# Pilot-Scale Water Treatment Efficacy Study of PFAS Removal

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# Nomenclature

| | |
|---|---|
| 6:2 FTAB | 6:2 fluorotelomer sulfonamide alkylbetaine |
| 6:2 FTSA | 6:2 fluorotelomer sulfonate |
| BV | Bed volume |
| DOC | Dissolved organic carbon |
| EBCT | Empty bed contact time |
| FASA | Perfluoroalkyl sulphonamides |
| FHxSA | Perfluorohexanesulfonamide |
| GAC | Granular activated carbon |
| GenX | Hexafluoropropylene oxide dimer acid |
| LC-MS | Liquid chromatography mass spectrometry |
| LMH | Litres per metre squared per hour |
| LPM | Litres per minute |
| LOQ | Limit of quantification |
| MWCO | Molecular weight cut-off |
| NF | Nanofiltration |
| PFAS | Per and poly fluoroalkyl substances |
| PFBA | Perfluorobutanoic acid |
| PFBS | Perfluorobutanesulfonic acid |
| PFCAs | Perfluoroalkyl carboxylic acids |
| PFECHS | Perfluoroethylcyclohexane sulfonate |
| PFHpA | Perfluoroheptanoic acid |
| PFHxA | Perfluorohexanoic acid |
| PFHxS | Perfluorohexanesulfonic acid |
| PFOA | Perfluorooctanoic acid |
| PFOS | Perfluorooctanesulfonic acid |
| PFPeA | Perfluoropentanoic acid |
| PFSAs | Perfluoroalkyl sulphonic acids |
| RO | Reverse osmosis |
| RSSCTs | Rapid small scale column tests |
| SMC | Surface modified clay |
| SPE | Solid phase extraction |
| UV254 | Ultraviolet light absorbance at 254 nanometres |
| WTWs | Water Treatment Works |

# 1. Introduction

This report follows on from work published by the DWI on the bench scale assessment of different water treatment processes for removal of per and poly fluoroalkyl substances (PFAS) from drinking water sources (DWI, 2025).

The bench-scale tests confirmed that granular activated carbon (GAC), surface-modified clay (SMC), ion exchange (IEX) resins, and membrane separation showed significant removal efficiency for PFAS. Nevertheless, these technologies yielded differing performance profiles across PFAS compounds of varying chain lengths and functional groups. Key observations included:

- GAC effectively removed PFAS that contained over 6 carbons and recorded breakthrough volumes between 10,000–13,000 BVs. GAC showed, however, limited effectiveness for short-chain PFAS (4 C) such as PFBA and PFBS.
- SMC showed promising removal potential for the limited testing that was carried out, though trials beyond 9,000 BVs were needed to establish broader effectiveness for a range of PFAS compounds.
- Among the IEX resins tested, Lewatit® TP109 MP provided the most robust removal performance. This resin achieved sustained removal over 83,000 BVs without breakthrough for most perfluoroalkanesulfonic Acids (PFSAs), perfluoroethylcyclohexane sulfonate (PFECHS) and F53B. Nevertheless, similar to GAC, no reliable removal of short-chain PFAS was observed over the long term for IEX.
- For membrane filtration, NF90 nanofiltration (NF) membrane and RO XLE reverse osmosis (RO) membrane consistently recorded removals exceeding 90% for most PFAS and maintaining over 80% removal even for short-chain compounds. The removal efficiency increased with chain length, often reaching >99% for longer PFAS molecules.

The bench-scale work also highlighted the variable influence of water types (groundwater, lowland, and upland surface waters). Effects such as competitive sorption, fouling, and concentration polarization strongly depended on the characteristics of dissolved organic matter and other constituents present in a specific water source.

These findings directly informed the experimental design for further pilot testing, leading to the selection of the following technologies for further evaluation under realistic operating conditions and over extended treatment durations:

- GAC (F400 grade) was selected to assess its capacity for longer-chain PFAS removal and to characterize breakthrough behaviour in larger-scale operation.
- SMC was included to build on the encouraging preliminary results and to evaluate long-term performance.
- The IEX Lewatit® TP109 MP resin was tested based on its high treatment capacity in the bench tests.
- Membrane nanofiltration was incorporated due to its consistently high removal efficiencies for a wide range of PFAS compounds, including short-chain species. While membrane NF90 showed the best results, spiral wound modules of this membrane are not available in the size required for the testing rig used. Therefore, following manufacturer recommendations, the TS80 model was used as it has similar properties to NF90.

To understand the impact of water quality on the treatments, three types of water matrices were tested in parallel: groundwater, lowland water, and upland water. The groundwater selected for testing was already contaminated by a range of PFAS compounds, so no further PFAS compounds were spiked into the water. For the lowland and upland water sources, selected PFAS compounds were spiked into the water because of the low background PFAS concentrations (Section 3). For each water type, a set of columns was operated concurrently to allow direct comparison of adsorbent performance under identical conditions. This experimental design facilitated systematic evaluation of PFAS removal across different technologies and water qualities, providing insights into the relative performance and suitability of each approach.

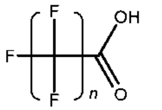
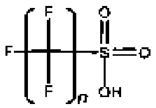
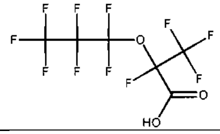
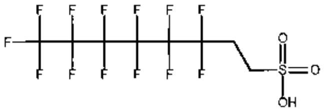
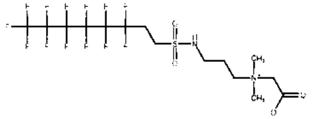
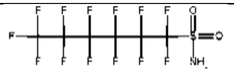
## 1.1. Aim and Objectives

The aim of this work was to determine the effectiveness of pilot-scale treatment processes for PFAS removal from drinking water. Processes were selected based on their performance in bench-scale experiments. Pilot-scale performance was assessed based on the type of PFAS compound removed and considered the influence of the background water matrix on removal by investigating three different water sources (upland and lowland surface waters spiked with PFAS, and a groundwater already contaminated by a range of PFAS compounds).

## 2. PFAS model compounds

The ten PFAS selected for testing in this work reflected prevalent compounds in UK source waters and provided a comprehensive range of chemical functionalities and physicochemical properties to test (Table 1). This selection included variations in PFAS chain length, hydrophobicity, charge, and functional groups, all of which are known to influence treatability and removal mechanisms such as size exclusion, diffusion, electrostatic interactions, and hydrophobic interactions.

**Table 1.** Selected PFAS model compounds and physicochemical properties.

| Group | Subgroup | Functional group | compound | Molecular weight g/mol | Mass concentration for 0.4 nM (ng/L) | number of C | Charge | pKa | Log Koc | Structure |
|---|---|---|---|---|---|---|---|---|---|---|
| **Perfluoroalkyl acids** | Perfluoroalkyl carboxylic acids (PFCAs) | $-COO^-$ | Perfluorobutanoic acid (PFBA) | 214.04 | 85.6 | 4 | negative | 1.07 | 1.9 |  |
| | | | Perfluorohexanoic acid (PFHxA) | 314.05 | 125.6 | 6 | negative | -0.78 | 1.3 | |
| | | | Perfluorooctanoic acid (PFOA) | 414.07 | 165.6 | 8 | negative | -4.2 | 1.89-2.63 | |
| | Perfluoroalkyl sulphonic acids (PFSAs) | $-SO3^-$ | Perfluorobutanesulphonic acid (PFBS) | 300.1 | 120.0 | 4 | negative | -3.31 | 1.2-1.79 |  |
| | | | Perfluorohexanesulphonic acid (PFHxS) | 400.11 | 160.0 | 6 | negative | -3.32 | 2.4-3.1 | |
| | | | Perfluorooctanesulphonic acid (PFOS) | 500.13 | 200.1 | 8 | negative | -3.32 | 2.4-3.7 | |
| **Perfluoroalkyl ether acids** | Perfluoroalkyl ether carboxylic acids | $-COO^-$ | Hexafluoropropylene oxide dimer acid (GenX) | 330.19 | 132.1 | 6 | negative | -0.77 | 1.7 |  |
| **Fluorotelomer substances** | Fluorotelomer sulphonic acids | $-CH_2CH_2SO_3^-$ | 6:2 fluorotelomer sulphonate (6:2 FTSA) | 428.16 | 171.3 | 8 | negative | -2.72 | 2.1 |  |
| | Fluorotelomer sulphonamide alkyl betaine | (-CO2-) | 6:2 fluorotelomer sulphonamide alkylbetaine (6:2 FTAB) | 570 | 228.0 | 6 | Zwitterionic | 2.81 | 2.4 |  |
| **Perfluoro alkane sulphonamides** | Perflorohexane sulphonamide | $-SO_2NH_3$ | Perfluorohexanesulphonamide (FHxSA) | 399.13 | 159.7 | 6 | neutral | 3.37 | 2.3 |  |

# 3. Water characterisation

Three water types were used in this study: a groundwater source, an upland surface water source, and a lowland surface water source (Table 2). The groundwater and upland water contained similar concentrations of dissolved organic carbon (DOC), 2.5 and 2.1 mg/L, respectively. The lowland surface water had 4.1 mg/L DOC, which was between 1.6-1.9 times higher than the other two source waters. The upland water was more acidic than the other sources and was softer, only having a conductivity of 32 µS/cm. The upland and lowland water source were spiked with a nominal concentration of 0.4 nM of each of the 10 PFAS compounds (Table 3). The groundwater was not spiked because it already contained a range of PFAS compounds.

**Table 2.** Characterisation of the water sources tested.

| **Source water** | **pH** | **$UV_{254}$ ($cm^{-1}$)** | **DOC (mg/L)** | **Turbidity (NTU)** | **Conductivity (µS/cm)** |
|---|---|---|---|---|---|
| Groundwater | 7.6 | 0.044 | 2.5 | 0.22 | 151 |
| Lowland water 2 | 8.3 | 0.130 | 4.1 | 0.24 | 586 |
| Upland water | 6.1 | 0.184 | 2.1 | 0.42 | 32 |

**Table 3.** Model PFAS concentrations. Spiked concentrations (molar and mass) for up- and lowland water sources and typical measured concentrations for the groundwater source.

| **Compound** | **Spiked Concentration** | | **Groundwater Concentration (ng/L)** |
|---|---|---|---|
| | **(nM)** | **(ng/L)** | |
| PFOS | 0.4 | 200 | 670 |
| PFOA | 0.4 | 166 | 176 |
| 6:2 FTAB | 0.4 | 229 | 142 |
| PFBA | 0.4 | 86 | 134 |
| PFBS | 0.4 | 120 | 186 |
| GenX | 0.4 | 132 | 43 |
| PFHxS | 0.4 | 160 | 564 |
| PFHxA | 0.4 | 126 | 391 |
| FHxSA | 0.4 | 160 | 178 |
| 6:2 FTSA | 0.4 | 171 | 172 |

To assess the accuracy of the spiking and to determine the range of PFAS concentrations present in the groundwater, the measured PFAS concentrations were compared with the nominal concentration for the spiked upland and lowland surface water (Figure 1). Groundwater samples showed elevated levels of some PFAS compounds, including PFOS, perfluoropentanoic acid (PFPeA) and perfluoroheptanoic acid (PFHpA) when comparing measured values with the concentrations spiked into the other water sources (Figure 1). This highlighted that the groundwater was an already significantly contaminated matrix. The upland water had more uniform box plots with mean concentration values close to the targeted PFAS concentrations, although some compounds such as GenX, FHxSA and 6:2 FTAB were underdosed (Figure 1). Whilst spike recoveries for most compounds were acceptable for the lowland surface water, higher concentrations of 6:2 FTAB were measured (Figure 1). This indicated that this source contained this specific compound prior to the spiking. However, due to the time lag between taking a sample for analysis and return of the result, it was decided that we would continue to spike 6:2 FTAB into the water, acknowledging that this compound was at an elevated level than originally intended.

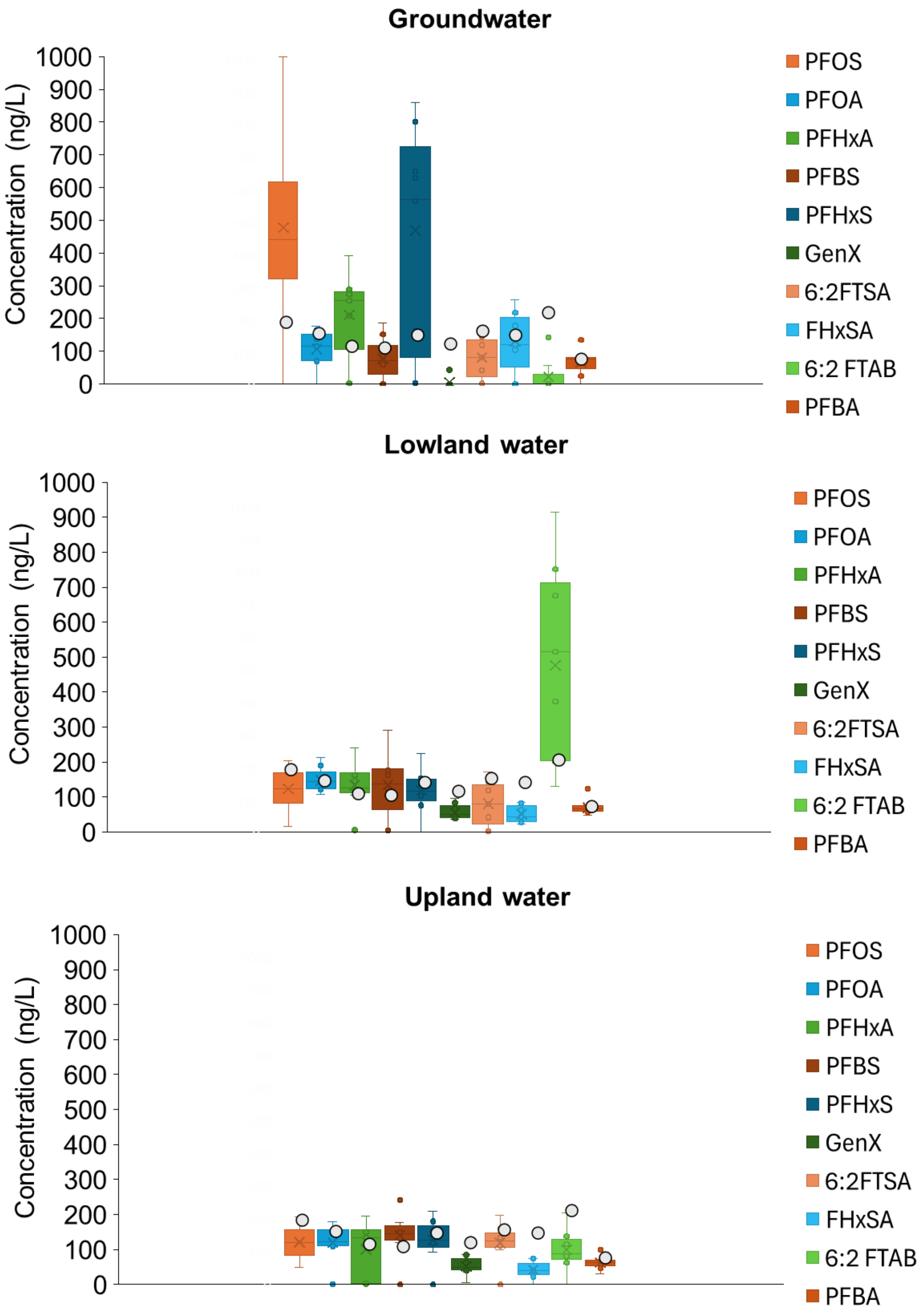


**Figure 1.** PFAS concentrations in groundwater, lowland water an upland water. Circles represent the targeted mass concentration based on spiking at 0.4 nM. In groundwater the circles are for reference and the water was not spiked.

# 4. PFAS analysis and chemicals

PFAS standards were supplied by Greyhound Chromatography (Wallasey, England). Samples were sent to RPS Laboratories (Bedford, England) for external analysis. The full set of model compounds were analysed by RPS following a UKAS accredited method for 51 fluorinated and perfluorinated compounds; the limit of quantification (LOQs) was 0.1 ng/L for most PFAS and the summed reporting limit for all 51 compounds was 25 ng/L. The method required a  sample volume of 250 mL; the samples were acidified, purified and further concentrated by solid phase extraction. Extracts were analysed by liquid chromatography mass spectrometry (LC-MS/MS) in negative ESI mode.

# 5. Adsorption and Ion Exchange Column Tests

Adsorption and IEX column tests were performed with groundwater, lowland and upland surface waters. The three waters were treated with GAC, SMC and IEX for 9 months. PFAS and fluorotelomer concentrations were monitored monthly and an interim DOC sampling was carried out to better understand the impact of DOC on PFAS removal.

## 5.1 Methodology

Adsorption tests were performed at the UKCRIC National Research Facility for Water and Wastewater Treatment (Cranfield University) using 10-cm internal diameter glass columns (Figure 2). Conventional bituminous coal-based activated carbon Filtrasorb 400 (F400), a modified bentonite clay media (SMC) and Lewatit® TP109 IEX resins were tested. The latter is a strong base anion exchange resin with a quaternary ammonium functional group. The media were rinsed with ultrapure water and dried at 35 °C before testing.  An empty bed contact time (EBCT) of 10 min was used for F400 and 5 min for the SMC and IEX, following bench-scale testing results and manufacturers recommendations. Water was pumped through the columns with centrifugal pumps operating at 26.1 ml/min and 30.9 ml/min for F400 and the SMC and IEX, respectively. Raw groundwater was delivered weekly, post-filtration lowland water was delivered bimonthly, and post-filtration upland water was delivered monthly.

The PFAS mass adsorbed onto the media at a given throughput was calculated using Equation 1 from the breakthrough curves data and solved using the Trapezoidal Rule approximation.

$$q_{PFAS} = \frac{m_{PFAS}}{m_{adsorbent}} = \frac{\int Q(C_0 - C_t)dt}{m_{adsorbent}} \qquad (1)$$

Where:

$q_{PFAS}$ is the PFAS load per mass of adsorbent (µg/mg)

$m_{PFAS}$ is the mass of PFAS adsorbed in µg

$m_{adsorbent}$ is the mass of media used (F400, SMC or IEX) in g

Q is the flow used during testing (L/d)

$C_0$ is the concentration of PFAS in the untreated water (µg/L)

$C_t$ is the concentration of PFAS at a given time (µg/L)

t is the testing time (d)

**Figure 2.** Column experimental set up.

## 5.2 Adsorption media comparison

Carboxylic acid PFAS (PFBA, PFHxA and PFOA) were the group showing the earliest breakthrough, regardless of the media and water type. Overall, the second PFAS group to breakthrough was the sulphonic acids (PFBS, PFHxS and PFOS), followed by a third group which included PFAS with other functional groups (fluorotelomers 6:2 FTSA and 6:2 FTAB, GenX and FHxSA) (Figures 3-5). For a given PFAS group (i.e. carboxylic acids) the order of breakthrough followed the length of the PFAS chain. These findings are in agreement with the results obtained in the bench-scale testing (DWI, 2025) and the literature (Park et al., 2020; Schaefer et al., 2020).

When using GAC (Figure 3), carboxylic acids were detected in the treated water after ~1,500 BVs for all water types. However, the rate of breakthrough (based on the slope of the breakthrough curve) was faster in the lowland water. The rate of breakthrough for the sulphonic acids was also faster in the lowland water. In the treated groundwater, PFBS (4C) broke through after PFHxS (6C) and at the same time as PFOS (8C), which did not follow the order of chain length. This was due to the PFBS concentration in the untreated groundwater being much lower than that of PFHxS. The average PFBS concentration in the groundwater was 77 ng/L, while PFHxS was present at a six times higher concentration at 469 ng/L. In contrast, the surface water concentrations were 146 ng/L for PFBS and 124 ng/L for PFHxS (Figure 1).

GAC showed higher efficiency and slower breakthrough rates for the PFAS group with other functionalities (fluorotelomers 6:2 FTSA and 6:2 FTAB, GenX and FHxSA) when treating the upland and groundwater sources. These PFAS compounds were detected in the treated water between 3,000-5,000 BVs but at low concentrations levels (typically, below 50 ng/L), and adsorption continued for most compounds. Less effective removal of this group of compounds was seen for the lowland water source. Breakthrough of these compounds was observed at similar BVs as was seen for the carboxylic and sulphonic acids (~1,200 BVs). It was evident that 6:2 FTAB was particularly problematic, reaching a concentration of 200 ng/L after 6000 BVs. This was due to the high loading of this compound onto the columns due to the already high concentration present in the source water.

For the SMC, very good removal of most of the sulphonated PFAS was observed (Figure 4), with low levels of PFAS being detected after 17,000 BVs for the three water sources. PFOS breakthrough was not recorded during the trial. The exception was PFBS which started to see breakthrough after 15,000 BVs for the surface waters, while all other compounds were still being efficiently removed for all other conditions. For the carboxylic acids, the picture was more mixed. PFOA was consistently very well removed from all source waters, only starting to be seen in the treated waters after 19,000 BVs. PFBA was not well adsorbed for any of the surface waters but showed some adsorption in the groundwater (until ~5,000 BVs). PFHxA was not well adsorbed on the SMC when treating the lowland water, and adsorption/desorption profiles were seen, with increasing and decreasing concentrations seen in the treated water with continuing BVs treated. The PFAS group with other functionalities were effectively removed, except for 6:2 FTAB which appeared in the treated water immediately for the upland and lowland surface waters. The latter was overall not well removed by the SMC, and fluctuations in the treated water concentrations suggested an adsorption/desorption process was taking place. GenX was detected after 12,000 BVs for the upland water, while FHxSA appeared in the groundwater after 17,000 BVs.

IEX treatment was overall the most effective treatment for removal of PFAS (Figure 5). Sulphonic acids and PFAS with other functional groups (excluding carboxylic acids) did not breakthrough for 17,000 BVs, apart from 6:2 FTAB, which was not well removed. This PFAS compound is positively charged and, therefore, was not expected to be well removed by the IEX resin used in this study, which includes a positively charged group. While removal of PFAS by IEX is a combination of charge and hydrophobic interactions, electrostatic repulsion limits the interaction of positively charged compounds with the resin. For the carboxylic acid group, PFOA continued to be adsorbed after 17,000 BVs. PFHxA was detected after ~2,400 BVs in the upland treated water, 5,000 BVs in the lowland water, and 12,200 BVs in the groundwater. PFBA had limited removal when treating the surface waters and showed a rapid breakthrough profile after 5,000 BVs.

Overall, removal efficiency was higher when using IEX, followed by SMC and GAC for a broad range of PFAS, although the best treatment option was dependent on the specific PFAS fingerprint in a water source. The total throughput tested in the trial was lower than adsorption systems usually run for other micropollutants (10,000 BVs for GAC, 19,000 BVs for SMC and 17,000 BVs for IEX). However, it was apparent that many PFAS compounds broke through into treated water much earlier than the throughputs tested here for the different media. The scope of the trial was therefore long enough to establish the PFAS groups with higher affinity for each media, the order of breakthrough and the influence of water source on these factors.

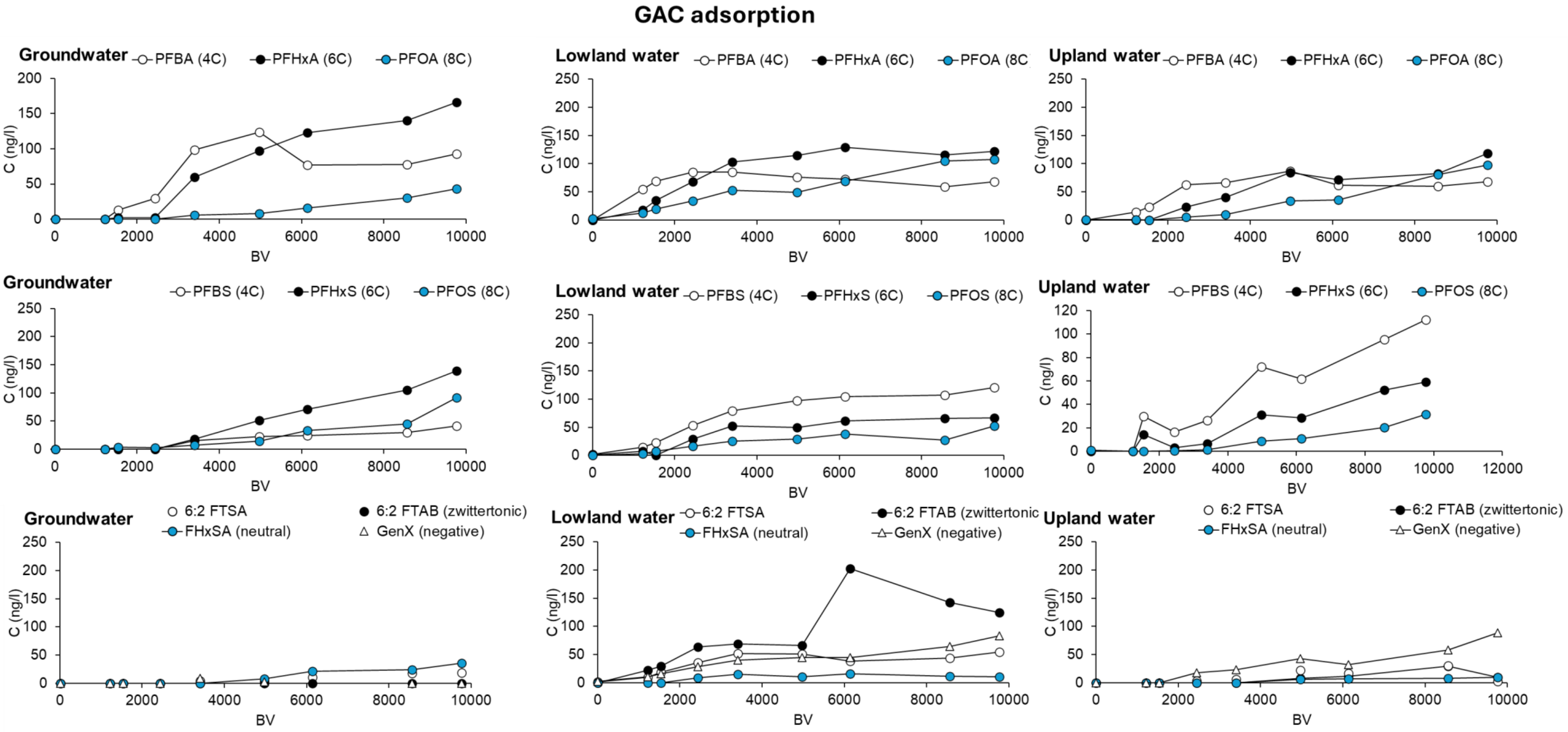


**Figure 3.** PFAS breakthrough profiles for GAC adsorption from groundwater, lowland and upland waters.

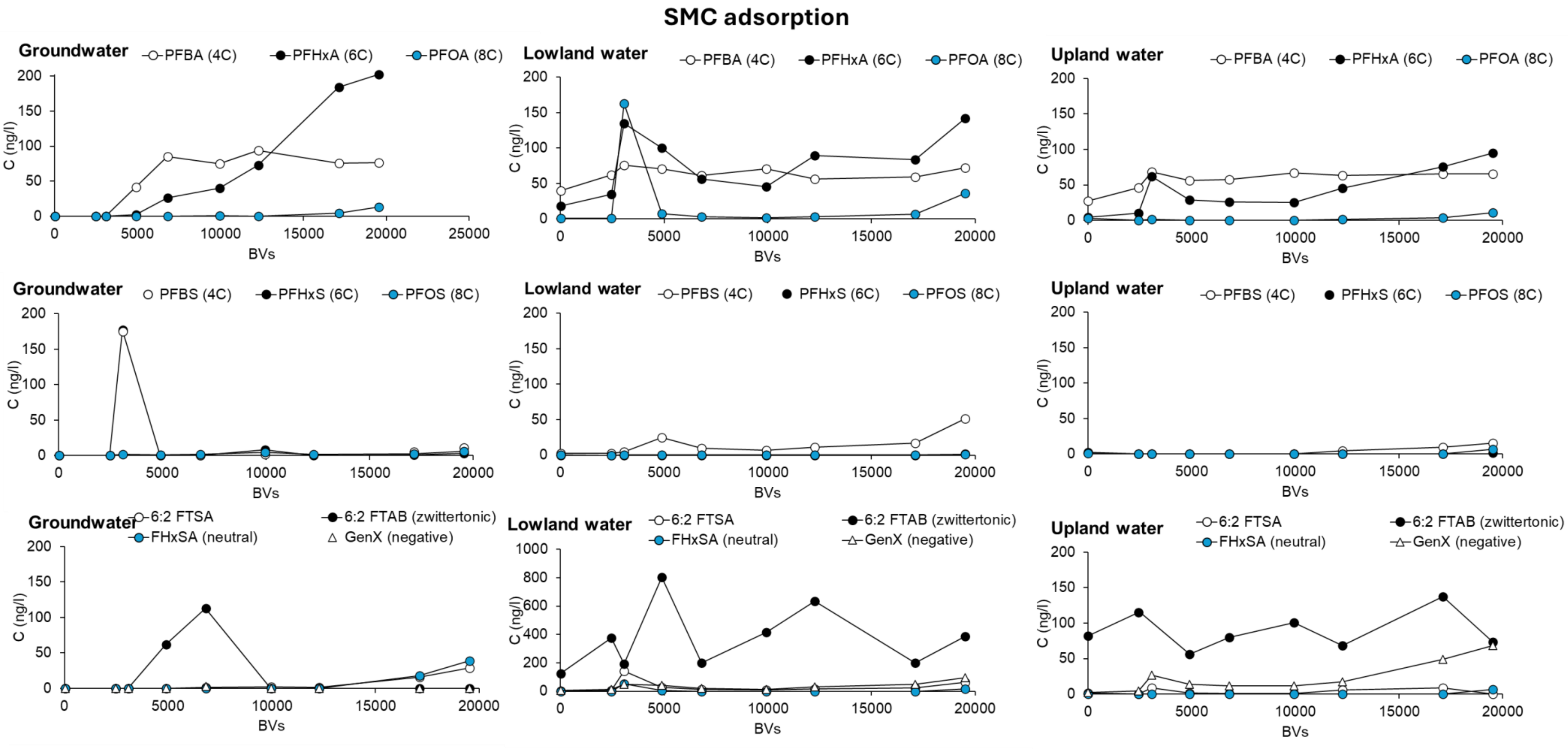


**Figure 4.** PFAS breakthrough profiles for SMC adsorption from groundwater, lowland and upland waters.

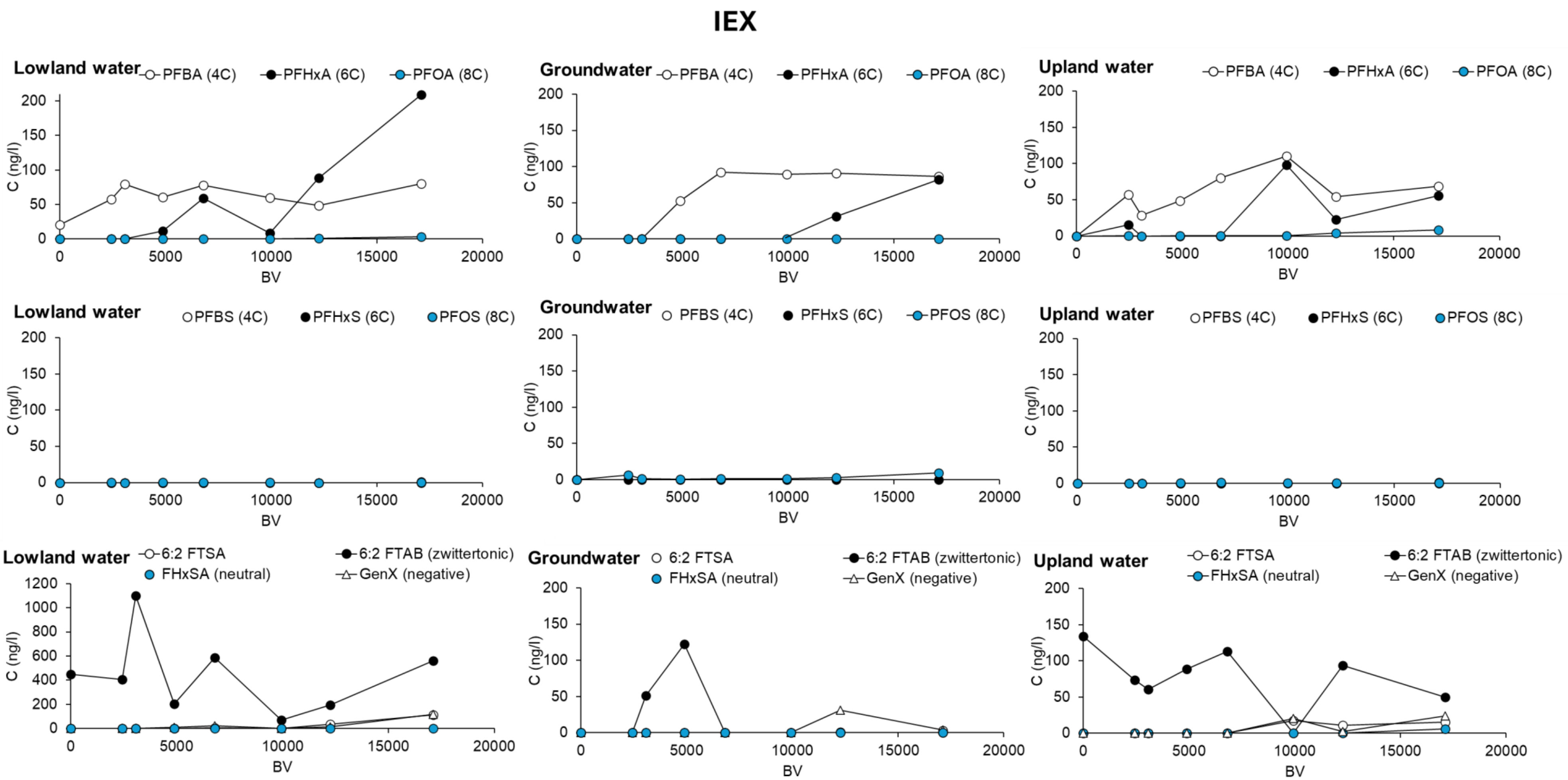


**Figure 5.** PFAS breakthrough profiles for IEX adsorption from groundwater, lowland and upland waters.

## 5.3 Impact of the water type on PFAS removal

This section reviews the influence of the water matrix on the adsorption efficiency for each media type.

### 5.3.1 4C and 6C PFAS

In general, sulphonic acid PFAS were better removed than carboxyic acids. PFBS, which is a 4C sulphonic acid, did not show breakthrough when using IEX, while only trace level of the compound was measured for SMC treated water (Figure 6). However, PFBS was detected at 1,200 BVs when using GAC. In this case, the rate of breakthrough was higher in the lowland water, followed by the upland water and then the groundwater. While this order of breakthrough was observed for other PFAS compounds (i.e. the rate of breakthrough was lowland water > upland water > groundwater), in this case the rate of breakthrough for PFBS concentration in groundwater was much lower than was seen for the other water sources. This was considered to be due to the lower DOC concentration present in this water souce when compared to the surface waters. Carboxylic acid PFBA was the most challenging to remove of all the tested PFAS compounds. It showed breakthrough before 3,000 BVs, regardless of the media and water type. For this compound the water source used did not have as significant impact on treatment efficiency because of the overall low capacity for this compound for all media.

Adsorption through GAC and SMC is largely driven by hydrophobic interactions. The length of the chain is proportional to hydrophobicity for a given PFAS group. For example, the water-octanol coefficient (log $K_{ow}$), which is related to the hydrophobicity of a compound, of 4C PFBA is 1.9 times lower than that of 8C PFOA (2.8 vs 5.3 ) (Table 2). Hydrophobic interactions are also important during removal by IEX, but charge-driven electrostatic forces also have a significant role (Lei et al., 2023). Other mechanisms involved in PFAS soprtion processes are to do with the formation of micelles. Due to their surfactant properties, PFAS can form micelles, which influence surface mechanisms (Leung et al., 2023). The properties of these micelles are influenced by the chain length, and longer-chain, more hydrophobic PFAS are more likely to form micelles. As a result, the formation of micelles can positively impact the removal of long-term PFAS through the formation of hydrophobic micelle aggregates. On the other hand, micelle formation can hinder the removal of shorter-chain, more hydrophobic PFAS as micelles can make them less accessible for interactions with the soprtion media.

**Table 2**. PFAS reported octanol partition coefficient (log $K_{ow}$)

| PFAS | Carbon atoms | log $K_{ow}$ |
|---|---|---|
| Perfluorobutanoic acid (PFBA) | 4 | 2.8[1] |
| Perfluorohexanoic acid (PFHxA) | 6 | 4.1[1] |
| Perfluorooctanoic acid (PFOA) | 8 | 5.3[1] |
| Perfluorobutanesulphonic acid (PFBS) | 4 | 3.9[2] |
| Perfluorohexanesulphonic acid (PFHxS) | 6 | 5.2[2] |
| Perfluorooctanesulphonic acid (PFOS) | 8 | 6.3[2] |
| Hexafluoropropylene oxide dimer acid (GenX) | 6 | 3.2[3] |
| 6:2 fluorotelomer sulphonate (6:2 FTSA) | 8 | 4.4[4] |
| 6:2 fluorotelomer sulphonamide alkylbetaine (6:2 FTAB) | 15 | NA |
| Perfluorohexanesulphonamide (FHxSA) | 6 | NA |

[1] Z. Wang, M. MacLeod, I.T. Cousins, M. Scheringer, K. Hungerbühler
Using COSMOtherm to predict physicochemical properties of poly- and perfluorinated alkyl substances (PFASs)
Environ. Chem., 8 (2011), pp. 389-398

[2] Concawe. Environmental fate and effects of poly and perfluoroalkyl substances (PFAS), 2011.

[3] Hopkins et al. Recently Detected Drinking Water Contaminants: GenX and Other Per- and Polyfluoroalkyl Ether Acids, 2018

[4] Concawe. Environmental fate and effects of poly and perfluoroalkyl substances (PFAS), June 2016

PFBA is a particularly problematic PFAS compound. Following regulatory restrictions, PFBA presence in the environment has been increasing due to its use as an alternative to PFOA (for example, in water- and stain-resistant coatings). As such, PFBA is one of the most prevalent compounds in water bodies in England and Wales, with an average concentration of 0.072 µg/L (DWI, 2022). As reported herein, PFBA was poorly removed to any degree by any of the media, including GAC and IEX. As such, where this contaminant is a concern, alternative adsorbents and different IEX media should be tested, as well as other types of treatment (e.g., membrane separation).

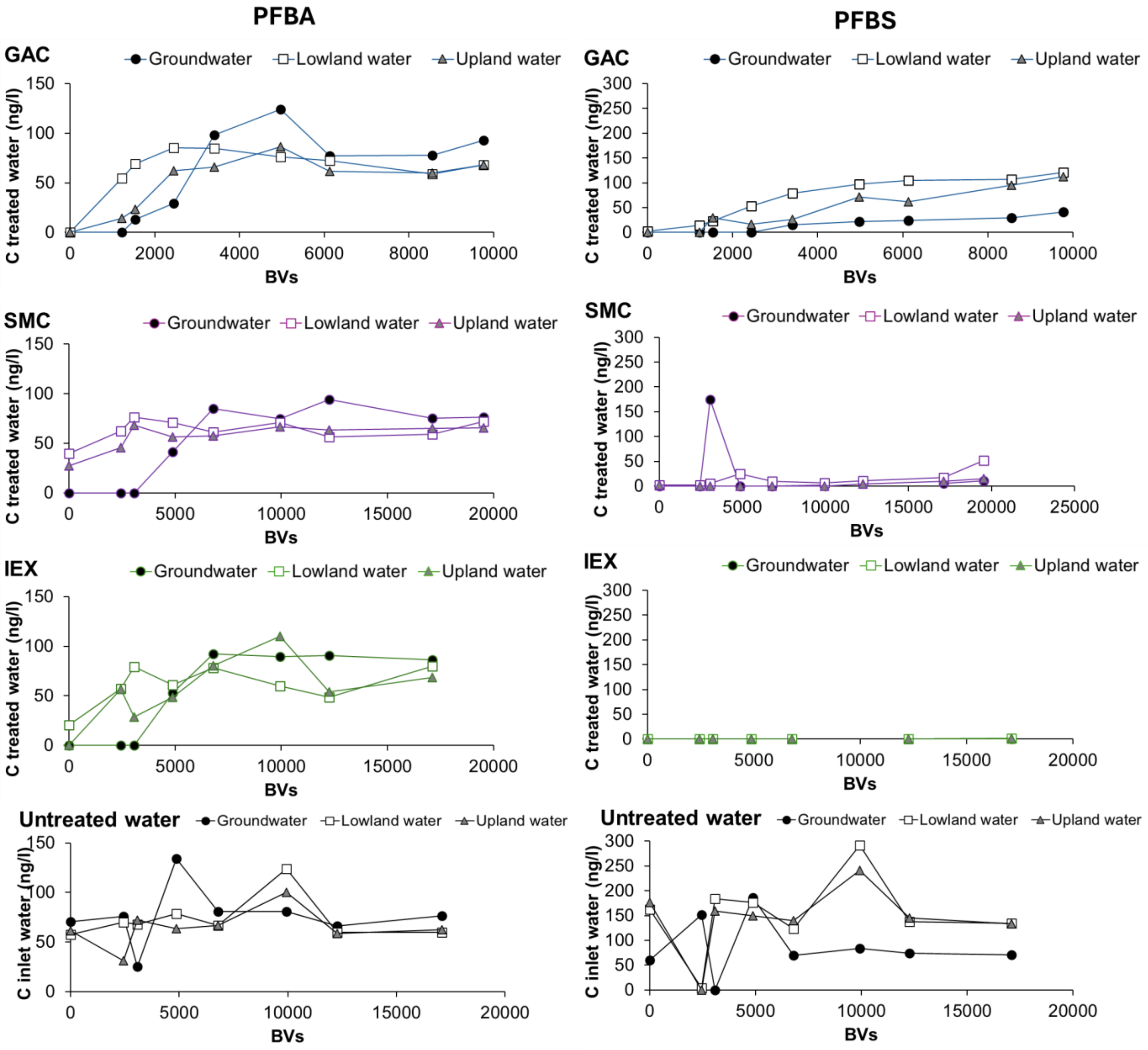


**Figure 6.** PFBA and PFBS breakthrough profiles during GAC, SMC and IEX adsorption in lowland, upland and groundwaters and the untreated water concentration profiles.

Within the 6C PFAS, PFHxA (carboxylic acid) had similar breakthrough curves for all water sources but removal was more efficient with IEX > SMC > GAC (Figure 7). It is worth noting that the concentration of carboxylic PFAS in the untreated groundwater was much higher than in the other two water sources for the samples corresponding to 5,000 BVs. However, this was not reflected in the breakthrough profiles, showing effective removal from the groundwater could be maintained regardless of the load onto the column. GenX is a 6C ether-based PFAS, where the main functional group is a carboxylic group. Its influent concentration profiles were similar to those recorded for PFHxA for the surface waters (GenX was not routinely seen in the groundwater), although breakthrough rates for this compound were faster than for PFHxA (Figure 7). This was linked to the difference in hydrophobicity between the two compounds, with PFHxA being more hydrophobic. To illustrate, log $K_{ow}$ for PFHxA and GenX was 4.1 and 3.2, respectively (Table 2). Although the GenX breakthrough curves were similar across the surface waters, treatment of the lowland water was slightly less efficient than in the upland water, likely reflecting the increased competition for adsorption with background organic compounds for the lowland source.

PFHxS (sulphonic acid) was well removed by all treatment processes, particularly for IEX and SMC for all water sources. This was even though the concentration in the groundwater source was high, reaching 800 ng/L. For GAC, breakthrough occurred after ~5,000 BVs, but the rate of breakthrough was much slower than that seen for the 4C PFAS (Figure 6). PFHxSA was also well removed and had a similar breakthrough profile to that seen for PFHxS. It was only detected in GAC and SMC towards the end of the trial (ca 8,000 BVs for GAC and 17,000 BVs with SMC), so no effect of water matrix was observed (Figure 7).

Both compounds have high hydrophobicity (log $K_{ow}$ of 5.17 and 4.2 for PFHxS and PFHxSA, respectively) (Table 2). Perfluoroalkyl sulphonamides (also known as FASAs) such as PFHxSA contain a sulphonamide as the main functional group. For a given chain length, FASAs are generally more hydrophobic than their corresponding sulphonic acid PFAS (i.e. PFHxS). This is because the sulphonamide group, although hydrophilic, is less so than the sulphonate group (Ma and Olivares, 2025). However, these differences were not reflected in the removal profiles observed for these two compounds, likely because both were already relatively hydrophobic and had high affinity for the media.

For the 6C PFAS group there was not a strong impact of the water source on PFAS breakthrough, despite the fact that the concentrations of some compounds were significantly higher in the groundwater. This was explained by the higher adsorption capacity of the three media for these 6C compounds. Longer throughputs may highlight more differences in removal for these PFAS compounds.

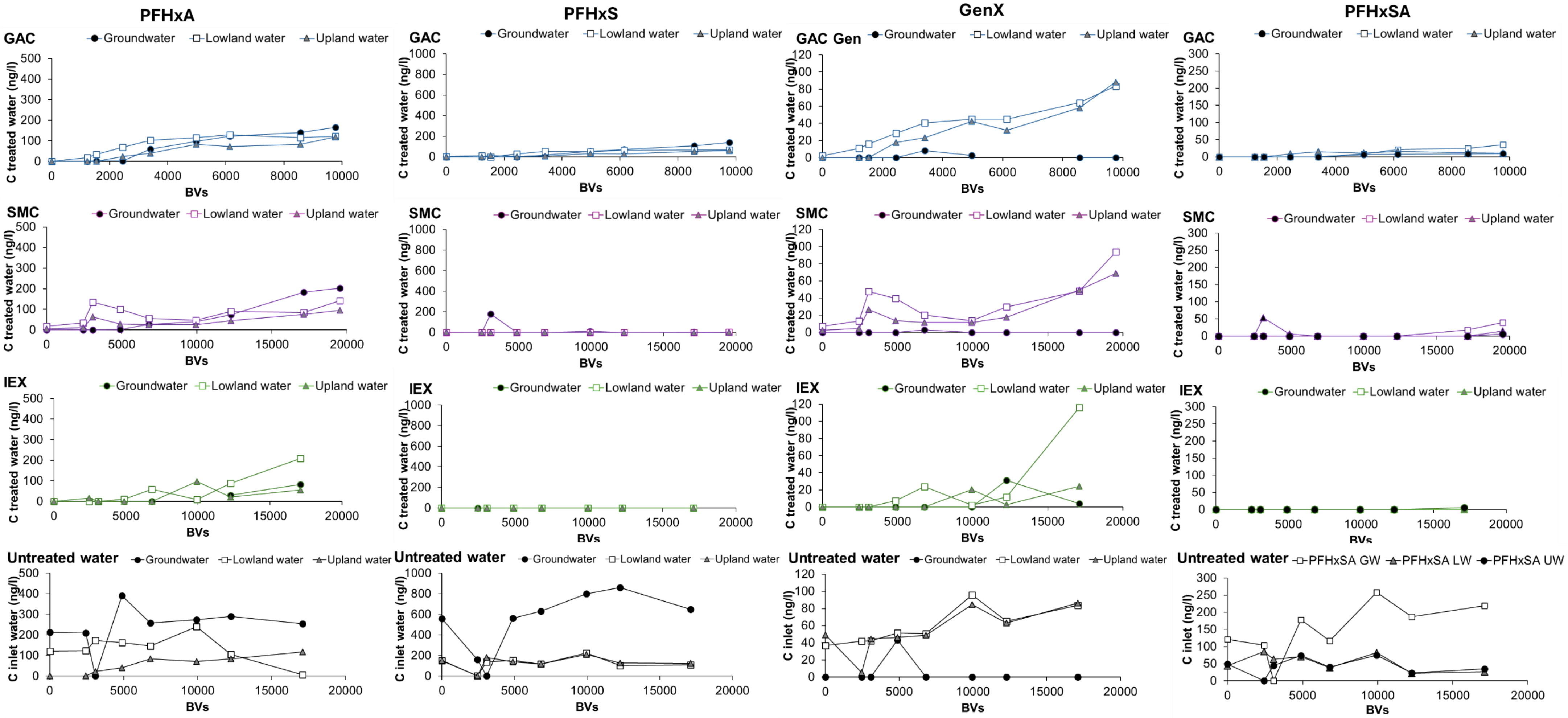


**Figure 7.** PFHxA, PFHxS, GenX and PFHxSA breakthrough profiles for GAC, SMC and IEX adsorption from lowland, upland and groundwaters, alongside the concentration of PFAS in the inlet water (untreated).

### 5.3.2. 8C PFAS and fluorotelomers

PFOA (carboxylic acid), PFOS (sulphonic acid) were the 8C PFAS tested in this study. Both 6:2 FTAB and 6:2 FTSA are fluorotelomers with 6C perfluoroalkyl chain linked to other groups. In the case of 6:2 FTAB the fluorinated chain is linked to a sulphonamide and alkylbetaine groups; while 6:2 FTSA is connected to a sulphonic acid group. Due to their longer chain length and hydrophobic properties, all four of these compounds were overall better removed than the 4C and 6C PFAS. PFOS had the best adsorption properties, and was not detected in the treated water throughout the trial with any media type (Figure 8). This was despite the high concentrations observed in the untreated waters (up to 1,000 ng/L in the groundwater). PFOA was also efficiently removed throughout the duration of the trial, particularly with SMC and IEX. However, it was detected after 1,200 BVs in the lowland water and 5,000 in the upland and groundwaters when using GAC. Towards the end of the trial with the surface waters (~10,000 BVs) the GAC was saturated with PFOA and the concentration in the treated waters was close to that in the untreated water.

6:2 FTAB was effectively removed by GAC, with no breakthrough observed for the upland water, while breakthrough was seen after 6,000 BVs for the lowland water. It should be noted that the concentration of 6:2 FTAB in the lowland water was much higher than the upland water due to the real contamination of the water on top of the spiked amount. SMC and IEX could not remove this compound from the lowland water, and lower levels of removal were also seen for the upland surface water by these media than was seen for GAC. The reduced removal of 6:2 FTAB for IEX was due to the electrostatic repulsion between the positive charge on the compound and the IEX media. For SMC, reduced removal was hypothesised to be due to the reduced affinity of the modified surface for this fluorotelomer. 6:2 FTAB was rarely detected in the groundwater, so the adsorption efficiency could not be compared for this water matrix.

6:2 FTSA was well removed from all water matrices and by all media types. The lowland water was again the one where breakthrough was observed first. This was more significant when using GAC (but some removal was still maintained), and towards the end of the trial for IEX (after 10,000 BVs). Breakthrough was recorded at ~3,000 BVs when using SMC, matching a spike in the untreated water, but high adsorption rates continued to be seen afterwards.

6:2 FTAB is another PFAS that affords significant challenges for removal in water treatment. The betaine group in this compound has a quaternary ammonium group with a positive charge and is linked to a carboxymethyl group. These positive quaternary ammonium groups are often the functional groups present on many anionic IEX resins. Therefore, when searching for removal options for 6:2 FTAB processes other than anionic IEX should be considered. Little is known of the hydrophobicity of this compound and a log $K_{ow}$ value has not been reported in the literature, but based on the results in this study, activated carbon and other processes that rely on hydrophobicity-based removal mechanisms, cationic exchange or size separation with membranes are likely to be more effective.

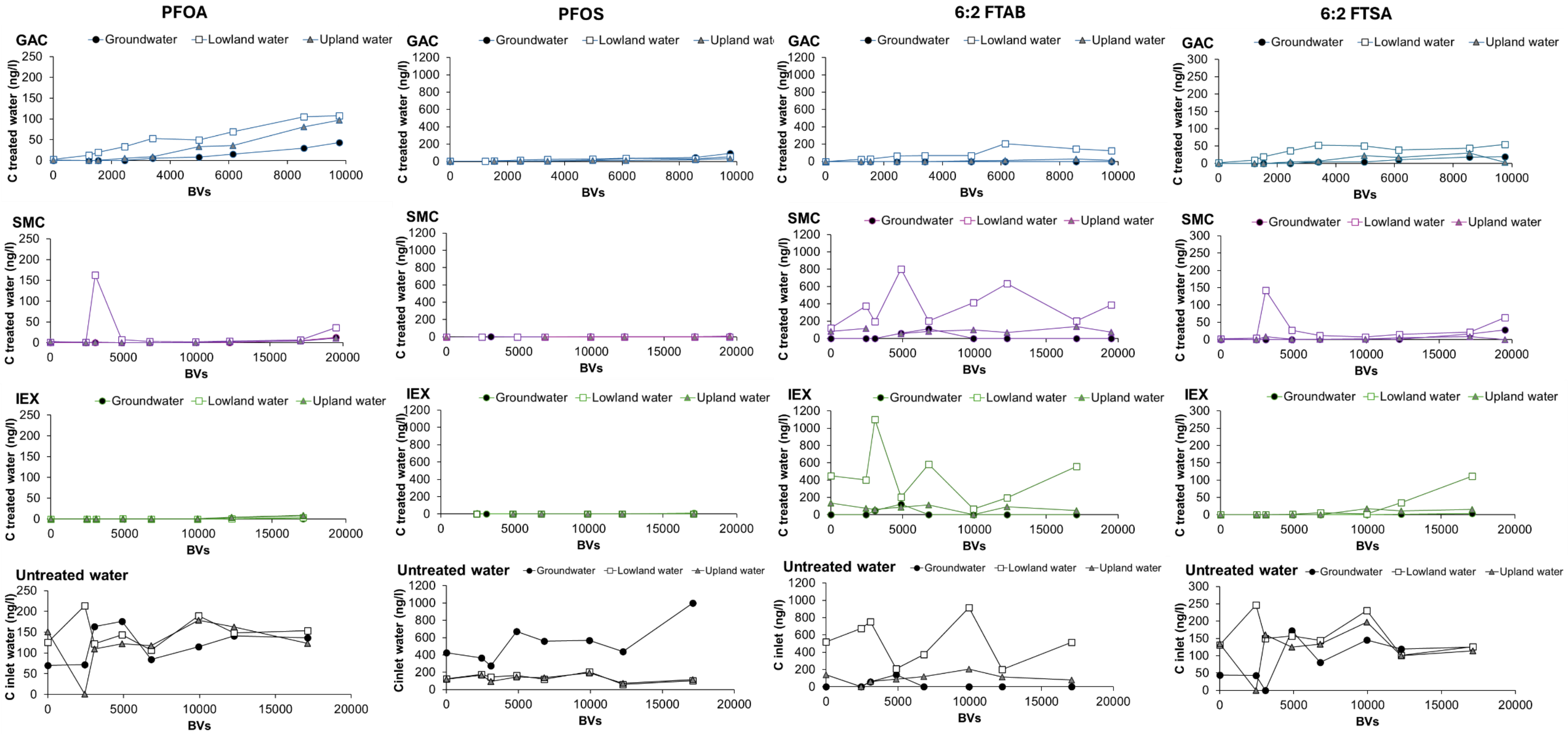


**Figure 8.** PFOA, PFOS, 6:2 FTAB and 6:2 FTSA breakthrough profiles during GAC, SMC and IEX adsorption in lowland, upland and groundwaters and the untreated water concentration profiles.

## 5.4 Interim Dissolved Organic Carbon sampling

The adsorption results showed that the background matrix has an important influence on the removal of PFAS. This was particularly so when the affinity of the media for a PFAS compound is low, such as was seen for the 4C PFAS, or when the media capacity is more limited (GAC adsorption). Organic matter can influence PFAS adsorption through competition for the active sites in the adsorption media. This is due to the physicochemical properties of certain fractions of organic matter and because it is present at much higher concentrations than PFAS and other micropollutants. For IEX, organic matter is equally relevant, although other inorganic anions (such as sulphate, chloride or bicarbonate) can also compete for the active sites.

To shed light on to the impact of organic matter concentration on PFAS removal, an interim DOC sampling was conducted during the trial. Analysis of the untreated water indicated high variability in the groundwater DOC (up to 1.5 mg/L variability), with an average value of 2.5 mg/L (Figure 9). The lowland water had the highest DOC content at 4.1 mg/L, and the upland water had the lowest concentration at 2.1 mg/L.

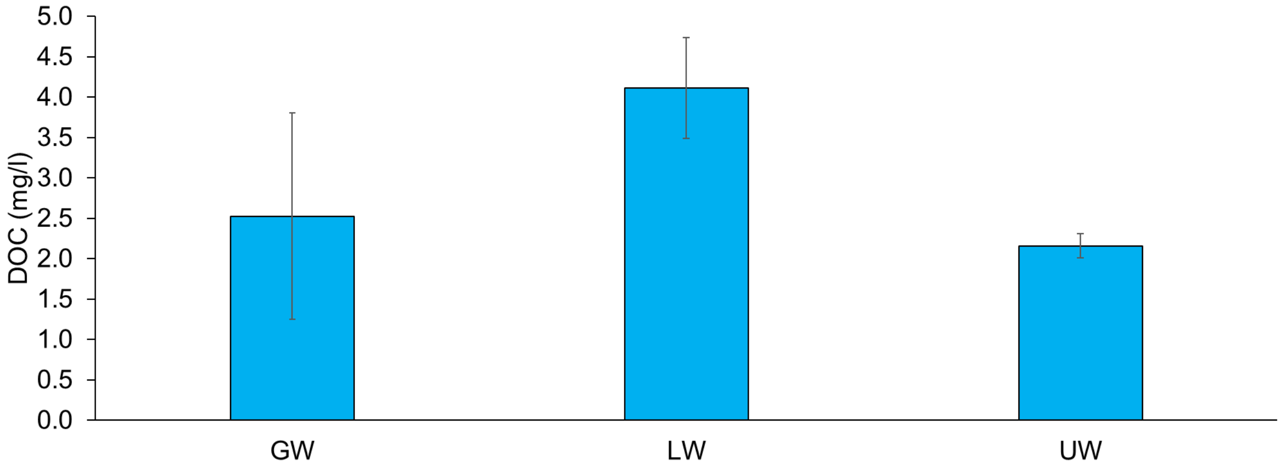


**Figure 9.** Average DOC concentration in the untreated water during the interim DOC sampling in the groundwater, lowland water and upland water.

For the lowland water, GAC, SMC and the IEX resins continued to remove approximately 1 mg/L of the DOC from the water up until 7,000-8,000 BVs (Figure 10). After this time, the concentration of DOC in the treated water was similar to that of the untreated water and there was no overall net removal. This water had the highest DOC concentration, but despite this, some PFAS compounds were still being removed after this point, particularly those with high affinity for the carbon. For example, many of the 8C and 6C PFAS were still being adsorbed after 5,000 BVs (Figures 7 and 8). This would suggest that the role of organic matter on adsorption is more relevant for the removal of shorter-chain PFAS, for which the adsorption media has lower affinity and capacity.

In the case of the upland water, there was some DOC removal by all media up to 5,000 BVs (approximately 0.5-1.0 mg/L). After this, the SMC and GAC saw no further overall net removal of DOC. For the SMC the treated water also recorded higher values than the untreated water, suggesting that there may have been some desorption of DOC, consistent with the desorption effects seen for some of the PFAS compounds from the breakthrough curves (Figure 4). IEX maintained a small amount of continued DOC removal up until ~13,000 BVs, after which the influent and effluent values were similar to one another. As for the lowland water, some PFAS were still removed after this apparent DOC saturation, suggesting that adsorption niches were still available for removal of these contaminants.

All media continued to remove DOC from the groundwater throughout the duration of the trial (between 0.5 and 1.0 mg/L) for the BVs tested, showing that the background organic matter did not saturate the media. This was despite the fact that the DOC level in this water was similar to that in the upland water. This draws attention to the need to better understand the properties of DOC in each water source and its interaction with PFAS during adsorption.

The inference from this is that individual water sources need to be assessed on a case-by-case basis when considering the potential for a particular adsorbent to be applied for PFAS removal. DOC will influence PFAS removal through a number of potential ways: 1) competition for adsorption sites on the media; 2) interaction with PFAS in the water, potentially increasing or decreasing interactions with the sorbent; 3) interaction between PFAS and organic matter already adsorbed. All of these factors are complex and not yet fully understood.

In the present case, the water source with the highest DOC (lowland water) saw the earliest breakthrough for most PFAS compounds when compared to the other two water sources. However, the upland water and groundwater had similar DOC profiles, but the PFAS breakthrough was typically seen earlier in the upland water than when compared to the groundwater. Comparisons need to be made carefully, since slightly different PFAS loadings were present for the two different water sources, but there were also differences in the nature of the organic matter for these water types. The groundwater had lower UV absorbance when compared to the upland water, indicating that the organic matter was more hydrophilic for the groundwater. There may, therefore, be less competition with the PFAS for adsorption sites than was seen in the surface waters.

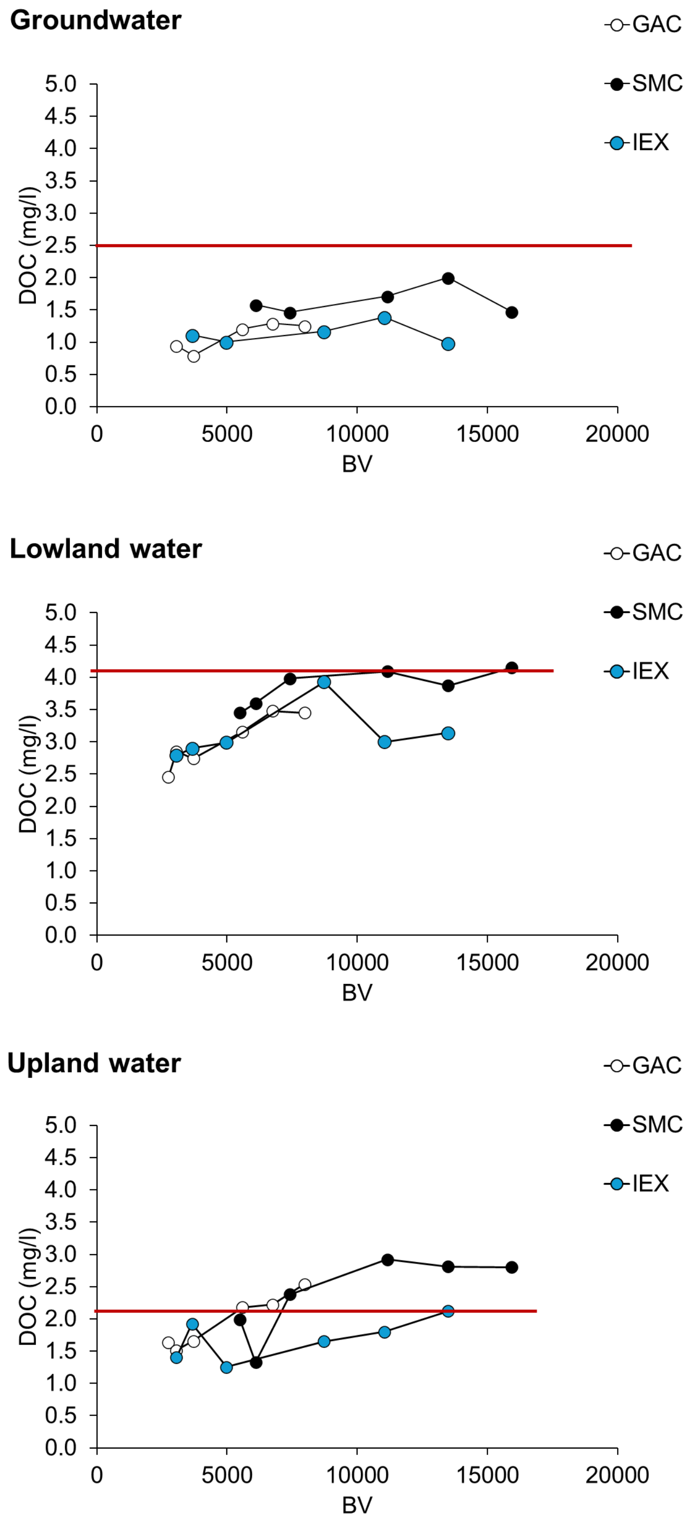


**Figure 10.** DOC concentration in the treated water for the groundwater, lowland water and upland water during the interim sampling. The red line denotes the average DOC concentration in the untreated water.

## 5.5 PFAS load on the media

The capacity of an adsorbent for a contaminant represents the mass of contaminant adsorbed per mass of adsorbent when the latter reaches saturation. This value is specific for a given contaminant and water treated, and it is useful to compare across PFAS and water types. For those PFAS where full column saturation was not reached, the mass of PFAS accumulated per mass of adsorbent at the end of the trial was calculated (Table 3).

All media were fully saturated with PFBA, regardless of the water type (Table 3). GAC also showed saturation for all 6C PFAS after ~10,000 BVs when treating the surface waters, except for PFHxSA, which was close to saturation when treating the groundwater (Figure 3). Although saturation was not recorded, breakthrough was observed for all other PFAS when using GAC. Comparison across GAC capacity values suggested that for a given PFAS compound, the type of surface water (upland or lowland) did not have a significant impact on capacity. GAC capacity values ranged from 90 to just under 2,700 ng/g for 4C and 6C PFAS, and the mass of PFOS and 6:2 FTAB adsorbed was >8,083 and >6,520 ng/g, respectively.

SMC reached saturation with 6C GenX and PFHxA when treating the surface waters. IEX recorded saturation for these two compounds only for the lowland water, although breakthrough was detected in the upland water. Both IEX and SMC media were saturated or showed breakthrough of 6:2 FTAB. Most other compounds were not detected in the first 10,000 BVs of trial, except for PFBS and PFOA when treating the lowland water with SMC, and PFHxSA in the lowland water (Table 3).

The capacity of the SMC media ranged from 204 to 2442 ng/g for those PFAS where saturation of the column was reached. For longer-chain PFAS the mass adsorbed was >3,000 ng/g for many 6C-8C compounds, and in the case of sulphonic acids PFHxS and PFOS, >13,000 ng/g. Overall, the capacity of IEX was similar or higher than of SMC for a given PFAS and water type. For example, the mass of PFHxA removed by IEX when treating the lowland water was >3,657 vs >1876 ng/g with SMC. This is further illustrated in Figure 11, where the IEX:GAC and the SMC:GAC mass ratios are shown.

The SMC:GAC mass ratio at 10,000 BVs (Figure 11) was slightly higher for most PFAS in the lowland water than for the upland water (1.3-1.6 for the lowland water, and 1.1-1.4 for the upland water), excluding 6:2 FTAB, which was best removed by GAC. For the groundwater the adsorbed mass was similar or higher when using GAC than the SMC. This is because this water source, which was already contaminated, had higher content of longer-chain PFAS rather than shorter-chain ones, which are efficiently removed by GAC.

The difference in PFAS capacity was generally higher when using IEX (Figure 11). In the lowland water IEX had between 1.3 and 2.3 times more capacity than GAC. The highest ratios were obtained for short chain PFAS such as PFBS (1.6), PFHxA (2.3) and GenX (2.4). This would suggest that two stage processes combining IEX and GAC would be effective at targeting a broad range of short and long chain PFAS compounds for this water. These trends were also seen in the lowland water, but the capacity ratios were lower (between 1.1-1.4), and more similar to those of SMC:GAC.

**Table 3.** Adsorbed mass of PFAS per mass of GAC, SMC and IEX at the end of the column trial (ng PFAS/g media)[1,2,3]

| | **PFBA (4C)** | **PFBS (4C)** | **PFHxA (6C)** | **PFHxS (6C)** | **GenX (6C)** | **FHxSA (6C)** | **PFOA (8C)** | **PFOS (8C)** | **6:2FTSA** | **6:2 FTAB** |
|---|---|---|---|---|---|---|---|---|---|---|
| **GAC** | | | | | | | | | | |
| **Groundwater** | **90** | >1095 | >2674 | >8771 | **Not present** | >2391 | >1768 | >8083 | >1702 | **Not present** |
| **Lowland water** | **146** | **1544** | **941** | **1514** | **323** | >725 | >1544 | >1779 | >2023 | >6520 |
| **Upland water** | **193** | **1751** | **1144** | **1951** | **403** | >721 | >1685 | >1999 | >1997 | >1802 |
| | | | | | | | | | | |
| **SMC** | | | | | | | | | | |
| **Groundwater** | **316** | >1800 | >4433 | >14559 | **Not present** | >3851 | 3027 | >13056 | >2684 | **Not present** |
| **Lowland water** | **341** | >4104 | **1876** | >3531 | **729** | >1253 | >3509 | >3404 | >3464 | **2442** |
| **Upland water** | **204** | >4008 | **2088** | >3674 | **880** | >1212 | >3454 | >3318 | >3333 | >618 |
| | | | | | | | | | | |
| **IEX** | | | | | | | | | | |
| **Groundwater** | **326** | >2413 | >5062 | >16338 | **Not present** | >4448 | >3457 | >14452 | >3205 | **Not present** |
| **Lowland water** | **422** | >4591 | **3657** | >3695 | **1553** | >1459 | >4096 | >3714 | >4180 | **Not adsorbed** |
| **Upland water** | **70** | >4227 | >2413 | 3825 | >1138 | >1256 | >3556 | >3563 | >3080 | **Not adsorbed** |

[1] Values in bold are the capacity of media for a given PFAS compound, calculated at the column saturation point (when the concentration in the treated water matches that of the untreated water). In any other case, column saturation was not reached, and capacity values are provided as "greater than".

[2] Red was used when column saturation was achieved; yellow represents when breakthrough was observed before 10,000 BVs but column saturation was not reached during the trial (ie some removal was still being seen); and green was used when the PFAS compound was not detected in the treated water during the trial. 10,000 BVs was the throughput treated with GAC in the trial.

[3] Capacity values calculated for the duration of the trial (ca 10,000 BVs for GAC, ca 17,000 BVs for EX and ca 20,000 BVs for SMC).

'not present' means that the PFAS was not detected in the real ground water (not spiked)

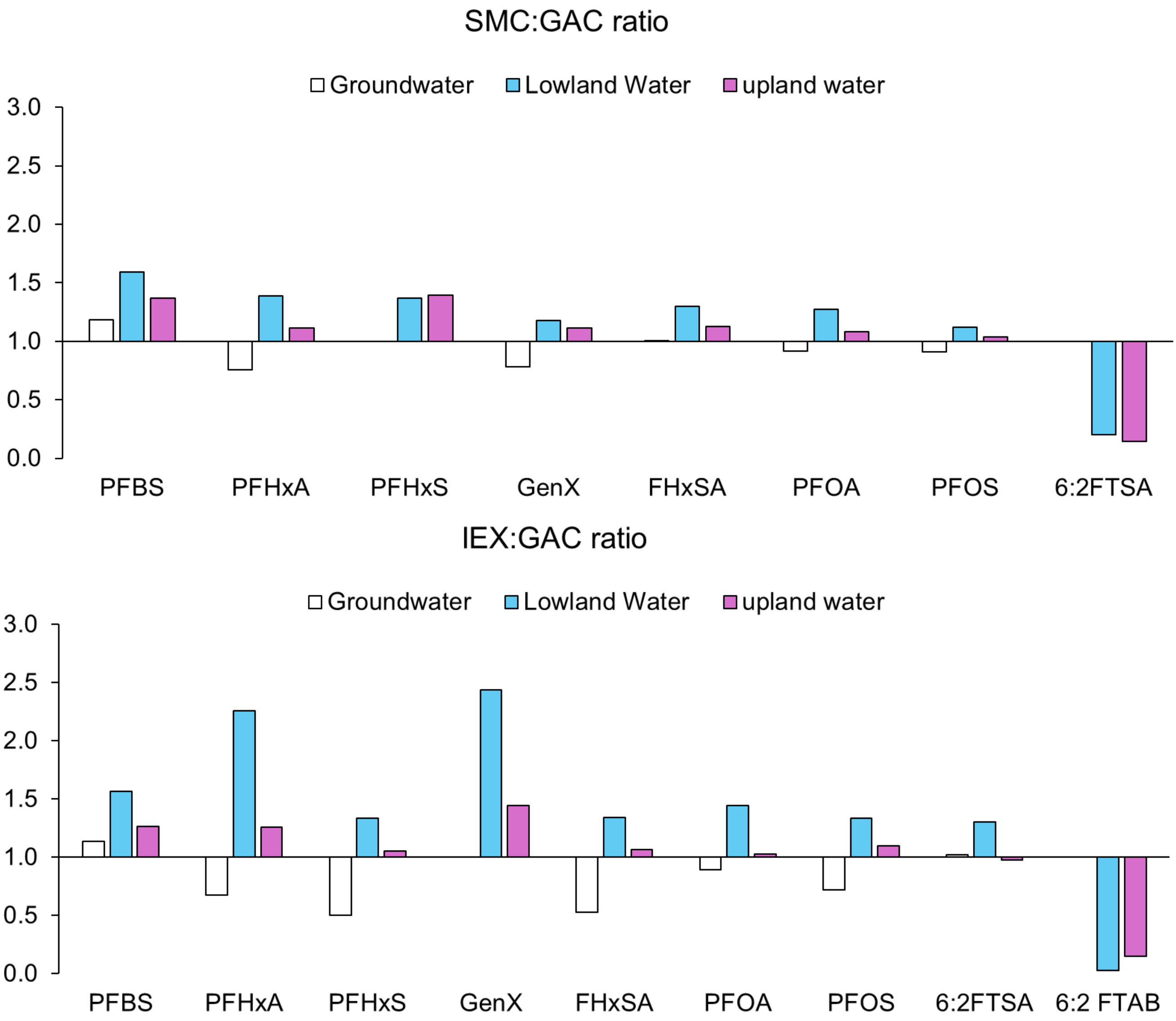


**Figure 11.** SMC:GAC (top) and IEX:GAC (bottom) adsorbed mass ratios at 10,000 BVs throughput.

## 5.6 Summary and key findings for GAC, SMC and IEX

In agreement with other studies and the bench scale tests, the results from the longer-term operational trial showed that most long chain PFAS were more efficiently removed than the short chain compounds. Sulphonated PFAS were more easily removed than the corresponding carboxylic acid compounds.

The short chain carboxylic acid PFAS were the most challenging group to remove, with breakthrough occurring as early as a few hundred BVs. PFBA (4C) was particularly difficult to remove, and it was not efficiently treated over a long period of operation by any of the media types tested.

Most PFAS compounds were removed for longer throughputs by IEX, followed by SMC and GAC. One exception was for 6:2 FTAB, which was best removed by GAC. While longer chain PFAS for all PFAS types were overall better adsorbed by all media, the poor removal of 6:2 FTAB with IEX was an exception due to its positive charge preventing ionic exchange separation. SMC showed breakthrough profiles that could be associated with adsorption/desorption effects for certain PFAS, such as 6:2 FTAB and GenX.

In the present example, most PFAS were preferentially removed from the different water sources in the order: Groundwater > Upland water > Lowland water. From the interim DOC sampling, results suggested complex competition and interactions effects exist between PFAS and background organic matter during adsorption, and that further understanding of the important physicochemical properties of organic matter is required, rather than looking at absolute DOC values.

Further research should also explore the impact of cations on the adsorption of PFAS since they affect surface mechanisms important to the total treatment removal efficiency such as $K^+$, $Ca^{2+}$, $Na^+$ and $Mg^{2+}$ (Lei et al., 2023). For IEX treatment, understanding exchange competition with anions such as chloride, sulphate and bicarbonate will also support the development of models for PFAS removal.

The main PFAS removal mechanisms are driven by hydrophobic interactions for adsorption, and a combination of electrostatic and hydrophobic interactions when using IEX treatment. As such, the octanol-partition coefficient of PFAS can provide a high-level indication of the potential for compound removal and the order of breakthrough for a PFAS compound relative to other PFAS in the water.

While GAC was less efficient in removing shorter chain PFAS, the fact that this treatment process is already frequently present at many surface WTWs, alongside existing infrastructure for regeneration, mean that it is likely to be a front-running option for PFAS treatment. However, the short regeneration frequencies that will be required for treating many PFAS compounds needs to be mapped against national capability for regeneration. This needs to be considered alongside the incurred OPEX when compared against other adsorption treatment processes that are likely to have longer run times. On the other hand, the use of emerging sorbents would need to explore the possibility of retrofitting the media into existing assets, consider waste management options if regeneration is not possible, and have confirmed approval for their use in drinking water treatment systems.

Other operational considerations for adsorption systems includes the management of other process waste streams, such as backwash water resulting from media cleaning cycles. This aspect was not assessed in this study, with more research required to understand whether PFAS are present in this waste stream. This is increasingly relevant where these types of streams are returned upstream of WTWs or discharged to the environment.

Exploration of an increased range of EBCTs with novel and conventional adsorption media is also recommended to ensure that operation is optimised for PFAS treatment, rather than for other micropollutants such as pesticides, which is typically the case for application of GAC. Two-stage adsorption processes could also be considered to reduce competition effects between organic matter and PFAS, with an initial stage designed to target bulk DOC removal, and the second focusing on PFAS removal.

# 6. Membrane separation

Dense membranes processes reject PFAS molecules through a combination of mechanisms including size exclusion, electrostatics and hydrophobic interactions. The predominant mechanism is size exclusion but in the case of smaller molecular size compounds, electrostatic repulsion and hydrophobic interactions also impact performance. The previous bench scale tests revealed that certain RO and NF membranes were effective across all PFAS. In fact, the NF90 membrane, delivered a similar performance to the best RO membrane tested but enabled a higher permeability. Importantly, the NF90 trials demonstrated this membrane to be an effective barrier for all PFAS chain lengths, all functional groups and charges. Removal exceeded 90% for all PFAS with the exception of the 4C PFAS, which recorded removals of 82% for PFBA and 88% for PFBS during the bench scale testing, which used groundwater 1 (DWI, 2025).

Comparison against the different water sources showed lower removals for a number of compounds indicating the importance of the background water chemistry. This lowered the removal of PFBA to around 50% in the case of groundwater 3 and lowland water 1 (DWI, 2025). Reductions were seen for other PFAS molecules, but PFBA represented the lowest removal seen in every trial. In part this was connected to the presence of trivalent ions which in a series of separate experiments were shown to lower PFBA removal to around 50%.

Accordingly, pilot trails were planned for the NF90 membrane against the same three waters as trialled for the sorption processes: groundwater, lowland and upland surface waters. Unfortunately, a suitable NF90 module was unable to be sourced. Discussion with the supplier identified an equivalent membrane, TS80, which was used as an alternative. The aim of the trials was to validate the findings of the bench scale and assess the impact on hydraulic performance.

## 6.1 Methodology

The nanofiltration membrane module was used in a standard test platform (Figure 12). Experiments were run at a set operating pressure of 10 bar using flowrates of 6 and 8 litres per minute (LPM).

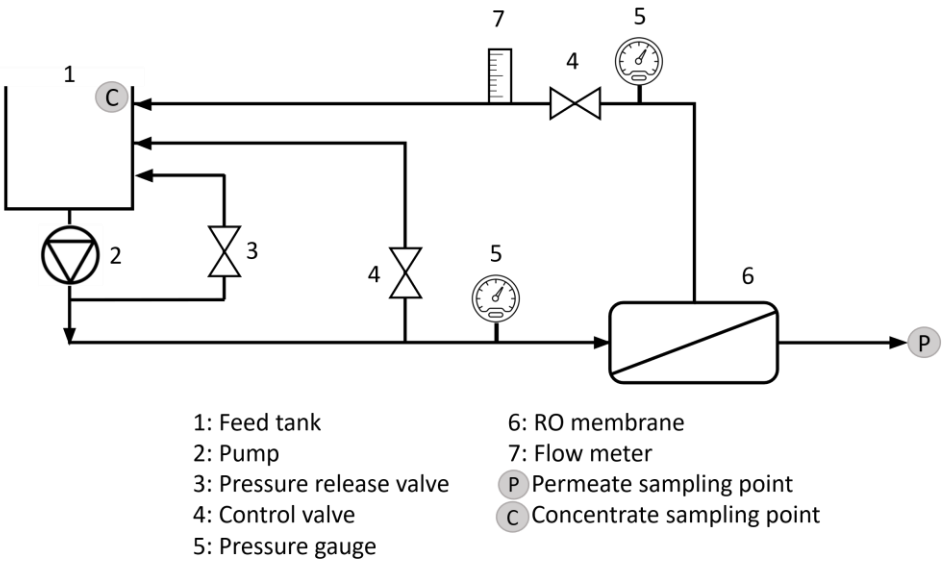


**Figure 12**. Set up for the pilot membrane experiments

Samples were taken at 50%, 70% and 90% recovery points to ascertain the impact of recovery. In addition, the reject line was sampled to ascertain the concentration factors of the different PFAS molecules. Before and after each run, pre and post deionised water tests were carried out to determine the clean water flux ($J_w$) (Equation 2) and the membrane permeability ($L_w$) (Equation 3):

$$J_w = \frac{Q_p}{A} \tag{2}$$

$$L_w = \frac{J_w}{Pf} \tag{3}$$

Where $Q_p$ is the permeate flow, A is the effective area and $P_f$ the feed pressure.

The membrane tested is a TS80 polyamide thin film composite membrane from Trisep with an approximate molecular weight cut off of 150 Da and a salt rejection (NaCl) of 80-90%. This compares to the NF90 membrane with a similar molecular weight cut-off (MWCO) range and a salt rejection of 90-95%. Hence, the result of the pilot trial can be viewed as conservative (Table 4).

**Table 4**. Comparison of the properties of the NF90 and TS80 membranes.

| **Membranes** | **Manufacturer** | **MWCO (Da)** | **Polymer material** | **Chlorine resistance** | **Max temp.** | **pH range** | **Flow GPD** | **NaCl rejection** |
|---|---|---|---|---|---|---|---|---|
| TS80<br>(pilot scale) | Trisep | ~150 Da | Polyamide TFC | <0.1 ppm | 50 C | 3-10 | 20 | 80-90% |
| NF90<br>(bench scale) | Dow/Filmtech | 100-200 | Polyamide TFC | <0.1 ppm | 45 C | 3-10 | 21.6 | 90-95% |

Quality assurance testing of the membrane during the trial revealed consistent salt rejection confirming the suitability of the membrane. Salt rejection varied across the test from 83.5% to 90.3% consistent with the manufacturer's data (Figure 13).

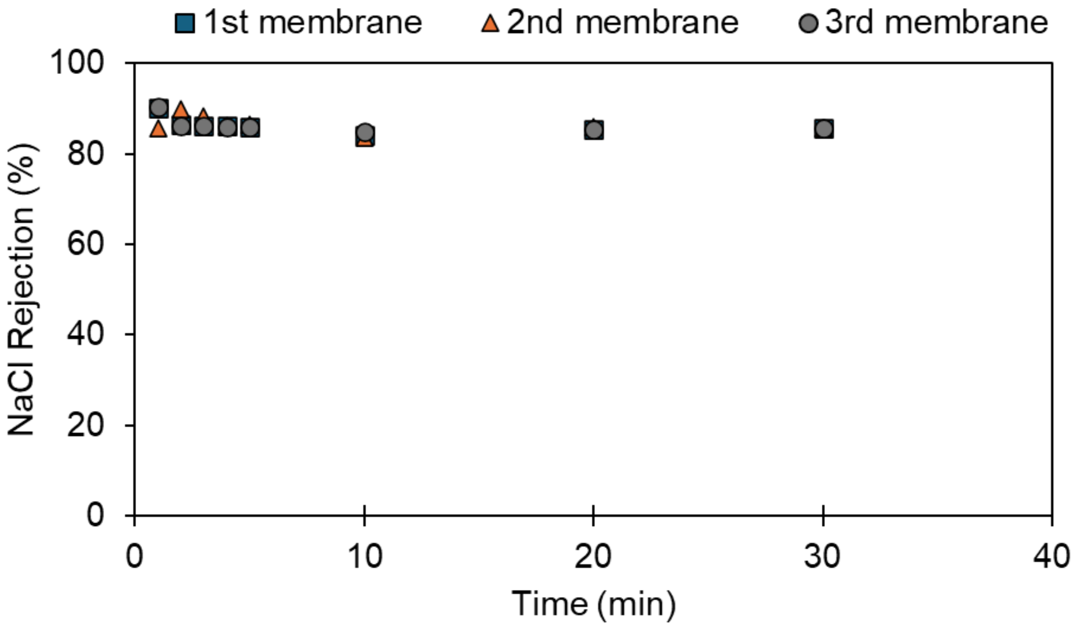


**Figure 13.** Salt rejection of the TS80 membrane.

## 6.2 Membrane treatment results

### 6.2.1 Bench scale test for TS80

The comprehensive bench scale trial of the new membrane (TS80) was conducted using the same equipment as previously used for the NF90 bench scale trials, treating the lowland source water. All PFAS molecules were rejected by 90% or more with the exception of PFBA at 53% and PFBS at 88% (Figure 14). Both are 4C PFAS which are poorly removed by the alternative treatment, showing the enhanced performance of the membrane options. This included 6:2 FTAB which was highlighted to also be difficult to remove by SMC and IEX, and with only GAC offering significant effectiveness. Direct comparison with the previous bench scale testing with the NF90 must proceed with caution as the water sources are different. However, the removal profiles reveal similar performance with rejections above 90% for all but the two 4C PFAS, with removal efficiencies of 52% and 90% for PFBA and PFBS, respectively, when treating the lowland 1 source water (DWI, 2025). This indicates that the two membranes have performed similarly, and confidence can be assigned to the change.

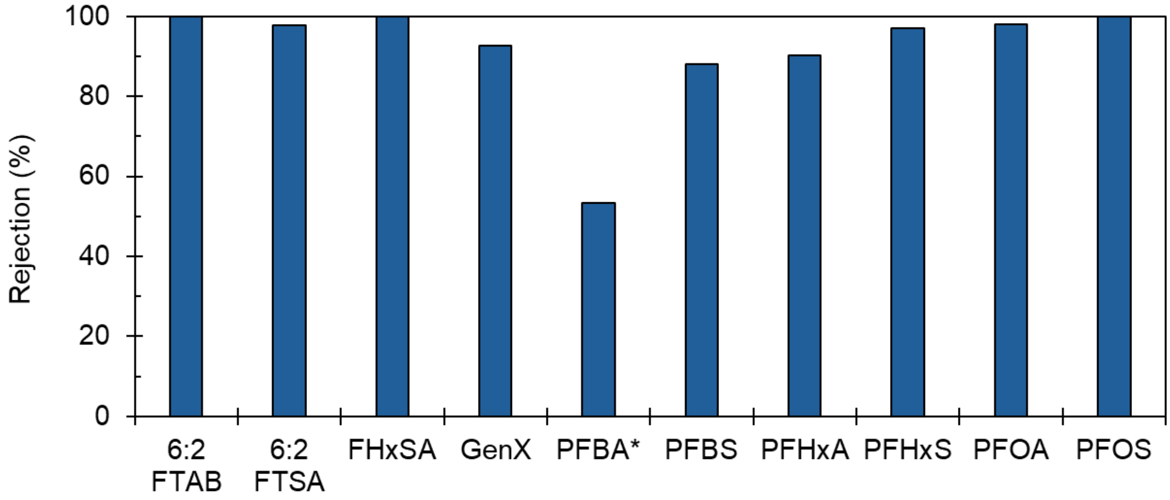


**Figure 14.** Bench scale trial of the TS80 nanofiltration membrane treating the lowland surface water source.

### 6.2.2 Pilot scale trials

Removal of PFAS molecules during the pilot trials confirmed the positive results of the bench scale trials. In fact, the performance of the TS80 membrane exceeded the previous trials with non-detectable levels of 6:2 FTAB, 6:2 FTSA, FHxSA, GenX, PFBS, PFHxA, PFHxS, PFOA and PFOS (Figure 15) during the upland water tests, and 6:2 FTAB, 6:2 FTSA, GenX, PFBS, PFHxA, PFHxS, PFOA and PFOS from the lowland water source (Figure 16). The

PFAS compounds where a residual was detected in the permeate were PFBA, PFBS and PFHxSA. In the case of PFBA, during the upland water trial, the residual was 10 ng/L for PFBA from a feed concentration of 85.6 ng/L, representing an 88.3% rejection. This level of rejection is considerably higher than that observed during the bench scale trials of the TS80 membrane (Figure 14). The same residual was measured for PFBA during the trials on the lowland water source (Figure 16), which was the same water as was trialled during the bench scale tests. In additional, residuals of <5 ng/L for PFHxSA and between 1.1 and 2.3 ng/L for PFBS were observed, representing rejections of greater than 95% and 98% respectively. These results are improved compared to the bench scale trials indicating that the bench scale trails potentially represent a conservative assessment of treatability.

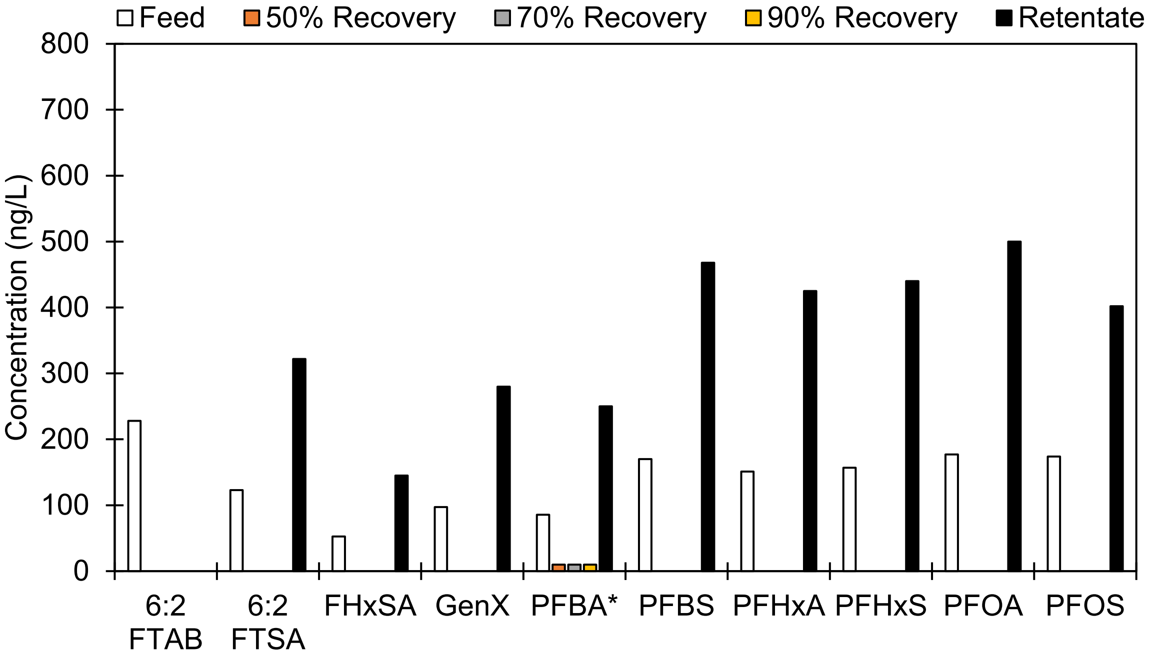


**Figure 15.** PFAS removal from the upland source water using the TS80 nanofiltration membrane.

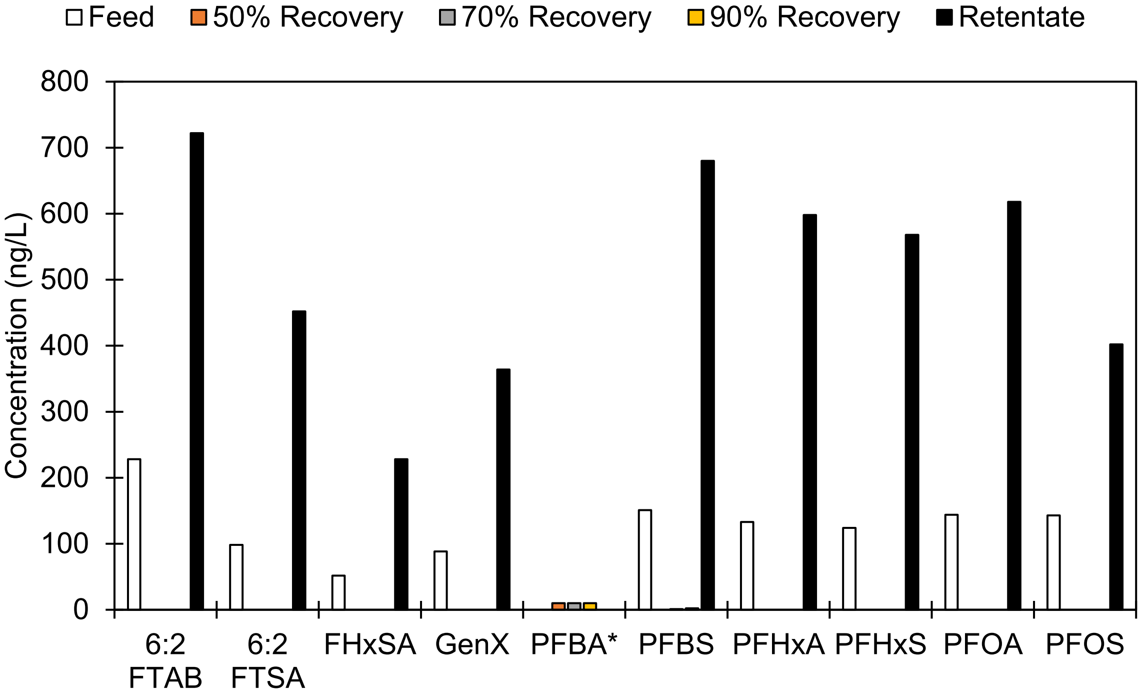


**Figure 16**. PFAS removal from the lowland source water using the TS80 nanofiltration membrane.

The performance was consistent across all recovery levels apart for with PFHxSA and PFBS, where the residual level slightly increased as the recovery level increased. The action of the membrane rejects the PFAS compounds that leaves more concentrated levels in the retentate. Concentration factors of between 2.3 and 2.9 were observed when treating the upland water source and between 2.8 and 4.6 when treating the lowland source water (Figures 15-17). The difference in levels requires investigation but is most likely due to differences in the PFAS retained in the fouling layers. Such suggestions are congruent with the lower concentration factors being associated with the upland water source which contains more hydrophobic organic matter compounds. These are known to be able to complex with PFAS. Further work is required to properly understand the fate of PFAS during membrane treatment and the respective accumulation of PFAS within fouling layers, which are then removed through chemical cleaning.

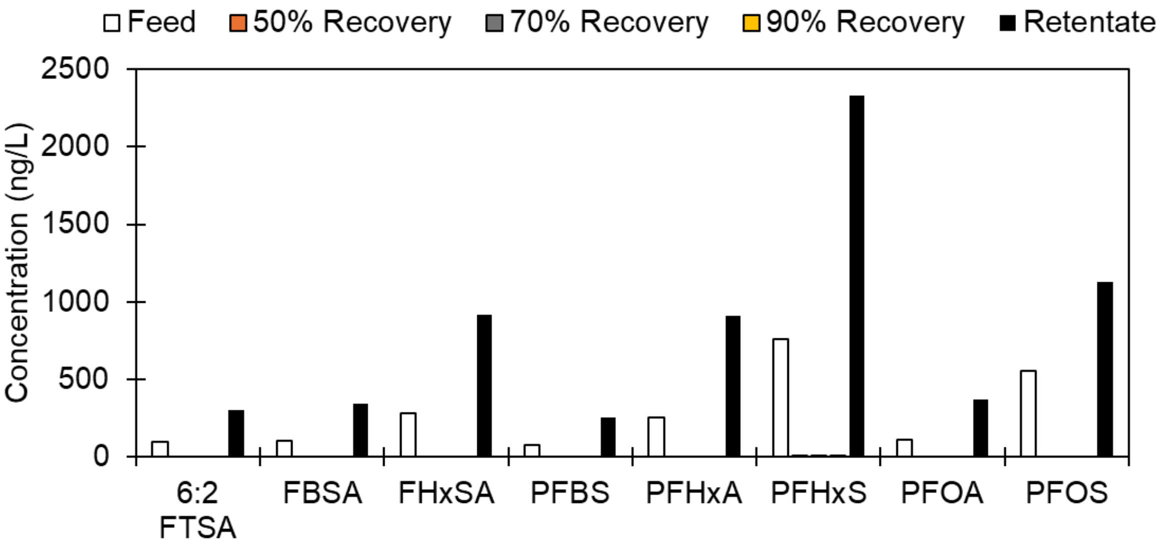


**Figure 17.** PFAS removal from the groundwater source using the TS80 nanofiltration membrane.

The performance of the nanofiltration membrane in treating the groundwater (Figure 17) was based on the environmental levels of PFAS present in the water. The water did not contain measurable levels of 6:2 FTAB, GenX or PFBA, so these compounds cannot be compared. 6:2 FTSA, PFBS and PFOA were at lower concentrations than in the upland and lowland source spiked waters, with PFBS at 77 ng/L compared to 170 ng/L and 151 ng/L for the upland and lowland spiked source water, respectively. Further, FHxSA, PFHxA, PFHxS and PFOS were at higher concentrations with an increase of 168% for PFHxA, 317% for PFOS, 484% for PFHxS and 530% for FHxSA compared to the highest in either of the other two source waters.

Irrespective of the higher concentrations, the TS80 NF membrane completely rejected all PFAS molecules with the exception of PFHxS, with a maximum residual of 4.4 ng/L, representing 99% rejection. The performance was once again independent of recovery levels and led to concentration factors of between 1.8 and 3.6. The reported levels sit between those of the upland and lowland spiked water sources. The comparative concentration factors do not mirror the reported DOC in the feed water, further indicating the importance of the fouling layer and its organic make up in the fate of the PFAS molecules.

### 6.2.3 Hydraulics

The operation of the membrane system generated pseudo stable flux profiles with flux levels of 35.8 and 43.5 litres per square metre per hour (LMH) when treating the upland source water (Figure 18a), 55.0 and 52.1 LMH for the lowland source water (Figure 18b) and 29.3 and 36.3 LMH for the groundwater source (Figure 18c). The differences between source water will be based on the differences between concentration and character of the organic and inorganic compounds in the background water. To illustrate, the measured flux was higher for the lowland source water compared to the upland source water despite the DOC of the lowland source water being 4.1 mg/L compared to 2.1 mg/L for the upland source water. The corresponding permeability is between 3.6 and 4.4 LMH/bar when treating the upland water source, 5.2 and 5.5 LMH/bar when treating the lowland water source and 2.9 and 3.6 when treating the groundwater source. This compared to the manufacturers quoted clean water permeability of 5.2 LMH/bar, indicating that the lowland water source has generated minimal loss in permeability, congruent with a very high concentration factor. The other two sources water generated a drop in permeability of 69% and 56% for the upland and groundwater sources respectively.

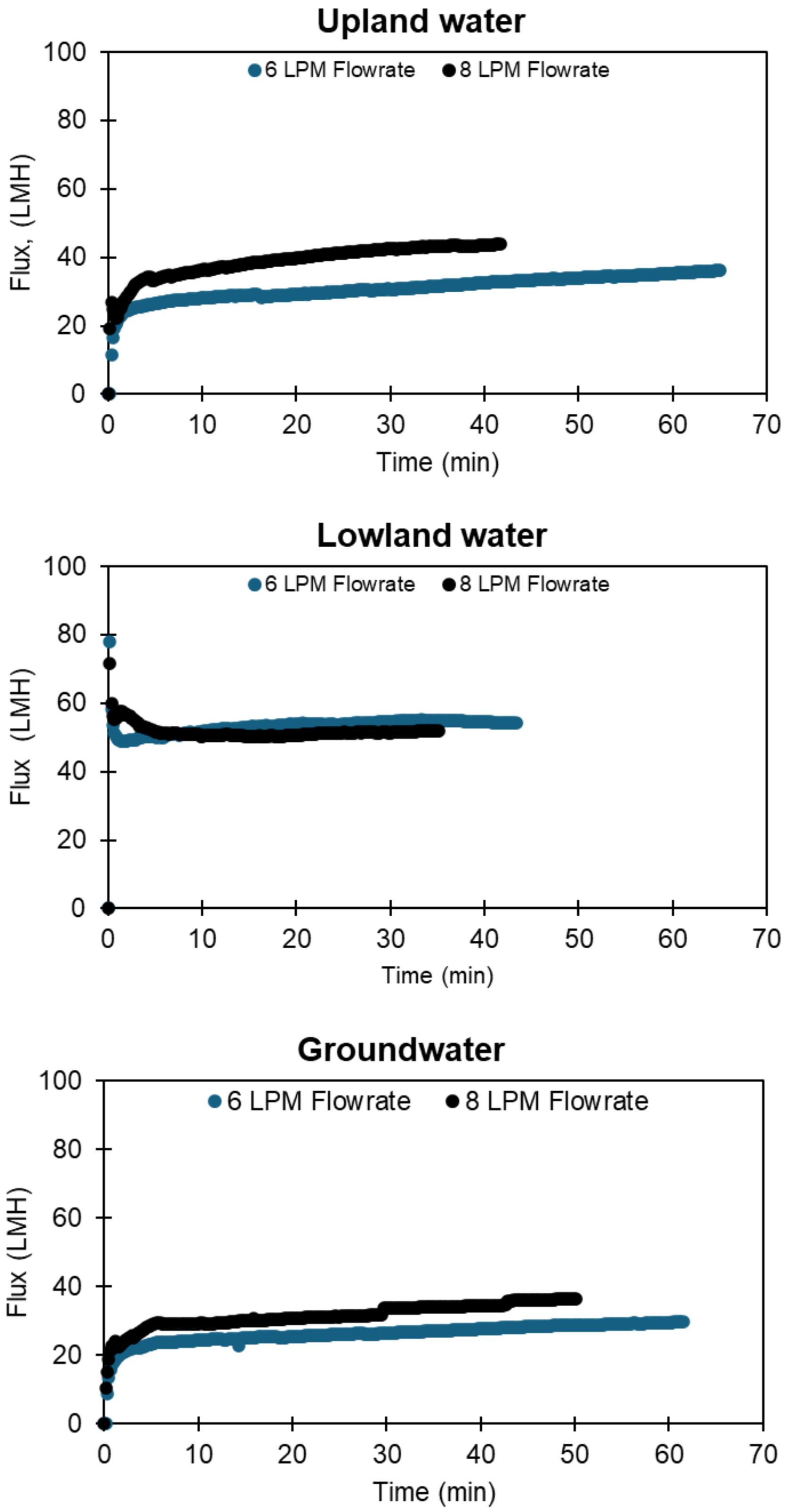


**Figure 18**. Hydraulic performance for the TS80 membrane at 6 and 8 LPM against the upland, lowland and groundwater sources.

## 6.3 Summary and key findings for membrane tests

The pilot trial of the TS80 NF membrane against the three source waters confirmed previous bench scale data and literature in the overall efficacy of the NF membrane. Importantly, the membrane was effective for PFAS molecules where the sorption processes showed lower efficiency, particularly for the 4C PFAS compounds (PFBA and PFBS) as well as 6:2 FTAB. Further, the ability to remove the PFAS molecules from the water was independent of the source water type or the level of recovery. Ultimately, the work indicates the universal applicability of the NF membrane for PFAS removal and represents a good option for source waters rich in PFAS molecules that are more difficult to remove using sorption media or produce short operational cycles.

Whilst the performance data is encouraging, it should be noted that the trials were short term in nature and did not cover multiple operating cycles including the required fouling amelioration actions, such as chemical cleaning. As a consequence, the fate of the PFAS molecules appears source water dependent along with the operational flux achievable. The latter can be enhanced through the use of ceramic NF membranes, which are now becoming available. Current research on drinking water application indicates ceramic membranes can operate at 3-4 times the flux for an equivalent molecular weight cutoff and pressure. The material can also afford more intense cleaning if needed and have considerably longer lifetime. Together these attributes transform the whole life cost of the technology and hence its relative position compared to the alternatives.

# 7. Conclusions and future research

This report presents the findings from a 9-month pilot scale adsorption trial comparing the performance of GAC, SMC and IEX for the removal of PFAS from groundwater, upland and lowland waters. The report also includes the results of NF membrane trials conducted using the same water sources. The key outcomes from the trials are summarised as follows:

- In agreement with other studies and bench scale tests previously performed (DWI, 2025), the results from the adsorption trials showed that most long chain PFAS were more efficiently removed than the short chain compounds.
- Sulphonated PFAS were more easily removed through adsorption than the corresponding carboxylic acid compounds.
- The short chain carboxylic acid PFAS were the most challenging to remove, regardless of the water type. In particular, PFBA was not efficiently treated by any of the media tested in this trial.
- Most PFAS compounds were removed for longer throughputs by IEX, followed by SMC and GAC. The exception was fluorotelomer 6:2 FTAB due to the positive charge in its structure, and which was best removed by GAC.
- SMC showed breakthrough profiles that could be associated with desorption effects for certain PFAS such as 6:2 FTAB and GenX.
- Most PFAS were preferentially removed by the media from groundwater > upland water > lowland water. It is postulated that this is partly due to the complex competition and interaction effects between PFAS and background organic matter during adsorption.
- The main PFAS removal mechanisms are driven by hydrophobic interactions for adsorption, and a combination of electrostatic and hydrophobic interactions for IEX.
- NF membrane results confirmed previous bench scale data and literature in the overall efficacy of the NF treatment to remove PFAS. Very high removal of all PFAS was observed during the pilot scale trials.
- Although the NF membrane rejection was not as high for 4C PFAS compounds as for longer chain ones, it still provided more efficient separation against them than sorption processes. NF separation was also an effective barrier against 6:2 FTAB.

The higher efficiency of GAC for longer chain PFAS, and that of IEX for shorter chain PFAS, highlights the merit of investigating the combination of both processes to effectively target a broad range of PFAS in water. Further research into the impact of EBCTs on PFAS removal will aid in optimising both new and existing infrastructure. A deeper understanding of how background organic matter and inorganic ions influence the performance of adsorption and membrane processes is essential to accurately assess their effectiveness against PFAS. Long-term membrane trials, including the use of ceramic membranes, are necessary to establish performance sustainability and cleaning requirements, thereby supporting decision making on the implementation and operation of membranes for PFAS removal.

Future research should also consider waste management strategies for both liquid and solid waste from adsorption and membrane processes, as well as potential regeneration options for novel sorbents. This would guarantee a pathway towards PFAS destruction rather than only separation.

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