PFAS in the natural environment and human infrastructure

Behaviour in Natural Environments

Soil and Vadose Zone

After use, PFAS in the spent firefighting foams infiltrate into the ground, which can consist of hard manmade surfaces (typically asphalt and concrete) or unsaturated soil. Sorption in unsaturated soil is driven by hydrophobic interactions of the fluorinated "tails" with soil organic matter and by accumulation at air-water interfaces (Guelfo & Higgins, 2013). Anionic PFAAs are electrostatically repelled by negatively charged clay minerals, yet multivalent cations, such as calcium ions, can bridge PFAS to mineral surfaces, increasing retention. Air-water interfacial sorption, especially for long-chain PFOS and PFHxS, can retard downward migration for years (Guelfo & Higgins, 2013). Sorption increases with chain length, so short-chain PFHxA and PFBS leach rapidly, whereas PFOS often remains concentrated in the top 0–50 cm, acting as a continuing source that slowly desorbs during rainfall events (Douglas et al., 2023). Several other factors, such as pH and the presence of co-contaminants may also be important.

Groundwater

Once PFAS reach the saturated zone they behave as highly soluble, conservative solutes, forming plumes that may extend miles downgradient (Guelfo & Higgins, 2013). Differential mobility separates compounds by chain length (partly through differential sorption by chain length: PFBA and PFHxA form the leading edge, while PFOS and PFHxS lag nearer the source. Because natural degradation is negligible, and slow transport from unsaturated zones, plumes persist for decades; modelling suggests century-scale time frames for significant attenuation in the absence of intervention (ITRC, 2021).

Groundwater discharges to rivers and wetlands transfer PFAS to surface waters, broadening exposure pathways (Reinikainen et al., 2022).

Surface Water and Sediments

Stormwater runoff can carry foam residues from concrete pads, asphalt, and contaminated soil into drainage ditches and streams (Kali et al., 2025) (although a proportion can stick to the surface and slowly leach through). In open water, short chain acids remain dissolved. PFOS and PFHxS partition partly to suspended solids and, along with other longer chain PFAS, settle into sediments, creating long-term benthic reservoirs (ITRC, 2021). Aquatic foams formed during wave action can enrich PFAS by orders of magnitude, redistributing them to shorelines or volatilising droplets into the air (Reinikainen et al., 2022).

Air

Many (but not all) AFFF-derived PFAS have negligible vapour pressure; however, spraying creates aerosols that transport PFAS locally. Additionally, volatile precursors such as fluorotelomer alcohols present in some formulations can evaporate, undergo atmospheric oxidation, and redeposit as PFAAs, contributing to regional background loads (Brunn et al., 2023).

Biota

Long-chain sulphonates and carboxylates bind to blood proteins and bioaccumulate. PFOS and PFHxS display long half-lives in humans (4–8 years and 5–7 years, respectively), whereas PFHxA and PFBS are cleared within weeks to months (Wee & Aris, 2023).Half lives can vary greatly

between and within species. In aquatic food webs near AFFF sites, PFOS concentrations in fish can exceed advisory thresholds, posing dietary risks (Reinikainen et al., 2022).

Interactions with Built Environments

Concrete and Asphalt

Porous concrete absorbs PFAS-laden foam. Core samples from fire-training pads routinely contain 10^2 – $10^5 \mu g kg^{-1}$ of PFOS and 6:2 FTS, with highest levels in the top centimetres (Douglas et al., 2023). Leaching experiments show that only a few percent of total mass is released per rainfall simulation, indicating decades-long secondary emissions. Asphalt runways also sorb PFAS; PFOS binds to the bitumen phase and can be mobilised by run-off, though initial leaching percentages are somewhat higher than for concrete (Li et al., 2024).

Stormwater Infrastructure

Drains, oil / water separators, and retention basins capture AFFF runoff and themselves accumulate PFAS in sediments and biofilms (Kali et al., 2025). Storm events resuspend these deposits, creating episodic pulses to downstream waters. Remediation therefore must consider physical infrastructure as an integral part of the PFAS mass balance (ITRC, 2021).

Key Transport and Redistribution Processes

- **Leaching:** Vertical migration of dissolved PFAS through soil into groundwater; rate controlled by chain length and interfacial sorption (Guelfo & Higgins, 2013).
- **Runoff:** Overland flow mobilises surface-deposited PFAS to bodies of water and into drains (Kali et al., 2025).
- **Sorption/Desorption:** Reversible binding to soils and concrete dictates long-term release; stronger for PFOS/PFHxS than for PFHxA/PFBS (Douglas et al., 2023).
- Air-Water Interfacial Adsorption: Major retention mechanism in unsaturated soils, particularly for long-chain PFAS (Guelfo & Higgins, 2013).
- **Precursor Transformation:** Partial fluorinated molecules such as 6:2 FTS biodegrade slowly to terminal PFAAs (NASF, 2019).
- Aerosolisation: Foam spray droplets carry PFAS short distances downwind. Volatile precursors contribute to long-range deposition after atmospheric oxidation (Brunn et al., 2023).

Persistence

Fully fluorinated PFAAs (e.g., PFOS, PFHxS, PFOA) resist hydrolysis, photolysis, and microbial attack, giving environmental half-lives that exceed centuries in groundwater and sediments (Brunn et al., 2023). Precursors transform rather than mineralise, ultimately increasing the pool of persistent acids. Consequently, AFFF plumes function like conservative tracers, and contaminated infrastructure or soil continues to release PFAS indefinitely unless physically removed or isolated (ITRC, 2021).

Implications for Management

Because natural attenuation is negligible, mitigation must focus on source removal or immobilisation. Options include excavation of contaminated soil, in-situ stabilisation (e.g., soil

blending with sorbents), pump-and-treat of groundwater, and removal or surface-sealing of PFAS-laden concrete and asphalt (Douglas et al., 2023). Stormwater capture and treatment; often with granular activated carbon or ion exchange resins; may be important for preventing offsite transport (Kali et al., 2025). Knowledge of differential mobility is useful: short-chain PFAS require hydraulic containment, whereas long-chain PFAS demand strategies targeting sorbed phases in soil or hardstanding.

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Chemistry of AFFF

Aqueous Film Forming Foams (AFFF) used for firefighting prominently incorporate PFAS because of the ability of these strong surfactants to spread an aqueous film over fuel fires and thus effectively extinguish the fire. Over the years, however, concerns over the bioaccumulation and environmental persistence (discussed below) of certain PFAS in the AFFF have led to shifts in chemistries of AFFF.

PFAS Types in AFFF

Historically the 3M PFAS-based foams (developed in the late 1960s) containing long-chain PFSAs (especially PFOS and PFHxS) and PFCAs (notably PFOA) dominated the market with socalled fluorotelomer-based AFFF (developed in 1970s) having a small share. Since 3M discontinued the manufacture of AFFF in 2002, there was a shift to the use the fluorotelomerbased AFFF (developed in the 1970s). The fluorotelomer-based foams contain shorter chain PFAS. In recent years, there has been a shift towards fluorine-free alternatives in many countries and regions; Sweden, for example. The key characteristics of PFAS-containing foams are:

- Legacy (ECF) AFFF. Historical "Light Water" type formulations contained complex mixture of fluorosurfactants including mixtures of long chain perfluoroalkane sulfonates, dominantly PFOS with lesser PFHxS (Backe et al., 2013). Long-chain perfluoroalkyl carboxylates such as PFOA are also often present in smaller amounts (Guelfo & Higgins, 2013).
- Fluorotelomer AFFF. These foams were developed in the 1970s and used alongside the 3M AFFF products, but only had a relatively small share of the market. Post-2002, these foams often replaced the previous use of 3M AFFF and began to dominate the market. They contain complex mixtures of fluorotelomer-based fluorosurfactants. They are based on C6 fluorotelomer chemistries and components include PFHpA, PFHxA, PFPeA and 6:2 fluorotelomer sulfonate (FTS).
- **Complex mixtures**. Analyses of commercial foams reveal hundreds of individual PFAS and up to 40 distinct PFAS classes, including anionic, cationic, and zwitterionic species (Backe et al., 2013). The main fluorosurfactant components are so-called precursors which break down in the environment and form PFSAs and PFSAs. Field contamination therefore reflects overlapping generations of usage, with long-chain sulfonates and carboxylates from 3M foams and short-chain acids from newer foams often co-located in soil and water (ITRC, 2021). The complex mixtures of substances in AFFF are unique to a particular product. Understanding the "chemical fingerprints" of these products in environmental samples, which include the presence of specific isomers or breakdown products, is crucial for tracking and mitigating environmental contamination.

The only AFFF products which have PFOS and PFHxS as markers are 3M Lightwater AFFF products. These were initially used in the US, from 1967, and were also used in Jersey and elsewhere. The presence of those specific PFAS in Jersey, suggest that it is these products, rather than something else, that are the primary source of PFOS and PFHxS contamination. Most users transitioned away from 3M Lightwater to fluorotelomer-based products in the early 2000s when 3M discontinued manufacture of AFFF in 2002. Some later transitioned to fluorine-free foams (3F). PFOA is a marker of both 3M Lightwater AFFF products and the older fluorotelomer-based AFFF products. A unique marker of fluorotelomer-based foams is 6:2 FTS.

Bioaccumulation

The tendency of PFOS, PFHxS and PFOA to bioaccumulate in biological tissues raises concerns about their long-term health impacts. Regulatory bodies have identified long-chain PFAAs as particularly bioaccumulative (as they bioaccumulate in humans), emphasizing the need for stringent regulatory controls for long-chain PFAAs (i.e. PFSAs with six carbons and greater (PFHxS and upwards) and PFCAs with 8 carbons or greater (PFOA and upwards)). PFSAs are relatively more bioacculative than PFCAs with equivalently long perfluorinated chains because of the special effect of the sulfonate functional group (Brunn et al., 2023; Buck et al., 2011).

Environmental Accumulation

The chemical stability of PFOS, PFOA, and PFHxS due to the strength of the carbon-fluorine bonds leads to their extreme persistence in the environment, posing significant challenges for remediation. Efforts to monitor and reduce environmental levels of these pollutants are ongoing, but their inert nature complicates effective degradation and removal strategies (Cousins et al., 2020; Prevedouros et al., 2006). This will be explored in more detail in Report 4.

Regulatory and environmental concerns

PFAS are notably persistent in the environment, which complicates their management and regulation. In response, Denmark, Germany, the Netherlands, Norway, and Sweden have proposed a REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) restriction for all PFAS meeting the OECD definition (with some exceptions). It is recognized that PFAS have diverse properties (e.g. different toxicities), but they are all highly persistent in the environment. Additionally, there is an ongoing restriction proposal specifically targeting PFAS in firefighting foams, with a proposed 10-year phase-out, which is also currently under consideration at the European Commission.

PFAS encompass a broad and complex category of chemicals that pose significant challenges due to their persistence, bioaccumulation, and widespread use. Ongoing research and regulatory efforts aim to better understand and mitigate their health and environmental impacts. Looking forward, enhanced analytical methods and comprehensive toxicological data are increasingly important for guiding effective policy and industry practices.

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Testing for PFAS

Testing (sampling and analysis) for per- and polyfluoroalkyl substances (PFAS) in the environment is challenging for four main reasons. Firstly, PFAS have an extraordinary chemical diversity. More than 12 000 PFAS moieties exist, but routine methods quantify only a few dozen; many others lack reference standards so they escape targeted detection (Wang et al., 2017). Second, parts-per-trillion concentrations. Regulatory limits for drinking water are now moving towards the single-digit ng L⁻¹ range, demanding methods and instruments that can distinguish a handful of PFAS molecules from background contamination (Zahra et al., 2025). Third, strong matrix effects. (ITRC, 2023). Matrix effects in PFAS analysis occur when components in complex environmental samples (soil, wastewater, or blood) interfere with the detection of PFAS by either suppressing or enhancing the analytical signal. This can lead to inaccurate quantification unless properly corrected using internal standards, matrix-matched calibration, or rigorous sample cleanup. Fourth, ubiquitous background contamination. Fluoropolymer tubing, waterproof clothing, and even ambient laboratory air can introduce trace levels of PFAS into samples. Studies have shown that field blanks frequently contain detectable PFOS or PFOA, prompting laboratories to adopt PFAS-free materials and implement rigorous blank controls to ensure data integrity. (ITRC, 2023). Together, these factors mean that analysts must combine rigorous sampling protocols, isotope-dilution mass spectrometry, and multiple quality-control layers to produce reliable PFAS data.

How samples are taken

Because PFAS are ubiquitous outdoors and indoors, laboratories follow strict sampling rules to avoid adding contamination or losing the chemicals before they reach the instrument. Field teams avoid equipment and materials which are known to contain PFAS, and meticulously rinse all equipment (e.g., sampling tools and containers) with PFAS-free water between samples. They also collect field blanks (sealed bottles that are exposed in the field, which travel together and are analysed with the real samples) to prove the chain of custody is clean and also to determine quantification limits.

Surface water and drinking water

Scientists collect water samples in high-density polyethylene or polypropylene bottles (glass can adsorb PFAS). Samples are chilled, transported the lab and analysed as soon as possible. In the lab a solid-phase extraction (SPE) cartridge is typically used to concentrate PFAS from the water and to transfer the PFAS into a solvent. The solvent is concentrated down to a few hundred microliters of solvent and a tiny fraction is injected onto the instrument for analysis. Clean-up procedures may be applied to separate PFAS from the sample matrix which can interfere with the analysis.

Surface sea water and sea foam

Near some coasts wind and wave action can lead to the formation of foam on the sea surface. This foam is naturally produced from the many natural surfactants present in natural organic matter present in sea water. Research over the last few years shows that foam can concentrate PFAS many thousandfold above the surrounding water concentrations; single sea-foam samples from the US Atlantic coast recently contained almost one million nanograms per litre of PFOS (Enders J, 2025). To capture foam, scientists skim it gently with a pre-cleaned stainless steel ladle into PFAS-free bottles or plastic bags and treat it like a highly contaminated water sample (the foam rapidly collapses in sample bottles to brown-coloured water), diluting a measured portion with reagent water before applying SPE.

Soils and sediments

A stainless steel spade or core sampler takes at least 100 g of material to a specific depth (depending on the objectives of the study), which is stored chilled or even frozen before analysis. In the laboratory, a subsample is typically taken (e.g., 2 to 5 g are weighed), spiked with isotope-labelled standards, and then extracted with a solvent. The extract is concentration and cleaned to remove interring matric (e.g., humic and fulvic acids present in soil organic matter) prior to injection on an instrument for analysis.

Air sampling

High-volume air samplers (which are like powerful vacuum cleaners) draw hundreds of cubic metres of air through quartz fibre filters (to trap particles and particulate-bound PFAS) followed by polyurethane foam or XAD resin (to trap vapour-phase PFAS). The samples are extracted with solvent, concentrated and cleaned up prior to analysis on an instrument.

Wildlife tissues

Biological tissues (e.g., fish muscle, bird liver or invertebrate homogenates) are weighed, spiked with isotope labelled standards and then extracted with a solvent. The samples are extracted with solvent, concentrated and cleaned up prior to analysis on an instrument.

Core laboratory methods

Once in solvent, all sample types converge on similar analytical approaches: liquid chromatography coupled to tandem mass spectrometry (LC–MS/MS) for targeted analysis or liquid chromatography coupled to high resolution mass spectrometry for broader screening. The liquid chromatograph separates PFAS compounds based on their interactions with the column, while the mass spectrometer identifies and quantifies them. Triple-quadrupole instruments are the industry standard for targeted analysis, capable of detecting individual PFAS compounds in complex mixtures by monitoring specific ion transitions, even in the presence of thousands of other chemicals.

Each extract typically receives a set of stable isotopically labelled internal standards — PFAS compounds that are chemically similar to the targets but slightly heavier due to atoms like carbon-13 or deuterium. These internal standards mimic the behaviour of native PFAS during extraction and analysis, helping to correct for losses, matrix effects, and variability in the mass spectrometer. This approach enhances accuracy and quantification, though its effectiveness depends on how well the labelled compounds match their target analytes.

How sensitive are laboratory tests?

- Limit of detection (LOD) is the smallest amount of a chemical that produces a signal distinguishable from background noise. Below the LOD the laboratory cannot reliably determine the presence of the chemical.
- Limit of quantification (LOQ) is higher than the LOD. It is the smallest amount the laboratory can measure with acceptable accuracy and precision (often defined as a relative standard deviation below 20 %). Between the LOD and LOQ a result is reported as "detected but not quantified."

Using today's instruments and one-litre water samples, typical LODs for PFOS, PFOA and PFHxS are 1 to 2 parts per trillion (ng/L) and LOQs are roughly 3-5 ng/L. High end systems concentrating five litres of water have pushed LOQs below 0.5 ng/L, while some research groups have demonstrated part-per-quadrillion capability for PFOS in ultrapure matrices (Ackerman Grunfeld et al., 2024). In soils and wildlife tissues, matrix interferences are stronger; so realistic LOQs are 0.5 to 2 parts per billion (ng/g): still sensitive enough to flag most contaminated hotspots.

Special challenges for PFOS, PFHxS and PFOA

All three target molecules ionise well in the mass spectrometer, which helps sensitivity, but they also stick to glass and Teflon surfaces, so sample bottles, pipette tips and tubing must be fluoropolymer-free. PFOS and PFHxS were historically manufactured as mixtures of linear and branched isomers. Modern analytical methods either sum the isomers or employ calibration standards that reflect the specific isomeric distribution in the sample.

PFHxS is shorter and more water-soluble than PFOS, making it more difficult to trap on traditional C18 cartridges used for extraction. However, weak anion exchange (WAX) sorbents introduced in 2019 have significantly improved recovery. PFOA, while phased out in many countries, continues to cause laboratory contamination problems due its ubiquitous presence (even in laboratory solvents). Therefore, quality control steps, such as reagent blanks, are vital to identify and correct for potential false positives during sample preparation.

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Treatment technologies for removing PFAS from drinking water

This section focuses on mature technologies that have been implemented for the removal of PFAS from municipal drinking water treatment plants. Emerging or developing technologies are not included, as they are not yet sufficiently advanced for implementation by Jersey Water in the near future. For the purposes of this review, mature technologies are defined as those that have been successfully applied in the field by multiple organizations across various sites, with well-documented results either in practice or in peer-reviewed literature.

In the context of drinking water treatment, mature PFAS removal technologies fall into two primary categories: sorption and high-pressure membranes. The main types of sorption technologies are granular activated carbon (GAC) and ion exchange resins (McCleaf, Englund et al. 2017, Ross, McDonough et al. 2018), though some novel sorbents have also been explored (Burkhardt, Speth et al. 2025). The high-pressure membrane category includes nanofiltration (NF) and reverse osmosis (RO) (Li, Duan et al. 2020) (McCleaf, Stefansson et al. 2023, Sadia, ter Laak et al. 2024) (Safulko, Cath et al. 2023).

Additionally, foam fractionation may be used to concentrate and remove a portion of PFAS from drinking water(McCleaf, Stefansson et al. 2023). However, this method is typically employed as a pretreatment step, followed by further purification using sorption technologies.

A summary of these technologies is provided below.

Granular activated carbon (GAC)

Sorption is a physical process where PFAS molecules bind to a sorbent media. GAC is a widely used sorbent that effectively removes many organic contaminants, including long-chain PFAS such as PFOA, PFHxS and PFOS, reducing their concentrations to low ng/L levels until breakthrough occurs (i.e., the media becomes saturated and cannot continue to sorb PFAS). Because GAC is non-selective, it also adsorbs co-contaminants and even natural organic matter (NOM), which can accelerate the saturation of the GAC and increase replacement frequency (McCleaf, Englund et al. 2017, Ross, McDonough et al. 2018).

GAC removal capacity and breakthrough time vary by PFAS type. Longer chain PFAS (e.g., PFOS) are more effectively removed than shorter chain PFAS (e.g., PFHxA), which tend to break through more quickly and require more frequent changes of the GAC. Adsorption efficiency is also influenced by PFAS functional groups; sulfonates generally have higher affinity than carboxylates of the same chain length. PFAS precursors can also compete for adsorption sites and degrade to other PFAS (e.g., PFAS) over time (McCleaf, Englund et al. 2017, Ross, McDonough et al. 2018).

Most full-scale GAC systems focus on treating drinking water and have demonstrated reliable removal of PFOA and PFOS (McCleaf, Englund et al. 2017). However, data on other PFAS, especially short-chain compounds, are more limited. Research continues towards improving GAC performance for shorter chain PFAS and integrating complementary technologies such as ion exchange. Hybrid treatment systems, where GAC precedes ion exchange, are also adopted. In such setups, GAC removes longer chain PFAS and non-PFAS organics, while ion exchange also targets the shorter chain PFAS (McCleaf, Englund et al. 2017).

GAC system design should be based on lab treatability tests (e.g., rapid small-scale column tests) as well as pilot-scale testing to refine system parameters under site-specific conditions. Conditions that should be tested include bed depth (vertical height of the GAC in the treatment vessel), empty bed

contact times (how long the water stays in contact with the GAC), how often the GAC needs to be replaced, and treatment cost. Testing with actual site water is critical, as water chemistry greatly influences performance.

GAC is produced from materials such as bituminous coal, lignite, and coconut shells. Bituminous coal-based GAC is most commonly used for PFAS treatment due to its good performance in testing (Westreich, Mimna et al. 2018). Media selection should consider PFAS type, co-contaminants, material performance, availability, and cost. GAC systems use relatively long empty bed contact times (McCleaf, Englund et al. 2017), which means that they require larger vessels compared to alternatives like ion exchange.

Spent GAC can be thermally reactivated and reused (Baghirzade, Zhang et al. 2021), reducing life cycle costs, though this option depends on access to reactivation facilities. Currently, Jersey lacks such facilities, and regulatory limitations may restrict off-island transport of PFAS-laden GAC, potentially classifying it as hazardous waste.

Some challenges associated with GAC systems include: energy-intensive production and transport, large vessel requirements due to longer empty bed contact times, increased space and infrastructure needs and transportation and disposal or reactivation of spent media. These factors must be accounted for during system planning and lifecycle analysis to ensure cost-effectiveness, environmental compliance, and operational resilience.

Ion-exchange resins

Ion exchange involves replacing charged ions in the water with non-toxic ions (e.g., chloride), targeting the ionized functional groups in PFAS molecules (e.g., the sulfonate group in PFOS). PFAS are removed by ionic bonding between the negatively charged sulfonate or carboxylate heads and the positively charged resin, along with hydrophobic interactions between the PFAS tail and the resin matrix. Similar to GAC, PFSAs are more easily removed compared to PFCAs of the same chain length, with longer-chain PFAS more strongly retained (McCleaf, Englund et al. 2017, Ross, McDonough et al. 2018).

Ion exchange resins have long been used in drinking water treatment for contaminants such as nitrate, perchlorate, and arsenic. For PFAS removal, both single-use and regenerable ion exchange resins are available (Ellis, Boyer et al. 2023). Single-use resins are operated until breakthrough at a target concentration, then disposed of; typically via incineration or landfilling. They are commonly used for drinking water applications due to ease of use and high removal capacity, often achieving low ng/L levels. Regenerable resins, on the other hand, are cleaned in place using brine or solvent-brine regenerant solutions, restoring some of their PFAS removal capacity. These systems are more suitable for sites with high influent PFAS concentrations where the cost of resin replacement would be high.

Ion exchange resin systems are typically designed using "lead-lag" vessels (Rodowa, Knappe et al. 2020). The lead vessel treats the incoming water and the lag vessel, placed directly after the lead, sorbs PFAS that break through the lead vessel. A third polisher vessel ("lead-lag-polisher" design) may also be added to extend operational capacity and reduce change-out frequency. Empty bed contact times are much shorter than for GAC, allowing for smaller treatment systems which require less space on site. Selective ion exchange resins can treat large bed volumes, depending on PFAS type and influent water quality.

Co-contaminants such as natural organic matter, sulfate, and nitrate can reduce PFAS removal capacity by competing for resin sites or causing fouling. Pretreatment may be necessary, especially for surface waters with high organic matter or groundwater with iron and manganese. Influent characterization is critical to assess pretreatment needs and resin performance. Lab-scale and pilot testing are required to predict resin performance, including breakthrough curves and loading capacities.

Challenges for the operational implementation of ion exchange resin technologies are the energy and materials required for continual replacement and disposal of the resins (in the case of single-use resins) and for regeneration, transportation and disposal of spent resins and regenerant waste (in the case of regenerative resins). Finally, although ion exchange resins have high selectivity for PFAS, their lifetimes are ultimately determined by competition from more abundant anions present in the water such as chloride, sulfate, and bicarbonate.

Regenerable systems may offer long-term cost benefits for high-load sites, especially when centralized regeneration facilities are available. Regenerant waste can be concentrated and managed as solid waste (incinerated or landfilled), or possibly treated via emerging destruction technologies. However, Jersey is not a high-load site and regeneration facilities are not currently available for ion exchange resins on island. In a recent study, reactivated GAC was shown the have lower lifecycle impacts compared to both single-use GAC and regenerable ion exchange resins (Ellis, Boyer et al. 2023).

Nanofiltration (NF)

Nanofiltration is a pressure-driven membrane technology that offers high water flux at low operating pressures and typically rejects polyvalent ions and larger molecules (including PFAS), while allowing smaller molecules and monovalent ions, such as sodium and chloride, to pass through. Data on PFAS removal with NF comes from laboratory-scale tests, with limited application at full-scale drinking water treatment plants. Membranes tested include NF-270, NF-200, NF-90, DK and DL membranes (Steinle-Darling and Reinhard 2008, Appleman, Dickenson et al. 2013) (McCleaf, Stefansson et al. 2023). These membranes generally show over 95% rejection of PFAS. Full-scale removal of PFAS to low ng/L levels has been confirmed (McCleaf, Stefansson et al. 2023).

Recent research has focused on modifying membrane surfaces to enhance PFAS selectivity (Johnson, Hoffman et al. 2019). Additionally, the disposal or treatment of the membrane concentrate stream is an important consideration, particularly when applying high-pressure membranes in inland communities (coastal communities often emit the reject water into the sea). NF technologies are usually part of a treatment train (e.g., (McCleaf, Stefansson et al. 2023)) given the potential for membrane fouling, and pre-treatment prior to applying NF is recommended.

The use of fluoropolymers in membrane manufacturing raises concerns given that fluoropolymers are a type of PFAS. It is well known that low molecular weight PFAS are emitted during fluoropolymer lifecycles (during manufacturing and potentially during disposal), (Lohmann, Cousins et al. 2020) but it was not expected that PFAS could be released during use of the membranes. It was thought that fluoropolymer membranes contained negligible leachable impurities of low molecular weight PFAS. Recent research has shown, however, that low molecular PFAS can potentially leach out of membranes used in membrane technologies such as RF and RO (Sadia, ter Laak et al. 2024).

Reverse osmosis (RO)

Reverse osmosis is an effective technology for removing multiple types of contaminants, including PFAS, from water by applying pressure to push water through a semipermeable membrane (Safulko, Cath et al. 2023, Sadia, ter Laak et al. 2024). RO works mainly through size exclusion, with contaminants being rejected based on their size. For PFAS removal, RO membranes effectively reject a wide range of compounds, achieving >98.3% removal for a wide range of perfluoroalkyl chain lengths (Safulko, Cath et al. 2023). Similar to NF, RO systems produce a PFAS-concentrated reject water. RO is often part of a treatment train, with pretreatment necessary for optimal performance.

RO removes not only contaminants but also beneficial minerals like calcium and magnesium. This can result in less palatable and less healthy drinking water for long-term consumption. Remineralization may be required to restore taste and essential minerals (Vingerhoeds, Nijenhuis-de Vries et al. 2016).

Recent advances have improved efficiency of RO, notably by reducing pressure requirements and lowering costs. RO is widely used for drinking water purification and is also currently used for desalination on Jersey. Compared to GAC and ion exchange, RO has higher capital and operating costs (Tow, Ersan et al. 2021).

Challenges for implementation of RO include energy consumption for the high-pressure pumps, the disposal of PFAS-enriched reject water, the need for remineralization and the high water loss (15-30%). This high water loss is a particular drawback on Jersey where there is water scarcity in times of drought. Effective treatment of the reject water from RO is also needed and is an area of ongoing research.

NF usually has slightly higher water recovery than RO under similar conditions. Moreover, NF membranes operate at lower pressure than RO, reducing energy needs and potentially increasing sustainability. Finally, NF does not require remineralization of the drinking water. NF is therefore the most promising among the membrane techniques for drinking water treatment on Jersey.

Foam fractionation

Foam fractionation is a physical separation method which uses rising gas bubbles in a water column to remove amphiphilic substances like PFAS, which adsorb to the bubble surface and concentrate in the foam formed on the surface (Buckley, Xu et al. 2022). This foamate, which is concentrated with PFAS, is then collected for further treatment or disposal. Removal efficiency varies with PFAS type, water chemistry, and other operational parameters (Buckley, Xu et al. 2022).

Foam fractionation has progressed from bench to full-scale application for PFAS removal. It has achieved removal of PFAS down to low ng/L levels of PFOS and PFOA. Full-scale systems are in operation in Europe as well as in other regions. Removal of shorter chain PFAS remains a challenge, with ongoing research to improve performance. Cationic surfactant additives can enhance short-chain PFAS removal, but effectiveness varies depending on water properties (Buckley, Karanam et al. 2023). Foam fractionation performs consistently across diverse water types and concentrations (ng/L to mg/L) and is less affected by background contaminants than the abovementioned treatment technologies and thus has been used as a pre-treatment option (McCleaf, Stefansson et al. 2023). Similar to the other treatment techniques, site-specific testing is recommended to optimize performance. Key factors influencing performance include: water characteristics affecting natural foaming, hydraulic retention time, bubble size, bubble formation, foamate removal method, column height, and which chemical additives to enhance foaming or removal of short-chain PFAS.(Buckley, Xu et al. 2022)

Research has shown elevated airborne PFAS concentrations near foam fractionators, with aerosolized PFAS matching foamate composition. Emission controls and enclosures are recommended to minimize exposure risks of workers at water treatment facilities (Smith, Lewis et al. 2023).

Collected foamate can be further treated using adsorptive media or destruction technologies (e.g., electrochemical oxidation) (Smith, Lauria et al. 2023). Energy use is moderate in comparison to, for example, RO (Molzahn, Collins et al. 2024). The volume of foamate is lower compared to RO concentrate. Disposal or treatment of the foamate produced from foam fractionation would be a key consideration for Jersey.

Conclusions

For Jersey, where water scarcity during periods of drought and options for waste disposal/treatment are constraints, selecting the right PFAS treatment technology requires balancing effectiveness, sustainability, and operational feasibility. GAC and ion exchange resins are both proven for removing long-chain PFAS, including PFOS, PFHxS and PFOA. GAC systems require more space and face disposal challenges. Ion exchange resins are more compact and effective for short-chain PFAS, though performance can be affected by co-contaminants and natural organic matter. Ion exchange could potentially be used in a treatment train following GAC. GAC cannot be reactivated on Jersey, which means that spent GAC would have to be disposed of on island (e.g., by incineration or landfill), if it is not possible to ship the waste GAC off island for reactivation. Modern regenerable ion exchange resins could potentially be regenerated on Jersey, but this would require investment in specialized regeneration systems. Otherwise, spent ion exchange resins would also need to be disposed of on Jersey by incineration or landfill.

Among membrane technologies, nanofiltration (NF) is the most suitable for Jersey. It offers high PFAS removal, lower energy use, higher water recovery, and no need for remineralization, making it more sustainable than reverse osmosis (RO), which suffers from high water loss (15–30%), energy demand, and mineral removal. There are some questions raised regarding leaching of PFAS from the fluoropolymer membranes.

Foam fractionation is a useful pre-treatment step to reduce PFAS load, but requires careful management of concentrated waste and emissions, and there are considerations regarding worker safety.

A hybrid approach, such as GAC followed by NF or ion exchange, may prove to be suitable. Final decisions cannot be made through a review of the literature alone and must be guided primarily by lab-scale testing followed by pilot testing using site-specific water. Jersey Water have already commenced the necessary testing pf multiple state-of-the art treatment combinations.

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International limits for drinking water

There are a number of guideline values for drinking water around the world. The picture is a little complex: Some are guidance values, some are enforceable standards which must be met. Some are for individual PFAS such as PFOA or PFOS, some for the sum of various numbers of PFAS with the concentrations added up. Some are based on toxicity estimated from animal tests, some from epidemiology and others based mainly on what is judged practicable to achieve. They vary between country and in larger countries such as the US, between states. Generally, however, they have been falling over time and we will summarise here key current or planned PFAS drinking water limits which have been recently established.

Limits reflecting the EFSA risk assessment

Several countries have drawn on the influential intake limits recommended by EFSA, the European Food Safety Agency (EFSA et al., 2020). They established a "tolerable weekly intake" (TWI) from all exposure sources of 4.4 ng/kg/week for people for the sum of 4 PFAS (PFOA, PFOS, PFHxS, PFNA, called PFAS-4 in this section). By dividing by 7 this can be expressed as the equivalent tolerable daily intake (TDI) of 0.63 ng/kg/day. The effect of interest that this limit seeks to avoid is reduction in vaccine effectiveness in young children, exposed via their mothers. But prevention of other effects of concern such as reduced birth weight or increased cholesterol also pointed to similar limits of exposure. This target value has been accepted by many authorities as a base for setting exposure limits.

Converting this TDI into a limit for drinking water requires two further assumptions: how much water is drunk per person by the population (in terms of litres per kg per day) and how important is water as a source of PFAS compares to other sources (mainly in food). In the case of water intake, sometimes the average water intake is used, whereas in other cases an upper percentile is used to be more protective. Consideration of other exposure sources relative to water intake from other sources, tips the total intake over the limit. To address this, the Relative Source Contribution (RSC) is applied. The RSC represents the proportion of the TDI that can be attributed to drinking water, with the goal that the guideline concentration remains within safe limits when all sources are considered.

water limit (ng per L) = $\frac{\text{target TDI } (ng \text{ per } kg \text{ per } day) \times \text{RSC } (fraction)}{\text{daily water intake } (L \text{ per } kg \text{ per } day)}$

The choice of the RSC is guided by surveys in the general population, away from local sources contaminating drinking water, which demonstrate that non-drinking water exposure sources account for most of the PFAS intake. For example a number of studies in the general population provided estimates varying from 1% to 35% of PFOS intake coming from drinking water (IARC, 2025). A 2015 Swedish study in the general population estimated that 11% of PFOS intake on average came via the drinking water (Gebbink et al., 2015).

In the past a wide range of RSC percentages have been applied in setting limits but since 2022 the WHO Guidance recommends assuming 20% for the RSC, and where water accounts for nearly all the intake, the RSC can be up to a maximum of 80%. "In the absence of adequate exposure data or where documented evidence is available regarding widespread presence in

one or more of the other media (i.e. air, food, soil or consumer products), the normal allocation of the total daily intake to drinking-water is 20% (floor value), which reflects a reasonable level of exposure based on broad experience, while still being protective. This value reflects a change from the previous allocation of 10%, which was found to be excessively conservative." (WHO 2022). The default value of 20% is also used by the USEPA (Greene et al 2025).

The proportion of PFAS coming from water varies and can be very high where the water is highly contaminated. So for example in Ronneby Sweden, those who had been drinking the AFFF contaminated drinking water averaged 262 ng/ml serum PFOS compared to a control area with clean water and average PFOS of 3 ng/ml (Xu et al). These samples were taken a short time after the contaminated water had been replaced, but it is clearly evident that before, when they were drinking the contaminated water, the water source accounted for about 99% of their PFOS intake.

Sweden has applied this RSC value of 20% and derived a limit from the EFSA TDI of 4 ng/L for PFAS-4. This is a guidance value and will be a binding limit from 2026. (Livsmedelsverket (2022) KEMI (2024)). Also they have modified slightly the 100 ng/L limit for 20 EFSA by adding one more PFAS (6:2 fluorotelomer sulfonic acid) to the 20 specified in the EU directive and listed below. They have a standard of 100 ng/L for the sum of these 21.

Denmark also derived a limit based on the EFSA TDI, but noting that the TDI was in general exceeded in Denmark from food alone, decided (before the 2022 guideline on 20%) to set a stricter water guideline based on a 10% RSC, leading to a limit of 2 ng/L for PFAS-4 Vinggaard, A.M. and Olesen, P.T. (2021).

Germany, following a widespread survey of drinking water which revealed that many water supplies exceeded 2 or even 4 ng/L for PFAS-4, decided to set a more achievable limit of 20 ng/L with a delay till 2028 to allow implementation. Borchers, U. et al (2022).

The Netherlands has also established some guidance values for drinking water limit on the EFSA TDI, but with a key difference. EFSA assumed that the 4 PFAS had equal potency, but the Netherlands have introduced a weighting by degree of potency. They have set a threshold for PFAS-4 of 4.4 ng/L expressed as PFOA equivalents (<u>https://www.rivm.nl/pfas/drinkwater</u>). The PFOA equivalents are derived by comparing the toxicity of each type of PFAS to that of PFOA. For example, PFOS is assigned a Relative Potency Factor of 2, meaning it is twice as potent as PFOA. For PFHxS the relative potency is 0.6 and for PFNA it is 10. Thus the target concentration of the sum of PFAS depends on the balance of mixture, and would be lower for mainly PFOS and less strict for mainly PFHxS.

UK standards

Closest to home, the UK standards have been based on a tiering system with increasing obligations from concentrations of 10 ng/L to above 100 ng/L. These guidance values were reviewed last year and a report on that review also includes a helpful summary of drinking water limits in some other countries (DWI 2024).

Similar requirements used to apply to individual PFAS, PFOA and PFOS, but now they are applied to the sum of 48 specified PFAS. The table below summarises the detailed guidance

required at each level of PFAS. These are not binding regulations, but the guidance seeks to ensure that no water supplied is above 100 ng/L (for the sum of PFAS), and progressive efforts are encouraged to reduce contamination down to below 10 ng/L. While these limits are for the sum, concentrations of individual PFAS should also be considered in the risk assessment. Water companies are also asked to report any other PFAS (outside of the 48) that are above tier guidance levels. Also it worth noting as many results are likely to be below detection, that these values (reported as not detected or 'less than' the reporting limit) are not counted in the sum of PFAS.

Table 1

Tiered guidance values for PFAS in UK

| Item | Guidance values (sum of PFAS or individual*) | Minimum action to be taken by water companies |
|--------|--|--|
| Tier 1 | 10 ng/L | Conduct a regulation 27 risk assessment. Monitoring frequency for PFAS should initially be quarterly to establish a baseline for risk assessment, accounting for temporal variation, then may be reduced to allow periodic validation of the risk assessment (minimum annual). Consider further actions required where sites are likely to breach tier 2 |
| Tier 2 | <100 ng/L | Increase PFAS monitoring frequency, generally to between monthly and quarterly, sufficient to enable predictive modelling. Higher frequency monitoring may be appropriate where concentrations could breach tier 3. Ensure regulation 27 risk assessments are up to date and under continuous review. Review existing control measures, including the effectiveness, validation, and monitoring of that measure. Identify additional control measures required to reduce PFAS concentrations. Water company will discuss the likelihood of a tier 3 breach with the DWI to determine whether the occurrence should be a reportable event. <u>Consult/discuss with UKHSA and local health authorities</u>. Prepare emergency contingency measures to prevent the supply of water to consumers with ≥0.1 µg/L PFASshould existing control measures become inadequate. Design a proactive and systematic risk reduction strategy which shall include a prioritised mitigation methodology to progressively reduce PFASconcentrations in drinking water. Appropriate mitigation may include catchment management, direct or indirect treatment, or process-controlled blending. |
| Tier 3 | ≥ 100 ng/L | Notify as an event any results ≥0.1 µg/L in water supplied to consumers, any raw water results that are likely to cause results ≥0.1 µg/L in water supplied to consumers, or any failure of PFAS treatment where raw water sources are in tier 3 (under the provisions of the current Water Industry (Suppliers' Information) Direction [the Information Direction]). |

| | Notify UKHSA and local health authorities and determine what action (beyond monitoring) is appropriate to reduce exposure via drinking water supplies. Factors such as population demographics or consumer groups at particular risk should be considered. Action may, for example, include provision of alternative supplies to the affected area. Sample investigation to include, as a minimum: resample point of contravention, individual raw water sources, blended or combined raw water points and final water from water treatment works. Consider sampling at treated water blending point (if applicable) and/or water quality |
|--|---|
| | Cones. Samples should be fast-tracked. Ongoing enhanced monitoring should be established at locations and frequencies to understand the impact for the specific supply situation. A minimum of one year of monthly samples at raw and final water points, timed to take account of any changes in hydrological conditions, such as precipitation, surface or groundwater flows and pumping regimes. Review existing control measures, including the effectiveness, validation, and monitoring of that measure. Implement emergency contingency measures to prevent the supply of water with $\ge 0.1 \ \mu g/L \ PFAS$ to consumers. Review the catchment risk assessment, including PFASsource information, within 3 working days of receiving the result. Prioritise site within the company's PFAS risk reduction strategy for medium/long term mitigation. This list of actions is not exhaustive; all necessary actions to investigate the source of the PFAS and reduce |
| | actions to investigate the source of the PFAS and reduce concentrations to below 0.1 μ g/L in water supplied to consumers must be taken in the short term, with a longer- term strategy designed and implemented to progressively reduce PFAS concentrations in drinking water. |

*Sum of PFAS refers to the sum of the concentrations of 48 compounds which water companies are required to test for.

Scotland has adopted the EU limit for the sum of 20 PFAS of 100 ng/L, described below.

European Union standards

In the EU there is an overall limit to be complied with by January 2026 for the sum of 20 PFAS (listed in the table below), of 100 ng/L, and a further overall standard for total PFAS of 500 ng/L. Countries can implement more strict standards within that overall target (as Denmark, Sweden and Germany have done). <u>https://eur-lex.europa.eu/eli/dir/2020/2184/oj</u>

North American standards

In the US, the EPA established in 2024 health-based, non-enforceable Maximum Contaminant Level Goals (MCLGs) for six PFAS, being zero for PFOA and PFOS based on the conclusion that they are "likely to be carcinogenic to humans" and 10 ng/L for PFHxS, 10 ng/L for PFNA, and 10 ng/L for GenX. They also established enforceable Maximum Contaminant Levels in drinking water, based more on practicability of measurements and control, of 4 ng/L for PFOA, 4 ng/L for PFOS, while the enforceable Maximum Contaminant Levels for PFHxS, PFNA, and GenX were

set at the health-based goals of 10 ng/L (EPA 2024). Additionally, an MCL based on a Hazard Index which considers the relative toxicity of the individual PFAS was set for mixtures of PFHxS, PFNA, GenX, and PFBS. More recently the EPA confirmed that the MCLs for PFOA and PFOS at 4 ng/L will be retained, but that they intend to rescind and reconsider the MCLs for other four PFAS including PFHxS. The date by which these MCLs have to be met has now been extended from 2029 to 2031. (EPA 2025)

Summary of international limits

A convenient summary of a number of national standards provided in the DWI report (DWI 2024), is reproduced here:

Table 2

International standards for PFAS. *Includes PFOS, PFOA, PFNA, PFHxS, PFBS, PFBA and PFHpA (DWI 2014) [need to convert to ng/L for consistency]

| Country / Region | Value (µg/L) | Comments | | | | | |
|-------------------|--------------------------|---|--|--|--|--|--|
| England and Wales | Guideline value of 0.1 µ | Guideline value of 0.1 µg/L for sum of 48 named PFAS* (DWI, | | | | | |
| - | 2024) | | | | | | |
| Scotland | 0.1 | Sum of 20 named PFAS* | | | | | |
| | | (Scottish Government, 2023) | | | | | |
| EU | 0.1 | Sum of 20 named PFAS* (EU, | | | | | |
| | | 2020) | | | | | |
| | 0.5 | Total PFAS | | | | | |
| Germany | 0.02 | Sum of PFOS, PFOA, PFNA and | | | | | |
| | | PFHxS. In addition to the EU | | | | | |
| | | requirement | | | | | |
| Denmark | 0.002 | Sum of PFOS, PFOA, PFNA and | | | | | |
| | | PFHxS. In addition to the EU | | | | | |
| | | requirement | | | | | |
| Sweden | 0.0044 | Sum of PFOS, PFOA, PFNA and | | | | | |
| | | PFHxS. In addition to the EU | | | | | |
| | | requirement | | | | | |
| Netherlands | 0.004 | For individual PFAS - PFOS, | | | | | |
| | | PFOA, PFNA and PFHxS. In | | | | | |
| | | addition to the EU requirement | | | | | |
| USA | 0.004 | PFOS | | | | | |
| | 0.004 | PFOA | | | | | |
| | 0.01 | PFNA | | | | | |
| | 0.01 | PFHxS | | | | | |
| | 0.01 | GenX | | | | | |
| | 0.01 | PFBS | | | | | |
| Australia | 0.07 | Sum of PFOS and PFHxS | | | | | |
| | 0.56 | PFOA | | | | | |
| | | A public consultation is | | | | | |
| | | underway to consider the | | | | | |
| | | following (NHMRC, 2024): | | | | | |
| | | ο PFOA – _0.2 μg/L | | | | | |
| | | o PFOS – _0.004 μg/L | | | | | |

| o PFHxS – _(| 0.03 µg/L |
|--------------|-----------|
|--------------|-----------|

• ο PFBS – _1 μg/L

• o GenX chemicals -

None

Sum of 25 named PFAS*. Objective to achieve as low as reasonably achievable and based on analytical and treatment achievability. Health Canada continues to develop or update guidelines and guidance for PFAS over the next few years (Government of Canada, 2024)

The most comprehensive list of limits is provided by the ITRC website (<u>https://pfas-</u><u>1.itrcweb.org/fact-sheets/</u>) follow link to the "PFAS Environmental Media Values Table Excel file". As well as national standards, many state-level regulations in the USA are summarised there, and it is frequently updated.

PFAS groupings

The table below lists the specific PFAS that are included in different regulations. The DWI requires water companies to sample for 48 PFAS and guidance values are for the sum of these 48 (Col 4). A number of countries such as Denmark focus on the sum of the 4 PFAS often present in human blood samples and used for the risk assessment by EFSA (Col 5). In the sampling carried out in Jersey of blood in 2022, 3 of these were detected in all samples (Col 6), and up to 8 PFAS were found in at least one of the samples, often at low levels. The sum of PFAS used for eligibility for potential interventions in report 3, is calculated for these same 8 PFAS (Col 7). The US National Academies of Science and Medicine (NASEM) recommended using a slightly different sum to identify levels of human exposure that are of potential concern (Col 8). The EU rules are based on the sum of 20 PFAS and this has also been used for regulations on drinking water in Scotland (Col 9). Abbreviations are provided for all of these 20 PFAS. In the most recent Arcadis Jersey water report, 8 PFAS were above detection limits in drinking water samples (Col 10).

The three PFAS of most concern for Jersey up to now for human health (PFOA, PFOS, PFHxS) are present in all of these lists of PFAS used for calculating sum of PFAS indices. In the sampling of PFAS in drinking water in Jersey 8 compounds were detected above detection limits, this list of 8 falls within the DWI 48 and the EU 20, so for these Jersey samples, the total under either of those ways of summing would be the same.

<u>Table 3</u>

Groupings of PFAS

| 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|----|
| | | | | | | | | | |

Canada

0.03

| Parameter name | | CAS code | DWI | PFAS- 4 (EFSA) | Jersey all blood samples | Jersey some blood samples | NASEM LIST | EU/ Scotland PFAS-20 | Jersey water detections |
|--|---------|-----------------|-----|----------------------|--------------------------------|---------------------------------|---------------|----------------------------|-------------------------------|
| Perfluorooctane Acid | PFOA | 335-67-1 | у | у | Y | у | У | у | у |
| Perfluorooctane Sulfonate | PFOS | 1763-23-1 | у | у | Y | у | у | у | у |
| Perfluorohexane sulfonic acid | PFHxS | 355-46-4 | У | У | Y | У | У | У | у |
| Perfluorononanoic acid | PFNA | 375-95-1 | у | у | | У | У | у | |
| Perfluorobutane sulfonic acid | PFBS | 375-73-5 | у | | | | | У | У |
| Perfluorobutanoic acid | PFBA | 375-22-4 | у | | | | | У | у |
| Perfluoroheptanoic acid | PFHpA | 375-85-9 | у | | | | | У | у |
| Perfluorohexanoic acid | PFHxA | 307-24-4 | у | | | | | У | у |
| Perfluoropentanoic acid | PFPeA | 2706-90-3 | у | | | | | у | у |
| 2-(N-Methyl perfluorooctane | MeFOSAA | 2355-31-9 | у | | | У | У | | |
| Perfluoropentane sulfonic acid | PFPeS | 2706-91-4 | у | | | у | | У | |
| Perfluorodecanoic acid | PFDA | 335-76-2 | У | | | У | У | У | |
| Perfluoroheptane sulfonic acid | PFHpS | 375-92-8 | У | | | У | | У | |
| Perfluoroundecanoic acid | PFUnDA | 2058-94-8 | у | | | | У | У | |
| Perfluorodecane sulfonic acid | PFDS | 335-77-3 | У | | | | | У | |
| Perfluorododecane sulfonic acid | PFDS | 79780-39-5 | У | | | | | У | |
| Perfluorododecanoic acid | PFDA | 307-55-1 | у | | | | | У | |
| Perfluorononane sulfonic acid | PFNS | 68259-12-1 | У | | | | | У | |
| Perfluorotridecane sulfonic acid | PFTrDS | 791563-89- 8 | | | | | | У | |
| Perfluorotridecanoic acid | PFTrDA | 72629-94-8 | У | | | | | У | |
| Perfluoroundecane sulfonic acid | PFUnDS | 749786-16-1 | У | | | | | У | |
| Perfluoro-3- | | 377-73-1 | У | | | | | | |
| Perfluoro-4-ethyl cyclohexane sulfonic acid | | 646-83-3 | У | | | | | | |
| Perfluoro-4- methoxybutanoic acid | | 863090-89- 5 | У | | | | | | |
| Perfluoro(2-ethoxyethane) sulphonic acid | | 113507-82- 7 | У | | | | | | |
| Perfluorobutyl sulfonamide (perfluorobutane | | 30334-69-1 | у | | | | | | |
| Perfluorohexadeconoic | | 67905-19-5 | у | | | | | | |
| Perfluorohexane sulfonamide | | 41997-13-1 | У | | | | | | |
| Perfluorooctadecanoic acid | | 16517-11-6 | у | | | | | | |
| Perfluorooctane sulfonamide | | 754-91-6 | У | | | | | | |
| Perfluorotetradeconoic acid | | 376-06-7 | У | | | | | | |
| 11-Chloroeicosafluoro-3- oxaundecane-1-sulfonic acid | | 763051-92- 9 | У | | | | | | |
| 2-(N-Ethyl perfluorooctane sulfonamido) acetic acid | | 2991-50-6 | у | | | | | | |
| 2H,2H,3H,3H- Perfluorodecanoic acid | | 812-70-4 | у | | | | | | |

| 2H,2H,3H,3H- Perfluorooctanoic acid | 914637-49- 3 | у |
|---|-----------------|---|
| 3-Perfluoropropyl propanoic acid | 356-02-5 | у |
| 4,8-Dioxa-3H- perfluorononanoic acid | 919005-14- 4 | у |
| 4:2 Fluorotelomer sulfonic acid | 757124-72- 4 | у |
| 6:2 Fluorotelomer sulfonamido propyl betaine | 34455-29-3 | У |
| 6:2 Fluorotelomer sulfonic acid | 27619-97-2 | У |
| 8:2 Fluorotelomer sulfonic acid | 39108-34-4 | У |
| 9-Chlorohexadecafluoro-3- oxanonane-1-sulfonic acid | 756426-58- 1 | У |
| Hexafluoropropylene oxide dimer acid (Gen X) | 13252-13-6 | у |
| Hexafluoropropylene oxide trimer acid | 13252-14-7 | у |
| N-Ethyl perfluorooctane sulfonamide | 4151-50-2 | у |
| N-Ethyl-N-(2-hydroxyethyl) perfluorooctane sulfonamide | 1691-99-2 | У |
| N-methyl perfluorooctane sulfonamide | 31506-32-8 | у |
| N-Methyl-N-(2- hydroxyethyl) perfluorooctane sulfonamide | 24448-09-7 | У |
| Nonafluoro-3,6- dioxaheptanoic acid | 151772-58- 6 | У |

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Modelling blood concentrations from environmental exposure

Tony Fletcher 22 June 2025

This section discusses the potential contribution of drinking water intake to the average blood serum levels in addition to likely background levels from general exposure, mainly from food consumption.

We are all exposed to PFAS and some PFAS are so persistent and ubiquitous that it is nearly impossible to completely avoid them and there is some level of PFAS in almost everyone's blood. Thus population samples of people with no obvious direct exposure to PFAS always show low but clearly measurable levels of various PFAS, the top two nearly always being PFOS and PFOA.

As discussed in section X.X, most of the intake of PFOS and PFOA in general population surveys is from non-water sources, principally food. For PFHxS and shorter chain PFAS, drinking water intake becomes relatively more important than food. In populations without a significant local contamination source, 10% to 20% of exposure of PFOS and PFOA is estimated to be from drinking water, but with heavily contaminated water the proportion from water can be high, rising well above 90%.

The likely concentrations of serum PFAS can be estimated if we know how much people are ingesting, from food and from drinking water. The average predicted concentrations will depend on how long people are exposed and the concentrations are hard to predict precisely as intakes from food and drinking water vary over time. However, we can get a reasonable approximate prediction based on average values.

For estimating the non-water intake we can take the average serum levels across multiple recent general population studies, estimate the corresponding intake, and subtract the 10 or 20% attributable to drinking water.

For water intake the relationship between intake and serum levels is well understood in pharmacokinetic models and so it is straightforward to predict average intake and consequent impact on serum levels related to water intake in particular, given some assumptions.

These predictions are presented below for 3 scenarios, assuming constant exposure and after reaching steady state: likely background serum concentrations assuming a stricter limit is met and we have used the example of the Swedish limit of 4 ng/L for PFAS-4; likely average serum levels from recent measured water concentrations in Jersey water; possible average serum levels based on a higher exposure scenario using the highest water concentrations measured.

Background levels derived from general exposures

General population exposure is primarily through food which contains PFAS accumulated through agricultural food chains (e.g., soil-grass-cow-human, soil-potatohuman, etc.) or has absorbed it from food packaging, and from drinking water. Food as a source has been shown to account for 85% or more of general population intake for PFOS and PFOA (Vestergren & Cousins, 2009; Vestergren et al., 2008). Therefore we can assume that typical background serum levels are mainly an indicator of average PFOS and PFOA intake mainly via food. Such serum data have been available in the US for some years (Botelho et al., 2025), but not for the UK. In the absence of local data we can draw on the multiple studies of European general population serum levels assembled by the HBM4EU project (Govarts et al., 2023) (HBM4EU). Both US and European data show that PFOS and PFOA serum levels have fallen since 2000, reflecting efforts to phase out their use. For PFAS, the HBM4EU database assembles a range of studies across Europe, and taking the more recent ones (data collected during 2017-2021), there are data from 9 studies summarised. For PFOS the median levels reported varied from 1.0 to 2.7 ng/ml, an average of 1.8 ng/ml. For PFOA the average was 1.1 (range 0.7 to 1.9). For PFHxS the average was 0.4 (range 0.2 to 0.6). For the others among the 8 PFAS of primary concern in Jersey (see ISAP Report 3), average background levels were 0.2 for PFNA, 0.1 for PFDA, and zero or below detection for PFPeS, PFHpS and MeFOSAA.

We will assume that these background intake and serum levels also apply to the Jersey population, and that the default 20% RSC (Relative Source Contribution, being the percentage of daily intake coming from drinking water) applies to the contributions from PFPS and PFOA. The sum of these values is 3.6 ng/ml (1.8+1.1+0.4+0.2+0.1) and we would therefore estimate the non-drinking water element is 20% less for PFOS and PFOA, that is 3.0 ng/ml (1.4+0.9+0.4+0.2+0.1) for PFOS, PFOA, PFHxS, PFNA, PFDA respectively.

Water concentrations in the Jersey water supply

In Jersey the average PFAS in drinking water has been summarised in the annual reports on drinking water. These data have been assembled with some older measurements and provided to the panel and the averages per year for each of the two water treatment works are as follows:

| Year | PFOS (µ | g/l) | PFOA (µg/l) | | PFHxS (µg/l) | |
|------|---------|--------|-------------|--------|--------------|--------|
| | Handois | Augres | Handois | Augres | Handois | Augres |
| 2024 | 0.004 | 0.003 | 0.007 | 0.005 | 0.004 | 0.003 |
| 2023 | 0.006 | 0.005 | 0.007 | 0.005 | 0.004 | 0.003 |
| 2022 | 0.010 | 0.010 | 0.006 | 0.007 | 0.006 | 0.004 |
| 2021 | 0.012 | 0.008 | 0.005 | 0.004 | 0.007 | 0.004 |
| 2020 | 0.011 | 0.008 | 0.007 | 0.004 | 0.006 | 0.004 |
| 2019 | 0.022 | 0.015 | 0.005 | 0.004 | 0.008 | 0.006 |

| 2018 | No data | | | | | |
|------|----------|-------|-------|-------|----------|-------|
| 2017 | <0.01 | <0.01 | <0.01 | <0.03 | <0.01 | <0.01 |
| 2016 | <0.01 | <0.01 | <0.03 | <0.03 | <0.01 | <0.01 |
| 2015 | <0.01 | <0.01 | <0.03 | <0.03 | 0.021 | <0.01 |
| 2014 | <0.01 | <0.01 | <0.03 | <0.03 | 0.014 | <0.01 |
| | | | | | | |
| 2002 | not | | | | not | |
| 2002 | detected | | | | detected | |
| 1000 | not | | | | | |
| 1999 | detected | | | | | |

Values marked as <0.01 are below the reporting level, either not detected or too low to be accurately reported. Both water treatment centres provide water so a reasonable estimate of the average concentration in water supplied to the population is the average of the two annual figures. We can assume that recent and current exposure is the average of the two last measurement years (2023-4). Predictions based on recent measurements would predict the serum levels in the future if current intake levels persisted. However, we know that past water concentrations were higher, so the current serum levels would reflect those higher water related exposures.

For the past high exposure scenario we take the highest annual average in either treatment works for each PFAS. Some of the reported measurements are lower than that so the expected average serum levels should lie between the predictions based on the high levels and the recent levels.

For the scenario of being below the 4 ng/L for the sum of these PFAS, we model based on 1.3 ng/L for each of the 3. That would be the expected serum levels after the water concentrations had fallen to these values and some years had passed so that steady states are reached.

| | Low Total | Recent water | High past |
|-------|-------------|---------------|---------------|
| | less than 4 | concentration | water |
| | ng/L | ng/L | concentration |
| | | | scenario ng/L |
| PFOS | 1.3 | 4.4 | 22 |
| PFHxS | 1.3 | 3.5 | 21 |
| PFOA | 1.3 | 5.9 | 8 |

Water concentration scenarios:

Estimating water concentration impact on serum levels

With some further assumptions we can estimate the impact of such concentrations on serum concentrations. If someone is exposed to a constant level of contamination from drinking water then serum levels start to build up and as these concentrations increase so does the rate of excretion, until they are same: at steady state the concentration is stable and reflects the daily intake, the excretion reflecting the serum concentration and the intake reflecting the drinking water exposure. These can be estimated from the following equations:

the <u>daily excretion</u> in $\mu g/day/kg = k \times Cs \times Vd$, where k is a excretion rate constant which is related to half life (k=log_e(2)/half-life per year or =log_e(2)/half-life/365 per day); Cs is the serum concentration in $\mu g/L$ and Vd is the volume of distribution a measure of how much volume in our body the PFAS is spread through, expressed in terms of litres of plasma per kg body weight.

the <u>daily intake</u> in μ g/day/kg = (Cw x Vw)/BWt = where Cw is the water concentration in μ g/L, Vw is the volume of water drunk and BWt is the average body weight.

At steady state the intake equals excretion. Rearranging the above equations we can estimate (at steady state) the conversion between drinking water concentration and the impact of that serum concentration on serum concentration:

$$Cs = \frac{Cw \times Vw}{Wt \times k \times Vd}$$

For PFOS we have used an average half life of 3 years, from which k is 0.231 per year and k per day for this equation is 0.000633 (Li et al., 2022).

The volume of distribution for PFOS has a number of published estimates ranging from about 100 to about 400 (Chiu et al., 2022; Fletcher et al., 2023; Thompson et al., 2010) and a mid value of 200 ml/kg is used here;

The volume of water drunk varies between individuals but a reasonable average value for adults is 1.5 L/day (Guelinckx et al., 2015).

Body weight also varies and we have used an average value of 70kg.

So based on the assumed average values for volume of water drunk per day, body weight, half life and volume of distribution (3 years for PFOS, 4.5 years for PFHxS, 2.5 years for PFOA), we can estimate how much serum concentrations you would expect from drinking water contamination and adding that contribution to the average background levels.

The following table summarises the results for blood concentrations in terms of μ g/L, which is equivalent to ng/ml, for the scenario of recent average concentrations

| PFAS | Serum | Water | Serum | Sum in serum |
|--------------|---------------|---------------|--------------|--------------|
| | Contribution | concentration | Contribution | μg/L |
| | from non- | μg/L | from water | |
| | water sources | | μg/L | |
| | μg/L | | | |
| PFOS | 1.4 | 0.0044 | 0.7 | 2.1 |
| PFHxS | 0.4 | 0.0035 | 0.9 | 1.3 |
| PFOA | 0.9 | 0.0059 | 0.8 | 1.7 |
| Sum of 3 | 2.7 | | 2.4 | 5.1 |
| Other 5 PFAS | 0.3 | Assume zero | | 0.3 |
| Sum of 8 | 3.0 | | | 5.4 |

The water table noted that some earlier measurements of PFHxS were higher so the following table shows the predicted serum levels in a worst case scenario:

| PFAS | Serum | Water | Serum | Sum in serum |
|--------------|---------------|---------------|--------------|--------------|
| | Contribution | concentration | Contribution | μg/L |
| | from non- | μg/L | from water | |
| | water sources | | μg/L | |
| | μg/L | | | |
| PFOS | 1.4 | 0.022 | 5.0 | 6.4 |
| PFHxS | 0.4 | 0.021 | 7.1 | 7.3 |
| PFOA | 0.9 | 0.008 | 1.5 | 2.4 |
| Sum of 3 | 2.7 | | 13.6 | 16.3 |
| Other 5 PFAS | 0.3 | Assume zero | | 0.3 |
| Sum of 8 | 3.0 | | | 16.6 |

The future scenario assuming that drinking water concentrations of each of the PFAS fell to 1.3 ng/L but the non drinking water element remained the same would lead to the following predictions:

| PFAS | Serum | Water | Serum | Sum in serum |
|------|---------------|---------------|--------------|--------------|
| | Contribution | concentration | Contribution | μg/L |
| | from non- | μg/L | from water | |
| | water sources | | μg/L | |

| | μg/L | | | |
|--------------|------|-------------|-----|-----|
| PFOS | 1.4 | 0.0013 | 0.2 | 1.6 |
| PFHxS | 0.4 | 0.0013 | 0.3 | 0.7 |
| PFOA | 0.9 | 0.0013 | 0.2 | 1.1 |
| Sum of 3 | 2.7 | | 0.7 | 3.4 |
| Other 5 PFAS | 0.3 | Assume zero | | 0.3 |
| Sum of 8 | 3.0 | | | 3.7 |

It should be stressed that these predictions depend on several assumptions, thus if the body weight or daily water intake differs so would the predictions, and these are average predictions. Individuals can have much shorter or longer half lives than the average, so these predictions are for the population average, not for individuals. Also these average half life are for people without any interventions. If people adopt interventions such as taking bile acid sequestrant medicine as discussed in our third report, their serum levels should be much lower.

All the above estimates are based on models using assumed constant intake, steady state, average weight, average half life, average background levels, average water consumptions etc. Obviously these are simplifying assumptions and these parameters vary between individuals and water concentrations vary over time. To illustrate this uncertainty, these results are compared to those from an online tool available in the US which predicts serum concentrations from drinking water levels.

https://www.atsdr.cdc.gov/pfas/blood-testing/estimation-tool.html. The methods are explained in detail in a recent publication (Lynch et al., 2023). Below is a screen shot of the prediction for PFOA, from a constant water concentration of 5.9 ng/L as our recent water concentration model, for a 60 year old adult. Their model predicts a serum level of 1.7 ng/ml in the blood, based on the US national average of 1.4, plus the additional PFOA in serum due to water intake of 0.3 ng/ml. Our model predicted 1.7 ng/ml but from 0.8 on top of a background average of 0.9 (as US PFAS levels have been generally higher than European levels). So the predicted increases due to water in the two models are a somewhat different. They take slightly different starting values for half life, daily intake etc, and don't take into account the water contribution in the US national average values, but the results are broadly similar. In addition to estimating the average expected value, they also model the uncertainty due to variability in all these starting values (for intake, half life and so on) and the bar in the chart illustrates that uncertainty: although the average expected serum is 1.7 for PFOA the possible values expected in an individual's blood test, range from 0.7 to 3.5 ng/ml.



*Based on survey data from 2017 and 2018. Levels of PFAS in the U.S. population are from CDC's <u>National</u> <u>Health and Nutrition Examination Survey (NHANES</u>). NHANES is a program of studies designed to assess the health and nutritional status of adults and children in the United States.

In conclusion, we would expect based on the above modelling, that the average for sum of 8 PFAS in blood serum of Jersey adults outside the plume area, from the above scenarios would be currently between 5 and 16 ng/ml (µg/L), and that there would be variability for individuals around the average. Looking ahead we would expect the water concentrations to fall from current levels and so general average levels to fall to under 5 ng/ml. It has been recommended to measure a sample of the general population in Jersey to establish current background levels, and an assessment carried out on food sources of PFAS, which should allow improvements of this prediction model. References

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