Points indicate the model-based estimated geometric mean concentration at the median site, while horizontal bars represent estimated concentrations at the 10th and 90th percentile sites. Percentiles are based on ranked site concentrations. Values reflect spatial variation only. Blue mussel data are shown using diamond symbols to distinguish them from fish, reflecting the smaller number of detectable PFAS substances within groups for mussels.***

Based on the mean concentrations shown in Figure 15, Figure 16 illustrates the relative ranking of PFAS groups in biota.

For all animals sampled, PFAS accumulation follows a broadly consistent pattern that contrasts with that observed in water. Longer-chain PFAS generally occur at higher concentrations than shorter-chain compounds, with LC-sulfonyl and -carbonyl PFAS contributing most strongly to overall PFAS burdens in freshwater, estuarine and coastal fish and mussels (Figure 15; Figure 16). Although some variation in ranking is observed between sample type and environments, the overall accumulation hierarchy remains similar across fish and mussels.

The higher levels of long-chain PFAS in fish and mussels, compared with short-chain PFAS, reflect their strong affinity for proteins, high bioaccumulation potential, and

persistence, leading to greater retention in biological tissues and accumulation through aquatic food webs.

Fluorotelomer PFAS generally occur at concentrations between those of long-chain and short-chain PFAS in fish and mussels. These substances accumulate less than long-chain PFAS but can act as precursors, transforming into more persistent PFAS over time and contributing indirectly to observed concentrations.

SC-carbonyl and -sulfonyl PFAS are typically present at lower concentrations, consistent with their higher solubility, faster elimination rates and lower bioaccumulation potential. Phosphate PFAS occur at the lowest concentrations in freshwater and estuarine fish, likely reflecting lower bioaccumulation potential and low ambient concentrations, consistent with their absence from the water data presented in Section 7.2.1.

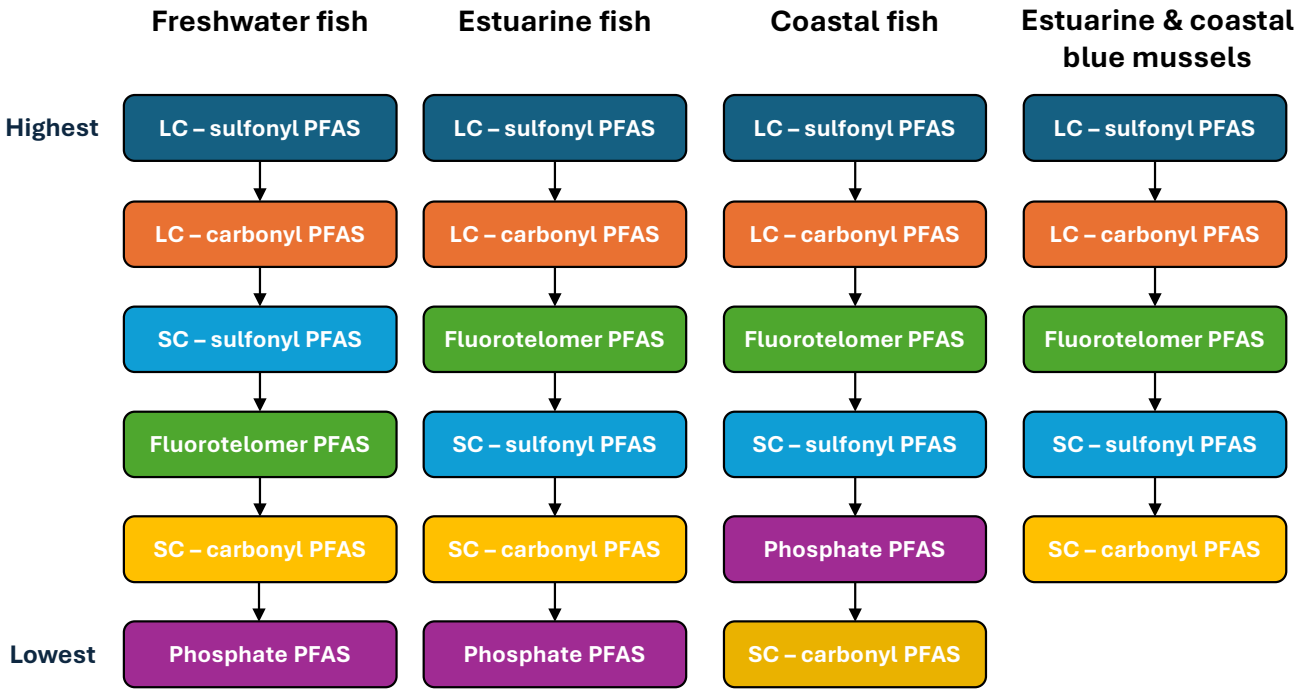


Figure 15: PFAS group rankings in biota (highest to lowest concentration).

Overall, the broadly consistent PFAS patterns observed across fish and mussels suggest similar accumulation behaviour across organisms and environments. Higher concentrations of long-chain PFAS in fish and mussels compared with water reflect their persistence and stronger biological retention, whereas lower concentrations of short-chain and phosphate PFAS are consistent with limited retention by these organisms.

7.3.2 Dominant PFAS in fish and mussels

Figure 17 shows estimated mean concentrations of individual PFAS in fish and mussels and highlights differences between sample types and PFAS groups. Numerical values for these concentrations are provided in Table A7 in Appendix 6.

PFAS concentrations vary across biota types. Fish generally have higher concentrations than mussels, particularly for long-chain PFAS, although this pattern is not uniformly

observed across all substances. In fish, concentrations of many PFAS decrease from freshwater to estuarine to coastal environments, but this varies between substances.

PFOS contributes a substantial proportion of the overall PFAS burden in fish and represents the highest concentrations observed in the dataset, increasing from around 1.1 µg/kg in coastal fish to 1.8 µg/kg in estuarine fish and 5.7 µg/kg in freshwater fish. Other long-chain sulfonyl PFAS occur at lower concentrations. In contrast, mussels show a different PFAS profile, with lower PFOS concentrations of around 0.068 µg/kg and PFDSA occurring at the highest concentrations, approximately 0.13 µg/kg, reflecting species-specific uptake and retention.

The continued dominance of PFOS in fish is notable given the restrictions placed on its manufacture and use. However, this pattern is consistent with the environmental persistence, strong protein binding and relatively high bioaccumulation potential of PFOS compared with many other PFAS (see Table 7 for a summary of key PFAS properties; Brunn et al., 2023). Combined with extensive historical use and the continuing influence of legacy contamination, these characteristics help explain why PFOS remains a dominant contributor to PFAS burdens in fish despite declining concentrations in the water environment (Environment Agency, 2024; Muir and Miaz, 2021).

These same properties also help explain the marked difference between PFOS and PFOA concentrations in fish. Both substances can bioaccumulate, but PFOS generally bioaccumulates more strongly in aquatic food webs owing to stronger protein binding and slower elimination from organisms (Brunn et al., 2023; Sun et al., 2022).

Concentrations of LC-carbonyl PFAS are generally higher in freshwater fish sampled than in estuarine and coastal fish, often by several-fold, although this pattern varies between substances.

The contrasting PFAS profiles observed between fish and mussels are consistent with other studies demonstrating fundamental differences in PFAS bioaccumulation between these organisms (Giffard et al., 2022; Jones et al., 2025; Pandelides et al., 2026; Steevens et al., 2023; Teunen et al., 2021).

Mussels typically show a greater relative contribution of short-chain PFAS and PFOA, whereas PFAS body burdens in fish are dominated by PFOS and long-chain perfluoroalkyl carboxylic acids reflecting biomagnification at higher trophic levels (Teunen et al., 2021). Laboratory studies confirm that bioaccumulation of PFOS and long-chain PFAS is lower in mussels than in fish under comparable conditions (Steevens et al., 2023), while field observations indicate that mussels more closely reflect ambient water concentrations, whereas fish integrate both waterborne and dietary exposure pathways (Jones et al., 2025).

These differences may explain why PFAS concentrations are typically lower for bivalves than for fish (Pandelides et al., 2026) and demonstrate that mussels and fish represent distinct bioaccumulation endpoints within aquatic food webs.

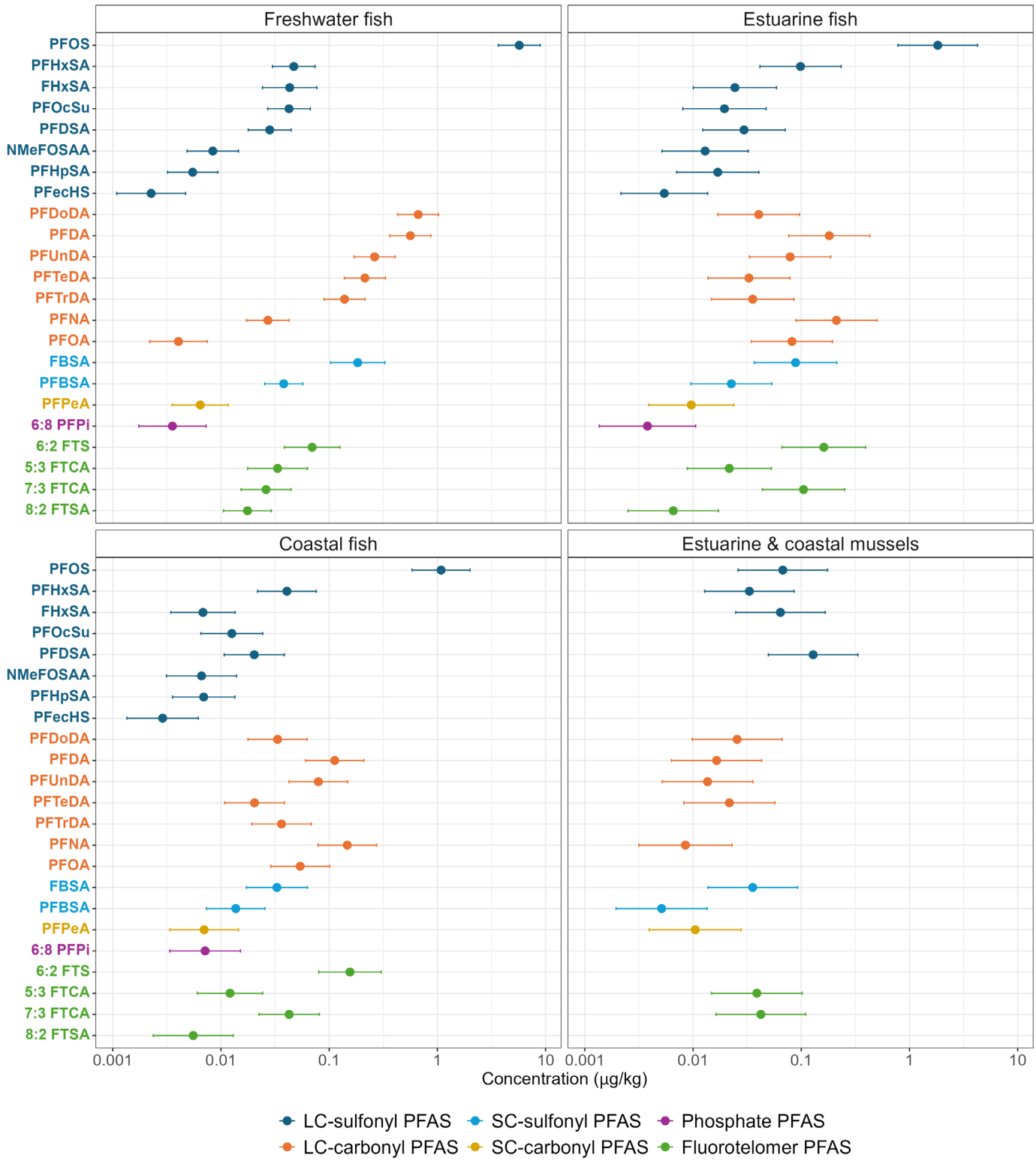


Figure 16: Geometric mean concentrations of individual PFAS in freshwater fish, estuarine and coastal fish, and blue mussels (µg/kg), averaged across the period 2018-2024. ***Points represent model-based estimated mean concentrations and horizontal bars represent 95% confidence intervals. Concentrations are displayed on a logarithmic scale. Colours indicate PFAS groupings. Numerical values corresponding to the figure are provided in Table A7 in Appendix 6.***

Comparison of PFAS patterns in water and biota

Overall, observed PFAS profiles differ between water and biota samples. In water, short-chain carbonyl PFAS dominate, whereas PFAS burdens in fish and mussels are dominated by long-chain sulfonyl and carbonyl PFAS.

These differences do not necessarily indicate different sources. Many PFAS detected in water and biota are likely to originate from the same environmental inputs and pathways. However, differences in physicochemical properties influence the fate of PFAS in the environment. Short-chain PFAS tend to remain in the water column, whereas long-chain PFAS are more likely to accumulate in biota and, in some cases, sediments. As a result, water monitoring primarily reflects transport and distribution within the aquatic environment, whereas biota monitoring reflects the selective accumulation of PFAS with greater bioaccumulation potential and provides an integrated measure of longer-term exposure.

Changes in PFAS production and use over time may also influence these patterns. Restrictions on some long-chain PFAS, such as PFOS, have led to increased use of alternative replacement substances, including short-chain and fluorotelomer compounds. Consequently, these compounds are often more prevalent in the water column, while legacy long-chain PFAS continue to contribute to PFAS burdens in biota. Together, these findings highlight the importance of monitoring both water and biota to understand PFAS behaviour, exposure pathways and assess ecological and human health risk.

7.4 Key monitoring insights from chapters 6 & 7

1: Declining PFOS and PFOA trends reflect regulatory restrictions, with some evidence of ongoing legacy contamination

Our long-term monitoring shows statistically robust declines in PFOS and PFOA concentrations in English rivers between 2016 and 2025. These findings are broadly consistent with the regulatory restrictions placed on these substances and suggest that reductions in use and emissions are contributing to measurable environmental improvements.

Evidence from estuarine and coastal waters and lakes is less consistent. PFOA concentrations show declining trends across these environments, although the magnitude of change is more uncertain than in rivers because monitoring records are shorter and datasets are smaller. In contrast, PFOS concentrations have neither increased nor decreased overall but exhibit greater year-to-year variability than those observed in rivers.

The absence of a clear declining trend for PFOS in estuarine and coastal waters and lakes does not necessarily indicate ongoing emissions. Instead, the available evidence suggests that environmental processes associated with legacy contamination may continue to play an important role. In estuarine waters in particular, sediment disturbance, changes in salinity and the remobilisation of historically contaminated sediments may influence

concentrations and contribute to short-term fluctuations in concentration. These patterns are therefore more consistent with the continued influence of legacy contamination than with significant new sources of PFOS.

Monitoring in fish and mussels shows a similar pattern. PFOS concentrations in freshwater fish have declined substantially over the past decade, mirroring the trends observed in rivers, while concentrations in estuarine and coastal fish have remained relatively stable. PFOA concentrations in fish are generally low throughout the monitoring period. Importantly, national mean PFOS concentrations in the freshwater, estuarine and coastal fish monitoring networks remained below the PFOS biota EQS throughout the monitoring period*. This provides additional evidence that biological exposure to PFOS is declining at a national scale, despite some water bodies continuing to exceed the corresponding water-column EQSs.

* *These national mean concentrations should not be interpreted as site-specific assessments of PFOS biota EQS compliance. Individual water bodies may exceed the EQS despite the national mean remaining below it.*

2: Dominance of short-chain PFAS in water reflects persistence and ongoing formation, not just substitution

The widespread dominance of short-chain PFAS in rivers, lakes and groundwater is consistent with their high environmental mobility compared with many long-chain PFAS. Changes in use, including substitution away from some long-chain substances to shorter chain PFAS following regulatory restrictions, have influenced PFAS profiles over time. However, evidence from long-term monitoring programmes and recent peer-reviewed research indicates that sources of short-chain PFAS to the environment are also sustained through the degradation of fluorotelomer precursors, atmospheric transformation processes (including from volatile fluorinated compounds), and waste-management pathways (Ackerman Grunfeld et al., 2024; Bossi and Ellermann, 2023; Evich et al., 2022; Thackray et al., 2020).

Atmospheric long-range transport and subsequent deposition are potentially important mechanisms sustaining environmental concentrations of short-chain PFAS, including in locations remote from direct emission sources (Thackray et al., 2020). The widespread occurrence of short-chain PFAS across rivers, lakes and groundwater in this report is consistent with transport through multiple pathways. However, the current analysis does not allow the relative contributions of atmospheric deposition, wastewater discharges, catchment sources or precursor transformation to be determined.

This means that even in the absence of direct emissions, short-chain PFAS are likely to remain widespread in the environment. While these substances generally show lower bioaccumulation than many long-chain PFAS, their persistence and, particularly, their mobility mean they can contribute to long-term environmental exposure and are relevant to groundwater protection and drinking-water management. These findings reinforce the need for group-based and precautionary approaches, rather than reliance on regulating individual substances.

When considered on a molar basis rather than the mass-based concentrations used here, this dominance implies that many more short-chain PFAS molecules are present in the environment compared with long-chain PFAS, as short-chain substances are lighter (meaning the same mass represents more molecules). This indicates that differences between short- and long-chain PFAS may be even more pronounced on a molar basis, which may have implications for environmental exposure.

3: Divergent PFAS behaviour in water and biota highlights distinct exposure pathways

Environmental monitoring data show a consistent and pronounced divergence between PFAS profiles in water and those in fish and mussels, reflecting differences in the physico-chemical properties of these substances. In rivers, lakes and groundwater, short-chain carbonyl PFAS dominate in the water column, reflecting their high solubility and resulting mobility in water, as well as their continued formation from the breakdown of precursor substances. In contrast, long-chain sulfonyl PFAS, particularly PFOS, dominate fish and mussel body burdens.

This divergence demonstrates that water and biota represent fundamentally different PFAS exposure pathways. Concentrations measured in the water column primarily reflect transport and dispersal as well as persistence within the water environment, whereas biota concentrations capture persistence, protein binding and food-web accumulation. As a result, water and biota data provide complementary lines of evidence, highlighting the value of considering both when assessing PFAS in the environment.

8. Monitoring and analytical method development

Chapters 6 and 7 present results from the Environment Agency's routine water and biota monitoring programmes using established fully quantitative analytical methods. This chapter describes non-routine monitoring and analytical approaches that are being developed to strengthen our future PFAS surveillance capability.

The work includes targeted investigations, analytical method development and exploratory studies, including assessments of PFAS in raw landfill leachate, application of the Total Oxidisable Precursor (TOP) assay and exploratory monitoring of trifluoroacetic acid (TFA) in groundwater and surface water. These studies are designed to address evidence gaps, improve understanding of PFAS occurrence in the environment and support the future evolution of the Environment Agency's monitoring capability and evidence gathering.

8.1 Analysis of raw landfill leachate

Landfills are recognised as a potentially significant source of PFAS to the environment. To better understand the PFAS content of landfill leachate, we analysed raw leachate samples collected from landfill sites across England. Data from treated leachate and final discharge effluents are not included in the analyses presented here.

The leachate samples were analysed by an external laboratory using a PFAS suite that is more limited in scope than the Environment Agency suite applied to freshwater and groundwater samples. The analytical suite comprised 54 PFAS, of which 31 were detected in more than 1% of samples and included in the assessment. These covered key short-chain and long-chain PFCAs and PFSAs, fluorotelomer PFAS and selected PFOS precursors, including PFBA, PFBS**,** PFHxA, PFHxS, PFOS, PFOA, 5:3 FTCA, N-EtFOSAA and N-MeFOSAA (see Appendix 4 for further information on PFAS names and abbreviations included in the Environment Agency analytical suite).

PFAS results have been grouped and interpreted using the same approach applied to the water data in Section 7.1. However, the Environment Agency water monitoring programme includes more than 60 PFAS determinands, spanning a wider range of precursor, ether-based, phosphate and fluorotelomer PFAS. Consequently, the leachate dataset represents a narrower range of substances, and PFAS group summaries are not directly comparable with the water-compartment analyses presented in Chapter 7.

8.1.1 PFAS patterns in raw landfill leachate

43 landfill sites were sampled between 2021 and 2024. Some sites were visited on multiple occasions (91 site visits in total), and in some cases more than one sample was collected per visit, resulting in a total of 125 leachate samples. Sites were selected to provide coverage across a range of landfill types, including operational, closed and historic

landfills, different waste ages and containment systems, and therefore provide broad representation of English non-hazardous landfill sites.

Figure 18 illustrates estimated PFAS concentrations in raw landfill leachate. Data are summarised as total concentration for each PFAS group, at the median site (central dot) and the 10th and 90th percentile sites (range of horizontal bar). The horizontal bars represent spatial variation across the sites, not uncertainty. Concentrations are displayed on a logarithmic scale spanning 1 to 10000 ng/L, reflecting multiplicative rather than linear differences in concentration.

Overall distribution and ranking of PFAS groups

Based on the estimated concentrations shown in Figure 18, Figure 19 provides a simplified ranking of PFAS groups according to their central estimates at the median site. Short-chain carbonyl and fluorotelomer PFAS have the highest central estimates, while ether-based PFAS have the lowest. The remaining groups occupy intermediate positions, with substantial variation between landfill sites and considerable overlap in their concentration ranges. The observed differences may reflect a combination of PFAS use and disposal, waste composition, precursor transformation processes and physicochemical properties.

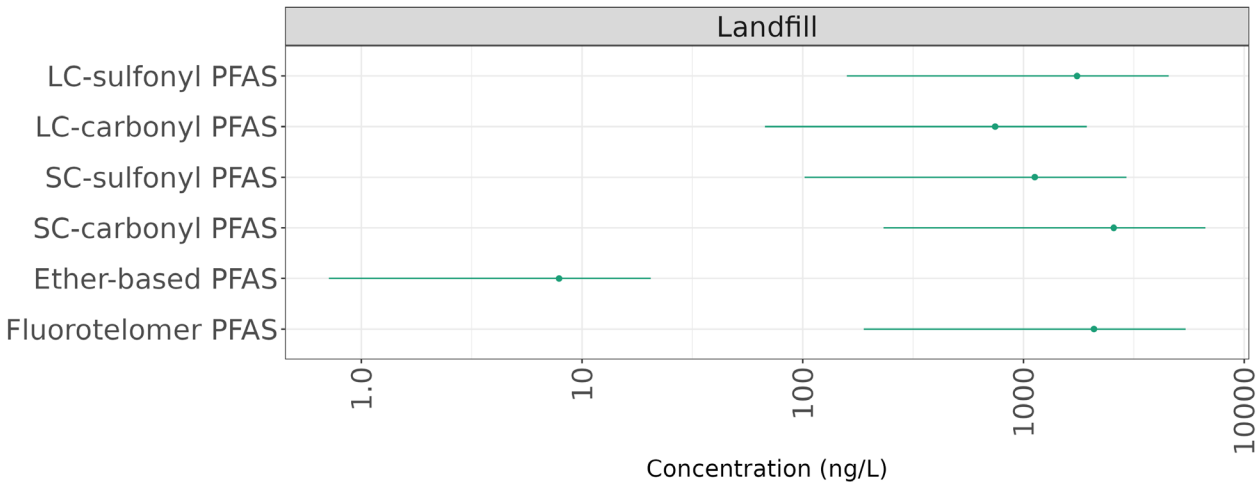


Figure 17: Spatial variability in PFAS group concentrations (ng/L) in raw landfill leachate taken from multiple sites across England. ***Points indicate the model-based estimated geometric mean concentration at the median site, while horizontal bars represent concentrations at the 10th and 90th percentile sites. Percentiles are based on ranked site concentrations. Values reflect spatial variation only.***

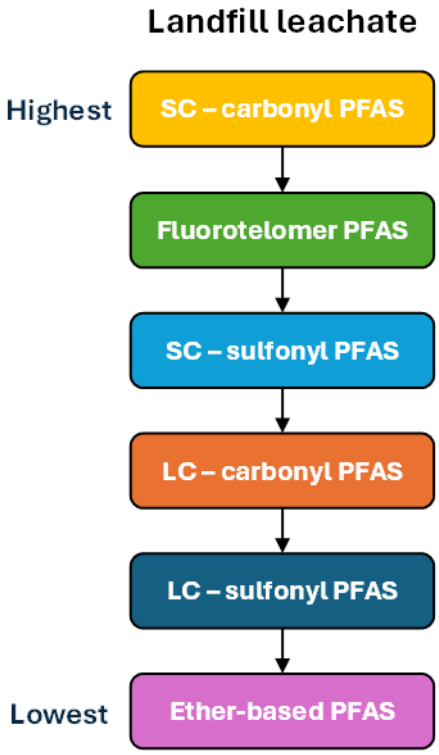


Figure 18: PFAS groups measured in landfill leachate, ranked by estimated concentration at the median site.

Short-chain carbonyl and fluorotelomer PFAS had the highest central estimates, while ether-based PFAS had the lowest. The remaining PFAS groups occupied intermediate positions, with considerable variation between landfill sites and substantial overlap in their concentration ranges, as shown in Figure 18.

The comparatively high concentrations of short-chain and fluorotelomer PFAS are consistent with their mobility and, for some substances, formation through degradation of precursor compounds within landfill environments. Overall, the results indicate that PFAS profiles in landfill leachate vary by chemical group, reflecting a combination of PFAS use and disposal, waste composition, precursor transformation and differences in environmental behaviour.

The prominence of short-chain and precursor-derived PFAS highlights the potential for landfill leachate to act as a source of mobile PFAS, particularly at older sites designed on dilute-and-disperse principles. At engineered landfills with leachate collection and management systems, releases can be controlled through appropriate leachate management.

8.1.2 Dominant PFAS in raw landfill leachate

Mean concentrations of individual PFAS in raw landfill leachate are summarised in Figure 20, expressed as means with associated 95% confidence intervals. Numerical values corresponding to the figure are provided in Table A8 in Appendix 6. The data indicate a complex PFAS profile dominated by a mixture of short-chain PFAS, selected long-chain

PFAS, and several fluorotelomer-derived compounds, with concentrations spanning more than three orders of magnitude. This variability likely reflects both the mixed nature of landfill inputs and the differing environmental persistence and mobility of individual PFAS chemistries.

Short-chain PFAS were prominent, with PFBS, PFHxA and PFBA among the most abundant substances, with mean concentrations of 650 ng/L, 610 ng/L and 530 ng/L respectively. These findings are consistent with the relatively high aqueous mobility of short-chain PFAS and their increasing use as replacements for regulated long-chain analogues in consumer and industrial products disposed of in landfills.

Among the long-chain PFAS, sulfonyl compounds were generally more prominent than carbonyls. PFHxS (160 ng/L) and PFOS (110 ng/L) were the dominant long-chain sulfonyl compounds substantially exceeding other LC sulfonyl species such as PFHpS or PFOSA, despite regulatory restrictions on their use. Several PFOS-related precursors, including N-ethyl-FOSAA and N-methyl-FOSAA, were also detected at comparatively high concentrations (38 and 36 ng/L, respectively), indicating the continued presence of PFOS precursor compounds within landfill leachate. In contrast, long-chain carbonyls such as PFNA and PFDA occurred at lower concentrations (<10 ng/L), with the notable exception of PFOA, which reached a mean concentration of 440 ng/L, highlighting its historical prevalence and persistence in landfill-derived waste streams.

Fluorotelomer PFAS form a distinct and quantitatively important component of the leachate profile in this study. In particular, 5:3 fluorotelomer carboxylic acid (5:3 FTCA) was the single most abundant substance measured in landfill leachate, with a mean concentration of 900 ng/L. Other fluorotelomer acids and sulfonates, including 6:2 FTSA and 6:2 FTAB, were also present at tens to hundreds of ng/L. The prominence of these substances suggests ongoing degradation of fluorotelomer-based consumer products within the landfill environment, leading to the release of both parent compounds and transformation products into leachate.

These results fall within the range reported for non-hazardous landfill sites internationally (US EPA, 2024) and are similar to that reported by the California Water Boards, which includes historic, closed and operational sites. However, our study identified several PFAS that are rarely reported in other international studies. Notably, substantial concentrations were observed in raw leachate for several fluorotelomer precursor compounds, particularly 7:3 FTCA, 5:3 FTCA and 6:2 FTAB. Where such precursors have not been included in analytical suites, international studies may substantially underestimate both the range of total PFAS present and the overall PFAS burden in raw leachates.

Overall, the PFAS profile observed in landfill leachate samples analysed in this study was characterised by dominance of short-chain PFAS and fluorotelomer-derived substances, alongside persistent long-chain sulfonyl compounds such as PFOS and PFHxS. Short-chain carbonyl and sulfonyl compounds account for a substantial fraction of the total PFAS burden, reflecting their high solubility and resistance to removal processes within landfill systems.

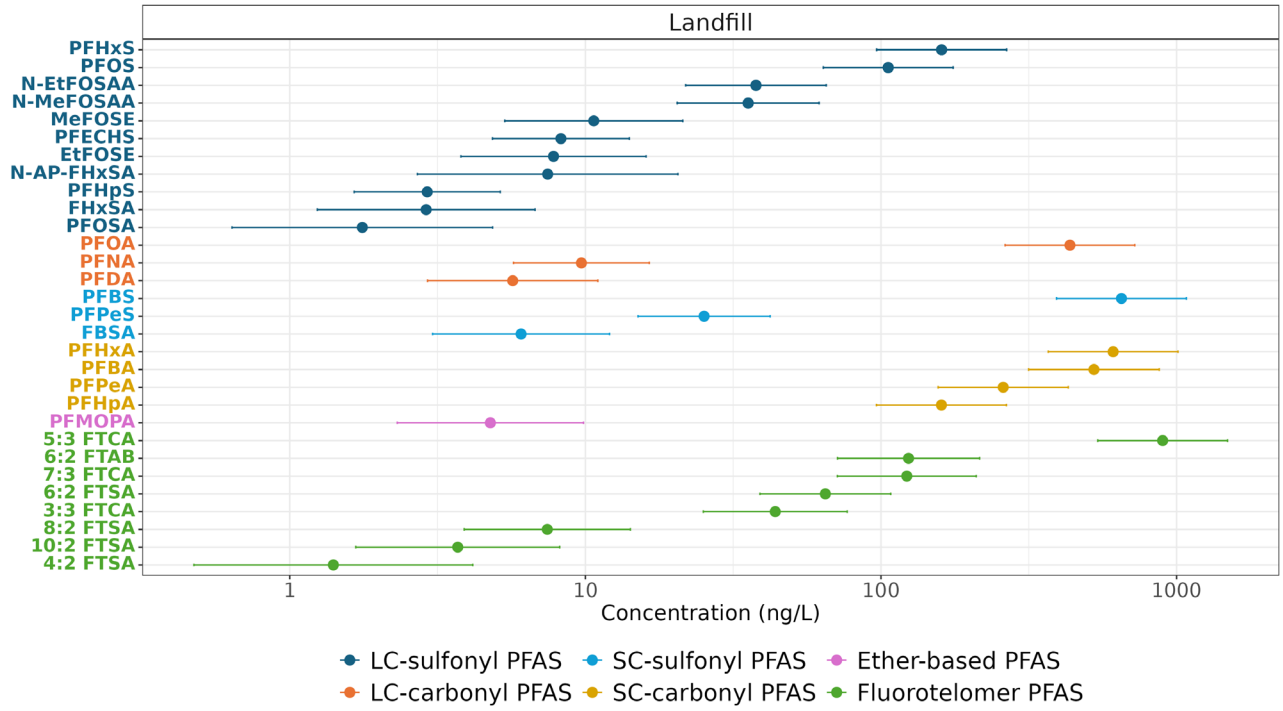


Figure 19: Geometric mean concentrations of individual PFAS in landfill leachate (ng/L) for 2025. ***Points represent model-based estimates of mean concentrations and horizontal bars represent 95% confidence intervals. Concentrations are displayed on a logarithmic scale. Colours indicate PFAS groupings. Numerical values corresponding to the figure are provided in Table A8 in Appendix 6.***

The fate and transformation of PFAS within landfills is complex and influenced by a wide range of factors, including climate, waste inputs, biodegradation and sorption, resulting in considerable variability between individual landfills. The presence of both legacy PFAS and newer replacement chemistries underscores the role of landfills as ongoing potential sources of a broad spectrum of PFAS to the aquatic environment where landfills were based on dilute and disperse principles, or where leachate emissions are not effectively controlled. In England, landfill emissions are controlled through the environmental permitting regime, with landfill operators responsible for managing leachate and preventing unacceptable releases to the environment, overseen and regulated by the Environment Agency.

It is important to recognise that high PFAS concentrations in raw landfill leachate do not necessarily translate directly into equivalent concentrations in receiving waters. Modern permitted landfills are designed with engineered containment measures, including liners, basal drainage and leachate collection systems and treatment or management controls intended to minimise releases to the environment. Leachate is commonly treated on-site or transferred to wastewater treatment works prior to discharge. Consequently, the environmental significance of landfill-derived PFAS depends not only on leachate concentrations but also on treatment performance, discharge pathways, containment effectiveness and receiving-water dilution. Assessing these factors was beyond the scope of the present analysis.

8.1.3 Comparison of PFAS concentrations in raw landfill leachate and water samples

Direct comparison between landfill leachate and water datasets is limited by differences in analytical methods, the substances measured and detection limits. The analytical suite used for these leachate samples included fewer PFAS than the Environment Agency's water monitoring programme. Comparisons between leachate analysis and water samples should be interpreted cautiously, as indicative broad patterns only and not directly or quantitatively.

PFAS concentrations are generally higher in landfill leachate, reflecting its concentrated nature. The lower concentrations of PFAS observed in groundwater, rivers and lakes reflect different sources, and the influence of environmental processes such as dilution and attenuation.

For landfill leachate, groundwater, rivers and lakes, short-chain carbonyl PFAS are the most prominent group, reflecting their high solubility and widespread use, with additional contributions from precursor breakdown in landfill leachate. Fluorotelomer PFAS show contrasting behaviour, dominating leachate samples, observed in groundwater at much lower concentrations but ranking among the least prominent groups in rivers and lakes. This likely reflects slower degradation under low-oxygen conditions, particularly in landfill settings where organic-rich, often anaerobic conditions promote prolonged transformation of fluorotelomer-based precursor compounds and the accumulation of intermediate degradation products. In contrast, long-chain PFAS occur at relatively low concentrations in leachate, likely due to sorption to solid matrices but are more prominent in rivers, lakes and biota, where they persist and bioaccumulate.

Collectively, these patterns are consistent with a source–pathway–receptor perspective, with PFAS composition varying across different environmental media as a result of transport, transformation and differential environmental behaviour.

8.2 Total Oxidisable Precursor Assay (TOP Assay)

Quantitative analysis of individual PFAS as described in Chapter 7 is useful but does not reflect the total environmental PFAS burden (Megson et al., 2024). Undetected PFAS can arise from manufacturing impurities, commercially confidential PFAS for which structural or compositional information is not publicly available, or as intermediates and breakdown products of widely used PFAS.

To address this gap, a number of analytical methods have been developed to estimate total PFAS, each with their own strengths and limitations (see Appendix 8 for examples). One method is the Total Oxidised Precursor (TOP) assay (Houtz and Sedlak, 2012). This uses hydroxyl radicals to chemically break down precursor PFAS through oxidation, converting them into terminal PFCAs.

The concentration of PFCAs in the sample is measured before and after oxidation. The difference between these measurements provides an estimate of the quantity of oxidisable

precursor PFAS present, although this approach is inherently limited to substances that oxidise to PFCAs (HSE, 2025a). It can also provide indicative information on perfluoroalkyl chain length, but interpretation is challenging and depends on the PFAS included in the analytical suite.

8.2.1 Preliminary TOP assay study on environmental water samples

To evaluate how much of the oxidisable PFAS present in environmental samples is captured by the PFAS routinely measured within our target PFAS suite, in 2024 we undertook a preliminary TOP assay study on 100 river and groundwater samples collected from 98 sites. We analysed samples using our target PFAS suite, both before and after oxidative treatment using modified TOP assay methods based on (Houtz and Sedlak, 2012). We applied minor modifications to the published TOP assay method to support routine application and quality assurance.

At the time of analysis, our analytical method did not include all substances now routinely monitored (for example, 6:2 FTAB); targeting only 49 PFAS compared with the current list of 68 substances.

In the samples analysed, 31 of the 49 PFAS analysed for were either not detected (both pre or post oxidation) or were detected in only a few samples at very low concentrations. Consequently, data analysis used the 18 adequately detected PFAS. This is described in more detail in Appendix 7 and consisted of a modified version of the full PFAS analyses undertaken in Chapter 7.

Figure 21 shows estimated pre- and post-TOP total PFAS concentrations on a molar basis (nmol/L) using our 2024 target PFAS suite. The results indicate that, across the sites sampled, our 2024 target PFAS suite captured, on average, 46% of the total oxidisable PFAS, with a 95% confidence interval of 41%–51%.

PFBA, PFHxA and PFPeA showed the largest increases in concentration following oxidation using the TOP assay, indicating a substantial contribution from oxidisable precursors that degrade to short-chain PFCAs.

The estimated percentage of total oxidisable PFAS captured by the 2024 target PFAS suite is strongly influenced by the calculation method. Alternative calculation approaches produce higher estimates and are summarised in Table 8. These differences arise from a combination of how concentrations below the MRV are handled, how results are aggregated across sites, and whether concentrations are expressed on a molar or mass basis. Approaches that calculate by summing concentrations across all sites before and after oxidation yield higher values than site-averaged estimates, as they are more strongly influenced by a small number of high-concentration sites.

Calculations based on mass concentrations (µg/L) consistently give higher estimates than molar-based calculations, reflecting differences in how higher- and lower-molecular-weight PFAS contribute to the total. This range of estimates highlights the importance of methodological choices when interpreting and comparing TOP assay estimates.

The model-based molar estimate (46%) is used as the reference value for reporting here (highlighted in Table 8) because it appropriately accounts for censored values below the MRV. Furthermore, a molar basis provides a more chemically meaningful representation of precursor transformation during the TOP assay and reduces the influence of differences in molecular weight between PFAS.

Table 8: ***Estimated percentage coverage of total PFAS captured by the EA target suite (2024) using different calculation approaches. Coverage values are percentages of total oxidisable PFAS revealed by the TOP assay. Calculation approaches differ in <MRV treatment, aggregation across sites (site-averaged versus total-sum), and whether concentrations are expressed on a molar or mass basis. CI = confidence interval, SD = standard deviation.***

Calculation Approach	Treatment of <MRV values	Basis of calculation	Estimated Coverage (%)
Model-based (GAM)	Modelled (censored data)	Molar (nmol/L)	**46** (SE = 2.5, CI = 41–51)
		Mass (µg/L)	55 (SE = 2.5, CI = 50–60)
Site-averaged raw data	<MRV = 0	Molar (nmol/L)	47 (SD = 30)
		Mass (µg/L)	57 (SD = 35)
Total-sum raw data	<MRV = 0	Molar (nmol/L)	60
		Mass (µg/L)	73

Our current (2026) PFAS suite includes additional substances such as 6:2 FTAB, reflecting growing evidence of its occurrence in environmental investigations, particularly at sites impacted by AFFFs (Macorps et al., 2023). This may increase the percentage of total oxidisable PFAS captured by the analytical suite in future TOP assay studies. In addition, the measurement of TFA (Section 8.3) is an additional consideration. We have not included TFA in this evaluation of the TOP assay as its substantially higher concentrations would dominate the results. Further data collection and analysis will help refine estimates of total PFAS as estimated by the TOP assay and better characterise variability across sites and substances.

Our results indicate that our targeted PFAS suite captures only part of the oxidisable PFAS present in monitored samples. While some of the difference may relate to PFAS that have subsequently been added to the EA monitoring programme after 2024, the identity of the remaining oxidisable PFAS cannot be determined from the current analysis and should not be assumed to arise solely from known PFAS not included in the 2024 target suite.

In addition, the TOP assay provides information only on oxidisable PFAS and does not comprehensively account for ultra-short-chain PFAS such as TFA, nor for non-oxidisable PFAS. The coverage estimates presented here should therefore be interpreted as an assessment of oxidisable PFAS captured by the target analytical suite rather than total PFAS present in the environment.

As there is currently no single standardised approach for calculating total oxidisable PFAS from TOP assay data, estimates obtained using different TOP assay protocols or calculation methods should be compared cautiously.

These findings highlight the complementary value of targeted PFAS analysis and TOP assay approaches. Together they provide insight into both known PFAS and the broader burden of oxidisable PFAS, while also indicating that additional unidentified or unmeasured PFAS may be present within environmental samples. Other total PFAS methods (see Appendix 8) may also provide complementary information to help further characterise PFAS not captured through targeted analysis or the TOP assay.

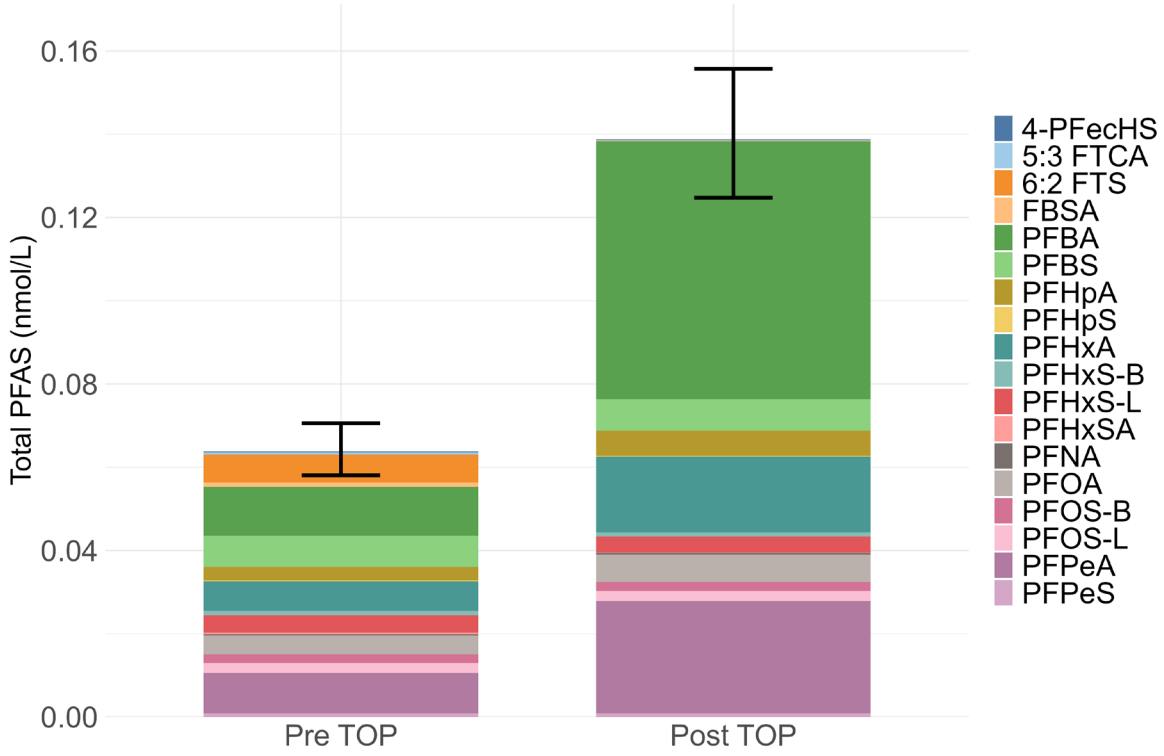


Figure 20: Model-based estimated total PFAS concentrations before and after TOP assay oxidation (molar basis). ***Stacked bar chart showing model-based estimated total PFAS concentrations expressed on a molar basis (nmol/L) before and after TOP oxidation. Error bars indicate 95% confidence intervals around the total concentrations, derived from model simulations. Colours within each bar represent the contribution of individual PFAS substances to the total concentration pre- and post-oxidation.***

8.3 Trifluoroacetic acid (TFA)

There are growing concerns about the environmental persistence, mobility and potential health impacts of ultra-short-chain PFAS compounds such as trifluoroacetic acid (TFA) (Arp et al., 2024). Recent research indicates that global TFA concentrations may exceed those of other PFAS in the environment by several orders of magnitude and continue to rise.

Our fully quantitative PFAS suite does not include TFA, which requires a separate analytical assay. To address this knowledge gap, in 2024 we validated an analytical method for TFA analysis in freshwaters. Here we present analysis and interpretation of an exploratory investigation of TFA in English waters.

Samples were collected between March and July 2024 from 49 groundwater and 50 river sites from a broad area across central England. Sites were selected from the Environment Agency's PFAS monitoring programme and were chosen to represent a range of different environments. Locations were chosen to provide a broad coverage of different land uses and geologies.

8.3.1 TFA concentrations

Concentrations of TFA measured in groundwater and surface water (rivers) are summarised in Table 9 and Figure 22. All surface water TFA concentrations were measurable above the MRV of 0.04 µg/L, whereas six out of the 49 groundwater samples were recorded below the MRV (Table 9). Summary statistics for TFA concentrations in groundwater and surface water were calculated using Regression on Order Statistics (Helsel, 2011). As only six groundwater samples and no surface water samples were below the MRV, the influence of censored observations on the summary statistics is expected to be small.

Table 9: ***Summary statistics for groundwater and surface water (river) TFA concentrations estimated using Regression on Order Statistics. Values are reported to two significant figures.***

Material	Min., µg/L	Max., µg/L	Median, µg/L	Mean, µg/L	Std. Dev.	n	Censored %
Groundwater	0.075	10	1.7	2.8	2.7	49	12
Surface water	0.24	6.4	1.7	2.6	1.9	50	0

Median TFA concentrations were identical in groundwater and surface water at 1.7 µg/L, demonstrating no difference in median concentrations between sample types when aggregated nationally. However, a wider range of concentrations were observed in groundwater than surface water (Table 9), with both the greatest maximum and smallest

minimum concentrations occurring in groundwater. These TFA concentrations are higher than those of other PFAS measured in the samples as part of our target PFAS analytical suite, typically by several orders of magnitude.

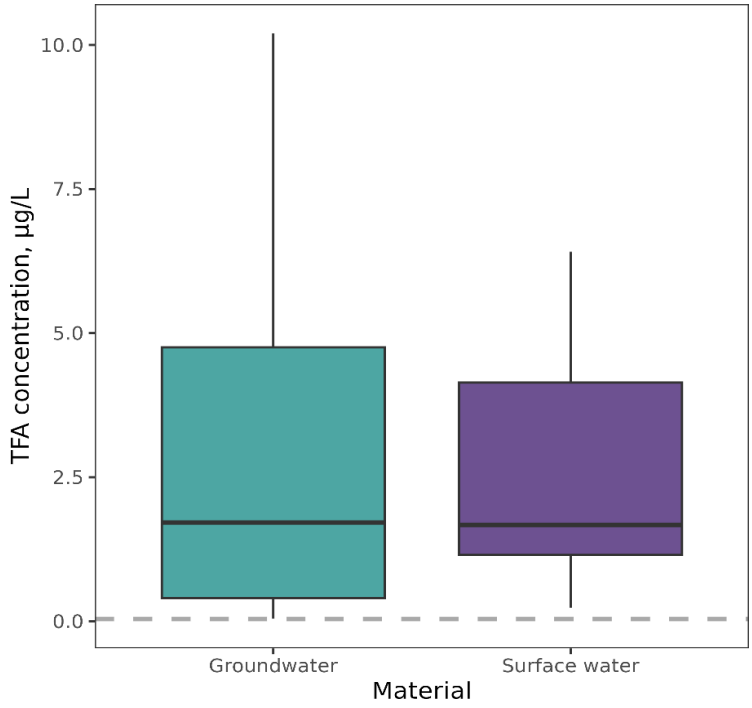


Figure 21: Boxplot of TFA concentration in groundwater and surface water (rivers) estimated using Regression on Order Statistics. ***Dashed horizontal line represents the analytical limit of detection (0.04 µg/L). The solid horizontal lines represent the median concentration, and the upper and lower limits of the boxes represent the 25th and 75th percentile values. The ends of the whiskers (vertical lines) represent 1.5x those percentiles.***

8.3.2 Spatial patterns

TFA concentrations appear to vary between catchments, particularly in groundwater (Figure 23). In surface waters, this pattern is less clear, with concentrations varying across regions and land-use types. Interpretation of these spatial patterns is complicated by the uneven geographical coverage of the monitoring network. Groundwater monitoring locations are concentrated in parts of England where monitoring resources were available, resulting in better coverage in some regions than others. As a result, some areas known to experience elevated PFAS concentrations in groundwater are underrepresented in this exploratory dataset. Consequently, the maps presented here should not be interpreted as a comprehensive national assessment of TFA occurrence, but rather as a representation of the locations sampled.

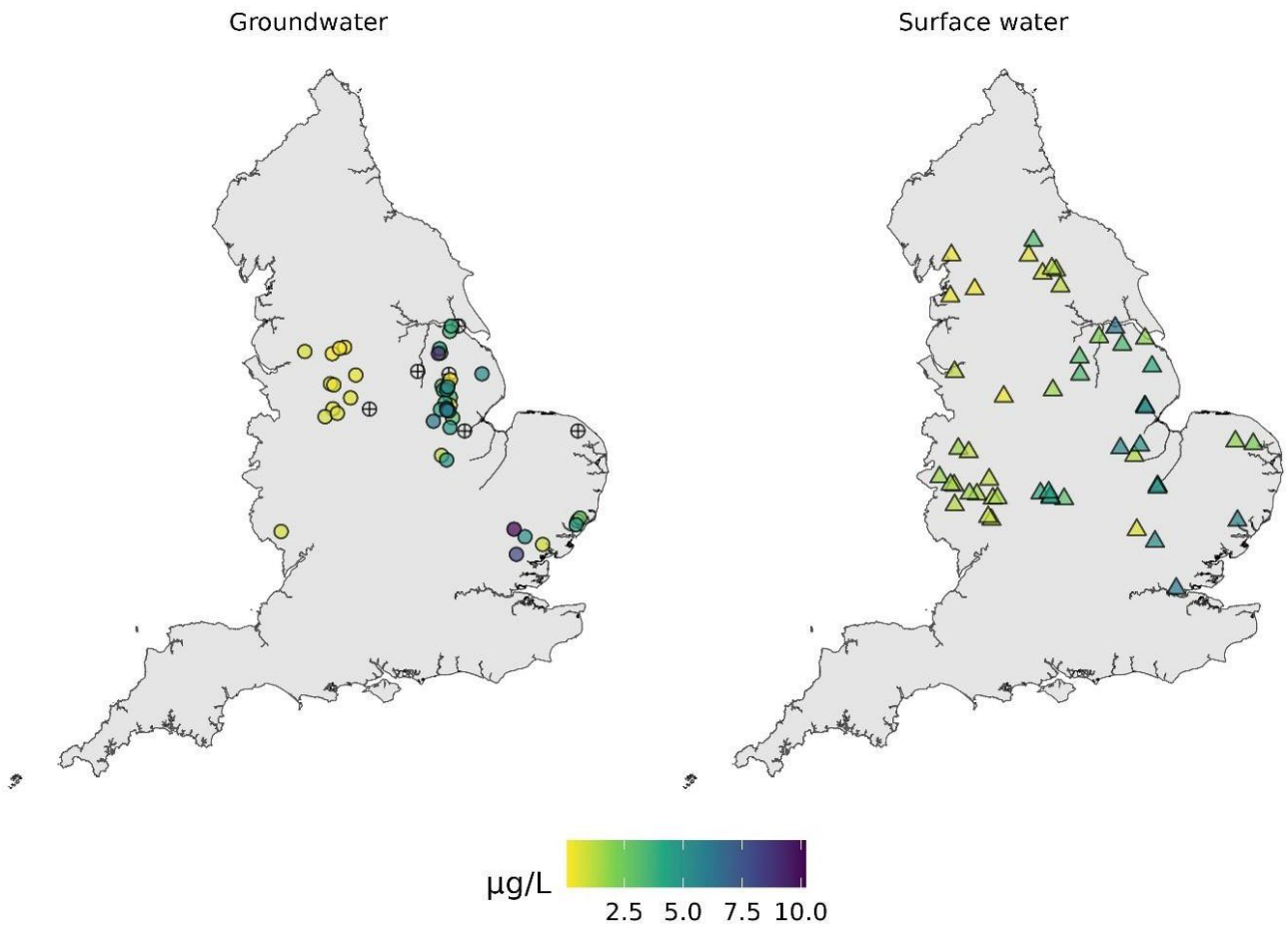


Figure 22: Spatial distribution of TFA measurements. ***Concentrations are represented by colour, with values below the MRV shown as unfilled symbols.***

TFA was detected across groundwater with various bedrock geologies (Figure 24) and censored values were present in chalk, mudstone and superficial hydrostratigraphic units (geological layers grouped according to their groundwater characteristics). Although differences between hydrostratigraphic units are apparent within the dataset, their interpretation remains uncertain. Observed patterns may reflect a combination of source distribution, groundwater vulnerability, depth to aquifer, overlying superficial deposits and local hydrogeological conditions, rather than bedrock geology alone. The current dataset does not allow these factors to be separated, and further monitoring would be required to determine their relative importance.

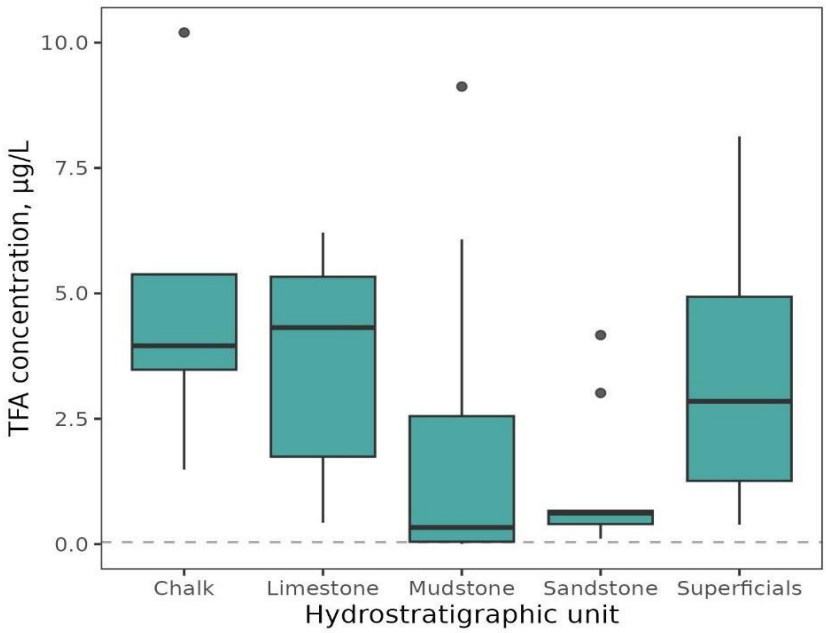


Figure 23: Regression on Order Statistics - estimated groundwater TFA concentrations grouped by hydrostratigraphic unit.

8.3.3 TFA discussion

Concentrations of TFA measured in groundwater and surface water in this exploratory study (Table 9) are consistent with those reported internationally. For example, broad-scale measurements in Germany indicated that TFA was widely detected in the range of 0.1 to 10 µg/L, with occasional exceedance of 10 µg/L in groundwater and surface waters (Sturm et al., 2023).

A recent report of TFA measurements in surface waters in the UK indicates similar patterns, with high detection frequencies (98 % of sites), an overall mean TFA concentration of 2.2 µg/L, and a maximum of 75 µg/L at one location (Fidra, 2025). This suggests that while TFA is ubiquitous, local pressures have the potential to drive spatial differences in concentrations.

Monitoring more widely for TFA in the surface water environment could help to elucidate spatial and seasonal patterns of TFA distribution. In the UK, there are currently no regulatory standards for TFA with which we can compare our data. Ecotoxicology data for TFA indicate that concentrations measured in this study are of low concern for aquatic ecosystems (ECHA, 2023; NORMAN, 2025).

TFA remains under active regulatory review. In June 2026, the European Chemicals Agency's Committee for Risk Assessment (RAC) adopted a scientific opinion supporting harmonised classification of TFA and relevant inorganic salts for reproductive toxicity Category 1B and as persistent, mobile and toxic (PMT) and very persistent and very mobile (vPvM). These were the first RAC opinions to apply the EU's newer PMT and vPvM hazard classes.

The RAC opinions inform the subsequent European Commission process but do not, by themselves, make these classifications legally binding under the EU CLP Regulation. In

GB, the hazard assessment performed for the UK REACH firefighting foam restriction notes that HSE will consider the mandatory classification of short-chain PFCAs, including TFA in due course under the GB CLP Article 37 process (HSE, 2025b). PMT and vPvM are not currently hazard classes within the GB mandatory classification and labelling framework.

The recast EU Water Framework Directive includes TFA within the group of 25 PFAS for which an EQS is specified for the sum of their concentrations (European Parliament and Council of the European Union, 2026). The revised EU PFAS EQS applies a relative potency factor to account for combined toxicity for the sum of PFAS present. The relative potency factor (RPF) approach expresses the toxicity of chemicals relative to a reference compound for a specified toxicity endpoint and species.

The reference compound is selected based on factors such as toxicity, data availability, or environmental relevance. Further explanation of this approach is provided in Critical review of approaches for determining PFAS thresholds (Environment Agency, 2026f). TFA is assigned an RPF of 0.002 relative to PFOA, reflecting its substantially lower toxicity. This means TFA is considered approximately 500 times less potent than perfluorooctanoic acid (PFOA) when concentrations are expressed as PFOA equivalents.

It remains uncertain whether observed patterns in groundwater are primarily attributable to variations in the properties of hydrostratigraphic units or whether land-use practices exert a significant influence. Further monitoring would be required to elucidate the origin of TFA in groundwater and to clarify whether observed patterns in relationships with hydrostratigraphic characteristics hold across wider spatial scales.

Interpretation of apparent differences between hydrostratigraphic units should be treated cautiously, as sample numbers within individual units are relatively limited and TFA was detected across most geological settings examined. The apparent absence or lower occurrence of TFA in some geological categories should therefore not be interpreted as evidence that TFA is absent from those environments. Additional monitoring would be required to determine whether observed differences reflect genuine hydrogeological controls or sampling variability.

In addition to local sources, atmospheric degradation of fluorinated precursor substances is increasingly recognised as an important diffuse source of TFA. These precursors include hydrofluorocarbons (HFCs) and hydrofluoroolefins (HFOs), which are used in refrigeration, air-conditioning and other applications (Section 4.1.6).

Recent research indicates that atmospheric degradation of some HFOs may contribute substantially to environmental TFA formation, with atmospheric transport and subsequent deposition via precipitation providing a widespread input pathway to aquatic environments (Garavagno et al., 2024; Khan et al., 2026). Consequently, the relative contribution of direct local sources, diffuse atmospheric inputs and precursor transformation pathways to TFA concentrations observed in England remains uncertain and warrants further investigation.

Overall, TFA is a substance with many sources, detectable in English rivers and groundwaters at concentrations comparable to those reported internationally. Our understanding of sources of TFA to the environment, their relative significance and the risks posed by TFA to ecosystem and human health is evolving. A better understanding of transformation pathways, kinetics and environmental factors influencing formation will be valuable in determining effective and targeted interventions for risk management. Additional monitoring of TFA in both surface water and groundwater would be beneficial to improve understanding of its spatial distribution, concentration trends, and potential environmental implications within England’s water environment.

9. Current work and future evidence needs

Managing risks to human health and the environment from PFAS is complex and requires coordinated action across regulators, industry and users throughout the full life cycle, from manufacture through to disposal. Effective management relies on understanding sources, exposure pathways, environmental fate and (eco)toxicology, supported by ongoing advances in monitoring, analytical capability, risk assessment and regulation.

Over the past five years, the Environment Agency has developed a highly specialised PFAS capability, bringing together scientific, regulatory and policy expertise to support national delivery and provide advice across, non-governmental organisations, academia and industry. This capability underpins the Environment Agency's role as a key delivery partner for the Government's PFAS Plan, the UK's first cross-government plan to address PFAS risks to human health and the environment.

The PFAS Plan sets out a coordinated framework for action across three pillars: improving understanding of PFAS sources, pathways and exposures; tackling PFAS pathways including strengthening regulatory controls; and reducing and managing ongoing exposure including from legacy pollution. Within this framework, we are continuing to monitor PFAS, strengthen analytical capability where needed, develop regulatory approaches through emission limits and toxicological thresholds, and contribute to the national evidence base that will inform future restriction, permitting and enforcement decisions.

In addition to progressing actions set out in the PFAS Plan, the Environment Agency continues to develop an England-relevant evidence base, addressing critical evidence gaps identified through monitoring, regulatory activity and scientific review. This is supported through collaboration across government, regulators, industry and academia and includes:

- Further developing the evidence base needed to better understand PFAS occurrence, sources, pathways and risks.
- Continuing to improve the tools, methods and data needed for effective PFAS regulation.
- Providing evidence and technical advice to inform regulatory and policy development.
- Supporting risk-based prioritisation of PFAS issues through monitoring, investigation and evidence generation.
- Improving understanding of environmental exposure to PFAS and the implications for environmental protection.
- Continuing to support the development of approaches for managing PFAS in waste, contaminated land and legacy contamination.
- Working collaboratively with partners to improve consistency, transparency and accessibility of PFAS evidence.

This programme of work supports a collaborative approach to managing PFAS across different environmental media and between regulators. Despite significant progress on this complex topic, our understanding of PFAS continues to evolve and important evidence

gaps remain relating to PFAS occurrence, behaviour, exposure and risks. A key ongoing challenge is defining what successful long-term PFAS management looks like in practice, including how progress should be measured.

9.1 Analytical methods

Reliable analytical methods are essential for identifying PFAS across environmental media and supporting the protection of human health and the environment. However, fully characterising PFAS in environmental samples remains challenging due to the large number of PFAS and the complexity of their use and production. Analytical methods also continue to evolve rapidly as understanding of PFAS exposure and behaviour develops (Androulakakis et al., 2022; Balasuriya et al., 2026; Idowu et al., 2025; RSC, 2023).

PFAS analysis can broadly be grouped into four main approaches: fully quantitative targeted analysis, semi-quantitative target screening, non-target (suspect) screening, and fluorine mass-balance methods used to estimate total PFAS burden. Advantages and limitations of these methods are summarised in Appendix 8. Several reviews provide a useful overview of these approaches, including Koch et al., (2020), McDonough et al., (2019) and De Silva et al., (2021), with additional detail provided by the ECHA PFAS restriction proposal (ECHA, 2025) and HSE PFAS in FFF restriction proposal (HSE, 2025b).

However, no single analytical method can quantify all PFAS, and current approaches provide incomplete coverage of novel PFAS, precursors, and degradation products, often requiring multiple complementary techniques. Although analytical capability and standardisation continue to improve, challenges remain around method harmonisation, quality assurance and quality control, and reliable analytical standards, particularly for total PFAS approaches. Additional analytical challenges include measurement of very low environmental concentrations, the complexity of some environmental media, contamination risks during sampling and analysis, limited availability of analytical standards, and associated cost constraints. As a result, there is no single analytical approach suitable for all regulatory decision-making across monitoring, permitting and remediation.

The Environment Agency is the chair of a UK cross-regulator analytical working group, whose purpose is to strengthen capability and develop guidance on method selection to support proportionate and risk-based application of PFAS analytical approaches across different regulatory needs. Several international initiatives and guidance resources also support discussion and development of PFAS analytical methods, including the OECD portal on PFAS , US EPA analytical methods development and sampling research on PFAS and the ITRC (Interstate Technology and Regulatory Council) resource on PFAS.

9.2 Monitoring

As described in Chapter 5 we have expanded PFAS monitoring across water in recent years, improving our understanding of PFAS occurrence and distribution in English rivers, lakes, groundwater and estuarine environments. This has generated an improved evidence base that supports risk assessment, source identification and regulatory action.

Through targeted investigations and long-term surveillance monitoring, we have developed a substantially stronger evidence base on PFAS occurrence, trends and distribution within English waters. This evidence is already supporting regulatory decision-making, risk assessment and prioritisation of further investigation. Important evidence gaps remain, particularly in relation to source apportionment, and PFAS occurrence in soils, sediments, marine environments, airborne PFAS, leachate and biosolids.

Although long-term monitoring datasets are now becoming available, continued monitoring is important for understanding trends in space and time and evaluating the effectiveness of regulatory interventions. Changes in environmental PFAS concentrations following regulatory action may take many years to become evident. We therefore continue to review and refine our monitoring approach to ensure it remains proportionate and aligned with evolving scientific understanding and regulatory priorities, recognising that maintaining long-term monitoring programmes requires sustained investment.

9.3 Data analysis and interpretation

While this report has applied current best-practice statistical approaches to analyse the available PFAS monitoring data, further method development will continue to improve interpretation of environmental patterns and risks. This includes improving approaches for analysing and interpreting data on co-exposure of multiple PFAS, especially as our datasets continue to increase in size, strengthening links between environmental exposure and ecological and/or human health risk, and improving understanding of how hydrology and environmental processes influence PFAS distribution across the environment. Continued development of screening and prioritisation tools, including the integration of additional monitoring and hazard and exposure information, would further support prioritisation, risk assessment and regulatory decision-making.

PFAS-relevant datasets are held across multiple organisations and generated through monitoring programmes developed to meet different regulatory and operational objectives. While coordination and data sharing continue to improve, opportunities remain to further integrate these datasets, providing a more coherent and comprehensive evidence base.

9.4 Ecological impacts and environmental thresholds

Our understanding of the ecological effects and long-term environmental behaviour of PFAS continues to develop, particularly for novel PFAS and precursor compounds for which (eco)toxicological information remains limited.

Our monitoring programmes provide an increasingly detailed understanding of PFAS occurrence in England, including their sources, pathways and exposure patterns across rivers, lakes and estuarine and coastal waters. However, translating these observations into assessments of ecological and human health risk remains challenging.

An important tool in environmental management is the use of regulatory thresholds and environmental standards to define acceptable limits in water, soil and air for ecosystem and human health. They are concentration limits used to inform decision-making in the assessment and management of risks associated with PFAS (and other chemicals) in the aquatic environment and discharges to water. For example, Environmental Quality Standards (EQSs) are a key tool in water quality legislation in England (Appendix 2). At the time of writing there is only one statutory EQS; for PFOS. Our monitoring shows that a much wider range of PFAS are present in rivers, estuaries and coastal waters. Consequently, we have undertaken research to determine suitable approaches for deriving thresholds for PFAS in water and air to support wider risk assessment (Environment Agency, 2026f, 2026b).

Deriving thresholds and environmental standards for PFAS is challenging for several reasons: (i) there are many PFAS, and they are chemically diverse; and (ii) sufficient reliable and relevant (eco)toxicity data are only available for a few substances. As a result, regulators globally have adopted a range of approaches to derive PFAS thresholds (Environment Agency, 2026f). These include thresholds for individual PFAS, groups of specific PFAS and total PFAS. Some approaches have used toxicology data while others are based on analytical capability. Our research (Environment Agency, 2026f) has explored the strengths and limitations of these different approaches to threshold derivation for PFAS considering both scientific basis and practical considerations to inform future regulatory approaches.

Further, we have recently derived toxicologically based water thresholds for four commonly occurring PFAS in water: PFOA, PFHxS, PFNA and PFOS (Environment Agency, 2026b). Further work is ongoing to explore the feasibility of developing thresholds for additional PFAS for which sufficient data are available.

The monitoring data presented in this report are drawn from a wide range of locations and programmes across England and are aggregated to provide a national scale overview of PFAS occurrence, sources and exposure patterns. Consequently, in this report we have not compared environmental observations with these newly derived water thresholds. Concentration limits are available for only a limited number of PFAS, and regulatory approaches continue to evolve. Where appropriate we have included the PFOS EQS for context (Section 6.1.1 and 6.2.1).

To better understand airborne sources of PFAS, we have recently reviewed evidence on emission characteristics of PFAS from key EA-regulated sources and airborne monitoring methods, to support potential future airborne/inhalation exposure assessment work (Environment Agency, 2026a). Additionally, toxicological evidence has been used to develop Environment Assessment Levels (EALs) for selected PFAS. These represent

pollutant concentrations in ambient air below which minimal or no appreciable risks to human health are expected and are used in setting permit limits.

While laboratory studies and field investigations have demonstrated that some PFAS can accumulate in organisms and may cause adverse effects, uncertainty remains about how these effects translate to populations and ecosystems under real-world environmental conditions. Further evidence is therefore needed to better understand the relationships between environmental exposure, bioaccumulation and ecological effects, and their significance for environmental risk assessment. Several major UK research programmes, including work involving the Environment Agency, are helping to address these and other evidence gaps. These include the UKRI-funded UNSaFE (Understanding the Scale, Sources, Fate and Effects of PFAS Pollution) project, which is investigating PFAS occurrence, environmental behaviour, bioaccumulation and biological effects in UK waters, and developing evidence to support environmental risk assessment. The P-FASE (PFAS Fate and Speciation in the Environment) project is also investigating the environmental fate and behaviour of PFAS.

9.5 Treatment technologies and waste management

PFAS remediation remains technically complex and costly, with effectiveness influenced by chemical diversity, site-specific conditions, and the presence of mixtures and precursors. Preventing releases to the environment is always preferable to remediation once contamination has occurred.

Significant uncertainty remains around the long-term management and disposal of PFAS-containing wastes, including whether landfill and land spreading provide environmentally protective disposal routes, and the development of appropriate waste acceptance criteria. Existing regulatory frameworks were not designed to manage highly persistent and chemically diverse substances such as PFAS throughout their life cycle, especially for extremely low concentrations. A lack of supply chain transparency and incomplete information on PFAS use in products can also make it difficult to identify sources of contamination, assess risks and manage PFAS effectively.

Sequestration technologies such as activated carbon, ion exchange and membrane filtration can effectively remove some PFAS from water. However, they do not destroy PFAS and instead generate concentrated waste streams that require further management. Some of these technologies may also be less effective for some short-chain PFAS (Environment Agency, 2025a).

Destructive treatment technologies continue to develop and show promise, although many remain at the pilot or early deployment stage. Challenges include cost, energy demand, scalability, by-product formation and the need for long-term validation under field conditions. More evidence is also needed on the movement of PFAS throughout the waste life cycle, from their use in consumer products to end-of-life pathways and their transfer through waste and water management systems.

The Environment Agency has recently developed a Lines of Evidence approach for assessing the effectiveness of PFAS remediation technologies. The framework has evolved alongside new and emerging techniques for treating both solids and liquids and now incorporates the important treatment-train concept (a sequence of treatment steps). This will establish a clear, consistent method for both regulators and industry to follow when preparing remediation options appraisals and remediation strategies, improving consistency, quality, and efficiency in both development and regulatory review.

Continued development of the evidence base, treatment technologies and regulatory approaches will be needed to support the effective long-term management of PFAS-containing wastes. However, preventing releases at source is likely to remain the most effective means of reducing long-term environmental risks and costs (Ling et al., 2026). Remediation and treatment technologies will continue to play an important role in managing higher-risk sources, contaminated sites and concentrated waste streams, alongside wider efforts to reduce emissions.

9.6 Regulatory challenges

Responsibilities for PFAS are distributed across multiple regulatory regimes and bodies (Appendix 2). As understanding of PFAS develops, further work will be needed to support more consistent and proportionate approaches across monitoring, permitting, remediation and wider regulation. This includes improving how emerging concepts and methods such as Persistent, Mobile and Toxic (PMT) classification approaches and total PFAS methods can be applied within existing regulatory frameworks (DEFRA, 2025).

The evidence base for PFAS has expanded considerably in recent years, supporting regulatory decision-making. However, implementing regulatory changes across a large and diverse group of substances can be challenging. There is ongoing discussion regarding the use of group-based regulatory approaches, essential-use concepts, and the extent to which PFAS with differing properties and uses should be managed together or separately.

Translating increasingly complex monitoring and analytical evidence into proportionate and practical regulatory measures also remains challenging, particularly where PFAS sources and pathways are diffuse or where risks vary between substances. Regulatory frameworks and risk assessment approaches will therefore continue to evolve alongside advances in scientific understanding, analytical capability and environmental monitoring. Applying the precautionary principle remains important where there is potential for serious or irreversible environmental harm.

While PFAS present significant and distinctive regulatory challenges, it is important that efforts to understand and manage PFAS do not divert attention from the wider range of chemical pressures affecting the environment and human health. PFAS should therefore be considered within the broader context of chemicals management rather than as a standalone issue.

Many of the approaches developed to address other forms of pollution, including source control, risk-based regulation, monitoring, permitting and remediation, are directly applicable to PFAS and can help inform proportionate regulatory responses. Equally, the scientific, regulatory and operational lessons emerging from PFAS management are likely to strengthen wider approaches to chemical pollution, supporting the development of more integrated and preventative chemicals management frameworks in the future.

10. Summary

This report brings together current evidence on the sources, pathways, occurrence and environmental behaviour of PFAS in England, alongside recent developments in monitoring, analysis, and operational and regulatory activities. Since publication of our previous report (Environment Agency, 2021), we have made substantial progress in understanding the occurrence and distribution of PFAS in the environment and in developing the tools needed to assess and manage risks.

Over the past five years, we have significantly expanded our PFAS capability. Monitoring has progressed from a relatively small number of legacy PFAS analysed using semi-quantitative methods to fully quantitative analysis of more than 60 PFAS in water and 45 PFAS in biota. The expanded analytical suite includes short- and long-chain carbonyl and sulfonyl PFAS, fluorotelomer compounds, ether-based PFAS and other precursors. Together, these developments provide one of the most comprehensive PFAS environmental monitoring programmes globally.

Our understanding of PFAS sources has also improved considerably. Through engagement with regulated industry, targeted investigations, Environmental Permitting Regulation 61 information notices and sector-specific assessments, we have developed a more detailed picture of PFAS use and emissions across England. This work has been supported by our PFAS Risk Screening Project, which developed a national GIS-based Risk Explorer tool. The Risk Explorer has been used to screen more than 40,000 sites across England and help identify and prioritise site types that may warrant further investigation and regulatory action. These include civilian and military airfields, firefighting training grounds, fire stations, wastewater treatment works, landfills and a range of industrial sites.

The identification of approximately 40,000 potentially relevant sites should not be interpreted as meaning that all of these sites require investigation. The purpose of the screening tool is to support prioritisation by identifying sites with characteristics associated with a higher likelihood of PFAS use or release. Depending on the threshold applied, the number of sites identified for potential follow-up is likely to be substantially lower, ranging from approximately 2,900 sites using the lower threshold to around 10,200 sites using the higher threshold.

Any subsequent investigation would need to be prioritised using a risk-based approach that considers source type, pathways, receptors, existing evidence and available resources. As the evidence base develops and our understanding of PFAS occurrence, exposure pathways and environmental risks improves, the prioritisation criteria and investigation requirements are likely to evolve.

Our long-term monitoring programme provides clear evidence of declines in PFOS and PFOA concentrations in English rivers over the past decade. Concentrations of these substances have declined by approximately 30% and 29%, respectively, since 2016. This corresponds to mean annual reductions of around 4% for PFOS and 3–4% for PFOA, with clear seasonal variation also evident in river datasets. We also observed similar reductions

in freshwater fish, where PFOS concentrations have declined by approximately 49% over the same period, with more modest declines observed in coastal fish. These trends provide strong evidence that restrictions on use, changes in industrial practice and targeted regulatory interventions are contributing to measurable environmental improvements.

The picture is less clear elsewhere. Lakes generally exhibit higher concentrations of legacy PFAS than rivers, which may reflect longer residence times and stronger sediment-water interactions. In estuarine waters, overall declines are evident for PFOA, but PFOS concentrations do not show the same clear pattern of decline. The relative influence of sediment remobilisation, hydrological conditions and local source inputs remains uncertain. Coastal waters generally show lower concentrations and broad indications of decline, although smaller datasets reduce confidence in trend interpretation. The persistence of PFAS and their redistribution through multiple environmental pathways mean that improvements in environmental concentrations are unlikely to occur uniformly across environments, even where emissions have been substantially reduced.

Our monitoring data indicate a changing PFAS profile across England. While concentrations of some legacy long-chain PFAS are declining, short-chain PFAS are dominating concentrations in rivers, lakes, groundwater and landfill leachate. These substances are highly mobile, persistent and resistant to many conventional treatment technologies. Groundwater and landfill leachate also show strong fluorotelomer signals, suggesting an important role for PFAS precursor compounds and their transformation products in these media. In contrast, biota monitoring demonstrates that PFOS remains the dominant contributor to PFAS burdens in fish.

Our monitoring data also demonstrate that PFAS rarely occur as individual substances. Instead, they are typically present as complex mixtures containing multiple substances with differing environmental behaviours. These findings support the growing recognition that grouping approaches are likely to provide a more effective basis for long-term risk management than assessments focused solely on individual substances.

We have also demonstrated the value of complementary analytical approaches. Application of the TOP assay indicated that our targeted PFAS suite captures approximately 46% of oxidisable PFAS. This suggests that a proportion of PFAS present in environmental samples may not be measured through targeted analysis alone. Similarly, exploratory monitoring of trifluoroacetic acid (TFA) showed that this ultra-short-chain PFAS is widely present in groundwater and surface waters across England, often at concentrations orders of magnitude higher than those of other monitored PFAS and broadly comparable with levels reported internationally.

The evidence developed to date is already supporting regulatory action, operational targeting and policy development, however important evidence gaps remain. Further work is needed to improve understanding of PFAS in soils, sediments, air and marine environments. This is particularly true for soils, where comparatively little monitoring data are currently available. However, while soils and sediments can act as important long-term

reservoirs of PFAS, their large-scale monitoring is technically challenging and resource intensive.

Working with others, our research and science support and inform continued regulatory development and risk management actions for PFAS, helping to improve understanding of priority evidence gaps.

Overall, the evidence presented in this report demonstrates that action to reduce PFAS emissions can deliver measurable environmental improvements. However, these improvements have taken many years to emerge, reflecting the persistence of PFAS and the continued influence of historic contamination.

Managing PFAS remains a complex challenge that will require continued collaboration between government, regulators, industry, academia and research organisations. Nevertheless, significant progress has been made. Through sustained investment in monitoring, research and regulatory development, the Environment Agency has helped transform understanding of PFAS in England, while also making important contributions to the international evidence base. This work has informed policy and regulation, improved understanding of sources, pathways and risks, and established the scientific and regulatory foundations needed to support effective long-term management of PFAS. In doing so, it has not only advanced the management of PFAS risks but also strengthened the evidence and approaches available to address chemical pollution more broadly.

Appendix 1: Definition of PFAS and overview of types of PFAS

The definition of PFAS has been widely debated in the scientific literature, and there is no single globally agreed definition. Most definitions are based on structural characteristics or the length of their carbon chains. Definitions have been refined over time by regulatory bodies and researchers to better reflect their properties, behaviour, and regulatory relevance. As a result, approaches vary between organisations meaning the number of substances defined as PFAS differs depending on the definition used. The following outlines key definitions:

Definition by Buck et al. (2011)
Defines PFAS as "*highly fluorinated aliphatic substances that contain 1 or more C atoms on which all the H substituents (present in the nonfluorinated analogues from which they are notionally derived) have been replaced by F atoms, in such a manner that they contain the perfluoroalkyl moiety* C_nF_{2n+1}*–.*"

OECD definition OECD (2018)
In 2018, the OECD published the results of efforts by the OECD and the United Nations Environment Programme (UNEP) Global PFC Group to identify how many PFAS were on the global market; a total of 4,730 PFAS were believed to be in worldwide use (OECD, 2018). These substances were defined as those that contain a perfluoroalkyl moiety with 3 or more carbon atoms (CnF2n–, n ≥ 3) or a perfluoroalkyl ether moiety with 2 or more carbon atoms (CnF2nOCmF2m−, n and m ≥ 1).

Updated OECD Definition OECD (2021)
In a subsequent update, the OECD (2021) revised its 2018 PFAS definition, recognising these substances as "*fluorinated substances that contain at least one aliphatic carbon atom that is both saturated and fully fluorinated (without any H/Cl/Br/I atom attached to it), i.e. any chemical with at least a perfluorinated methyl group (–CF3) or a perfluorinated methylene group (–CF2–) in its structure*." This definition encompasses a broad range of substances with diverse chemical structures and properties and expands the number of substances classified as PFAS.

However, fluorinated aromatic substances fall outside the current OECD (2021) definition, so may potentially be used as substitutes for them even though they might possess similar properties (e.g. high persistence and mobility in the environment). OECD (2021) recognised this gap and encouraged further work on this type of compound.

To address this, the Environment Agency undertook a scoping study of the uses and hazards of fluorinated aromatic compounds. Literature and regulatory database searches identified around 120 substances containing three or more fluorine atoms bound to an aromatic ring that could potentially be used in the UK, although only a small number appear to be commercially important. Reported uses included industrial intermediates, pesticides, biocides, pharmaceuticals and veterinary medicines. Available evidence does not suggest that these substances are being used as large-scale alternatives for PFAS.

Instead, they appear to be used primarily in the manufacture of biologically active molecules. While hazard data were limited, available evidence suggests that some fluorinated aromatic substances, or their transformation products, may exhibit persistent and potentially mobile properties, warranting further investigation (Environment Agency, 2026g).

US EPA Definition (2021)
The US EPA's Office of Pollution Prevention and Toxics (OPPT), which administers Toxic Substances Control Act (TSCA), has developed the following working definition of PFAS: "*a structure that contains the unit R-CF2-CF(R')(R''), where R, R', and R'' do not equal "H" and the carbon-carbon bond is saturated (note: branching, heteroatoms, and cyclic structures are included).*" This definition accounts for diverse structural variations, including branching, heteroatoms, and cyclic forms, while notably excluding substances with isolated $–CF_3$ groups, such as certain pharmaceuticals or TFA.

Discussion by Buck et al. (2021)
Buck et al. (2021) opposed broadening the PFAS definition to include "'*substances containing only an isolated –CF2– group in the carbon backbone*", as these would be expected to be readily susceptible to biodegradation leading to non-PFAS transformation products (HSE, 2023).

RMOA definition (HSE, 2023)
The HSE RMOA introduced a refined and narrower working definition of PFAS "*fluorinated substances that contain at least one fully fluorinated methyl carbon atom (without any hydrogen, chlorine, bromine or iodine atom attached to it), or two or more contiguous perfluorinated methylene groups (-CF_2-)*" thereby removing the criterion of a single -CF_2- group. This approach was used to narrow the number of PFAS down to hundreds. However, this does not preclude the use of alternative definitions in future risk assessment or management work.

Gaines et al. (2023)
Gaines et al. (2023b) propose that a substance should be classified as PFAS if it either contains one of four specific fluorinated chemical substructures, or if at least 30% of its atoms (by count) are fluorine. This hybrid approach moves beyond purely structural definitions, which the authors argue are inconsistent and hard to apply to complex molecules.

Non-polymeric PFAS

PFAS can be divided into two major classes: non-polymeric and polymeric (Figure A1). Polymers are large molecules made by bonding many repeating smaller subunits called monomers. Non-polymeric PFAS are molecules or compounds that do not have this repeating unit structure.

Non-polymeric can be further categorised into two sub-classes: perfluoroalkyl and polyfluoroalkyl substances, each further divided into many groups and subgroups.

Perfluoroalkyl substances - Perfluoroalkyl substances are characterised by a fully fluorinated carbon chain where all hydrogen atoms in the alkyl chain are replaced by fluorine atoms (C_nF_{2n+1})

Polyfluoroalkyl substances - Polyfluoroalkyl substances are aliphatic substances in which all hydrogen atoms attached to at least one (but not all) of the carbon atoms have been replaced by fluorine atoms in such a manner that they contain the perfluoroalkyl moiety (-C_nF_{2n}-). Polyfluoroalkyl substances have the potential to be transformed into perfluoroalkyl substances under appropriate conditions.

Fluorotelomer based substances are polyfluoroalkyl substances and are named using an “n:x” prefix where “n” indicates the number of fully fluorinated carbon atoms (n >2) and “x” indicates the number of carbon atoms that are not fully fluorinated (x > 1). Examples include fluorotelomer substances such as 8:2 fluorotelomer alcohol (8:2 FTOH), which has 8 fully fluorinated carbon atoms and 2 terminal carbon atoms with no fluorine atoms attached (Buck et al. 2011), and 10:2 fluorotelomer alcohol (10:2 FTOH).

Polymeric PFAS

Fluorinated polymers are considered to be PFAS if they contain per- or polyfluoroalkyl moieties (chemical groups or structural units). Polymeric PFAS are made from one or more PFAS monomers or involve the use of a fluorosurfactant processing aid (Buck et al., 2011). Polymeric PFAS typically have different physicochemical and toxicological profiles compared to non-polymeric PFAS, reflecting their larger molecular size and more complex structures. The most commonly identified groups of polymeric PFAS include side-chain fluorinated polymers (SCFPS), perfluoropolyethers (PFPEs), and fluoropolymers. However, they can be a source of lower molecular weight PFAS due to degradation of poly/perfluorinated side chains, or the presence of residual manufacturing impurities (such as processing aids) that are PFAS themselves.

Fluoropolymers – Fluoropolymers are typically made by the (co)polymerisation of unsaturated monomers at least one of which contains fluorine atoms bound to one or both of the unsaturated carbon atoms, to form a carbon-only polymer backbone with fluorine atoms directly attached (Buck et al., 2011). Examples include:

• polytetrafluoroethylene (PTFE);

• polyvinylidene fluoride (PVDF);

• fluorinated ethylene propylene (FEP);

• perfluoroalkoxyl polymer (PFA).

Henry et al. (2018) argued that fluoropolymers are distinctly different from other polymeric and non-polymeric PFAS due to their thermal, chemical, photochemical, hydrolytic, oxidative and biological stability. They have negligible residual monomer and oligomer content and low to negligible leachability. Fluoropolymers have very high molecular weights, are practically insoluble in water and are not subject to long-range transport (Henry et al., 2018). However, other researchers dispute this due to non-polymeric PFAS being emitted at other stages of fluoropolymer lifecycle such as manufacturing and

disposal (Lohmann et al., 2020). Given evidence suggesting that fluoropolymers differ from other PFAS, further work is warranted to better understand these differences (ECCC and Health Canada, 2024). Perfluoropolyethers differ from other fluoropolymers in that they have a C-O-C backbone.

Side-chain fluorinated polymers (**SCFPs**) - SCFPs have a non-fluorinated polymer backbone with fluorinated side chains, ending in $–C_nF_{2n+1}$. Examples are polymers derived from 8:2 FTOH. These fluorotelomer-based polymers may degrade into non-polymer PFAS through their lifecycle (Lohmann and Letcher, 2023).

Perfluoropolyethers (PFPEs) - have an ether polymer backbone with F atoms directly attached, having backbone units such as $-CF_2-$, $-CF_2CF_2-$, and $-CF(CF_3)\,CF_2-$ units separated by oxygen atoms (Buck et al. 2011). PFPEs are generally stable under environmental conditions, although some lower molecular weight PFPEs may be bioavailable, mobile and volatile. Under use-related conditions, PFPEs may degrade to form non-polymeric PFAS. Their environmental behaviour and potential risks vary according to molecular weight and structure (ECCC and Health Canada, 2024). Figure A1 below illustrates a simplified chemical taxonomy of polymer and non-polymer PFAS.

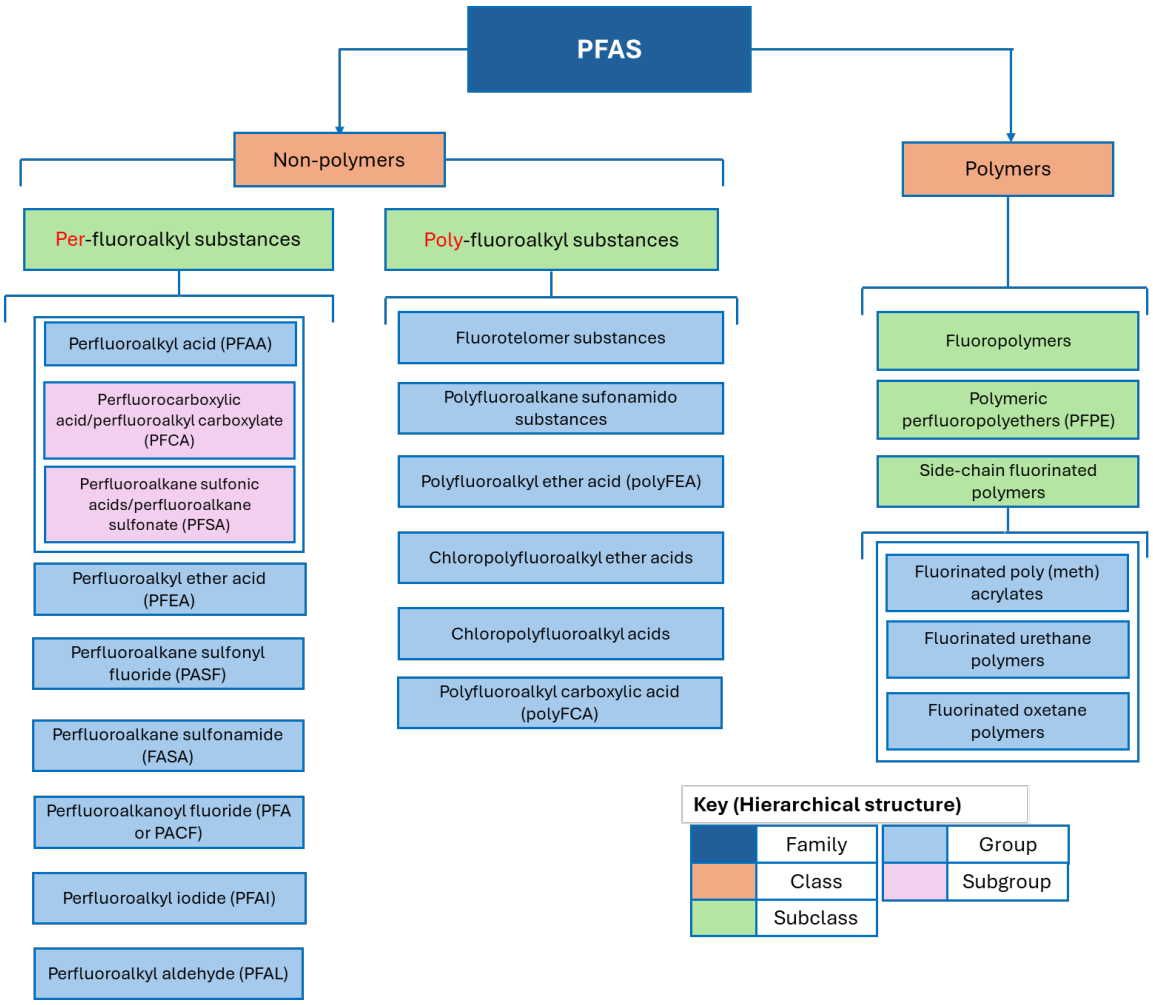


Figure A1. Simplified chemical taxonomy of PFAS, ranging from the family level to the subgroup level, modified and redrawn from (ITRC, 2023).

Appendix 2: PFAS Regulatory frameworks

The Environment Agency does not directly regulate products such as cosmetics, personal care products, toys, or other consumer goods, which are regulated by the Office for Product Safety and Standards (OPSS) and relevant Trading Standards authorities. However, the Environment Agency may have an enforcement role where such products contain restricted PFAS above relevant concentration thresholds under UK REACH or the POPs Regulations.

Local Authorities are the lead organisation for contaminated land under the contaminated land regulations (e.g. Part 2A), unless a site is designated as a special site, when the Environment Agency leads.

Central Government, particularly Defra, is responsible for introducing restrictions under UK REACH and setting statutory standards, but informed by the HSE, with support from the EA and other regulators. Other public bodies, including HSE and UK Health Security Agency (UKHSA) are responsible for assessing human health risks. The Environment Agency provides expert advice and evidence to support these processes.

The authorisation of plant protection products (PPPs) and biocides is regulated by the HSE, while pharmaceuticals and veterinary medicines are overseen by the Medicines and Healthcare products Regulatory Agency (MHRA) and the Veterinary Medicines Directorate (VMD), respectively.

The Drinking Water Inspectorate (DWI) is responsible for ensuring that public drinking water supplied by water companies is wholesome. This includes setting non-statutory guidance values for PFAS in drinking water, overseeing monitoring and reporting by water companies, and providing regulatory oversight of drinking water quality. Local authorities have a duty to regulate private water supplies.

The Food Standards Agency (FSA) is responsible for protecting public health in relation to food safety. This includes assessing and monitoring PFAS in food, undertaking surveillance programmes, and providing public health advice on potential dietary exposure to PFAS.

Centre for Environment, Fisheries and Aquaculture Science (Cefas) and the UK Centre for Ecology & Hydrology (UKCEH) provide scientific evidence, research and monitoring data to support policy development, environmental assessment and regulatory decision-making.

UK Registration, Evaluation, Authorisation and Restriction of Chemicals (UK REACH)

UK REACH regulates the manufacture, import, and use of industrial and consumer chemicals in Great Britain (GB) following the UK's exit from the EU. This means it applies to most uses of individual PFAS whether on their own, in mixtures, or in articles (i.e., finished goods such as clothing, furniture, and electronics). Other uses, such as in pesticides and pharmaceuticals, are covered by separate regulations. The Environment

Agency supports the implementation of measures adopted under UK REACH Regulations to protect the environment.

Despite this, the UK REACH registration database has significant limitations in the information it can provide about the overall PFAS market in GB. Substances manufactured or imported by individual companies in quantities below one tonne per year are not required to be registered, and polymers are exempt from registration. Rather than registering the polymer itself, manufacturers and importers are required to register the monomers and other reactants used to produce the polymer. The UK REACH registration database therefore does not provide information on the volumes or uses of fluoropolymers, which may act as ongoing sources of PFAS emissions to the environment through degradation, abrasion, or weathering over time. However, polymers are not exempt from other elements of REACH, including restriction. Consequently, polymers, including fluoropolymers and other polymeric PFAS, may still be subject to regulatory controls where risks to human health or the environment are assessed and warranted.

In addition, registration information remains incomplete due to transitional arrangements following the UK's exit from the EU. Full registration requirements for most PFAS are not expected to apply until 2029 to 2031. Furthermore, most PFAS are not currently classified as hazardous under the Classification, Labelling and Packaging (CLP) Regulation, meaning that even when they are registered, companies are generally not required to submit exposure scenarios or risk assessments.

There is currently no comprehensive mechanism for collecting information on individual PFAS present in articles or in imported polymers placed on the UK market, unless a substance is identified as a Substance of Very High Concern (SVHC). Several PFAS have been placed on the Candidate List of SVHC because of their hazardous properties. Some additional PFAS are being considered for addition during 2026/27, and the EA will draft the environmental hazard sections of the proposals where relevant. Candidate Listing triggers some obligations on REACH registrants and suppliers. Article importers must also inform HSE if they are importing an SVHC above a specific concentration limit in annual amounts above 1 tonne. However, there is no comprehensive mechanism for collecting information on the PFAS content of imported goods.

The PFAS regulatory management option analysis (RMOA) published by the HSE in collaboration with the Environment Agency, reported that 36 PFAS are officially registered, with a further 40 more potentially subject to registration at the time of reviewing the UK REACH registration database (June 2022) (HSE, 2023). These UK REACH PFAS are registered at volumes above 1 tonne annually. However, as discussed, a significant quantity of PFAS may remain unregistered due to low production volumes and imports, their presence in finished goods, or exemptions from registration requirements.

Currently, UK REACH restricts one PFAS (perfluorinated silane, entry number 73). The EA published non-statutory evaluations of nine PFAS in 2023 (Environment Agency, 2023d, 2023e, 2023f, 2023b, 2023g, 2023c, 2023h, 2023i, 2023j). These reports describe the life cycle of some representative substances produced or used by the main PFAS manufacturers in England, as well as F-53B, which was identified in an English river

despite not being associated with any manufacturing sites. They highlight REACH registrations have limited detail about uses and environmental exposure.

The HSE, in collaboration with the Environment Agency, has been commissioned by Government to develop a UK REACH restriction on the placing on the market and use of PFAS in firefighting foam. The final restriction recommendations are expected to be published later in 2026. Following this, the Government will decide whether the restriction should enter into force.

Stockholm Convention on Persistent Organic Pollutants (POPs)

The United Nations Stockholm Convention on Persistent Organic Pollutants (POPs) was adopted in 2001 and entered into force as international law in 2004. The evolution of PFAS regulation under the Stockholm Convention and POPs Regulations from 2016 is summarised in Figure A2.

As a signatory, the UK implements these obligations through domestic legislation. Listing a substance under the Stockholm Convention imposes global restrictions on its production and use and has implications for waste containing that substance. A substance can be nominated as a POP if it meets certain criteria, including persistence, bioaccumulation, toxicity and the potential for long-range environmental transport. Only a relatively small number of PFAS currently meet the criteria for listing. Candidate POPs may undergo a review process lasting several years before being added to the Convention's annexes. There are general exemptions which allow the selling of POPs to laboratories for research and development purposes.

Where waste contains a PFAS POP above the relevant concentration thresholds set out in Annex IV, it must be classified as POPs waste. Such wastes must be managed through disposal or recovery operations that destroy or irreversibly transform the POP content, typically through high-temperature incineration, preventing it from circulating further in the environment.

In 2009, PFOS, its salts, and PFOSF (a key precursor used to manufacture PFOS) were listed under Annex B (restricted use) of the Stockholm Convention. Since then, three additional PFAS have been added to Annex A (elimination), as follows:

- **2019**: Perfluorooctanoic acid (PFOA), its salts, and related compounds.
- **2022**: Perfluorohexane sulfonic acid (PFHxS) and PFHxS-related substances.
- **2025**: Long-chain perfluorocarboxylic acids (LC-PFCAs), their salts, and related compounds, with the ban scheduled to take effect in 2026 (Department for Environment, Food & Rural Affairs, 2026).

These decisions reflect an evolving global effort to address the risks posed by PFAS and protect human health and the environment. The UK implements its obligations under the Stockholm Convention through the Persistent Organic Pollutants (POPs) Regulations. Responsibility for enforcement is devolved, with the Environment Agency acting as the competent authority for England and equivalent environmental regulators fulfilling this role in the other UK nations.

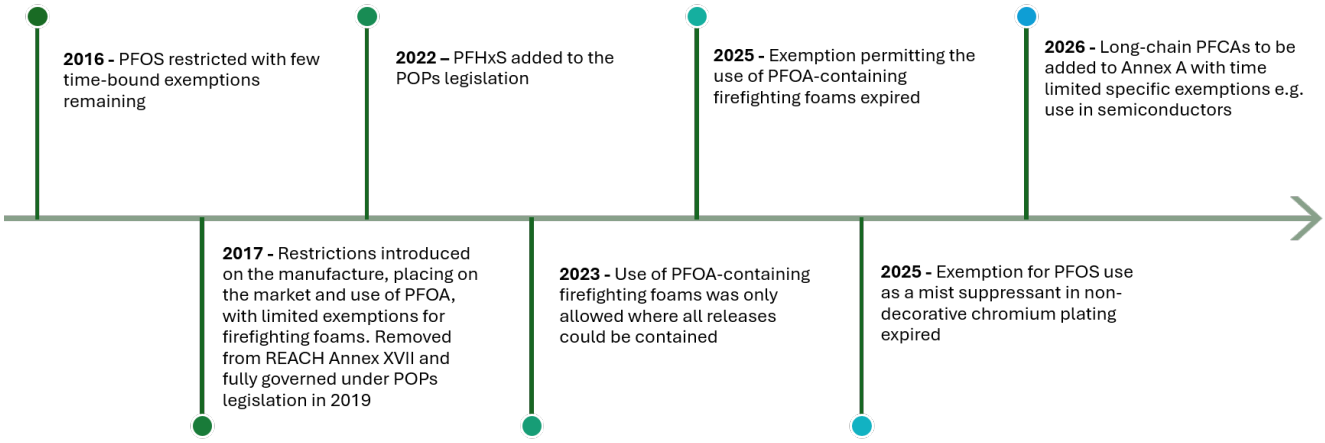


Figure A2. Regulatory milestones and interventions for PFAS under the POPs regulations between 2016-2026

Environmental Permitting Regulations

The manufacture of PFAS, among other activities, can indirectly fall within the scope of the Environmental Permitting Regulations 2016 (EPR). These regulations apply to a wide range of industrial installations, waste operations, water discharge activities and groundwater activities. An environmental permit is required to operate a “regulated facility”, and they are enforced in England by the Environment Agency.

EPR ensures that permitted activities arising from manufacturing, use, storage, testing, training, treatment, and disposal prevent pollution or, where that is not practicable, reduce emissions to a minimum to protect the environment and human health. However, not all activities involving PFAS fall within the scope of EPR-regulated activities.

Different regulatory measures may apply, depending upon the type of activity or site and the scale of environmental risk involved. Several sites that manufacture or use PFAS hold environmental permits in England. These permits are not generally required because PFAS are present; rather, they relate to the regulated activity being undertaken. However, environmental permits can be used to control PFAS emissions and associated environmental risks where these are identified.

However, applying permit conditions to PFAS can be challenging, as setting emission limits usually depends on having statutory standards, such as Environmental Quality Standards (EQS) together with sufficient risk assessment information.

Currently, statutory environmental standards for air exist for only a limited number of substances. Environmental Assessment Levels (EALs) are risk-based benchmark concentrations in ambient air at which no appreciable, or only minimal, risks to human health are expected. EALs are used by the Environment Agency when assessing emissions from permitted activities and determining whether proposed emissions may result in unacceptable risks to human health; they are not permit limits in themselves. (Environment Agency, 2025c). There are currently no PFAS-specific EALs. However, the

Environment Agency has reviewed toxicological evidence to support the development of EALs for selected PFAS, alongside reviewing evidence on emission characteristics from key EA-regulated sources and airborne monitoring methods to support future airborne and inhalation exposure assessments. Site-specific limits and monitoring requirements are being included in permits where there is evidence of risk.

EPR are also used to regulate emissions from installations within scope of the Industrial Emissions Directive (IED). IED applies an integrated environmental approach to regulating certain industrial activities. This means that emissions to air, water, sewer, and land along with other environmental impacts must be considered together.

Regulators are required to set permit conditions that achieve a high level of environmental protection. Permits require operators to use Best Available Techniques (BAT) to control emissions and undertake monitoring of discharges to the environment. It is the responsibility of operators to demonstrate what BAT is for their facilities. BAT represents the state-of-the-art and will change with time, for example taking into account technological advances and changes in scientific knowledge and understanding through research development and innovation.

Best Available Techniques (BAT) were adopted and applied across the EU through Best Available Techniques Conclusions (BATC) under the IED (Environment Agency, 2025d). Following the UK's withdrawal from the EU, existing EU BAT Conclusions continue to have effect through the European Union (Withdrawal) Act and remain available through the relevant Best Available Techniques Reference Documents (BREFs) (European Commission, 2025). Since 2021, new BAT requirements have been developed through the publication of UK Best Available Techniques Conclusions (BATC) (Environment Agency, 2025d).

PFAS are not specifically addressed in most BAT Conclusions developed under the IED framework (European Commission, 2025). At present, only three BREF BAT Conclusions documents, covering the textiles, surface treatment of metals, and waste treatment sectors, include requirements for monitoring PFAS emissions. In the UK, the draft BAT Conclusions for the textiles sector include PFAS monitoring under BAT 11, with reference to a targeted analytical method. Though these conclusions have not yet been adopted into law (ISO, 2019; Environment Agency, 2025d). They also require a regular review of all relevant inputs and outputs, development of a chemicals management system and development of a chemicals inventory (BAT 2-4), all of which will require consideration of PFAS and how best to limit/avoid.

The EA is working closely with regulated industry sector leads to consider the inclusion of PFAS in the periodic sector review, ensuring that emerging risks are addressed within the IED framework.

Further guidance on environmental permits and BAT can be found in Environmental permitting guidance: Core guidance - GOV.UK and Best available techniques: environmental permits.

Water Quality Standards

Environmental Quality Standards (EQSs) are an important tool in water quality legislation in England. They represent regulatory concentrations for substances in water, sediment or biota that are intended to protect aquatic ecosystems and, where relevant, human health (Environment Agency, 2026b). These standards are used to assess the potential impact of chemicals detected in the environment, to manage risks associated with their release, and to support classification of chemical status as part of the river basin planning process under the Water Environment (Water Framework Directive) (England and Wales) Regulations 2017.

In England, there is currently a statutory EQS for one PFAS, perfluorooctane sulphonate (PFOS). This EQS was established in the EU EQS Directive (2013/39/EU) and transposed into English legislation via The Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015. The biota-based EQS of 9100 ng/kg is designed to protect human health via the consumption of fish. An equivalent freshwater EQS for the water column has also been derived at 0.65 ng/L.

For groundwater specifically, 16 PFAS are designated as hazardous substances and these are published in the updated list of groundwater hazardous substances which can be found here. The Environmental Permitting Regulations 2016 (Schedule 22) requires the Environment Agency to take all necessary measures to prevent the input of hazardous substances to groundwater. In line with Defra's Guidance on Groundwater Activities, prevention of hazardous substances is deemed to be achieved if there is no discernible concentration of the substance in question either in the discharge, or in the groundwater immediately down-gradient of the discharge zone.

To address wider PFAS risks, the Environment Agency has recently established toxicological threshold concentrations for four commonly occurring PFAS in water: PFOA, PFHxS, PFNA and PFOS (Environment Agency, 2026b). These thresholds complement, rather than replace, existing statutory EQSs and provide additional guidance for risk assessment. Further work is ongoing to explore the feasibility of developing thresholds for additional PFAS.

Currently, there are no statutory standards for PFAS in drinking water in England. However, under the Water Supply (Water Quality) Regulations 2016, drinking water must be considered "wholesome" and must not contain substances at levels that could pose a risk to human health. The Drinking Water Inspectorate (DWI) oversees drinking water quality and has issued tiered guidance values. This includes a cumulative value of 0.1 µg/L (100 ng/L) for the sum of 48 specified PFAS. Water companies are expected to monitor against this value and, where concentrations exceed it, report this as a water quality incident and take prompt action to reduce levels. Under the UK PFAS Plan, the Government has announced its intention to consult on the introduction of a statutory limit for PFAS in drinking water.

Water companies are responsible for operating their wastewater treatment works in compliance with environmental permits issued by the Environment Agency. This includes managing the quality and quantity of inputs to their networks, including trade effluent, and

taking appropriate action where substances such as PFAS may affect permit compliance, treatment performance, receiving waters or sludge management. Trade effluent discharges to public sewer are controlled by sewerage undertakers under the Water Industry Act 1991, and these controls provide an important mechanism for managing industrial inputs to wastewater treatment works.

The Environment Agency regulates water company discharges through environmental permits under the EPR. Its role includes assessing environmental risk, issuing and reviewing permits, monitoring and inspecting compliance, investigating incidents and taking enforcement action where appropriate. For PFAS, the regulatory framework is developing. PFAS-specific permit limits are not routinely applied to wastewater discharges at present, but where PFAS presents a site-specific risk, the Environment Agency may consider appropriate regulatory controls, monitoring, operating techniques, improvement requirements, or enforcement responses on a case-by-case basis.

Contaminated Land and Land Affected by Contamination

In England, Local Authorities are the primary regulators for contaminated land under Part 2A of the Environmental Protection Act 1990. Their responsibilities cover land where historic (legacy) contamination may pose an unacceptable risk to human health or the environment, and where no other regulatory controls exist or the land is not being brought forward through redevelopment.

Inspections under the Part 2A regime aim to:

- Evaluate the nature, extent and risks posed by contamination
- Identify any remediation required to ensure the land is suitable for its current use

Certain sites may be designated as Special Sites, meaning that where land is formally determined as contaminated, the Environment Agency, rather than the Local Authority, becomes the lead regulatory authority. For sites that may meet the Special Site criteria, Local Authorities may request the Environment Agency to undertake preliminary inspections on their behalf.

Increasingly, the Part 2A regime is being utilised to address emerging contaminants such as PFAS arising from historic industrial or commercial activities, particularly where no alternative regulatory mechanism is available. This reflects the persistence of these substances in the environment and their potential to pose significant risks to human health and the wider environment.

For the purposes of this section, it is important to distinguish between “contaminated land” and “land affected by contamination”. *Contaminated land* has a specific legal meaning under Part 2A of the Environmental Protection Act 1990 and applies only where land has been formally determined to pose a significant risk to human health or the environment. In contrast, *land affected by contamination* is a broader, non-statutory term commonly used in the planning and redevelopment context to describe land where contamination is known or suspected but managed through the redevelopment process. The Environment Agency

also provides non-statutory advice during planning and redevelopment, which is how most contaminated sites, including brownfield land, are assessed and remediated.

Although the Environment Agency is not a statutory consultee for land contamination, it provides advice and guidance to Local Planning Authorities, developers and their consultants where contamination may pose a risk to the water environment, including groundwater and surface waters. Through this advisory role, the Agency supports a risk-based approach in accordance with Land Contamination Risk Management guidance, promoting sustainable remediation. This helps to ensure that land affected by contamination is made safe for its intended use as part of redevelopment.

Fluorinated Gases Regulation

The assimilated Fluorinated Greenhouse Gases Regulation places a cap on the quantity of HFCs that can be produced and imported into GB and provides for a progressive phase down.

The Fluorinated Greenhouse Gases Regulations 2015 (UK Government, 2015):

- establishes rules on containment, use, recovery and destruction of F-gases, and on related ancillary measures;
- imposes conditions on the placing on the market of specific products and equipment that contain, or whose functioning relies upon, F-gases;
- imposes conditions on specific uses of F-gases; and
- establishes quantitative limits for the placing on the market of HFCs.

Annex I of the Regulation is a non-exhaustive list of specific HFCs, PFCs and sulphur hexafluoride. Annex II is a non-exhaustive list of HFCs and HCFCs, fluoroethers, fluoroalcohols and other perfluorinated compounds, all of which are covered by reporting requirements. Further guidance is available here.

Appendix 3: Pollution Inventory data

Table A1: Summary of GB PFAS emissions estimates by industry sector. This table is extracted data from the Pollution Inventory and has been filtered to summarise the quantities of HFCs released to air per sector per year (kg). The reporting threshold for HFCs is 100kg.

	Year		
Hydrofluorocarbons (HFCs) by regulated industry	**2022**	**2023**	**2024**
Agriculture	-	-	0
Biowaste Treatment	-	0	0
Cement and Minerals	0	0	0
Chemicals	11623.54	10728.6	5648.59
Combustion	215	133.15	191.47
EfW	0	0	297
Food & Drink	16145.41	13725.06	13994.91
Landfill	0	0	2.4
Metals	15000	11696	11678
Nuclear	0	0	0
Paper and textiles	0	-	0
Refineries & Fuel	0	0	0
Waste Treatment	0	0	0

Table A2: Summary of GB PFAS emissions estimates by industry sector. This table is extracted data from the Pollution Inventory and has been filtered to summarise the quantities of PFCs released to air per sector per year (kg). The reporting threshold for PFCs is 10kg.

	Year		
Perfluorocarbons (PFCs) by regulated industry	**2022**	**2023**	**2024**
Agriculture	-	-	-
Biowaste Treatment	-	0	0
Cement and Minerals	-	-	-
Chemicals	16324	19109	15366
Combustion	-	-	-
EfW	-	-	-
Food & Drink	0	0	0
Landfill	0	0	0
Metals	834	0	0
Nuclear	-	-	-
Paper and textiles	-	-	-
Refineries & Fuel	0	0	0
Waste Treatment	-	-	-

Appendix 4: Substances included in fully quantitative targeted PFAS analysis - 2026

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
LC-sulfonyl PFAS	**Terminal**	Regulated	**1763-23-1**	Perfluorooctanesulfonic acid (Branched)	PFOS (B)	2960	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
				Perfluorooctanesulfonic acid (Linear)	PFOS (L)	2959	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
				Perfluorooctanesulfonic acid (Total)	PFOS (T)	2958	Ground Water	1	ng/L					
							Surface Water	1	ng/L					
			45298-90-6	Perfluorooctanesulphanoate (Branched)	PFOS anion (B)	2862	Biota - Fauna	0.004	ug/kg					
				Perfluorooctanesulphanoate (Linear)	PFOS anion (L)	2861	Biota - Fauna	0.03	ug/kg					
				Perfluorooctylsulphonate anion	PFOS anion	3375	Biota - Fauna	0.03	ug/kg	✓		✓	✓	
						9276	Ground Water	0.00009	ug/l					
							Saline Water	0.00003	ug/l	✓				
							Surface Water	0.00009	ug/l	✓				
			355-46-4	Perfluorohexanesulfonic acid (Branched)	PFHxS (B)	2860	Biota - Fauna	0.002	ug/kg					
						2944	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
				Perfluorohexanesulfonic acid (Linear)	PFHxS (L)	2859	Biota - Fauna	0.006	ug/kg					
						2964	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
				Perfluorohexanesulfonic acid (Total)	PFHxS (T)	3029	Biota - Fauna	0.007	ug/kg			✓	✓	

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
						2893	Ground Water	1	ng/L					
							Surface Water	1	ng/L					
		Un-regulated	**335-77-3**	Perfluorodecanesulfonic acid	PFDS	3035	Biota - Fauna	0.04	ug/kg			✓	✓	
						2969	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
			375-92-8	Perfluoroheptanesulfonic acid	PFHpS	3032	Biota - Fauna	0.01	ug/kg			✓		
						2967	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			646-83-3	Perfluoro-4-ethylcyclohexanesulfonic acid	PFECHS	2904	Biota - Fauna	0.008	ug/kg			✓		
						2973	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			68259-12-1	Perfluorononanesulfonic acid	PFNS	2901	Biota - Fauna	0.03	ug/kg					
						2963	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			749786-16-1	Perfluoroundecane sulfonic acid	PFUnDS	2953	Ground Water	4	ng/L					✓
							Surface Water	4	ng/L					✓
			79780-39-5	Perfluorododecane sulfonic acid	PFDoS	2968	Ground Water	5	ng/L					✓
							Surface Water	5	ng/L					✓
	Precursor	Regulated	**1691-99-2**	N-Ethyl-N-(2-hydroxyethyl)perfluorooctanesulfonamide	EtFOSE	2978	Ground Water	20	ng/L					✓
							Surface Water	20	ng/L					✓
			24448-09-7	N-Methyl-N-(2-hydroxyethyl)perfluorooctanesulfonamide	MeFOSE	2976	Ground Water	10	ng/L					✓
							Surface Water	20	ng/L					✓
			2991-50-6	2-(N-Ethylperfluorooctanesulfonamido)acetic acid	EtFOSAA	2917	Biota - Fauna	0.1	ug/kg					
				2-(N-Ethylperfluorooctanesulfonamido)acetic acid (Branched)	EtFOSAA (B)	2864	Biota - Fauna	0.006	ug/kg					
						2942	Ground Water	1	ng/L					✓

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
							Surface Water	1	ng/L					✓
				2-(N-Ethylperfluorooctanesulfonamido)acetic acid (Linear)	EtFOSAA (L)	2863	Biota - Fauna	0.02	ug/kg					
						2991	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
			31506-32-8	N-methylperfluorooctanesulfonamide	N-MeFOSA	2975	Ground Water	20	ng/L					✓
							Surface Water	2	ng/L					✓
			38850-51-0	(Perfluorohexanesulfonyl)[3-(trimethylazaniumyl)propyl]azanide	N-TAmP-FHxSA	2890	Ground Water	0.5	ng/L					
							Surface Water	0.5	ng/L					
			4151-50-2	N-Ethylperfluorooctanesulfonamide	EtFOSA	2977	Ground Water	20	ng/L					
							Surface Water	2	ng/L					
			41997-13-1	Perfluorohexanesulfonamide	PFHxSA	2902	Biota - Fauna	0.008	ug/kg			✓	✓	
						2965	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			50598-28-2	Perfluorohexane sulfonamido amine	N-AP-FHxSA	2874	Ground Water	0.5	ng/L					
							Surface Water	0.5	ng/L					
			754-91-6	Perfluorooctanesulfonamide	FOSA	3014	Biota - Fauna	0.02	ug/kg			✓		
						2961	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
		Un-regulated	**2355-31-9**	2-(N-Methylperfluorooctanesulfonamido)acetic acid	MeFOSAA	2916	Biota - Fauna	0.5	ug/kg			✓		
				2-(N-Methylperfluorooctanesulfonamido)acetic acid (Branched)	MeFOSAA (B)	2866	Biota - Fauna	0.005	ug/kg					
						2943	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
				2-(N-Methylperfluorooctanesulfonamido)acetic acid (Linear)	MeFOSAA (L)	2865	Biota - Fauna	0.5	ug/kg					
						2990	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
LC-carbonyl PFAS	Terminal	Regulated	335-67-1	Perfluorooctanoic acid	PFOA	3376	Biota - Fauna	0.02	ug/kg	✓		✓		
						8890	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			45285-51-6	Perfluorooctanoate anion	PFOA anion	9274	Ground Water	0.0006	ug/l					
							Saline Water	0.0002	ug/l	✓				
							Surface Water	0.0006	ug/l	✓				
		Un-regulated	16517-11-6	Perfluorooctadecanoic acid	PFODA	2962	Ground Water	2	ng/L					✓
							Surface Water	2	ng/L					✓
			2058-94-8	Perfluoroundecanoic acid	PFUnDA	3003	Biota - Fauna	0.02	ug/kg			✓	✓	
						8893	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
			307-55-1	Perfluorododecanoic acid	PFoDA	3033	Biota - Fauna	0.02	ug/kg			✓	✓	
						8894	Ground Water	1	ng/L					✓
							Surface Water	2	ng/L					✓
			335-76-2	Perfluorodecanoic acid	PFDA	3034	Biota - Fauna	0.02	ug/kg			✓	✓	
						8892	Ground Water	2	ng/L					✓
							Surface Water	1	ng/L					✓
			375-95-1	Perfluorononanoic acid	PFNA	3015	Biota - Fauna	0.02	ug/kg			✓	✓	
						8891	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			376-06-7	Perfluorotetradecanoic acid	PFTeDA	3006	Biota - Fauna	0.03	ug/kg			✓	✓	
						8895	Ground Water	6	ng/L					✓
							Surface Water	6	ng/L					✓
			72629-94-8	Perfluorotridecanoic acid	PFTrDA	3005	Biota - Fauna	0.1	ug/kg			✓		
						2955	Ground Water	2	ng/L					✓

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
							Surface Water	2	ng/L					✓
SC-sulfonyl PFAS	Terminal	Un-regulated	2706-91-4	Perfluoropentane sulphonic acid	PFPeS	3012	Biota - Fauna	0.02	ug/kg					
						2957	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			375-73-5	Perfluorobutane sulfonic acid	PFBS	3037	Biota - Fauna	0.008	ug/kg			✓	✓	
						2972	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			423-41-6	Perfluoropropanesulfonic acid	PFPrS	2873	Ground Water	0.5	ng/L					
							Surface Water	0.5	ng/L					
	Precursor		30334-69-1	Perfluorobutylsulfonamide	FBSA	2903	Biota - Fauna	0.01	ug/kg			✓	✓	
						2970	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
SC-carbonyl PFAS	Terminal	Un-regulated	2706-90-3	Perfluoropentanoic acid	PFPeA	3011	Biota - Fauna	0.02	ug/kg			✓	✓	
						8887	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			307-24-4	Perfluorohexanoic acid	PFHxA	3016	Biota - Fauna	0.01	ug/kg					
						8888	Ground Water	0.5	ng/L		✓			✓*
							Surface Water	0.5	ng/L		✓			✓*
			375-22-4	Perfluorobutanoic acid	PFBA	3036	Biota - Fauna	0.06	ug/kg					
						2995	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			375-85-9	Perfluoroheptanoic acid	PFHpA	3031	Biota - Fauna	0.02	ug/kg					
						8889	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			76-05-1	Trifluoroacetic acid	TFA	2872	Ground Water	40	ng/L					

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
							Surface Water	40	ng/L					
Ether-based PFAS	**Terminal**	Un-regulated	**113507-82-7**	Perfluoro(2-ethoxyethane)sulfonic acid	PFEESA	2924	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			13252-13-6	Hexafluoropropylene oxide dimer acid (Gen X)	HFPO-DA	2905	Biota - Fauna	0.05	ug/kg					
						2981	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
			13252-14-7	Hexafluoropropylene oxide trimer acid	HFPO-TA	2980	Ground Water	1	ng/L					
							Surface Water	1	ng/L					
			151772-58-6	Perfluoro-3,6-dioxaheptanoic acid	NFDHA	2922	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			377-73-1	Perfluoro-3-methoxypropanoic acid	PFMOPrA	2921	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			756426-58-1	9-chlorohexadecafluoro-3-oxanonane-1-sulfonic acid	9Cl-PF3ONS	2906	Biota - Fauna	0.008	ug/kg					
						2982	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			763051-92-9	11-chloroeicosafluoro-3-oxaundecane-1-sulfonic acid (F-53B Minor)	11Cl-PF3OUdS	2918	Biota - Fauna	0.007	ug/kg					
						2993	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
			863090-89-5	Perfluoro(4-methoxybutanoic) acid	PFMOBA	2923	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			919005-14-4	4,8-Dioxa-3H-perfluorononanoic acid (ADONA)	ADONA	2912	Biota - Fauna	0.02	ug/kg					
						2988	Ground Water	1	ng/L					✓
							Surface Water	1	ng/L					✓
Phosphate PFAS	**Precursor**	Regulated	**40143-79-1**	8:8 Perfluorophosphinic acid	8:8 PFPi	2907	Biota - Fauna	0.06	ug/kg					
						2879	Ground Water	10	ng/L					

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
							Surface Water	10	ng/L					
			610800-34-5	6:8 Perfluorophosphinic acid	6:8 PFPi	2909	Biota - Fauna	0.02	ug/kg			✓		
						2882	Ground Water	10	ng/L					
							Surface Water	10	ng/L					
		Un-regulated	40143-77-9	6:6 Perfluorophosphinic acid	6:6 PFPi	2910	Biota - Fauna	0.02	ug/kg					
						2883	Ground Water	4	ng/L					
							Surface Water	4	ng/L					
Fluorotelomer PFAS	Precursor	Regulated	120226-60-0	10:2 Fluorotelomer sulfonic acid	10:2 FTSA	2994	Ground Water	2	ng/L					
							Surface Water	2	ng/L					
			39108-34-4	8:2 Fluorotelomer sulfonic acid	8:2 FTSA	2908	Biota - Fauna	0.03	ug/kg			✓		
						2983	Ground Water	1	ng/L				✓	
							Surface Water	1	ng/L				✓	
			678-41-1	8:2 Fluorotelomer phosphate diester	8:2 diPAP	2880	Ground Water	4	ng/L					
							Surface Water	4	ng/L					
			70887-84-2	2H-Perfluoro-2-decenoic acid	8:2 FTUCA	2888	Ground Water	1	ng/L					
							Surface Water	1	ng/L					
		Un-regulated	171184-02-4	5:1:2 Fluorotelomer betaine	5:1:2 FTB	2887	Ground Water	0.5	ng/L					
							Surface Water	0.5	ng/L					
			171184-14-8	5:3 Fluorotelomer betaine	5:3 FTB	2886	Ground Water	0.5	ng/L					
							Surface Water	0.5	ng/L					
			27619-97-2	6:2 Fluorotelomer sulfonic acid	6:2 FTSA	2998	Biota - Fauna	0.4	ug/kg			✓		
						2984	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			34455-29-3	6:2 Fluorotelomer sulfonamide betaine	6:2 FtSAB	2985	Ground Water	1	ng/L					
							Surface Water	1	ng/L					

OECD Grouping	Precursor / Terminal	POPs Regulation (2025)	CAS Number	Determinand (Det)	Det abbreviation	WIMS code	Material	MRV	Units	PFOS/ PFOA Trends	Full PFAS - Water	Full PFAS - Fish	Full PFAS - Mussel	TOP Assay Substances
			356-02-5	3:3 Fluorotelomer carboxylic acid	3:3 FTCA	2914	Biota - Fauna	0.06	ug/kg					
						2926	Ground Water	4	ng/L					✓
							Surface Water	4	ng/L					✓
			57677-95-9	6:2 Fluorotelomer phosphate diester	6:2 diPAP	2986	Ground Water	4	ng/L					
							Surface Water	4	ng/L					
			757124-72-4	4:2 Fluorotelomer sulfonic acid	4:2 FTS	2911	Biota - Fauna	0.02	ug/kg					
						2987	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			80475-32-7	6:2 Fluorotelomer sulfonamide amine oxide (6:2 FTNO) (Capstone A)	NNOx-6:2FOSA	2841	Ground Water	1	ng/L					
							Surface Water	1	ng/L					
			812-70-4	3-Perfluoroheptylpropanoic acid	7:3 FTCA	2913	Biota - Fauna	0.02	ug/kg			✓	✓	
						2925	Ground Water	0.5	ng/L					✓
							Surface Water	0.5	ng/L					✓
			914637-49-3	2H,2H,3H,3H-Perfluorooctanoic acid	5:3 FTCA	2915	Biota - Fauna	0.02	ug/kg			✓	✓	
						2989	Ground Water	1	ng/L		✓			✓*
							Surface Water	1	ng/L		✓			✓*
			943913-15-3	6:2/8:2 Fluorotelomer phosphate diester	6/8:2 diPAP	2884	Ground Water	10	ng/L					
							Surface Water	10	ng/L	✓				

* PFAS with sufficient measurements above the minimum reporting value (MRV) for TOP assay

Appendix 5: Data analysis methods used for estimating PFAS concentrations in water and biota

Model-based approach to reporting

To summarise temporal trends and spatial patterns in PFAS concentrations consistently across environmental media (water and biota) and compartments, we fitted Generalised Additive Models with random intercepts (often referred to as GAMMs) to the raw monitoring data. These models are fitted using maximum likelihood, an approach which estimates model parameters by finding values that make the observed data most probable under the model. We applied variations on this approach to all analyses in this report with the exception of the analysis of TFA data presented in Section 8.3. This approach allowed us to account for key features of the monitoring data, including its hierarchical structure (uneven numbers of samples nested within monitoring sites), seasonal variation in the water datasets, longer-term temporal trends and the high proportion of concentration measurements reported below the minimum reporting value (MRV).

For most waterbody types, data were available from two main analytical methods: longer term datasets for PFOS and PFOA only, and more recent full-PFAS datasets covering an expanding suite of compounds (initially 47 substances, increasing to up to 68 over time). For the purposes of this report, the term "full PFAS suite" refers to the EA's analytical PFAS suite in use at the time of data analysis and report preparation. The full-PFAS data does also include PFOS (linear and branched isomers quantified separately) and PFOA: we have not undertaken a combined integrated analysis of PFOS/ PFOA across the two data sources. A similar structure is applied to the biota datasets, where longer time series are available for PFOS and PFOA, alongside shorter full-PFAS datasets covering up to 45 substances.

For river waters, we analysed data separately for sites monitored as part of the River Surveillance Network (RSN). The Environment Agency designed the RSN as a probability-based, spatially balanced survey of the English river network where monitoring sites are chosen using a Generalised Random Tessellation Stratified (GRTS) design (Stevens and Olsen, 2004). The network is intended to provide unbiased estimates of physical-chemical, chemical and ecological characteristics across the population of river waterbodies represented within the sampling frame. Because of its spatially balanced design, the network includes a relatively large proportion of smaller watercourses, although it does not sample the smallest headwaters which are covered by a separate Small Streams Network.

Consequently, the RSN should be considered broadly representative of its river-network sampling domain and will sample all conditions and environmental pressures in proportion to their occurrence (for more details see Environment Agency, 2025g). In contrast, the

other river monitoring sites included in this analysis are taken from the non-RSN Environment Agency monitoring networks, and are largely selected for statutory, operational or investigative purposes. These other networks therefore represent a targeted monitoring population and on average are likely to reflect higher pollutant concentrations.

The availability of data across environmental compartments is summarised in Table A3.

Table A3: ***Summary of major data availability across media and compartments for PFOS/ PFOA and full PFAS analyses***

Media	Compartment	PFOS/ PFOA	Full PFAS
Saline water	Estuarine	✓	×
	Coastal	✓	×
Freshwater	Lake	✓	✓
	River Surveillance Network (RSN)	✓	✓
	Rivers – non-RSN	✓	✓
	Landfill leachate	×	✓
Groundwater	All groundwater	(minimal data)	✓
Biota	Freshwater fish	✓	✓
	Estuarine / coastal fish	✓	✓
	Mussels	(minimal data)	✓

For PFOS and PFOA analysed individually, we modelled $\log_{10}$-transformed concentrations as the response variable and included the following model terms:

- a smooth term for sampling date, expressed as a decimal (year + fraction of year), to capture long-term temporal trends;
- for water data only, a cyclic smooth for month (1–12) to represent seasonal variation. For biota we did not model seasonal effects because samples were not collected evenly throughout the year and there was insufficient within-year variation in sampling dates to estimate seasonal patterns reliably;
- a random effect (random intercept) for site, to account for dependence between observations collected at the same location. This term also reflects the large number of distinct monitoring sites included in each dataset and recognises that the sampled sites represent a subset of the wider population of sites that could potentially be monitored; and

- for biota data only, an additional random effect to account for replicate samples collected during the same site visit.

As the datasets are relatively short, with large numbers of monitoring sites but relatively few samples per site, we focused on overall spatial variation and broad temporal trends rather than characterising trajectories at individual sites. Consequently, the reported temporal trends represent broad trends across the monitored network and are not intended to reflect changes occurring at specific individual sites or in specific catchments.

For the biota analyses, we analysed data from different species within each biota category together. Our objective was to describe broad patterns within each biota category, and we did not examine species-specific differences in PFAS concentrations and trends.

Handling data below the Minimum Reporting Value (MRV)

Many of the PFAS concentration measurements were reported as being below the MRV, meaning the true concentration was only known to lie somewhere between zero and that value. MRVs are compound-specific and even for the same compound, can change between samples, particularly in the longer PFOS / PFOA data series. To account for this, we fitted the models using a censored normal response family applied to $\log_{10}$ transformed concentration data. For measurements reported as below the MRV, the true (but unknown) value was treated as falling within a bounded interval. The upper bound was set equal to the reported MRV for the relevant sample and determinand, while the lower bound was defined as a very small non-zero value, set two orders of magnitude below the smallest MRV observed in the dataset. The use of a non-zero lower bound reflects the fact that we modelled the data on a logarithmic scale, for which zero values are not permitted. In our analysis we chose to retain values between the MRV and the Limit of Quantification (LoQ). These measurements provide quantitative information even though their uncertainty increases as concentrations approach the LoD. Retaining them avoids discarding information from measurements that can still be distinguished from analytical noise.

In the model, uncensored measurements contribute directly as point observations to the likelihood being maximised, whereas censored measurements contribute through the probability of the true value falling between their specified lower and upper bounds, calculated using differences in the cumulative distribution function. This approach reflects current best practice for analysing concentration data that include detection limits. It avoids the bias introduced by substitution methods (e.g. replacing <MRV with fixed fractions of the reporting limit), provides appropriate estimates of uncertainty, and allows all available information to be used consistently within the modelling framework. MRVs will vary by determinand and analytical method, and in some cases over time, and this variation is explicitly accounted for in the analysis.

Full PFAS data availability and modelling

Rates of detections for the full PFAS data vary according to media and compartment. For freshwater, typically only a few determinands are detected in more than 50% of samples,

with the remaining determinands forming a long tail of progressively lower detection frequencies. Many determinands are detected in only a handful of samples or are not detected at all. Groundwater detection rates are substantially lower than those observed in freshwater.

We provide the detection rate figures for the full PFAS data in water and biota in tables A6 and A7 in Appendix 6.

Across the study period, data were available for up to 68 PFAS determinands in freshwater and groundwater, reflecting an expansion of the analytical method from an initial suite of 47 compounds. In fish and mussels, data were available for up to 45 determinands. Prior to analysis, we filtered the datasets to exclude determinands with very low detection frequencies. Specifically, we excluded determinands with a detection rate of less than 1 % across all water media, while for biota data the corresponding threshold was fewer than three detected samples. In addition, for water datasets, we excluded sites with only a single sample from the analysis. Following this filtering, data for 18 PFAS determinands were available for freshwater and groundwater, 23 for fish, and 14 for blue mussels.

For the analyses, the model formulation was similar to the PFOS/ PFOA approach, but with additional complexity to handle the multiple PFAS determinands available. Models included a random effect (intercept) for site and, for water datasets only, a cyclic smooth for season (month of year). Although there are fewer years of data available, the total amount of data in each full-PFAS model was greater than for the equivalent PFOS / PFOA models because we analysed all PFAS determinands that met the above criteria together. We treated determinand as random intercept additive to the other model terms.

This approach optimised the precision of our estimates of mean concentration for the less frequently detected PFAS as well as the more frequently detected PFAS. However, this meant that this approach was not suitable to also model separate trends in individual PFAS over time. Given the relatively short duration of available data (2-5 years depending on environmental compartment), we deemed this an acceptable compromise. For water datasets, we described overall temporal variation across all PFAS using a factor denoting year of sample, allowing us to report estimates standardised to a common reference year (2025), regardless of the distribution of data across sites, determinands and years. In contrast, for biota, no explicit temporal term was included in the model and estimates therefore represent averages across the full period of available data.

To describe the nested structure of the data, we specified an additional random effect (intercept) for site visit (nested within site), and for biota data only, a further random effect was included to identify individual samples nested within site visits. These model terms were required to ensure determinands from the same samples were grouped together correctly. Under this formulation, overall concentration is expressed on the logarithmic scale as the additive effect of chemical identity and site, with additional terms describing temporal variation for the water datasets (year and month). The nested effects are set to zero for prediction.

This additive effects approach represents a simplification of reality in that the relative ordering of concentrations among PFAS determinands (expressed as ratios on the scale of the original data) is assumed to be consistent across sites and over time: this is the reason why it was not possible to consider chemical-specific temporal trends. Likewise in the model there is a single seasonality term across all PFAS, rather than allowing seasonality to vary with PFAS identity. Despite these simplifications, model fits were generally good, with deviance explained ranging from 66 % for groundwater to 86 % for lakes. This range is consistent with lakes being subject to greater mixing and groundwater exhibiting more spatially constrained behaviour.

For the purposes of this report, we considered the overall model fits to be adequate, and the model structure justified, when compared to the alternative of modelling every PFAS separately and assuming no correlation between them. The percentage figures for deviance explained demonstrate that at a national scale, concentrations of individual PFAS strongly correlate. The decisions on model structure are further supported by the ability of the models to maintain precision for the PFAS determinands with lower detection rates, and the relatively few years of data available. The approach is also robust to changes in the number of PFAS included over time: concentrations for determinands introduced later into the chemical analysis (subject to the filtering criteria described above) are effectively imputed for earlier samples. A variant of this modelling approach applied to TOP assay data is described in Appendix 7.

Model diagnostics

We checked the fit of each model using randomised Probability Integral Transform (PIT) residuals. This approach is appropriate for situations where we model partially censored data. Rather than treating censored observations as single values, we account for the range of values consistent with the censoring limits. This allows us to assess our estimates of both censored and uncensored observations together in a consistent way. If the model fits the data well, the residuals should follow a uniform distribution, allowing departures from model assumptions to be identified. Our model checks indicated that the models were fit for purpose. In some cases, they do not fully capture inter-annual variation in concentrations. These departures are discussed in the main text in Section 5.2; we highlight that this does not detract from our ability to characterise long-term trends. Characterisation of both long-term trend and inter-annual variation (which can help identify processes and sources) can be improved as we collect more years of monitoring data.

Post-model data processing

To produce the summary statistics for this report, we used each fitted model to generate predictions of monthly concentrations, with associated 95% confidence intervals, over the full period of observed data. We made predictions on the log scale for both the overall mean (across all sites) and for each monitoring site separately, with model terms selectively excluded depending on the reporting objective (as described below). We then back-transformed the estimated values and their confidence intervals (10^y, where y is each value predicted by the model on the log scale) to the original concentration scale. The confidence intervals indicate the range of mean values that are reasonably consistent with

the observed data and fitted model. Their width is influenced by both the size of the dataset and the amount of unexplained variation in the data.

A consequence of the transform-model-back-transform workflow is that the models estimate geometric mean concentrations. Where the fitted models explain a substantial proportion of the variation in the data, the residual variance on the log scale is small and the geometric mean is therefore expected to be close to the corresponding arithmetic mean, albeit with slightly narrower confidence intervals. A geometric mean will be less than or equal to its arithmetic mean counterpart.

The above distinction between geometric and arithmetic means is one of several reasons why the displayed confidence intervals should be regarded as approximate. We interpret them in a broadly frequentist manner as indicating 'the range of mean values that are reasonably consistent with the observed data and fitted model.'

For the trend and spatial pattern graphs, the confidence intervals are derived from the approximate posterior covariance matrix of the model coefficients, assuming the coefficient estimates are approximately multivariate normal. In turn, this relies on a local quadratic approximation to the likelihood around its maximum and hence on approximate symmetry of the likelihood in that region (Wood, 2017).

While censoring can introduce some asymmetry into the likelihood overall, we expect the approach to be adequate when inference is based on the fitted model in the vicinity of the optimum. Although the approach to interval calculation has been described as 'empirical Bayes', it has been shown to exhibit good frequentist coverage properties for smooth functions (Marra and Wood, 2012). Finally, the intervals are calculated conditional on the estimated smoothing parameters, treating them as fixed. This is considered a reasonable approximation for larger datasets (Wood, 2017).

The approach for derivation of confidence intervals for the statistics derived through simulation (such as percentage changes and peak to trough ratios) is different and described below.

Reporting of PFOS and PFOA

For PFOS and PFOA analysed individually, results are presented graphically in the following ways:

- **Temporal trends:** Long-term trends are shown with month and site effects are excluded from predictions (i.e. the predictions can be considered to be averaged over those variables). Periods where there was no or very minimal data (fewer than 20 samples per year) are presented as dotted lines to emphasise areas of greater uncertainty;
- **Seasonal patterns (water only):** Seasonal variation is presented as the fitted seasonal (month) component in the main report. In addition, the seasonal range is summarised as peak to trough ratio in Appendix 6. For these summaries, overall concentrations are predicted for the year 2025 with site effects excluded;

- **Spatial variation:** Site-specific concentrations are predicted for the mid-2025 for water and mid-2023 for biota. Spatial variability between monitoring sites is summarised in the main text using the ratio between the 90th and 10th percentile site concentrations, with larger ratios indicating greater differences between sites (see details below). Spatial patterns are illustrated in Appendix 6 using cumulative frequency plots, in which mean concentrations at individual sites are ordered from lowest to highest, with the x-axis labelled from 0 % (lowest concentration) to 100 % (highest concentration).

In all cases, spatial variation represents the largest source of variability in the concentration data, followed by seasonal variation (water data only) and then long-term temporal trends.

Results are also presented in tabular form as overall mean concentrations (with 95% confidence intervals) for mid-2025, along with the number of samples and monitoring sites used in each analysis.

Tables are also presented for percentage change in concentration over time and the ratio of the 90th to 10th percentile site mean concentrations. We calculated percentage change over the whole period from modelled concentrations at the start and end of the monitoring period, while mean annual percentage change was calculated as the average of the modelled year-on-year percentage changes over the same period. For situations with a broadly linear trend, we would expect strong consistency between the whole period and mean annual figures, but this was not always the case. For these calculations, we derived 1000 post-hoc simulations of mean concentration from each fitted model, from which we calculated mean values and 2.5th and 97.5th percentiles (to represent a 95% confidence interval). We used the same approach to calculate seasonality ratios – the ratio between the highest and lowest mean concentration over a year while holding the other model covariates constant.

Reporting of full PFAS dataset

For the full PFAS dataset, reporting followed a similar workflow, however we calculated estimates for each determinand within each environmental compartment. The PFAS grouping framework described in Section 7.1 was not incorporated into the statistical models and was used only for interpretation and reporting. We assigned the PFAS groups after modelling by aggregating determinand-specific estimates and summing concentrations within each group. For the reasons explained above, for water we calculated estimates for the reference year 2025 only; for biota the estimates are averages over the range of the data (September 2022 to November 2024 for fish, January 2023 to January 2025 for mussels).

We present results as total concentrations for each PFAS grouping, with the adapted OECD grouping used as the primary reporting framework. The graphical presentation focuses on spatial variation only. We show the geometric mean concentration at the median-ranked site (i.e. the site with 50 % of sites having lower concentrations and 50 % higher concentrations) along with concentrations at the 10th and 90th percentile sites. For

clarity, confidence intervals are not displayed for aggregate totals, although they can be derived using the same simulation-based approach.

Indicative maps of total PFAS concentrations are also presented for waterbody types in the main report and for biota in Appendix 6.

Reporting of PFAS associations with risk categories

To aid interpretation of the full PFAS water monitoring data, we identified which sites were within a 1,000m buffer of identified potential PFAS source sites. Source sites are defined using the Environment Agency's PFAS Risk Explorer and grouped into air transport sites, permitted landfill sites, and fire stations. No account was taken of the relative location of the source site and the monitoring point. This may particularly affect the mean concentrations across river sites where some sites may be located upstream of PFAS source sites. However, because such river monitoring sites will generally have been chosen to monitor PFAS from the source sites, we felt this was an acceptable compromise for this exploratory analysis.

We combined air transport sites and fire stations into a single category (which we denoted AFFF), as there were insufficient data for air transport sites alone and both are expected to be dominated by AFFF use. We categorised all sites not within the 1,000m buffer zones as 'lower risk'. All observations were then given the category for their associated site. We undertook this categorisation on the monitoring sites for groundwater and rivers; there were insufficient data for lakes and for River Surveillance Network sites. We also excluded 5 sites which were associated with both AFFF and permitted landfill. Overall, 18% of observations for rivers and 11% for groundwater fell into the higher-risk categories (AFFF and permitted landfill).

We then fitted a variant of the main full-PFAS water model in which we allowed variation in observed concentration among PFAS determinands (modelled as a random effect additive to the other model terms) to differ by risk category. We also allowed seasonal effects (month) to vary by risk category and included a main effect for risk category as is standard for this approach. Seasonality is an important effect overall; differences in seasonality among the risk categories were significant for rivers but not practically important for subsequent calculations.

Using the resulting models (fitted separately for rivers and groundwater), we estimated contrasts as differences between each higher-risk category (AFFF, permitted landfill) and our baseline / reference category (lower-risk). We obtained these contrasts from model-based predictions, with site- and sample-level random effects excluded to yield population-level estimates. We evaluated predictions for 2025 and month 7 (July), corresponding to the typical seasonal peak in PFAS concentrations for river sites. Seasonality was very weak for the groundwater sites. As we estimated contrasts as differences on the model (log10) scale we back-transformed these differences to the original scale, making each difference into a ratio. Hence a contrast between categories represents the multiplicative change in PFAS concentration relative to the lower-risk category. We estimated contrasts

simultaneously for each PFAS determinand, along with corresponding 95% confidence intervals.

Tools used for statistical analysis

We undertook all the above analyses in R (R Core Team, 2025). Aside from the TFA analysis we fitted models using Restricted Maximum Likelihood (REML) in the mgcv R package (Wood, 2017). We derived the contrasts for the full PFAS risk sites analysis (Section 7.2.3) using the marginal effects R package on models fitted with mgcv (Arel-Bundock et al., 2024). We produced all graphs using the ggplot2 R package. For post-hoc simulation of mean trend (for % change figures) and for model diagnostics using randomised PIT residuals we used bespoke R code which accessed data contained in mgcv model objects. For the TFA analysis, we calculated summary statistics in R using Regression on Order Statistics in the NADA package (Lee, 2025) owing to the presence of censored values and the small sample sizes (~50) within each sample material type and data skewness (Ahmadi et al., 2021).

Appendix 6: Supplementary Monitoring Data Reporting

Appendix 6 provides supplementary analyses that support and extend the findings presented in Chapters 6, 7 and 8, strengthening interpretation of observed PFAS monitoring patterns rather than introducing new headline results. It explores how spatial variability, seasonality, chemical properties, and analytical constraints influence PFAS behaviour across environmental compartments.

Total PFAS Map – Biota

Figure A3 shows the locations and total PFAS concentrations from the biota sampling programmes in coastal, estuarine and freshwater fish and estuarine & coastal mussels.

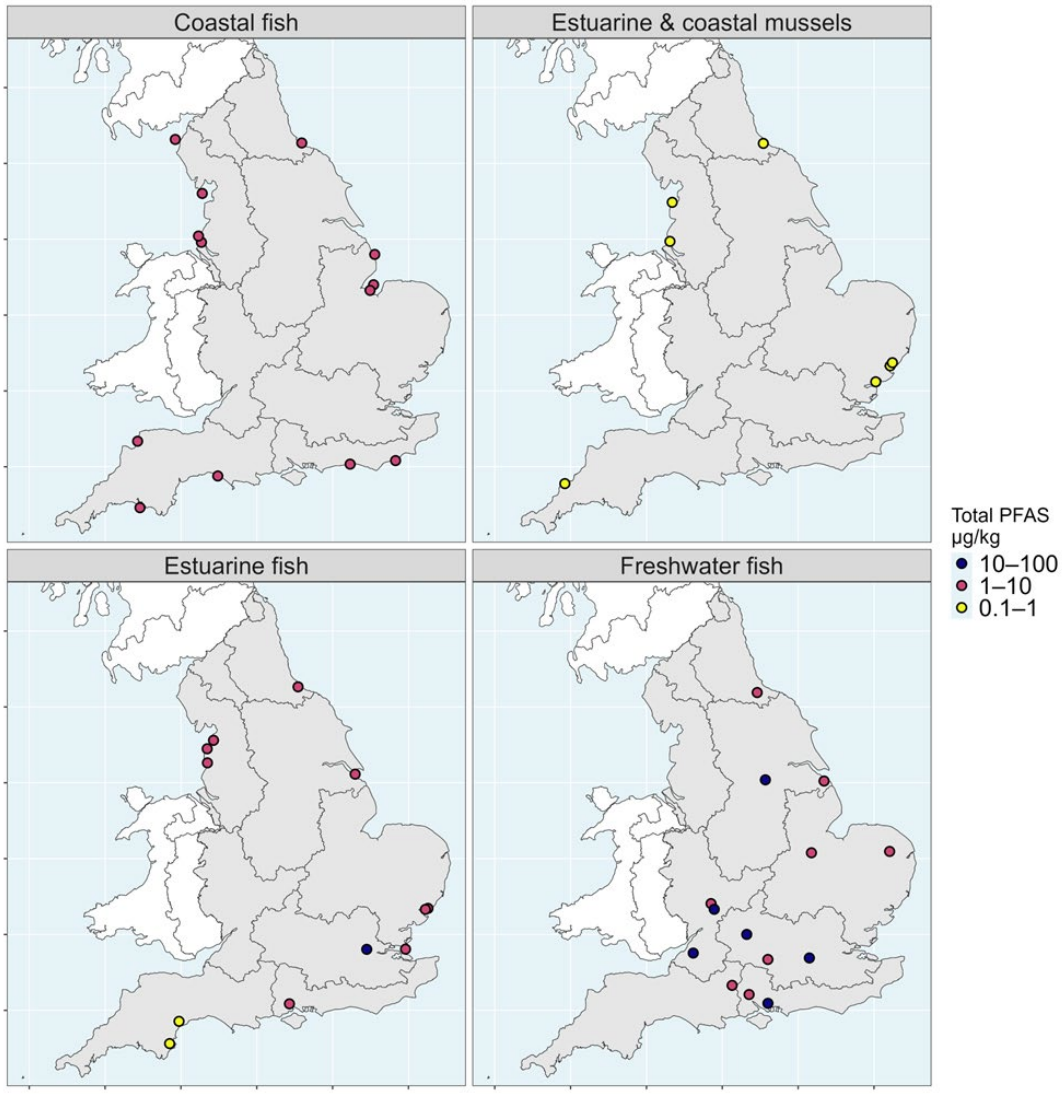


Figure A3: Map showing the locations of PFAS biota monitoring sites across the different waterbody types for fish and mussels. Additional sites monitored only for PFOS/PFOA are not shown. *Colours indicate model-estimated total PFAS concentrations (sum of all measured PFAS), expressed in µg/kg and pooled across data collected between 2022 and 2025.*

Spatial variation in PFOS and PFOA in surface waters

Figure A4 shows the estimated geometric mean concentrations of PFOS and PFOA measured at individual water quality monitoring sites across England, for estuaries, lakes and rivers (RSN and non-RSN). Coastal sites are not reported due to the limited data available. Each point represents the estimated concentration at the midpoint of 2025, with 95% confidence intervals. Concentrations are displayed on a logarithmic scale to capture the wide range of values observed. Sites are arranged from lowest to highest concentration along the percentile axis, creating a rank ordered distribution that highlights how levels vary across monitoring sites in estuaries, lakes, and rivers.

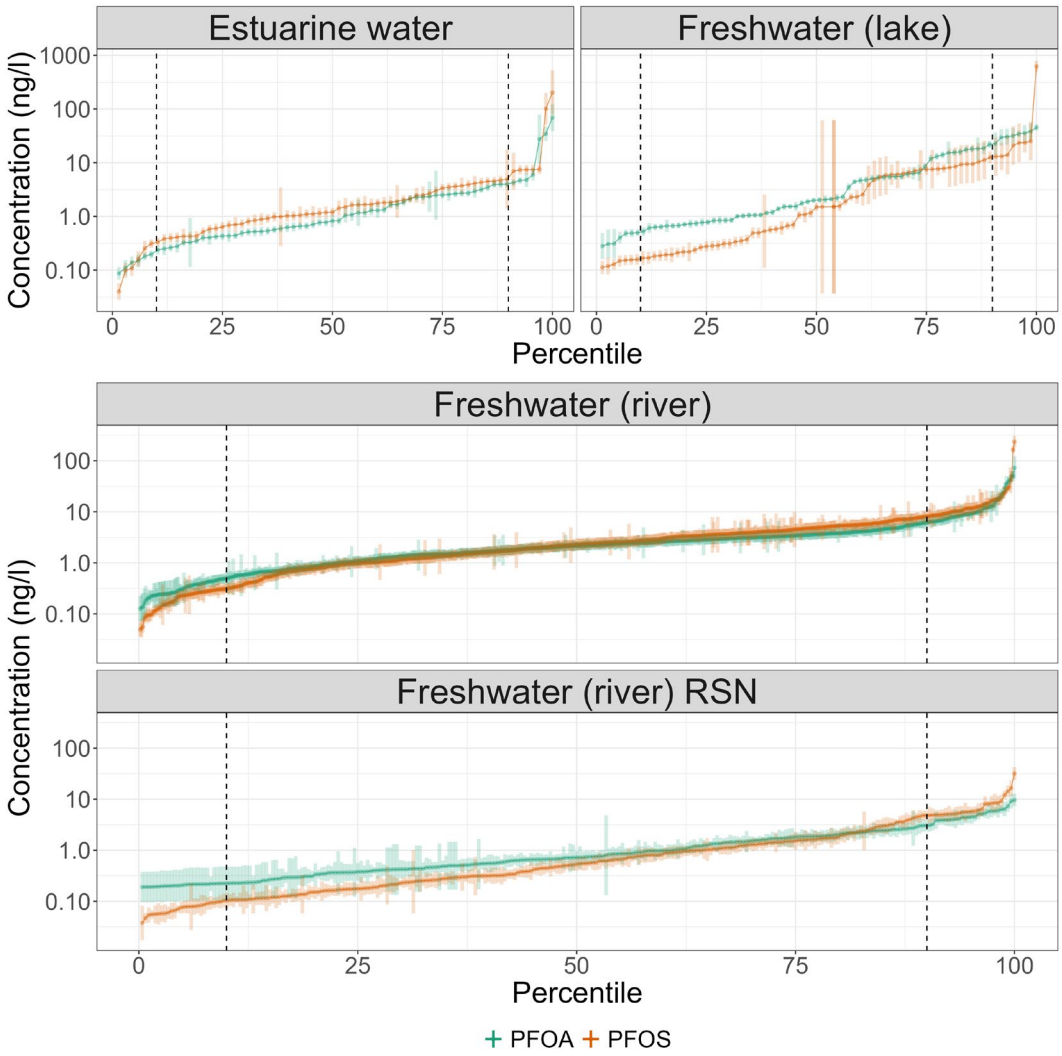


Figure A4: Cumulative distribution of geometric mean PFOS and PFOA concentrations at individual water monitoring sites across England. ***Each point represents the site-specific mean concentration estimated in the middle of 2025, vertical coloured bars show 95% confidence intervals. Concentrations are plotted on a log_{10} scale. Seasonal variation is excluded. Dashed black vertical lines denote the 10^{th} and 90^{th} percentile points used to derive ratios of spatial variation. Sites are ordered from low to high concentration along the percentile (x) axis to produce a rank-ordered cumulative distribution.***

Across all surface water compartments shown, the plots highlight substantial variation in concentrations between monitoring sites, with values spanning several orders of magnitude. For most compartments, concentrations are relatively low across a large proportion of sites, with progressively higher concentrations observed toward the upper percentiles of the distributions. These patterns provide visual context for the quantitative measures of spatial variability summarised in Section 6.1.3, where site-to-site variation is assessed using the ratio between concentrations at the 90th and 10th percentile sites.

Within this broader pattern, differences between PFOS and PFOA vary by waterbody type. At lower percentiles, PFOS concentrations are generally lower than PFOA in rivers and lakes, whereas in estuarine waters PFOS tends to be higher at the lower end of the distribution. Toward the upper percentiles, concentrations of both substances increase sharply and become more similar, with PFOS frequently exceeding PFOA at the extreme upper tail. These upper-tail behaviours reflect the presence of high-concentration hotspots, with the most extreme values occurring at a small number of sites. Maximum concentrations in lakes and estuarine waters reach around an order of magnitude higher than those observed at the highest river sites.

Spatial variation in PFOS and PFOA concentrations in biota

Figure A5 presents cumulative frequency plots of the estimated geometric mean concentrations of PFOS and PFOA at individual biota monitoring sites across England, for freshwater, estuarine and coastal fish. Concentrations are estimated for a common reference point in mid-2023. Each point represents a monitoring site, with vertical bars showing 95% confidence intervals, and concentrations are displayed on a logarithmic scale. Sites are ordered from lowest to highest concentration along the percentile axis.

Across all fish categories shown, the plots illustrate clear variation in concentrations between monitoring sites, with values spanning a wide range across the percentile distributions. Concentrations increase progressively from lower to higher percentiles in each environment, indicating substantial spatial heterogeneity within fish categories rather than uniform exposure across sites. These distributions provide visual context for the quantitative measures of spatial variation summarised in Table 6 in the main text.

Differences between fish categories are also evident in the shape and spread of the percentile distributions. Freshwater and estuarine fish show broader distributions, with greater separation between lower and upper percentiles, while coastal fish display a comparatively more constrained distribution. Overall, the cumulative frequency plots illustrate how site-specific concentrations are distributed within each fish type and support the spatial variability patterns quantified using 90:10 percentile ratios in Section 6.2.3.

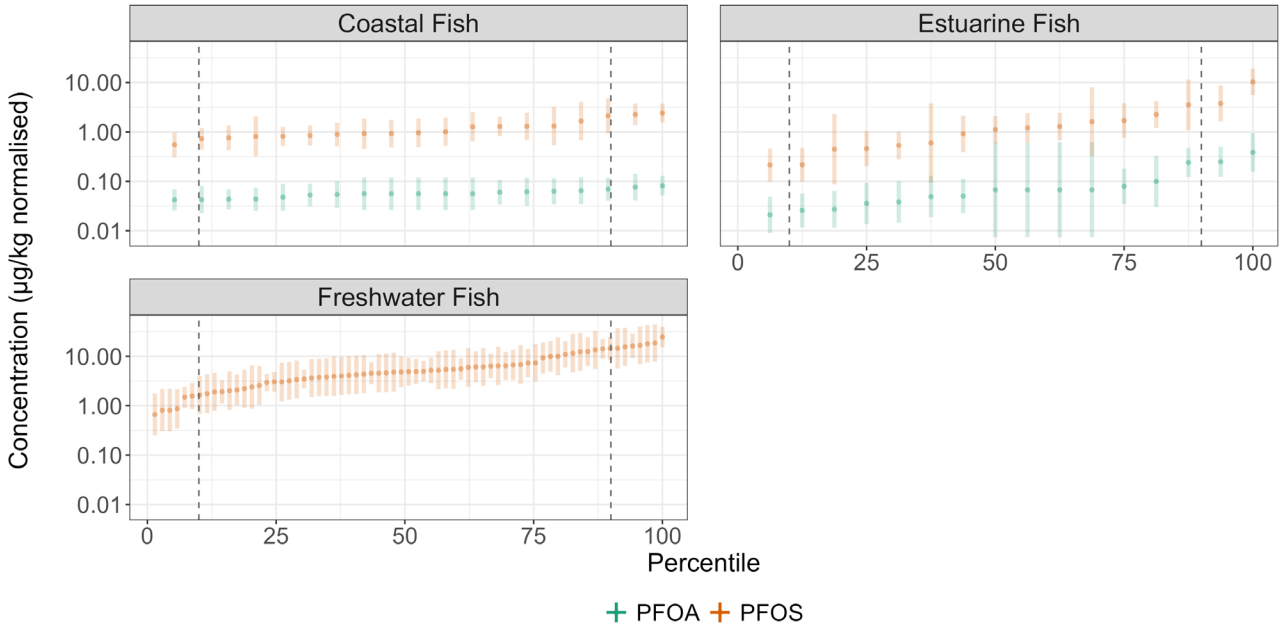


Figure A5: Geometric mean PFOS and PFOA concentrations in biota at individual monitoring sites across England in 2023. ***Each point represents the site-specific model-based estimated geometric mean concentration in 2023, with associated confidence intervals. Concentrations are plotted on a log$_{10}$ scale. Dashed black vertical lines denote the 10th and 90th percentile points used to derive ratios of spatial variation. Sites are ordered from low to high concentration along the percentile axis to produce a rank-ordered cumulative distribution plot, illustrating variability in PFOS and PFOA concentrations among monitored sites.***

Peak-to-trough Seasonality Ratios for PFOS and PFOA in surface waters

The seasonality ratios shown in Figure A6 quantify the strength of seasonal variation in PFOS and PFOA concentrations by comparing the estimated seasonal maximum to the seasonal minimum (peak-to-trough ratio). A ratio of 1 indicates no seasonality, meaning concentrations remain constant throughout the year. As ratios increase above 1, they indicate progressively stronger seasonal effects, with higher values reflecting a larger difference between seasonal highs and lows.

The numbers on the graph give a quantitative assessment of the seasonality shown in the main report in Figure 9; they are derived from the same estimates. The dots indicate the mean seasonal amplitude for each waterbody type, while the vertical bar ends of the horizontal lines show a 95% (2.5 to 97.5%) confidence interval. Wider intervals reflect greater uncertainty in the estimated seasonal signal. Greater seasonality is indicated by a higher ratio.

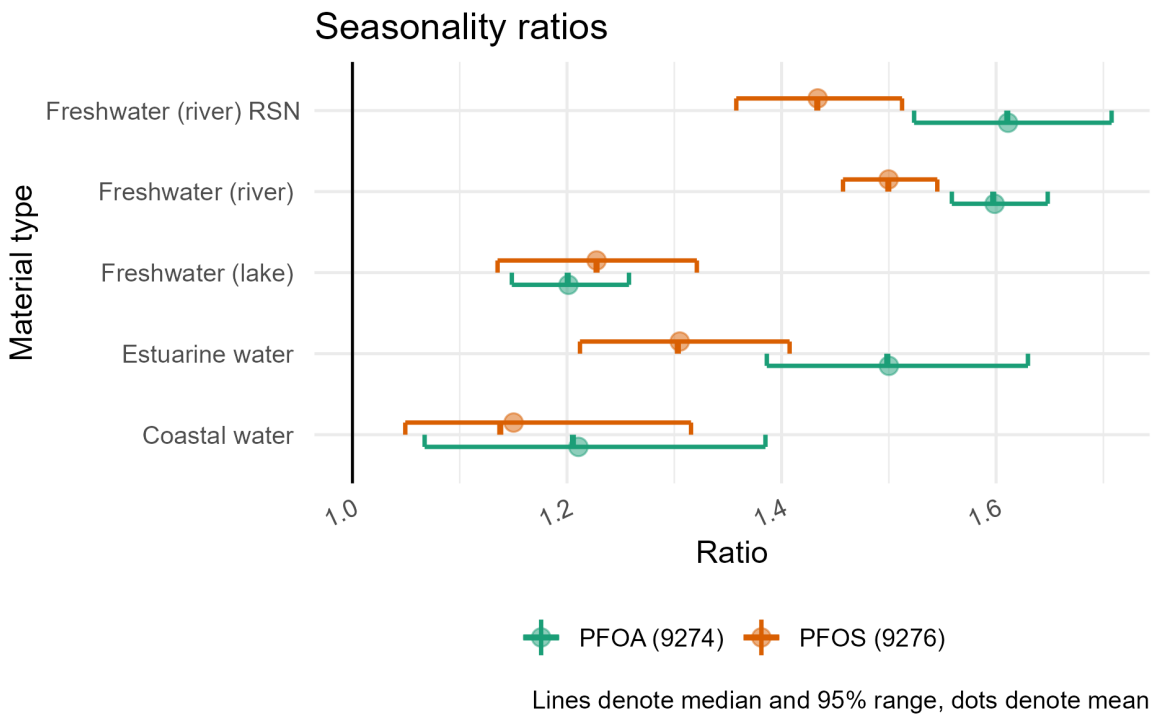


Figure A6: Peak-to-trough seasonality ratios for PFOS and PFOA in surface waters. ***Ratios represent the model-estimated seasonal maximum divided by the model-estimated seasonal minimum concentrations for each waterbody type. Dots represent the mean, horizontal lines show the median and 95% confidence intervals.***

Percentage detection figures for full PFAS monitoring data in water and biota

Table A4: Percentage of values above the detection limit (MRV) for multiple PFAS as analysed in water. These figures exclude determinands excluded from the analysis either because they were tested in very few samples or were detected in less than 1% of samples. Percentages are rounded to the nearest whole number.

Determinand	Groundwater (%)	Freshwater (river) (%)	Freshwater (river) RSN (%)	Freshwater (lake) (%)
PFBA	22	66	68	84
PFoctncAcid	24	71	67	97
PFhexncAcid	22	67	60	83
PFBS	21	57	52	84
PFpentncAcid	16	54	48	82
PFOS (B)	20	60	45	81
PFHxS-L	22	58	43	84
PFOS (L)	18	58	42	83
FBSA	8	34	26	67
PFheptncAcid	13	40	22	78
6:2 FTSA	23	28	20	12
PFHxS-B	13	23	17	22
PFPeS	9	15	12	42
PFHxSA	2	5	5	0
PFnonncAcid	2	5	2	5
PFecHS	3	10	2	28
PFHpS	3	2	1	6
5:3 FTCA	0	2	1	0

Table A5: Percentage of values above detection limit (MRV) for multiple PFAS as analysed in biota. These figures exclude determinands excluded from the analysis either because they were tested in very few samples or were detected in less than 1% of samples. Percentages are rounded to the nearest whole number.

Determinand	Coastal Fish (%)	Estuarine Fish (%)	Freshwater Fish (%)	Mussels (%)
FBSA	75	86	100	80
PFDoDA	63	56	100	77
PFOS	88	87	100	87
PFDA	93	89	97	53
FHxSA	22	67	96	77
PFUnDA	90	81	96	37
PFTeDA	49	51	94	40
PFTrDA	52	37	91	0
PFHxSA	75	78	61	90
PFNA	96	96	58	13
PFOcSu	22	24	57	0
PFDSA	35	37	52	80
PFBSA	40	48	51	20
5:3 FTCA	22	43	40	53
NMeFOSAA	10	17	24	0
7:3 FTCA	50	68	22	80
8:2 FTSA	4	9	22	0
PFNS	2	4	21	0
PFHpSA	21	45	19	0
6:2 FTS	30	35	12	0
PFPeA	7	18	9	17
6:8 PFPi	11	8	6	0
PFecHS	9	20	6	3
PFOA	58	53	6	0
8:2ClPFESA	7	3	2	3
PFBA	4	4	2	3
3:3 FTCA	7	0	0	3
4:2 FTSA	4	2	0	0
6:2ClPFESA	7	4	0	3
6:6 PFPi	2	10	0	0
8:8 PFPi	2	2	0	0
ADONA	4	4	0	0
NEtFOSAA	14	20	0	0
PFHpA	0	1	0	3

Individual PFAS concentrations from the full PFAS analysis

Table A6: Estimated *geometric mean concentrations of individual PFAS in surface water and groundwater, grouped by PFAS class. Figures estimated for 2025, expressed as ng/L, with 95% confidence interval shown in brackets. Mean concentrations are reported to two significant figures. Where available, branched and linear isomers are reported separately.*

Adapted OECD Grouping	Determinand	Groundwater (ng/L)	Freshwater lake (ng/L)	Freshwater river RSN (ng/L)	Freshwater river (ng/L)
LC-sulfonyl PFAS	PFHpS	**0.0079** (0.0065, 0.0096)	**0.062** (0.035, 0.11)	**0.019** (0.0097, 0.036)	**0.028** (0.024, 0.032)
LC-sulfonyl PFAS	PFHxS (Branched)	**0.033** (0.028, 0.039)	**0.30** (0.16, 0.54)	**0.092** (0.057, 0.15)	**0.12** (0.11, 0.14)
LC-sulfonyl PFAS	PFHxS (Linear)	**0.11** (0.10, 0.13)	**1.1** (0.68, 1.9)	**0.30** (0.19, 0.47)	**0.56** (0.50, 0.63)
LC-sulfonyl PFAS	PFHxSA	**0.0066** (0.0054, 0.0081)	**0.029** (0.011, 0.074)	**0.037** (0.021, 0.063)	**0.056** (0.049, 0.064)
LC-sulfonyl PFAS	PFOS (Branched)	**0.085** (0.073, 0.098)	**1.1** (0.66, 1.8)	**0.28** (0.18, 0.44)	**0.54** (0.48, 0.61)
LC-sulfonyl PFAS	PFOS (Linear)	**0.084** (0.072, 0.097)	**1.1** (0.64, 1.8)	**0.29** (0.18, 0.46)	**0.61** (0.54, 0.68)
LC-sulfonyl PFAS	PFECHS	**0.0094** (0.0077, 0.011)	**0.21** (0.12, 0.35)	**0.029** (0.016, 0.052)	**0.081** (0.072, 0.092)
LC-carbonyl PFAS	PFNA	**0.013** (0.011, 0.016)	**0.11** (0.062, 0.20)	**0.042** (0.023, 0.080)	**0.080** (0.070, 0.091)
LC-carbonyl PFAS	PFOA	**0.13** (0.11, 0.15)	**2.7** (1.6, 4.4)	**0.68** (0.43, 1.1)	**0.91** (0.81, 1.0)
SC-sulfonyl PFAS	FBSA	**0.020** (0.017, 0.024)	**0.30** (0.18, 0.50)	**0.16** (0.099, 0.25)	**0.18** (0.16, 0.21)
SC-sulfonyl PFAS	PFBS	**0.14** (0.12, 0.16)	**1.6** (0.93, 2.6)	**0.72** (0.46, 1.1)	**0.96** (0.86, 1.1)
SC-sulfonyl PFAS	PFPeS	**0.024** (0.020, 0.028)	**0.20** (0.12, 0.33)	**0.063** (0.038, 0.10)	**0.081** (0.071, 0.091)

Adapted OECD Grouping	Determinand	Groundwater (ng/L)	Freshwater lake (ng/L)	Freshwater river RSN (ng/L)	Freshwater river (ng/L)
SC-carbonyl PFAS	PFBA	**0.17** (0.15, 0.20)	**3.3** (2.0, 5.5)	**1.0** (0.66, 1.6)	**1.3** (1.2, 1.5)
SC-carbonyl PFAS	PFHpA	**0.082** (0.070, 0.096)	**0.83** (0.50, 1.4)	**0.25** (0.16, 0.40)	**0.49** (0.44, 0.55)
SC-carbonyl PFAS	PFHxA	**0.12** (0.11, 0.14)	**1.4** (0.84, 2.3)	**0.57** (0.36, 0.89)	**1.0** (0.93, 1.2)
SC-carbonyl PFAS	PFPeA	**0.14** (0.12, 0.16)	**1.8** (1.1, 3.0)	**0.69** (0.44, 1.1)	**1.2** (1.0, 1.3)
Fluorotelomer PFAS	5:3 FTCA	**0.0058** (0.0046, 0.0073)	**0.054** (0.019, 0.15)	**0.038** (0.020, 0.076)	**0.062** (0.053, 0.071)
Fluorotelomer PFAS	6:2 FTSA	**0.34** (0.30, 0.39)	**0.26** (0.15, 0.43)	**0.32** (0.20, 0.51)	**0.49** (0.44, 0.55)

Note: *The number of samples and sites contributing to PFAS concentration estimates varies between substances and environmental compartments. The distribution and coverage of monitoring sites for the environmental compartments shown here are presented in Figure 8 in the main report.*

Table A7: ***Geometric mean concentrations of individual PFAS in biota, grouped by PFAS class. Model-estimated mean concentrations based on data collected between 2018 and 2024, expressed as µg/kg, with 95% confidence intervals shown in brackets. Mean concentrations are reported to two significant figures.***

OECD Grouping	Determinand	Coastal fish (µg/kg)	Estuarine fish (µg/kg)	Freshwater fish (µg/kg)	Estuarine & Coastal mussels (µg/kg)
LC-sulfonyl PFAS	FHxSA	**0.0068** (0.0034, 0.014)	**0.024** (0.010, 0.059)	**0.043** (0.024, 0.077)	**0.064** (0.025, 0.17)
LC-sulfonyl PFAS	NMeFOSAA	**0.0066** (0.0031, 0.014)	**0.013** (0.0052, 0.032)	**0.0084** (0.0049, 0.015)	
LC-sulfonyl PFAS	PFDSA	**0.020** (0.011, 0.039)	**0.030** (0.012, 0.071)	**0.028** (0.018, 0.045)	**0.13** (0.050, 0.33)
LC-sulfonyl PFAS	PFHpSA	**0.0069** (0.0036, 0.013)	**0.017** (0.0071, 0.041)	**0.0055** (0.0032, 0.0094)	
LC-sulfonyl PFAS	PFHxSA	**0.041** (0.022, 0.076)	**0.098** (0.041, 0.23)	**0.047** (0.030, 0.074)	**0.033** (0.013, 0.086)
LC-sulfonyl PFAS	PFOS	**1.1** (0.58, 2.0)	**1.8** (0.78, 4.3)	**5.7** (3.7, 8.9)	**0.068** (0.026, 0.18)
LC-sulfonyl PFAS	PFOcSu	**0.013** (0.0065, 0.024)	**0.020** (0.0080, 0.047)	**0.043** (0.027, 0.067)	
LC-sulfonyl PFAS	PFECHS	**0.0029** (0.0014, 0.0062)	**0.0054** (0.0022, 0.014)	**0.0023** (0.0011, 0.0047)	
LC-carbonyl PFAS	PFDA	**0.11** (0.060, 0.21)	**0.18** (0.077, 0.43)	**0.56** (0.36, 0.87)	**0.017** (0.0063, 0.043)
LC-carbonyl PFAS	PFDoDA	**0.033** (0.018, 0.063)	**0.040** (0.017, 0.097)	**0.66** (0.43, 1.0)	**0.026** (0.0098, 0.066)
LC-carbonyl PFAS	PFNA	**0.15** (0.079, 0.27)	**0.21** (0.089, 0.50)	**0.027** (0.017, 0.043)	**0.0085** (0.0032, 0.023)
LC-carbonyl PFAS	PFOA	**0.054** (0.029, 0.10)	**0.082** (0.035, 0.19)	**0.0041** (0.0022, 0.0075)	
LC-carbonyl PFAS	PFTeDA	**0.020** (0.011, 0.039)	**0.033** (0.014, 0.079)	**0.21** (0.14, 0.33)	**0.022** (0.0082, 0.057)
LC-carbonyl PFAS	PFTrDA	**0.036** (0.019, 0.068)	**0.036** (0.015, 0.086)	**0.14** (0.090, 0.21)	

OECD Grouping	Determinand	Coastal fish (µg/kg)	Estuarine fish (µg/kg)	Freshwater fish (µg/kg)	Estuarine & Coastal mussels (µg/kg)
LC-carbonyl PFAS	PFUnDA	**0.080** (0.043, 0.15)	**0.079** (0.033, 0.19)	**0.26** (0.17, 0.41)	**0.014** (0.0052, 0.036)
SC-sulfonyl PFAS	FBSA	**0.033** (0.017, 0.063)	**0.089** (0.037, 0.21)	**0.18** (0.10, 0.33)	**0.036** (0.014, 0.092)
SC-sulfonyl PFAS	PFBSA	**0.014** (0.0074, 0.026)	**0.023** (0.0095, 0.053)	**0.038** (0.025, 0.057)	**0.0051** (0.0019, 0.014)
SC-carbonyl PFAS	PFPeA	**0.0070** (0.0034, 0.015)	**0.0096** (0.0039, 0.024)	**0.0064** (0.0035, 0.012)	**0.010** (0.0039, 0.028)
Phosphate PFAS	6:8 PFPi	**0.0071** (0.0034, 0.015)	**0.0038** (0.0014, 0.011)	**0.0036** (0.0017, 0.0073)	
Fluorotelomer PFAS	5:3 FTCA	**0.012** (0.0060, 0.024)	**0.022** (0.0088, 0.053)	**0.033** (0.018, 0.063)	**0.039** (0.015, 0.10)
Fluorotelomer PFAS	6:2 FTS	**0.16** (0.080, 0.30)	**0.16** (0.066, 0.39)	**0.070** (0.039, 0.13)	
Fluorotelomer PFAS	7:3 FTCA	**0.043** (0.022, 0.081)	**0.10** (0.044, 0.25)	**0.026** (0.015, 0.044)	**0.042** (0.016, 0.11)
Fluorotelomer PFAS	8:2 FTSA	**0.0055** (0.0024, 0.013)	**0.0066** (0.0025, 0.017)	**0.018** (0.011, 0.029)	

Note: *The number of samples and sites contributing to PFAS concentration estimates varies between substances and biota type. The distribution and coverage of monitoring sites for the different biota shown here are presented in Figure A3 in Appendix 6.*

Table A8: Model-estimated *mean PFAS concentrations for raw landfill leachate. Geometric mean concentration, expressed as ng/l, with 95% confidence intervals shown in brackets. Mean concentrations are reported to two significant figures. For full substance names refer to Appendix 4.*

OECD Grouping	Determinand & CAS Number	Concentration (ng/L)
LC-sulfonyl PFAS	EtFOSE: 1691-99-2	**7.8** (3.8, 16)
LC-sulfonyl PFAS	FHxSA: 41997-13-1	**2.9** (1.2, 6.8)
LC-sulfonyl PFAS	MeFOSE: 24448-09-7	**11** (5.3, 21)
LC-sulfonyl PFAS	N-AP-FHxSA: 50598-28-2	**7.5** (2.7, 21)
LC-sulfonyl PFAS	N-EtFOSAA: 2991-50-6	**38** (22, 65)
LC-sulfonyl PFAS	N-MeFOSAA: 2355-31-9	**36** (20, 62)
LC-sulfonyl PFAS	PFECHS: 646-83-3	**8.3** (4.9, 14)
LC-sulfonyl PFAS	PFHpS: 375-92-8	**2.9** (1.7, 5.2)
LC-sulfonyl PFAS	PFHxS: 355-46-4	**160** (97, 270)
LC-sulfonyl PFAS	PFOS: 1763-23-1	**110** (64, 180)
LC-sulfonyl PFAS	PFOSA: 754-91-6	**1.8** (0.64, 4.9)
LC-carbonyl PFAS	PFDA: 335-76-2	**5.7** (2.9, 11)
LC-carbonyl PFAS	PFNA: 375-95-1	**9.7** (5.7, 16)
LC-carbonyl PFAS	PFOA: 335-67-1	**440** (260, 720)
SC-sulfonyl PFAS	FBSA: 30334-69-1	**6.1** (3.0, 12)
SC-sulfonyl PFAS	PFBS: 375-73-5	**650** (390, 1100)
SC-sulfonyl PFAS	PFPeS: 2706-91-4	**25** (15, 42)
SC-carbonyl PFAS	PFBA: 375-22-4	**530** (320, 870)
SC-carbonyl PFAS	PFHpA: 375-85-9	**160** (97, 270)

OECD Grouping	Determinand & CAS Number	Concentration (ng/L)
SC-carbonyl PFAS	PFHxA: 307-24-4	**610** (370, 1000)
SC-carbonyl PFAS	PFPeA: 2706-90-3	**260** (160, 430)
Ether-based PFAS	PFMOPA: 377-73-1	**4.8** (2.3, 9.9)
Fluorotelomer PFAS	10:2 FTSA: 120226-60-0	**3.7** (1.7, 8.2)
Fluorotelomer PFAS	3:3 FTCA: 356-02-5	**44** (25, 77)
Fluorotelomer PFAS	4:2 FTSA: 757124-72-4	**1.4** (0.47, 4.2)
Fluorotelomer PFAS	5:3 FTCA: 914637-49-3	**900** (540, 1500)
Fluorotelomer PFAS	6:2 FTAB: 34455-29-3	**120** (71, 220)
Fluorotelomer PFAS	6:2 FTSA: 27619-97-2	**65** (39, 110)
Fluorotelomer PFAS	7:3 FTCA: 812-70-4	**120** (71, 210)
Fluorotelomer PFAS	8:2 FTSA: 39108-34-4	**7.4** (3.9, 14)

Appendix 7: TOP Assay reporting methods

This appendix describes the analytical reporting methods used in a preliminary, scoping assessment to evaluate the extent to which the EA's target PFAS suite captures total PFAS in the environment. The data collection, undertaken in 2024, applied the TOP assay to 100 surface water and groundwater samples collected from 98 sites across England, selected from the EA's PFAS surveillance programme.

We summarised the TOP assay data using multiple alternative calculation approaches to reflect the choices that could be applied in TOP assay assessments and the different options available for handling censored data. The primary GAM-based analysis explicitly accounts for concentrations below the method reporting value (<MRV) through statistical modelling (see Appendix 5 for further details).

For comparison, we performed alternative calculations in which concentrations below the MRV were set to zero, consistent with approaches used in some TOP assay studies. Under this approach, percentage coverage of total PFAS captured by the EA target suite can be calculated either by estimating coverage at each site and then averaging across sites, or by summing PFAS concentrations across all sites prior to calculating an overall coverage value.

In all cases, we calculated percentage coverage of the EA target PFAS suite as:

$$Coverage\ (\%) = \frac{\sum PFAS_{pre-TOP}}{\sum PFAS_{post-TOP}} \times 100$$

Model-based (GAM) TOP Assay Analysis

Analysis of the TOP assay data followed the same general modelling framework (GAM) as the full-PFAS analyses described in Appendix 5, but with a simplified structure reflecting the design of the TOP assay study.

Prior to modelling, we filtered the available TOP assay data to exclude PFAS with very low detection frequencies across the combined pre- and post-TOP datasets. We excluded substances detected in less than 1% of observations (fewer than three values above the MRV), resulting in 18 of the 49 PFAS being included in the analysis.

The TOP dataset comprised a single paired pre- and post-oxidation sample per site, with samples collected within a narrow time window. The model therefore focused on estimating differences between pre- and post-oxidation conditions, rather than temporal effects. We estimated concentrations on the $\log_{10}$ scale using a GAM, with a categorical fixed effect for oxidation condition (pre- or post-), a random intercept for site, and random intercept for PFAS determinand, the latter estimated separately for pre- and post-oxidation conditions. This structure allows PFAS-specific responses to oxidation to be captured while accounting for between-site variability.

Given that the primary aim of the analysis was estimation and prediction of total PFAS concentrations, we used default basis dimensions and penalisation was relied upon to

control model complexity. We encountered no convergence issues, and model diagnostics did not indicate any major violations of assumptions.

We generated 1,000 simulations from the fitted model to estimate concentrations for individual PFAS and total PFAS in the pre- and post-TOP assay groups, and to derive corresponding 95% confidence intervals for these estimates.

We undertook all analyses in R (R Core Team, 2025) with models fitted using REML in the mgcv package.

Appendix 8: Summary of PFAS analytical method options

Method	Advantages	Limitations	Potential Applications
Non-Target Analysis (NTA) & Suspect Screening	• Detects unknown, emerging, and novel PFAS • Broad chemical coverage beyond predefined target lists • Can support tentative structural identification of unknowns	• Requires high-resolution mass spectrometry (HRMS) and expert data interpretation • Generally qualitative or semi-quantitative • Resource-intensive and costly • Potential bias from sample extraction and data-processing work-flows •May not be able to identify all PFAS, with PFAS shorter than C_4 challenging to detect	Environmental surveillance, exploratory monitoring, source identification, research and early-warning studies
Targeted Analysis (Fully Quantitative)	• Highly specific and fully quantitative for predefined PFAS (typically legacy PFAS such as PFAAs) • Standardised and validated analytical methods available • Regulatory acceptance and certified standards typically available • High sensitivity (typically ~0.1–1 ng L^{-1})	• Limited to the prescribed list of target PFAS • Does not capture unknown PFAS or total PFAS burden • Variable recoveries for some compounds • Potential bias from sample extraction	Regulatory compliance monitoring, risk assessment, trend analysis, detailed characterisation of known PFAS

Method	Advantages	Limitations	Potential Applications
Target Screening (Semi-Quantitative)	• Wider PFAS coverage than targeted analysis • Useful where reference standards are unavailable • More cost-effective than non-target analysis	• Semi-quantitative only (concentration estimates) • Higher uncertainty due to use of surrogate or class-based calibration • Not suitable for regulatory compliance	Screening, prioritisation of samples or compounds for follow-up targeted analysis, environmental monitoring
Total Fluorine Content (TF)	• Rapid and broad indication of fluorine presence • Useful for initial screening	• Non-specific; includes inorganic fluorine • Does not distinguish PFAS from other fluorinated substances	Preliminary screening to flag samples for further PFAS-specific analysis
Inorganic Fluorine (IF)	• Differentiates inorganic fluorine contribution • Improves interpretation of total fluorine measurements	• Does not provide organofluorine or PFAS-specific information	Supporting interpretation of total fluorine results
Total Organic Fluorine (TOF)	• Indicates presence of total organofluorine, including PFAS • Lower analytical cost compared with LC-MS/MS • Useful screening tool	• Non-targeted; includes non-PFAS organofluorine substances (e.g. some pharmaceuticals) • Lack of standardised methodologies • Generally lower sensitivity than targeted PFAS analysis • Limited consensus on how to integrate TOF with targeted or TOP data	Screening for overall organofluorine burden and identifying gaps in targeted PFAS coverage
Method	**Advantages**	**Limitations**	**Potential Applications**

Extractable Organic Fluorine (EOF)	• Focuses on extractable fraction of organofluorine • Useful indicator of potentially mobile PFAS	• Extraction efficiency varies by matrix and compound • May exclude strongly bound or non-extractable PFAS	Screening and mass-balance assessments of organofluorine
Adsorbable Organic Fluorine (AOF)	• Targets organofluorine that adsorbs to activated carbon • Relevant for water treatment contexts	• May underestimate highly soluble or poorly adsorbing PFAS • Method-dependent selectivity	Screening and treatment-related assessments
Total Oxidisable Precursor Assay (TOP)	• Indicates presence of polyfluoroalkyl precursors that oxidise to terminal PFAAs • Useful for estimating potential PFAS burden beyond target lists as no standards needed • Low detection limits and formal reproducibility criteria	•Oxidation efficiency varies between precursor classes • Limited to PFAS that oxidise to measurable PFAAs • Challenging interpretation for regulatory compliance and risk assessment •No compound specificity •Does not indicate natural oxidation rates nor how long it may take for the post-oxidation composition to be achieved	Identifying oxidisable PFAS precursors not detected by targeted analysis, supporting conceptual site models

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List of abbreviations

Abbreviation	Definition
AFFF	Aqueous Film Forming Foam
BAT	Best Available Techniques
CIP	Chemicals Investigations Programme
CoT	UK Committee on Toxicity
Defra	Department for Environment Food and Rural Affairs
ECHA	European Chemicals Agency
EFSA	European Food Safety Authority
EPR	Environmental Permitting Regulations
EQS	Environmental Quality Standard
EU	European Union
FCMs	Food contact materials
FFF	Firefighting foams
FSA	Food Standards Agency
FTOH	Fluorotelomer alcohol
GAM	Generalised Additive Model
GAMM	Generalised Additive Mixed Model
GWQMN	Groundwater Quality Monitoring Network
HSE	Health and Safety Executive
HTI	High Temperature Incineration
ITRC	Interstate Technology and Regulatory Council
LC-carbonyl PFAS	Perfluoroalkyl carbonyl compounds (long chain)
LC-sulfonyl PFAS	Perfluoroalkane sulfonyl compounds (long chain)
MOD	Ministry of Defence

MRV	Minimum reporting value
N-EtFOSAA	N-ethyl perfluorooctane sulfonamidoacetic acid
N-MeFOSAA	N-methyl perfluorooctane sulfonamidoacetic acid
OECD	Organisation for Economic Co-operation and Development
OSPAR-RID	OSPAR Riverine Inputs and Direct Discharges
PFAAs	Perfluoroalkyl Acids
PFAS	Per- and polyfluoroalkyl substances
PFBA	Perfluorobutanoic acid
PFBS	Perfluorobutane sulfonate
PFCA	Perfluoroalkyl carboxylic acid
PFCAs	Perfluoroalkyl carboxylic acids
PFDA	Perfluorodecanoic acid
PFHxA	Perfluorohexanoic Acid
PFHxS	Perfluorohexane sulfonate
PFHpS	Perfluoroheptane sulfonic acid
PFNA	Perfluorononanoic acid
PFOS	Perfluorooctane sulfonic acid
PFOSA	Perfluorooctane sulfonamide
PFOA	Perfluorooctanoic acid
PFPEs	Perfluoropolyethers
PFSA	Perfluoroalkyl sulfonic acid
PFSAs	Perfluoroalkane sulfonic acids
PI	Pollution inventory
POPs	Persistent Organic Pollutants
PTFE	Polytetrafluoroethylene
RMOA	Regulatory Management Option Analysis

RSN	River Surveillance Network
SC-carbonyl PFAS	Perfluoroalkyl carbonyl compounds (short chain)
SC-sulfonyl PFAS	Perfluoroalkane sulfonyl compounds (short chain)
SSVs	Soil screening values
SVHC	Substance of Very High Concern
TFA	Trifluoroacetic Acid
TOP	Total oxidizable precursor
TULAC	The textiles, upholstery, leather, apparel, and carpets
UK REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
WFD	Water Framework Directive
WwTW	Wastewater treatment works
5:3 FTCA	5:3 fluorotelomer carboxylic acid
6:2 FTAB	6:2 fluorotelomer sulfonamidoalkyl betaine
6:2 FTSA	6:2 Fluorotelomer sulfonic acid

Glossary

Adsorb: A sorption process in which one substance becomes attached to another via adhesion to the surface.

Arrowheads: The formation of highly persistent final degradation products (perfluorinated terminal products)

Bioavailability: The extent to which a substance can be absorbed by a living organism

Bioaccumulation: The process by which a substance builds up in an organism over time from all routes of exposure.

Bioaccumulation Factor (BAF): The ratio of a substance's concentration in an organism to its concentration in the surrounding environment, used as an indicator of bioaccumulation potential.

Biomagnify: Through food chains, whereby organisms at higher trophic levels accumulate greater concentrations

Biosolids: Biosolids are the nutrient-rich organic materials resulting from the treatment of sewage biosolids. They can be used as fertilisers but may contain PFAS due to their presence in wastewater. Concerns regarding PFAS in biosolids highlight the need for careful management and potential treatment before application on land.

Best Available Techniques (BAT): The most effective and advanced stage in the development of activities and methods for preventing or minimising emissions and environmental impacts.

Biota: Living organisms used in environmental monitoring, including fish and other aquatic species.

Censored data: Analytical results reported as below a defined reporting or detection limit, where the true concentration is unknown but bounded.

Environmental Quality Standard (EQS): A legally defined concentration of a substance in the environment intended to protect human health or aquatic ecosystems.

Ecotoxicity: Is the negative impact of a substance or activity on a population of animals, plants, or microbes in the environment.

Estuarine waters: Transitional waters where freshwater from rivers mixes with seawater, characterised by variable salinity, strong hydrodynamic processes and complex chemical behaviour.

Fluorotelomer PFAS: A group of PFAS used in industrial and consumer applications that may degrade into more persistent perfluoroalkyl acids.

Generalised Additive Model (GAM): A flexible statistical modelling approach used to describe smooth relationships between a response variable (such as chemical concentration) and one or more predictor variables (such as sampling date or season). GAMs can also handle grouped data, for example measurements collected over time at multiple monitoring sites.

Groundwater: Water stored or flowing beneath the ground surface in permeable soil or rock formations.

Harmonised Monitoring Scheme (HMS): a long-term river water-quality monitoring network, established in the 1970s to provide consistent data on major rivers for assessing long-term trends in water quality. While HMS data are no longer reported by Defra as official statistics, the Environment Agency has continued to undertake monitoring at most HMS sites. More recently the Environment Agency has begun monitoring of some PFAS at some HMS sites. Some HMS monitoring sites overlap with OSPAR sites.

Headgroup: The headgroup is the hydrophilic part of a molecule, containing one or more functional groups (e.g., carboxylic acid or sulfonic acid) attached to a relatively long aliphatic tail or backbone. In PFAS, the headgroup influences solubility, reactivity, and environmental behaviour.

Hydrostratigraphic Unit: A body of geological material (rock, sediment, or soil) that has relatively uniform hydraulic properties meaning the way it interacts with groundwater is consistent throughout that unit. Hydrostratigraphic units are defined by their permeability, porosity, storage properties, and hydraulic conductivity.

Limit of Detection (LoD): The lowest concentration of a substance that can be reliably distinguished from analytical noise but not necessarily measured with a defined level of precision (see LoQ). It is determined statistically from the variability of replicate blank or low-level spiked samples.

Limit of Quantification (LoQ): The lowest concentration of a substance that can be quantified with a defined level of precision and accuracy. Under Environment Agency procedures, it is typically 2–3 times the LoD for fully-quantitative analyses (this includes all the data used in this report).

Mineralisation: refers to the complete chemical breakdown of PFAS into their simplest inorganic components primarily fluoride ions (F^-) and carbon dioxide (CO_2).

Minimum Reporting Value (MRV): The lowest concentration that a laboratory routinely reports as a quantitative result. For Environment Agency laboratories, the MRV is normally the statistically-defined LoD rounded up to one significant figure, although it may be set higher for operational or other reasons.

OSPAR: Oslo–Paris Convention, formally the Convention for the Protection of the Marine Environment of the North-East Atlantic. The Environment Agency undertakes monitoring for the purposes of reporting to OSPAR. This includes water quality monitoring close to the tidal limit of major rivers for the purpose of estimating river-borne inputs of pollutants to the sea termed RI (river input). Most OSPAR RI sites are also Harmonised Monitoring Scheme sites.

Perfluoroalkyl acids (PFAAs): Highly persistent PFAS that represent terminal degradation products and are often the focus of regulation and monitoring.

PIT (probability inverse transform) residual. A PIT residual is the probability, under the fitted model, of observing a value less than or equal to the one observed. They should be uniformly distributed if the model is correctly specified.

Precursors: PFAS that can transform in the environment or within organisms into more persistent terminal PFAS.

Random effect: A way of modelling differences between natural groups (such as monitoring sites) by assuming each group varies around a common average, rather than estimating each group completely separately.

Remediation: The process of removing, reducing, or eliminating contaminants from soil, water, air, or waste to protect human health and the environment. Methods include physical, chemical, biological, and thermal treatments to restore affected areas and prevent further pollution.

River Surveillance Network (RSN): A national monitoring network designed to provide representative information on river water quality across England.

Source-Pathway-Receptor Model: A conceptual framework used in environmental science to understand the movement of contaminants from their source through various pathways to potential receptors. In the context of PFAS, this model helps to identify potential sources of PFAS, pathways of contamination, and populations or ecosystems at risk.

Surface water: Open water bodies including rivers, lakes, estuaries and coastal waters.

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