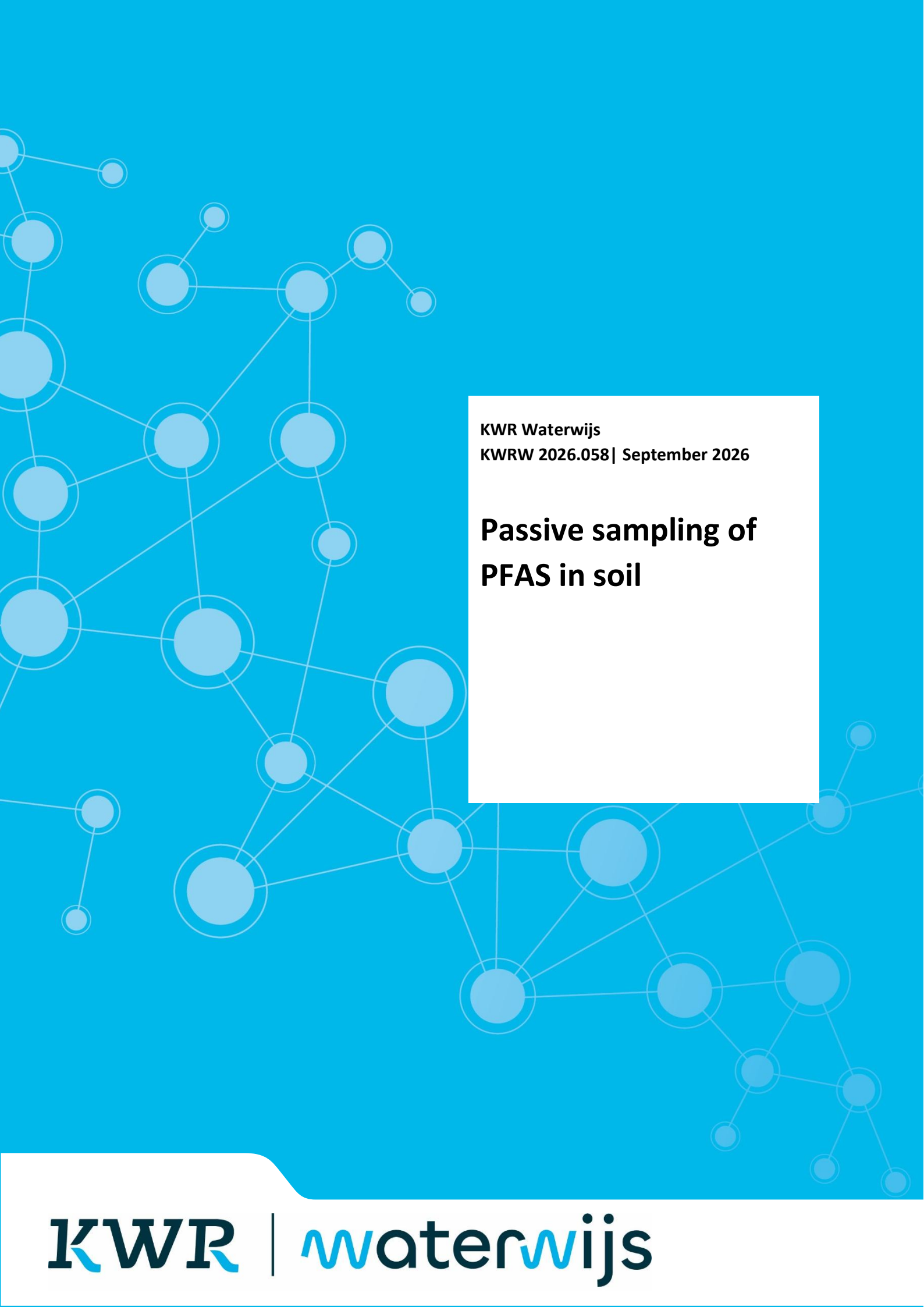




KWR Waterwijs

KWRW 2026.058 | September 2026

Passive sampling of PFAS in soil

Colophon

Passive sampling of PFAS in soil

This research was carried out as part of KWR Waterwijs, the collective drinking water research programme of KWR, the water companies and Vewin.

KWRW 2026.058 | September 2026

Project number

404300/031

Project manager

Renske Hoondert, PhD

Client

KWR Waterwijs – Kraamkamer

Author(s)

Elvio Amato, PhD

Quality Assurance

Thomas ter Laak, PhD

Sent to

This report has been distributed to participants in the KWR Waterwijs programme.

Publicity

This report is public.

Keywords

DGT; mobility; porewater; soil/water partition coefficient

Year of publishing

2026

More information

Dr. Elvio Amato
T 030-6069754
E elvio.amato@kwrwater.nl

PO Box 1072
3430 BB Nieuwegein
The Netherlands

T +31 (0)30 60 69 511
E info@kwrwater.nl
I www.kwrwater.nl



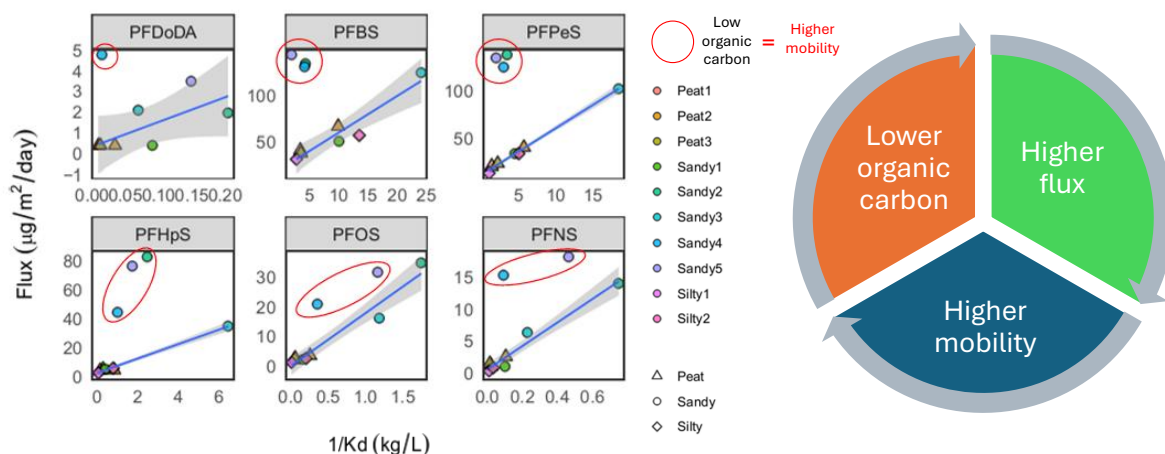
September 2026 ©

All rights reserved by the KWR Waterwijs participants. No part of this publication may be reproduced, stored in an automatic database, or transmitted in any form or by any means, be it electronic, mechanical, by photocopying, recording, or otherwise, without the prior written permission of KWR.

Samenvatting

Passive sampling voorspelt nauwkeurig de mobiliteit van PFAS in de bodem

Auteurs Elvio Amato



Aanleiding: Bodemverontreiniging vormt een risico voor drinkwater

Verontreinigingen in bodem (en sediment) kunnen de (drink)waterkwaliteit beïnvloeden doordat deeltjesgebonden verontreinigingen vrijkomen in de waterfase. Het beoordelen van de mobiliteit van verbindingen is belangrijk voor het beschermen van drinkwaterbronnen, vooral voor verbindingen die niet afbreken en dus langere afstanden in de bodem kunnen afleggen, zoals per- en polyfluoralkylstoffen (PFAS). De bodem-waterverdelingscoëfficiënt (K_d) is een belangrijke parameter om de mobiliteit te beoordelen. De sorptie van verontreinigende stoffen aan bodem wordt beïnvloed door factoren zoals organische koolstofgehalte, ijzeroxiden/hydroxidengehalte in bodem en de pH en deeltjesgrootte van de bodem of het sediment. Voor een nauwkeurige beoordeling van de mobiliteit van verontreinigingen moeten al deze factoren in beschouwd worden. Dit is arbeidsintensief (en kostbaar) omdat het verschillende manieren van monstervoorbewerking en meerdere analyses per bodemonmonster vergt.

Passive sampling is een alternatieve methode om verontreinigingen en hun mobiliteit in bodem en sediment te monitoren. Deze aanpak maakt gebruik van materialen die *in situ* verontreinigingen opnemen uit poriewater. De gemeten flux uitgedrukt in massa van de betreffende verontreiniging per oppervlak van het materiaal per dag ($\mu\text{g}/\text{m}^2/\text{dag}$) weerspiegelt de poriewaterconcentratie of de snelheid waarmee de stoffen vrijkomen uit de bodem. Deze poriewaterconcentratie bepaalt in belangrijke mate de mobiliteit en biologische beschikbaarheid van contaminanten in bodems (of sedimenten).

Aanpak: Vergelijking van bodem-waterverdelingscoëfficiënten van verschillende bodemtypen

Een set van 10 bodems met verschillende fysisch-chemische eigenschappen, zoals pH, deeltjesgrootte en het gehalte aan organische koolstof (OC), werd geselecteerd en blootgesteld aan 23 PFAS. Met behulp van passive sampling werden diverse PFAS uit de bodem geïsoleerd in een laboratoriumopstelling. De selectie van bodems was bedoeld om de verwachte variabiliteit in de mobiliteit van PFAS in relatie met de verschillende bodemkenmerken te bepalen. Vervolgens werden de met behulp van passive sampling verkregen resultaten vergeleken met de bodem-waterverdelingscoëfficiënt van de verschillende PFAS aan de verschillende bodems.

Resultaten: Passieve bemonstering voorspelt de mobiliteit van PFAS in de bodem

Poriewaterconcentraties van PFAS waren consistent hoger in zandbodems. Zandbodems hebben gemiddeld grotere deeltjes dan andere typen bodems. Dit illustreert de rol van deeltjesgrootte op de verdeling van verontreinigingen tussen bodem en poriewater. PFAS met een langere koolstofketens bonden sterker aan bodemdeeltjes dan verbindingen met een kortere koolstofketens, die meer in de waterfase bleven. Dit is in overeenstemming met hun toenemende hydrofobiciteit. De K_d varieerde van 0,15-2,36 L/kg in zandbodems, van 1,19-6,82 L/kg in veenbodems en van 0,40-0,94 L/kg in slibbodems. Deze waarden komen overeen met de in de literatuur gerapporteerde waarden. De hoogste mediane K_d gemeten in zandige bodems werd gevonden in het monster met de hoogste organisch koolstof-gehalte, wat laat zien dat het organisch materiaal een belangrijke sorptiefase voor PFAS is. De hoogste K_d -waarden werden over het algemeen waargenomen in veenbodems. Veenbodems bestaan vrijwel uitsluitend uit organisch materiaal.

Passive sampling met behulp van 'diffusive gradients in thin film' (DGT) apparaten correleerden sterk negatief met de K_d in verschillende bodemtypen. De Pearson's correlatiecoëfficiënten vielen gewoonlijk tussen 0,705 en 0,998. Dit komt overeen met het feit dat poriewaterconcentraties omgekeerd evenredig zijn aan de K_d . Uitzonderingen werden waargenomen voor zandige bodems met een laag OC-gehalte, waar de DGT-fluxen significant hoger waren dan op basis van K_d werd verwacht. Dit werd veroorzaakt door de hogere mobiliteit van PFAS in deze bodems als gevolg van het lage OC-gehalte, aangezien OC een belangrijke bindingsfase voor PFAS vormt. Dit wijst erop dat DGT de mobiliteit van PFAS in de bodem nauwkeurig weerspiegelt.

Passive sampling mat effectief fluxen van een reeks PFAS-verbindingen in diverse bodemtypen (waaronder zandige bodems, slibbodems en veen). De correlatie tussen gemeten fluxen en bodem-watervedelingscoëfficiënten (K_d) toont aan dat deze techniek waardevol is bij de beoordeling van mobiliteit, sorptie en biologische beschikbaarheid van PFAS in bodems. In tegenstelling tot poriewateranalyse werden DGT-metingen niet beïnvloed door matrixeffecten zoals de aanwezigheid van gesuspenseerd of opgelost organisch materiaal in het poriewater.

Vervolg: Verdere validatie

Toekomstige studies moeten zich richten op de verdere validatie van de passieve bemonsteringsmethode (inclusief de beoordeling van detectiegrenzen en extractierendementen) en toepassingen van deze methode in verontreinigde veldbodems en -sedimenten. Ook moeten overeenkomsten en verschillen met conventionele methoden verder worden onderzocht. Bovendien kan de methode worden getest voor een breder scala aan verontreinigingen en nutriënten dan in deze studie.

Implementatie: Toepassingen in de drinkwatersector

Het beoordelen van de mobiliteit van verontreinigingen in bodem en sediment is essentieel voor het evalueren van de verspreiding van de verontreinigingen in de bodem, de blootstelling van bodemorganismen en ook de potentiële risico's van deze verontreinigingen voor de drinkwaterproductie. Dit is met name relevant voor processen zoals duin- en oeverfiltratie, waarbij stoffen aan het bodemmateriaal binden maar ook weer vrij kunnen komen in het poriewater. Daarnaast is het ook relevant voor het opsporen van verontreinigingen die worden geïntroduceerd door atmosferische depositie (bijv. pesticiden of PFAS), het illegaal dumpen van chemisch afval van bijvoorbeeld de productie van drugs maar ook voor bodems die (in het verleden) zijn belast met bijvoorbeeld PFAS houdend brandblusschuim. Andere toepassingen zijn het beoordelen van de mobiliteit van verontreinigende stoffen in landbouwgrond en het monitoren van het vrijkomen van verontreinigende stoffen uit (historisch) vervuilde sedimenten naar oppervlaktewateren. Procedures voor de bereiding en analyse van passieve monstername-extracten zijn relatief eenvoudig en geschikt voor toepassing in drinkwaterlaboratoria.

Maatschappelijke impact: Menselijke gezondheid en natuurbehoud

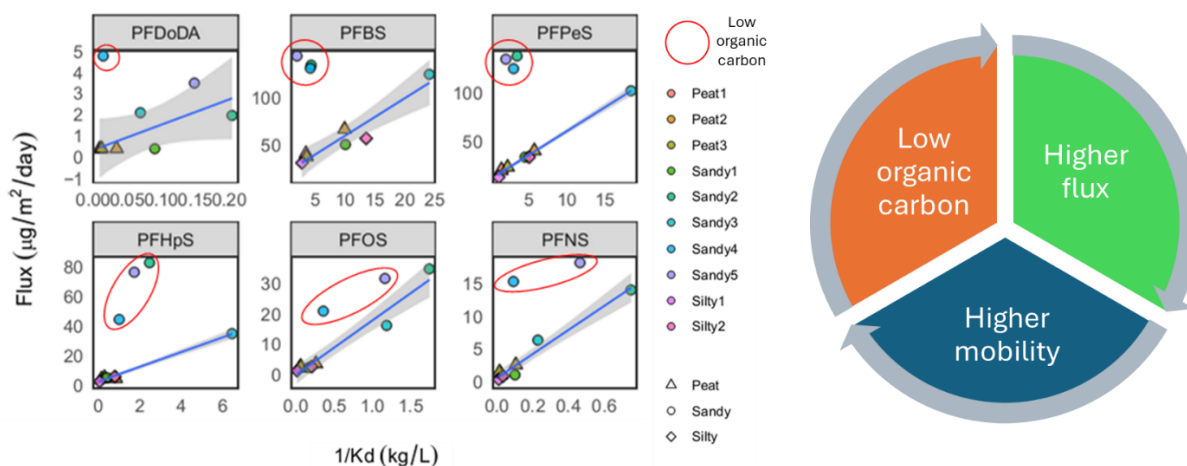
Passieve bemonstering biedt een eenvoudige methode voor het screenen van de mobiliteit van verontreinigende stoffen in verschillende milieucompartimenten, waaronder bodem en sediment. Dit maakt het mogelijk om de mobiliteit van de verontreiniging te monitoren en drinkwaterbronnen te beschermen. Daarnaast is het vermogen om

rekening te houden met de mobiliteit van verontreinigende stoffen bijzonder waardevol voor het beoordelen van biologische beschikbaarheid. Dit is een sleutelfactor zijn bij het begrijpen van hun potentiële impact op biota en het milieu.

Summary

Passive sampling accurately predicts PFAS mobility in soils

Author: Elvio Amato



Context: Soil contamination poses a risk to drinking water

Contaminants in soil and sediment can impact drinking water quality by releasing particle-bound pollutants into the water phase. The soil–water partition coefficient (K_d) is a key parameter used to assess this behaviour. Contaminant sorption is influenced by factors such as organic carbon, iron oxides/hydroxides, pH, and particle size. Accurate assessment of contaminant mobility requires considering all these factors, which involve several analyses that increase costs.

Passive sampling offers an alternative method for monitoring contaminants and their mobility in soil and sediment. This approach uses *in situ* devices that preconcentrate contaminants from porewater. The measured flux ($\mu\text{g}/\text{m}^2/\text{day}$) reflects both porewater concentrations and desorption kinetics, which control the equilibrium between solid and aqueous phases. Thus, passive sampling fluxes account for the same factors that influence contaminant mobility and bioavailability in soils and sediments.

Approach: Comparison with K_d over a wide range of soils

A set of 10 soils with varying physicochemical properties—such as pH, particle size, and organic carbon (OC) content—was selected to evaluate passive sampling across a broad range of conditions. This selection was intended to capture the expected variability in PFAS mobility in response to different soil characteristics. Passive sampling measurements were compared to K_d .

Results: Passive sampling predicts PFAS mobility in soil

Porewater concentrations of PFAS were consistently higher in sandy soils, emphasizing the role of particle size in contaminant partitioning. Longer-chain PFAS associated with soil particles to a greater extent than shorter-chain compounds, which remained more in the aqueous phase.

K_d varied across soil types, ranging from 0.15–2.36 L/kg in sandy soils, 1.19–6.82 L/kg in peat, and 0.40–0.94 L/kg in silty soils. These values align with ranges reported in the literature. K_d in silty soils were likely underestimated due to matrix effects affecting the quantification of hydrophobic compounds, for which coefficients could not be calculated.

K_d increased with increasing chain lengths, consistently with their greater hydrophobicity. The highest median K_d measured in sandy soils was found in the sample with the highest level OC, highlighting its influence on PFAS partitioning. The highest K_d values were generally observed in peats, reflecting their high OC content.

Passive sampling fluxes showed trends that generally matched porewater concentrations and strongly correlated with the inverse of K_d across various soil types, with Pearson's correlation coefficients typically ranging between 0.705 and 0.998. Exceptions were observed for sandy soils with low OC, where the DGT fluxes were significantly higher than those expected based on K_d . This was due to the higher mobility of PFAS in these soils, resulting from the low OC content — a strong binding phase for PFAS. This indicated that DGT accurately reflects the mobility of PFAS in the soil.

Passive sampling effectively measured PFAS fluxes across a range of compounds and soil types (including sandy soils, silty soils and peat), demonstrating its value in assessing PFAS mobility, partitioning, and bioavailability in soils and sediments. Unlike porewater analysis, DGT measurements were unaffected by matrix effects.

Follow-up: Further validation

Future studies should include validation of the passive sampling method and replicate analyses to account for variability. Comparisons with conventional methodologies should be further investigated, particularly in applications relevant to the drinking water sector. The applicability of this method can be extended to include a broader range of pollutants and nutrients.

Implementation: Applications in the drinking water sector

Assessing contaminant mobility in soil and sediment is essential for evaluating potential risks to drinking water production. This is particularly relevant for tracking and assessing the mobility of contaminants introduced by different sources, such as atmospheric deposition (e.g., pesticides, PFAS), illegal waste disposal, and aqueous film-forming foams (AFFF). Procedures for the preparation and analysis of passive sampling extracts are relatively simple and suitable for implementation at drinking water laboratories.

Social impact: Human health and nature conservation

Passive sampling provides a straightforward method for screening pollutants across various environmental compartments, including soil and sediment. This allows to monitor contamination and protect drinking water sources. Its ability to account for the chemical speciation of pollutants is particularly valuable for assessing bioavailability and toxicity, which are key factors in understanding their potential impact on biota and the environment.

Content

Colophon	2
Samenvatting	3
Summary	6
Content	8
1 Context	9
2 Research approach	10
2.1 Experimental set-up	10
2.2 Preparation of soil samples	10
2.3 Passive sampling deployment	10
2.4 Sample preparation and extraction	10
2.5 Instrumental analysis	11
2.6 Data analysis	11
3 Results	13
3.1 Soil physicochemical properties	13
3.2 Analytical interferences	13
3.3 PFAS in particulate soil	14
3.3.1 Sandy soils	Error! Bookmark not defined.
3.3.2 Peat and silty soils	Error! Bookmark not defined.
3.4 PFAS in porewater	Error! Bookmark not defined.
3.5 Soil–porewater partition coefficients (K_d)	14
3.6 DGT flux	16
3.7 DGT porewater concentrations	Error! Bookmark not defined.
4 Conclusions	23
4.1 Recommendations for follow-up	23
4.2 Application by drinking water companies	23
4.3 Relevance to other stakeholders and societal impact	24
I Appendix	27

1 Context

Contamination in sediment and soil may affect the quality of drinking water sources due to the release of particulate-bound pollutants. The extent to which these pollutants enter the water phase depends on their chemical speciation and mobility. Soil–water partition coefficients (K_d) are commonly used to investigate the fate and behaviour of contaminants in soil and sediment. This parameter is crucial for understanding the exchange between soil particles and water and predicting the risk of contamination to drinking water sources such as groundwater and surface water.

The sorption behaviour of contaminants in soil and sediment is influenced by parameters such as organic carbon (OC), iron oxides/hydroxides, pH and particle size (Agbaogun and Fischer, 2020; Fabregat-Palau et al., 2025; Hubert et al., 2023; Nguyen et al., 2020). These parameters affect the partition and exchange of contaminants to different extents: for instance, OC binds strongly with both organic compounds and cationic metals, reducing their mobility; in contrast, iron oxides/hydroxides provide weaker interactions with organic compounds and have a limited effect on their mobility. The pH affects the surface charge of these binding phases and the speciation of ionizable contaminants, thereby influencing the interactions between soil and contaminants. Particle size is also a very important factor regulating contaminant partition, as particles with a smaller size offer more binding surface per volume compared to larger particles. However, while K_d accounts for the contribution of these parameters, it only describes the distribution of contaminants between soil and water at the equilibrium. As a result, K_d poorly predicts the behaviour and mobility of contaminants under dynamic environmental conditions, requiring OC normalization to provide more accurate predictions of mobility.

An alternative approach for monitoring contaminants and their mobility in soil and sediment is provided by passive sampling. This approach uses *in situ* devices to preconcentrate contaminants dissolved in the porewater (i.e., interstitial water). The flux ($\mu\text{g}/\text{m}^2/\text{day}$) of contaminants from the soil–sediment to the device is influenced by the same desorption kinetics that regulate the equilibrium partition between aqueous and solid phases. As a result, passive sampling fluxes account for the same parameters that affect contaminants mobility (and bioavailability) in soils and sediments. Because passive sampling introduces a disturbance in the soil (i.e., it removes contaminants from the porewater), it creates dynamic conditions that better represent environmental systems.

Passive sampling has been widely used for measuring metals and hydrophobic organic compounds (e.g., polycyclic aromatic hydrocarbons (PAH) and polychlorobiphenyls (PCB)) in sediment/soil, however, measurements of PFAS have been rarely reported. PFAS pose a significant threat to drinking water sources due to their toxic effects to humans, poor degradation in the environment, and challenging removal from drinking water sources (Hofman-Caris, 2024; Veggel and Hofman-Caris, 2024). PFAS contamination in soil is frequently reported and occurs as a result of atmospheric deposition (Amato et al., 2024; Amato and van der Grift, 2024) as well as direct input from anthropogenic activities such as aqueous film-forming foam (AFFF) training (Barzen-Hanson et al., 2017b) or disposal of waste containing PFAS residues. The aim of this project is to evaluate the suitability of passive sampling as a tool for investigating PFAS mobility in soils. Passive sampling fluxes are expected to provide useful complementary information to better understand the behaviour of PFAS in soils (and sediments) and their potential impact on drinking water sources.

2 Research approach

2.1 Experimental set-up

A set of 10 soils with different physicochemical properties was selected for this test. Soils were previously characterised for pH, particle size, and organic carbon (OC) content. This selection was made to test passive sampling over a wide range of different soils, where PFAS mobility is expected to vary in response to different physicochemical parameters.

2.2 Preparation of soil samples

Soils (50 g, except for Sandy2 (100 g) and Peat3 (19 g)) were placed in plastic containers (polypropylene, 100 mL), wetted with ultrapure water (0–20 mL), and allowed to equilibrate overnight. Water saturation is required for sample homogenisation and passive sampling deployment. This was ensured by adding water to the soil and removing the excess of water from the soil surface. Samples included five soils characterized by coarse particles (Sandy1–5), three with fine particles (Silty1–2), and three rich in total organic carbon (Peat1–3). After one day, the excess water accumulated at the surface of the soil was removed. Each soil was spiked with a standard solution (2 mg/L, 300 µL) containing a mixture of 23 PFAS in methanol, to achieve a nominal concentration of approximately 0.01 µg/g (except for Peat3 and Sandy2 which had a concentration of approximately 0.02 and 0.005 µg/g, respectively). Soils were stirred with a clean metal spatula and transferred on a shaking plate placed in an incubator set to 20 °C. Shaking was performed for 7 days prior to the deployment of passive sampling devices. This step could be omitted because soils were rather firm, and shaking did not contribute much to the mixing; most of the homogenization resulted from spatula mixing.

2.3 Passive sampling deployment

Diffusive gradients in thin films (DGT) devices, loaded with a WAX agarose binding gel (0.5 mm thickness), an agarose diffusive gel (0.8 mm thickness), and a polyethersulphone filter membrane (0.14 mm thickness), were purchased from DGT Research (dgtresearch.com). A small amount of soil was smeared on the sampling window before gently pressing the probe into the soil, ensuring that the sampling window faced downward. Containers were closed with a lid to avoid water evaporation. Samples were placed in an incubator set to 20 °C for 24 h. No shaking was applied. Upon retrieval, DGT devices were rinsed with ultrapure water, placed in plastic bags, and stored refrigerated (5 ± 3 °C) until analysis.

2.4 Sample preparation and extraction

Soil sub-samples (20–40 g) were centrifuged in a plastic tube (50 mL) at 3000 rpm for 10 min to separate the solid phase from the porewater. The supernatant was transferred to a clean tube and store refrigerated (5 ± 3 °C) until analysis. A subsample of soil (1 g) was transferred to a clean plastic tube (15 mL) and spiked with 100 µL of internal standard (IS). The soil was extracted using 5 mL of methanol (0.25 % NH₄OH), followed by vortex for 1 min and sonication for 5 min. The sample was centrifuged for 10 min (3000 rpm) and the extract transferred to a glass tube. The extraction was repeated one more time and the extracts combined in the same tube. The extract was reduced to 0.5 mL under a gentle flow of nitrogen on an evaporation unit heated at 300 °C, resulting in a gas temperature of approximately 70–80 °C. The sample was diluted to 1 mL using ultrapure water, filtered (0.45 µm), and stored frozen (–20 °C) until analysis.

Porewater samples (0.8–7.7 mL) were spiked with internal standard (IS), evaporated to 0.5 mL under a gentle flow of nitrogen (on a unit heated to 300 °C, resulting in a gas temperature of 70–80 °C), and brought to 1 mL using methanol. Samples were filtered (0.45 µm) and stored frozen (-20 °C) until analysis.

DGT binding gels were transferred to plastic tubes (15 mL), spiked with IS, and extracted with 3 mL of methanol (0.25 % NH₄OH). Samples were vortexed for 1 min and sonicated for 5 min. The extract was transferred to a clean glass tube, and the extraction repeated one more time. The extracts were combined and evaporated to 0.5 mL under a gentle flow of nitrogen (on a unit heated to 300 °C). The sample was brought to 1 mL using ultrapure water, filtered (0.45 µm), and stored frozen (-20 °C) until analysis. Three blank probes (i.e., non-exposed DGT devices) were included in the procedure.

2.5 Instrumental analysis

PFAS concentrations in all extracts and water samples were determined using liquid chromatography coupled with high resolution mass spectrometry (LC-HRMS). Chromatographic separations were achieved using a Vanquish ultrahigh-performance liquid chromatography system (ThermoFisher Scientific, Breda, the Netherlands) equipped with an XBridge® BEH C18 (2.5 µm, 2.1 x 100 mm Column XP, Waters), and the mass spectrometer used was an Orbitrap FusionTMMass (Thermo Fisher Scientific). The injection volume and the flow rate were 10 µL and 0.3 mL/min, respectively. Initial eluent composition was A: 5 mM NH₄Ac in UP-water and B: 5 mM NH₄Ac in MeOH. Calibration standards were prepared in 50:50 MeOH:ultrapure-water %(v/v) at concentration levels of 0, 0.25, 0.50, 1, 2.5, 10, 25, 50 µg/L.

2.6 Data analysis

Soil/water partition coefficients (K_d , L/kg) were calculated using

$$K_d = \frac{C_{ps}}{C_{pw}} \quad \text{Eq. 1}$$

Where C_{ps} and C_{pw} are the concentrations of PFAS in the solid and aqueous (i.e., porewater) phases of the soil, respectively. DGT porewater concentrations (C_{DGT} , µg/L) were estimated using

$$C_{DGT} = \frac{M \Delta g}{DA t} \quad \text{Eq. 2}$$

Where M is the mass of PFAS measured in the binding gel (µg), Δg is the thickness of the diffusive gel layer (m), D is the diffusion coefficient of the PFAS (m²/s), A is the exposed surface area of the device (m²), and t is the deployment time (s). C_{DGT} concentrations are expressed as µg/L using an appropriate conversion factor. Temperature-dependent diffusion coefficients were taken from the literature (dgtresearch.com) based on the soil temperature recorded during the deployment. The DGT flux (µg/m²/day) was measured using

$$Flux = \frac{M}{At} \quad \text{Eq. 3}$$

Concentrations below the limit of quantification (LOQ) were set to half the LOQ (0.25 / 2 = 0.125 µg/L). When the IS did not match the set criteria, concentrations were not reported.

2.7 DGT theory in soils

In saturated soils, porewater concentrations are influenced by the equilibrium partition between the aqueous and solid phases of the soil. DGT accumulates compounds from the porewater altering the equilibrium between porewater and particulate soil. When weak binding phases are present in the soil, PFAS associated with these phases are released to re-establish the equilibrium partition between porewater and the solid phase. This generates a flux of labile PFAS from the solid phase to the porewater, and from the porewater to the device. In contrast, when compounds are strongly bound to the particulate phase, contributions to the porewater may be negligible (Macoustra et al., 2019; Nguyen et al., 2020).

Depending on the capacity of the soil to supply porewater concentrations, three distinct cases can be considered:

1. Fully sustained case: the release of contaminant from the particulate phase of the soil is sufficient to fully sustain porewater concentration. In this case, $C_{DGT} = C_{pw}$. This indicates that PFAS are highly mobile.
2. Partially sustained case: the flux of PFAS from the particulate phase to the porewater is significant but not sufficient to maintain porewater concentration. In this case, $C_{DGT} < C_{pw}$. This indicates that PFAS are moderately mobile.
3. Diffusion only case: there is no relevant supply of PFAS from the soil particles (due to slow or absent desorption), which indicates that PFAS are poorly mobile; the DGT device is supplied mainly by diffusion of contaminants from the bulk porewater. In this case, $C_{DGT} \ll C_{pw}$.

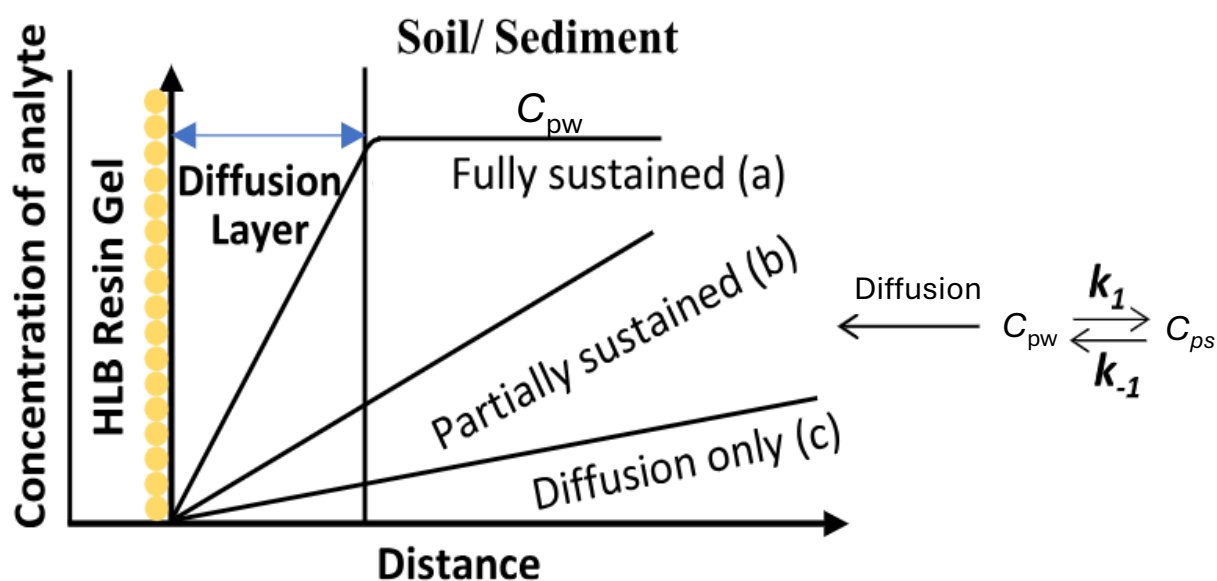


Figure 1. Schematic representation of a DGT device (loaded with HLB) deployed in soil. A concentration gradient exists between the concentration at the DGT/porewater interface ($C = 0$) and the bulk porewater (C_{pw}), which is indicated by straight diagonal lines. Depending on the interactions between PFAS and soil particles, governed by adsorption (k_1) and desorption (k_{-1}) kinetics, porewater concentrations can be fully sustained (a) by a large pool of weak binding phases, partially sustained (b) by a moderate pool of weak binding phases, or sustained by diffusion only (c), which results from migration of analytes from more concentrated and distant areas of the soil.

3 Results

3.1 Soil physicochemical properties

The percentage of particles sized between 63 μm and 2000 μm ranged from 76 to 99% in sandy soils, 30 to 57% in peaty soils, and 1.1 to 5.7%, in silty soils, respectively (Table 1). The concentration of total organic carbon (TOC) was the highest in peat (24.7–51.9%), followed by sandy soils (0.02–9.57%) and silty soils (1.1–5.7%). The pH of sandy soils ranged between 4.5 to 7.9, while peats had a narrow range of 5.0 and 5.3. Silty soils exhibited a considerably lower pH of 1.9 and 2.0, likely due to chemical reactions occurred during storage.

Table 1. Soil physicochemical properties.

Soil	TOC (%dw)	63-2000 μm (%)	pH
Sandy1	9.57	79	6.7
Sandy2	0.03	99	7.9
Sandy3	1.07	76	7.2
Sandy4	0.08	97	4.5
Sandy5	0.02	95	4.9
Peat1	51.9	30	5.0
Peat2	24.7	57	5.7
Peat3	49.5	55	5.3
Silty1	5.7	19	2.0
Silty2	1.1	14	1.9

3.2 Analytical interferences

Three different matrixes were analysed in this study, i.e., the soil particle phase, porewater, and passive sampling sorbents. Concentrations in particulate soil were always above the limit of quantification and no significant matrix effects were reported. In contrast, porewater concentrations were < LOQ in 9% of the total 230 analytical determinations (combinations of samples (10) and compounds (23)). Matrix effect resulting in non-reported results occurred for 8% of the measurements (Table 2). Non-reported results were due to the internal standard not matching set criteria (see Section 2.6).

For DGT, the frequency of missed detection due to extract concentrations < LOQ was 12%, while no matrix effects were observed (Table 2). This indicates that the analysis of porewater is affected by matrix effects more than the extractions of the soil particles and passive sampling devices, based on the analytical methods used in this study. It is important to note that matrix effects were observed predominantly in porewater from silty soils (17 times out of the 20 reported). These soils were characterised by very low pH (1.9–2.0) and fine particles. At pH $\sim$ 2 PFAS are protonated and more hydrophobic. Due to the large availability of binding sites provided by silty soils, adsorption of IS to soil particles may have occurred, thus removing IS from the porewater and contributing to the observed matrix effect. In addition, incomplete separation of the soil and pore water due to the fine particles might hamper pore water analysis and bias the determination of sorption coefficients. In Silty2, the soil particle concentrations were significantly lower than the concentrations expected from the amount of PFAS spiked in the soil. This may be due to the same matrix effect described above. As the extraction method used was not validated, further research is needed to assess the impact of pH and particle size on the extraction efficiency.

Table 2. Number of observations below the limit of quantification (LOQ) and non-reported (nr) results due to matrix effects. A total of 23 PFAS were analysed in 10 samples (resulting in 230 determinations per matrix).

	Particulate soil			Porewater			DGT		
	< LOQ	nr	< LOQ+nr	< LOQ	nr	< LOQ+nr	< LOQ	nr	< LOQ+nr
PFBA	0	0	0	0	2	2	0	0	0
4:2 FTS	0	0	0	0	0	0	0	0	0
6:2 FTS	0	0	0	0	0	0	0	0	0
8:2 FTS	0	0	0	0	1	1	0	0	0
FOSA	0	0	0	2	3	5	3	0	3
HFPO-DA	0	0	0	0	2	2	0	0	0
N-EtFOSAA	0	0	0	5	1	6	6	0	6
N-MeFOSAA	0	0	0	4	1	5	5	0	5
PFBS	0	0	0	0	0	0	0	0	0
PFDA	0	0	0	0	2	2	1	0	1
PFDoDA	0	0	0	3	2	5	5	0	5
PFDS	0	0	0	2	1	3	6	0	6
PFHpA	0	0	0	0	1	1	0	0	0
PFHpS	0	0	0	0	0	0	0	0	0
PFHxA	0	0	0	0	0	0	0	0	0
PFHxS	0	0	0	0	0	0	0	0	0
PFNA	0	0	0	0	1	1	0	0	0
PFNS	0	0	0	3	0	3	1	0	1
PFOA	0	0	0	0	1	1	0	0	0
PFOS	0	0	0	1	0	1	0	0	0
PFPeA	0	0	0	0	0	0	0	0	0
PFPeS	0	0	0	0	0	0	0	0	0
PFUnDA	0	0	0	2	2	4	3	0	3
Total	0	0	0	22	20	42	30	0	30
% of total	0	0	0	10	9	18	13	0	13

3.3 Soil–porewater partition coefficients (K_d)

Partition coefficients ranged between 0.15 and 2.36 L/kg in sandy soils, 1.19 and 6.82 L/kg in peat, and 0.40 to 0.94 L/kg in silty soils, respectively (Figure 2). These results were consistent with K_d values commonly reported in the literature (Fabregat-Palau et al., 2025; Hubert et al., 2023; Nguyen et al., 2020). A very consistent trend showing increasing K_d for increasing carbon atoms was observed for all PFAS and types of soils (Figure 6) (Hubert et al., 2023). This trend reflects the increasing hydrophobicity of PFAS characterised by longer carbon chains.

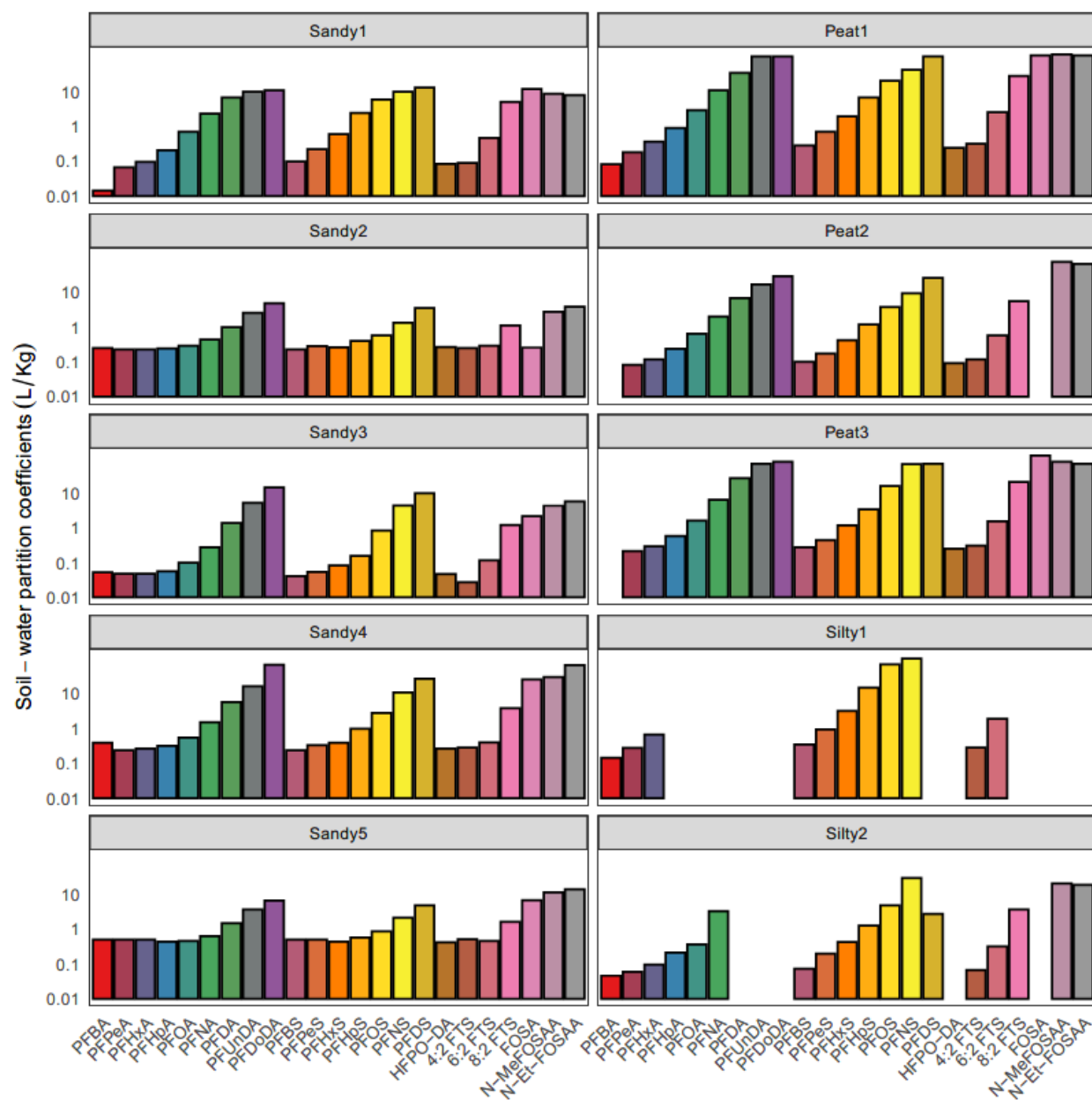


Figure 2. Soil-porewater partition coefficients of individual PFAS across different soils.

3.4 DGT fluxes

The DGT flux was typically higher in sandy soils compared to peat and silty soils, except for Sandy1, which had a flux similar to that of peats and silty soils (**Error! Reference source not found.**). Higher fluxes in sandy soils are likely due to higher porewater concentrations resulting from the relatively low surface area and low content of OC available for binding (Table 1). Lower fluxes were measured in Sandy1, consistently with its higher OC content, contributing to lower porewater concentrations (Table 1).

The DGT flux systematically decreased with increasing PFAS chain lengths, irrespective of the soil type (Figure 3). This flux was generally strongly correlated with the porewater concentration (Figure 4), except for a few compounds which largely deviated from the general trend, especially PFBA in Sandy1, 2, and 4, and Peat 2 and 3. This could be due to analytical interferences affecting PFBA determination in porewaters (Table 2; section 3.3.3). To better understand and interpret the relationship between the DGT flux and porewater concentrations, the mechanisms of uptake of DGT need to be considered.

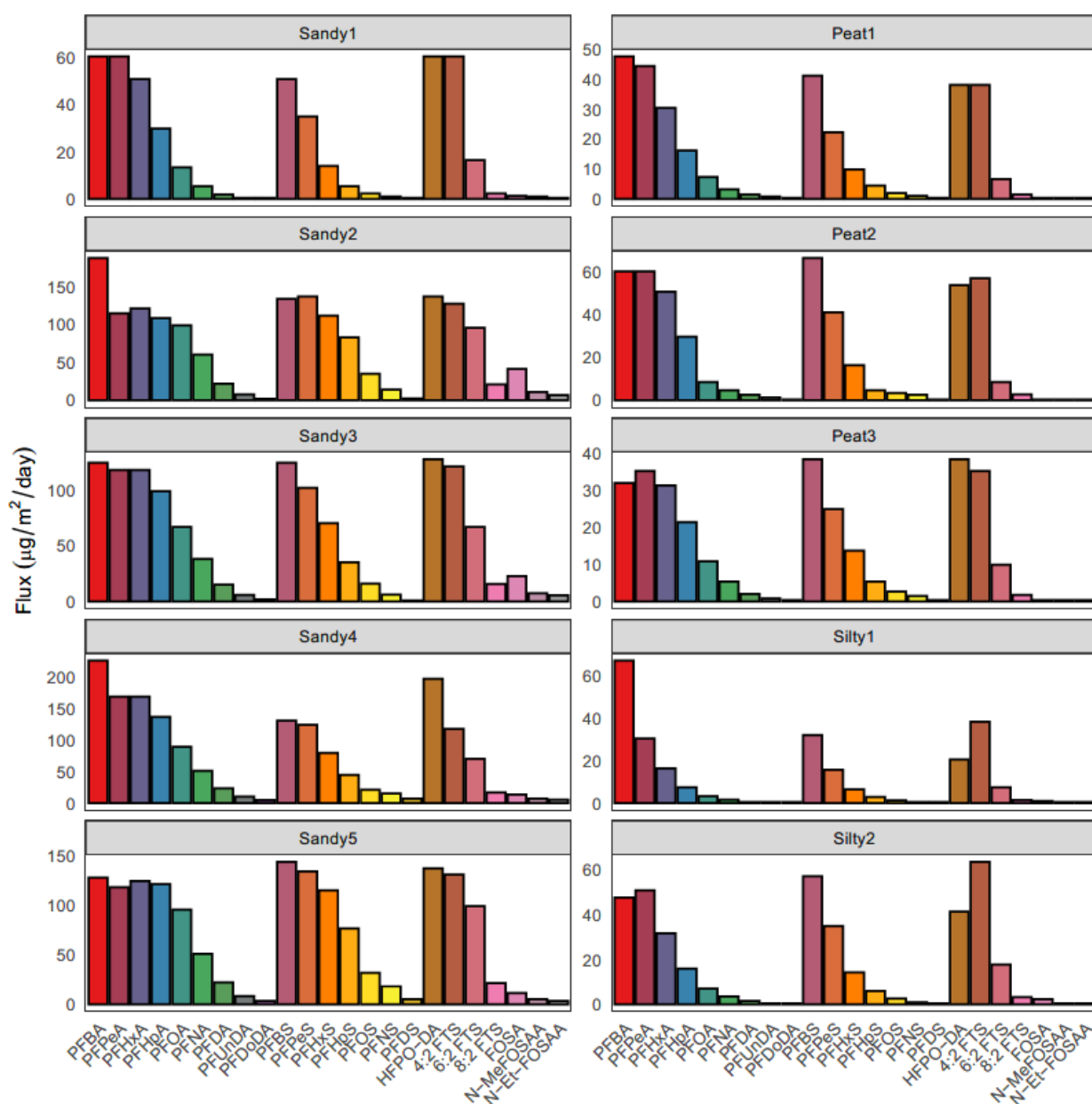


Figure 3. Individual PFAS DGT flux in different types of soil.

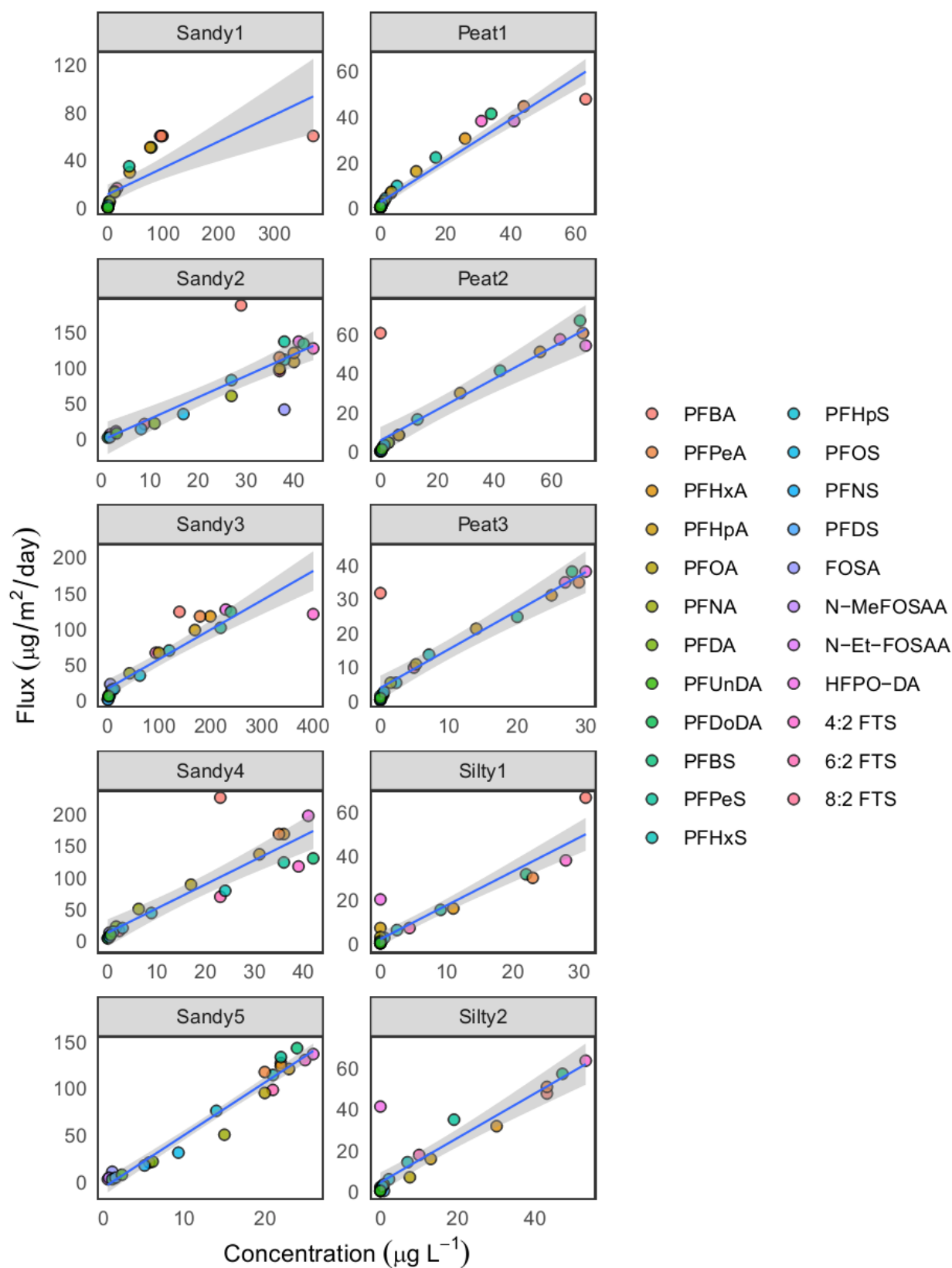


Figure 4. Correlations between the DGT flux and porewater concentrations in different soils, sorted by type of soil.

While the DGT flux correlated well with porewater concentrations in individual soils (Figure 4), new patterns emerged when these relationships were assessed for individual PFAS (Figure 5). In Sandy2, 4 and 5, the DGT flux was greater than expected based on the measured K_d for most of the compounds. This suggests that the mobility of these compounds was higher than predicted by K_d .

This was supported by the substantially lower (~1 to 3 orders of magnitude) TOC concentrations measured in these soils (Table 1). By excluding Sandy2, 4 and 5, correlations between the DGT flux and porewater were strong for most PFAS, ranging from 0.705 to 0.998 (Table 3), except for PFBA (-0.477) and PFDS (0.043). Porewater concentrations of PFBA were frequently above the calibration range and PFDS was frequently found < LOQ in DGT extracts (Appendix I.II and I.III).

Sandy2, 4 and 5 aligned progressively better with the general trend as the number of carbons in the PFAS aliphatic chains increased (Figure 9). This was consistent with greater partition of nonpolar molecules to the solid phase, resulting from stronger binding to the soil particles. Among these soils, Sandy2 had the highest pH (7.9; Table 1) and aligned the best with the general trend (Figure 9). At pH ~8, electrostatic attraction between PFAS and binding phases is minimal due to substantial deprotonation of carboxylic and hydroxyl groups, resulting in predominantly negative charges on these binding phases. In these conditions, hydrophobic interactions of the aliphatic chain become even more relevant, resulting in stronger attraction of hydrophobic PFAS compared to hydrophilic ones. This was consistent with previous studies showing that the impact of pH is greater for PFAS with longer chains (Barzen-Hanson et al., 2017a; Nguyen et al., 2020).

Based on these results, three conclusions can be drawn:

- 1) when strong interactions between PFAS and soil binding phases exist (for instance with OC), the release from soil particles is too slow or absent and does not significantly contribute to the DGT flux. This reflects the limited mobility of PFAS in these soils. Under these conditions, the DGT flux is proportional to the bulk porewater concentration;
- 2) when interactions between PFAS and soil binding phases are predominately weak, the DGT flux is sustained by PFAS released from the particulate phase, reflecting the greater mobility of PFAS in these soils; in this case, the DGT flux is higher than that resulting from diffusion from the bulk porewater only;
- 3) K_d describes the degree to which contaminants partition between the aqueous and particulate phases at the equilibrium but does not account for adsorption/desorption kinetics that affect their mobility. In contrast, the DGT flux provides a measure of the ability of the soil to release contaminants, which reflects their mobility.

In summary, the mobility of PFAS in soil depends on their interactions with the soil components: the weaker the interactions, the higher the mobility and the ensued DGT flux. Conversely, the stronger the interactions, the lower the mobility, and the smaller the DGT flux. Therefore, the DGT flux reflects the mobility of PFAS and the kinetics of the interactions that influence it, as opposed to K_d which is based on equilibrium concentrations that do not reflect kinetics of adsorption/desorption.

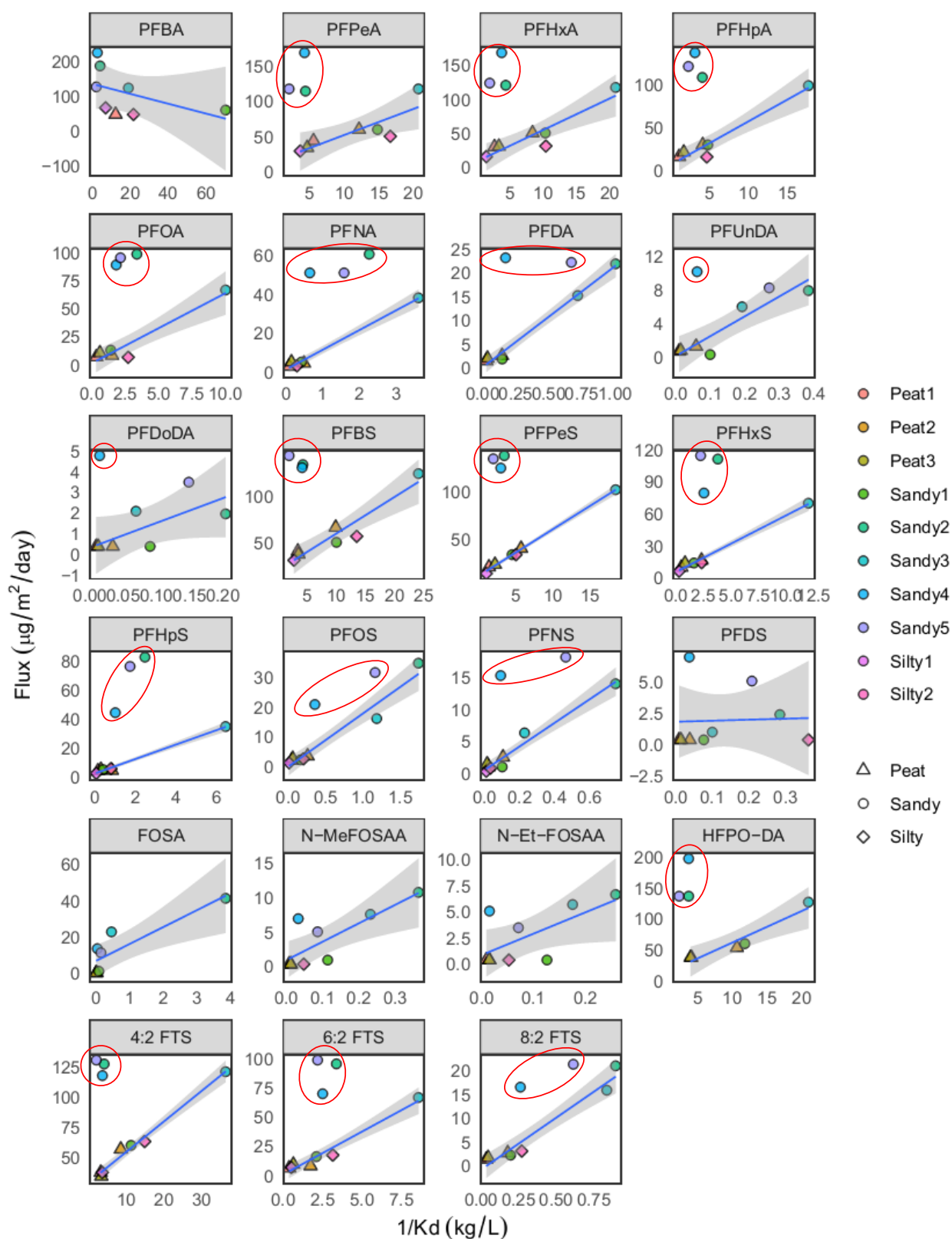


Figure 5. Correlations between PFAS DGT fluxes and porewater concentrations in different types of soils, sorted by individual PFAS. Red circles indicate datapoints excluded from the linear regression due to poor fit with the rest of the data (based on visual assessment). Pearson's correlation coefficients are reported in Table 3.

Table 3. Pearson's correlation coefficients from linear regressions between K_d vs DGT flux (Figure 9). Regressions were obtained excluding the soils highlighted in Figure 7.

Compound	Corr. Coefficient	No. of observations
PFBA	-0.477	8
PFPeA	0.839	7
PFHxA	0.926	7
PFHpA	0.973	6
PFOA	0.963	6
PFNA	0.995	6
PFDA	0.994	6
PFUnDA	0.924	5
PFDoDA	0.705	5
PFBS	0.952	7
PFPeS	0.998	7
PFHxS	0.993	7
PFHpS	0.995	7
PFOS	0.975	7
PFNS	0.983	6
PFDS	0.043	7
FOSA	0.875	5
N-MeFOSAA	0.809	5
N-Et-FOSAA	0.668	4
HFPO-DA	0.961	5
4:2 FTS	0.994	7
6:2 FTS	0.981	7
8:2 FTS	0.977	7

The DGT flux can also be used to estimate porewater concentrations (Eq. 2). The comparison between DGT concentrations (C_{DGT}) and porewater concentrations (C_{pw}) provides useful information about the mobility of PFAS. This comparison relies on the estimation of

$$R = \frac{C_{DGT}}{C_{pw}} \quad \text{Eq. 4}$$

which is a parameter reflecting the ability of the soil particles to release PFAS and sustain porewater concentrations. The 3 cases introduced in Section 3.5 (i.e., fully sustained, partially sustained, and diffusion only) (figure 6) can be directly linked to R :

1. if $R \geq 0.95$, the fully sustained case applies;
2. for $0.1 > R > 0.95$, the system is partially sustained;
3. for $R \leq 0.1$, diffusion from bulk porewater prevails.

In most soils, the R values ranged between 0.01 and 0.2, suggesting that PFAS were mainly associated with strong binding phases (Table 4 and 5). In contrast, for Sandy2, 4 and 5, R values were consistently higher than those of other soils, with ratios up to 0.74 (Table 3 and 4). These soils were characterised by relatively low organic carbon content (Table 1), consistently with reduced PFAS mobility in presence of OC (Nguyen et al., 2020). Data for sulfonamides, 4:2 FTS and 8:2 FTS were not reported due to lack of available diffusion coefficient necessary to estimated C_{DGT} (Eq. 1) and should be determined experimentally.

Table 4. *R* values estimated for perfluoroalkyl carboxylic acids (PFCA) and a perfluoroalkyl ether carboxylic acid (PFECA) in different soil types using Eq. 4. Only compounds with available diffusion coefficients are reported. na indicates non-available data (due to undetectable or non-reported data; for further details, see Section 3.2).

Soil	PFCA									PFECA
	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	HFPO-DA
Sandy1	0.00 ^a	0.01 ^a	0.01 ^a	0.02	0.03	0.04	0.05	0.02 ^c	0.02 ^c	0.02
Sandy2	0.13	0.06	0.06	0.06	0.07	0.07	0.07	0.10	0.07	0.08
Sandy3	0.02 ^a	0.01 ^a	0.01 ^a	0.01 ^a	0.02 ^a	0.03	0.06	0.10	0.12	0.01
Sandy4	0.19	0.10	0.10	0.10	0.14	0.25	0.49	0.72	1.80 ^b	0.12
Sandy5	0.11	0.12	0.12	0.12	0.13	0.10	0.13	0.14	0.15	0.13
Peat1	0.01 ^a	0.02	0.03	0.03	0.06	0.09	0.15	0.25 ^b	0.15 ^{b,c}	0.02
Peat2	na	0.02 ^a	0.02 ^a	0.03	0.03	0.05	0.09	0.11	0.07 ^c	0.02
Peat3	na	0.02	0.03	0.04	0.05	0.11	0.21	0.28 ^b	0.15 ^{b,c}	0.03
Silty1	0.04	0.03	0.03	na	na	na	na	na	na	na
Silty2	0.02	0.02	0.02	0.03	0.02	0.13	na	na	na	na

^a semi-quantitative results due to porewater concentrations or DGT fluxes exceeding the calibration range.

^b porewater concentration < LOQ (set to half the LOQ).

^c DGT extract < LOQ (set to half the LOQ).

na = not available (due to matrix effects reported for porewater analysis).

Table 5. *R* values estimated for perfluoroalkyl sulfonic acids (PFSA) and fluorotelomers (FT) in different soil types using Eq. 4. Only compounds with available diffusion coefficients are reported. na indicates non-available data (due to undetectable or non-reported data; for further details, see Section 3.2).

Soil	PFSA							FT
	PFBS	PFPeS	PFHxS	PFHpS	PFOS	PFNS	PFDS	6:2 FTS
Sandy1	0.01 ^a	0.02	0.02	0.04	0.05	0.03	0.02 ^c	0.02
Sandy2	0.06	0.08	0.07	0.08	0.06	0.05	0.05	0.07
Sandy3	0.01 ^a	0.01 ^a	0.01 ^a	0.01 ^a	0.03	0.05	0.03	0.02
Sandy4	0.06	0.07	0.07	0.12	0.19	0.41	0.74	0.08
Sandy5	0.12	0.13	0.12	0.14	0.09	0.11	0.11	0.12
Peat1	0.02	0.03	0.04	0.07	0.11	0.11	0.10 ^b	0.05
Peat2	0.02 ^a	0.02	0.03	0.04	0.07	0.08	0.04 ^c	0.03
Peat3	0.03	0.03	0.04	0.06	0.14	0.36 ^b	0.10 ^{b,c}	0.05
Silty1	0.03	0.04	0.06	0.12	na	0.09 ^{b,c}	na	0.04
Silty2	0.02	0.04	0.05	0.07	0.13	0.21 ^b	0.01 ^c	0.04

^a semi-quantitative results due to porewater concentrations or DGT fluxes exceeding the calibration range.

^b porewater concentration < LOQ (set to half the LOQ).

^c DGT extract < LOQ (set to half the LOQ).

na = not available (due to matrix effects reported for porewater analysis).

R generally increased with increasing chain lengths for both carboxylic and sulfonic acids (Table 4 and 5). This could be due to the lower porewater concentration of long-chain PFAS. As porewater concentrations significantly decrease with increasing chain length (Figure 4), the impact of the labile PFAS flux on porewater concentrations may be greater for long-chain PFAS than short-chain ones. The highest *R* were estimated for PFNA (0.25), PFNS (0.41), PFDA (0.49), PFUnDA (0.72) and PFDS (0.74) (excluding semi-quantitative or inaccurate results; Table 3). Despite low *R* values were obtained for most PFAS, considerable mobility was observed in sandy sediments characterised by low OC. This may

have important implications for drinking water production and ecological risk assessment. Additional applications of R include the assessment of desorption rates (k_{-1}), which can be obtained in combination with the use of numerical models that simulate the uptake of DGT (Li et al., 2021), and provide insight into exchange kinetics in soils and sediments, provided that $R < 0.95$.

4 Conclusions

The desorption behaviour of PFAS was tested for ten different soils. The impact of soil physicochemical properties such as particle size, pH, and organic carbon on PFAS mobility was accounted for by DGT, highlighting its suitability to investigate the fate and behaviour of PFAS over a wide range of soils. Because passive sampling reflects the chemical speciation of pollutants, its application is particularly relevant for bioavailability and ecotoxicological assessment (Amato et al., 2016, 2015, 2014). Despite the absence of direct testing, these findings and considerations are expected to be equally relevant to sediments.

The partition behaviour of PFAS varied greatly across the different soils investigated. Porewater concentrations were consistently higher in sandy soils, highlighting the impact of particle size on the partition of PFAS between the aqueous and particulate phases. The length of the fluorinated chain played a key role in defining the partition behaviour of PFAS, with compounds with longer chains generally partitioning more to the particulate phase, and those with shorter chains to the aqueous phase, respectively.

PFAS porewater fluxes measured by diffusive gradients in thin films (DGT) generally followed the same trend observed for porewater concentrations. Exceptions included 3 sandy soils, where DGT fluxes were significantly higher than expected based on estimated porewater concentrations derived from the K_d . These soils contained significantly lower levels of organic carbon, which strongly binds with PFAS reducing their mobility. This shows that the DGT flux accounts for adsorption/desorption kinetics that affect the mobility of PFAS in soil, whereas K_d does not provide this information. Poor correlations were obtained for PFBA and PFDS, likely due to analytical issues (i.e., porewater analysis exceeding the calibration range and fluxes below quantification limits).

In the context of assessing PFAS mobility in soil, passive sampling provided a level of insight equivalent to (or greater than) that achieved by combining several measurements (i.e., porewater concentrations, particulate soil concentrations, particle size, pH and organic carbon concentrations), making it an ideal approach for rapid screening of PFAS mobility in water-saturated soils and sediments. No matrix effects were observed for passive sampling measurements, whereas significant effects were reported for porewater measurements in two silty soils. This adds to the potential benefits of passive sampling over conventional approaches to determine porewater concentrations and mobility of chemicals in soils.

4.1 Recommendations for follow-up

Future studies should include a validation of the passive sampling method, as well as replicate analysis to account for experimental variability. The comparison with conventional methodologies should be further investigated, with a particular focus on applications that are relevant for the drinking water sector, such as determining the leaching potential and mobility of pollutants in soil. Furthermore, different types of pollutants other than PFAS can be tested. This may pave the way for *in situ* mobility assessment of organic compounds such as pesticides, phthalates and bisphenols from degradation of plastics in landfills, and chemicals associated with the dumping of waste from the production of illicit drugs.

4.2 Application by drinking water companies

The assessment of contaminant mobility in soil and sediment is critical to estimate their potential impact on drinking water production. This is particularly relevant to investigate the transport of contaminants introduced into the soil via different processes and sources, such as atmospheric deposition (i.e., pesticides and PFAS), (illegal) waste disposal

(historic landfills, illicit drug production waste), infiltration of surface water (aquifer recharge) or treated wastewater (reuse for irrigation purposes), application of water containing and aqueous film-forming foam (AFFF) (Firefighting practice and activities). Additional applications may also include the assessment of the release of (historic) contaminants from sediments into the water column of rivers and lakes.

4.3 Relevance to other stakeholders and societal impact

Passive sampling measurements offer a simple approach for screening pollutants in various environmental compartments, including soil and sediment. In addition, the ability of passive sampling to account for the chemical speciation of pollutants has important implications for the assessment of pollutant bioavailability and toxicity, which is essential to assess their impact on biota and the environment.

5 References

- Agbaogun, B.K., Fischer, K., 2020. Adsorption of phenylurea herbicides by tropical soils. *Environ. Monit. Assess.* 192, 212. <https://doi.org/10.1007/s10661-020-8160-2>
- Amato, E.D., Hockin, A., Van Leeuwen, J., 2024. Assessment of PFAS deposition in two drinking water production sites, BTO 2024.085. KWR Water Research Institute.
- Amato, E.D., Simpson, S.L., Belzunce-Segarra, M.J., Jarolimek, C.V., Jolley, D.F., 2015. Metal Fluxes from Porewaters and Labile Sediment Phases for Predicting Metal Exposure and Bioaccumulation in Benthic Invertebrates. *Environ. Sci. Technol.* 49, 14204–14212. <https://doi.org/10.1021/acs.est.5b03655>
- Amato, E.D., Simpson, S.L., Jarolimek, C.V., Jolley, D.F., 2014. Diffusive gradients in thin films technique provide robust prediction of metal bioavailability and toxicity in estuarine sediments. *Environ. Sci. Technol.* 48, 4485–4494.
- Amato, E.D., Simpson, S.L., Remaili, T.M., Spadaro, D.A., Jarolimek, C.V., Jolley, D.F., 2016. Assessing the Effects of Bioturbation on Metal Bioavailability in Contaminated Sediments by Diffusive Gradients in Thin Films (DGT). *Environ. Sci. Technol.* 50, 3055–3064. <https://doi.org/10.1021/acs.est.5b04995>
- Amato, E.D., van der Grift, B., 2024. Fate of PFAS during dune infiltration, KWR 2024.044. KWR Water Research Institute.
- Barzen-Hanson, K.A., Davis, S.E., Kleber, M., Field, J.A., 2017a. Sorption of Fluorotelomer Sulfonates, Fluorotelomer Sulfonamido Betaines, and a Fluorotelomer Sulfonamido Amine in National Foam Aqueous Film-Forming Foam to Soil. *Environ. Sci. Technol.* 51, 12394–12404. <https://doi.org/10.1021/acs.est.7b03452>
- Barzen-Hanson, K.A., Roberts, S.C., Choyke, S., Oetjen, K., McAlees, A., Riddell, N., McCrindle, R., Ferguson, P.L., Higgins, C.P., Field, J.A., 2017b. Discovery of 40 Classes of Per- and Polyfluoroalkyl Substances in Historical Aqueous Film-Forming Foams (AFFFs) and AFFF-Impacted Groundwater. *Environ. Sci. Technol.* 51, 2047–2057. <https://doi.org/10.1021/acs.est.6b05843>
- Fabregat-Palau, J., Ershadi, A., Finkel, M., Rigol, A., Vidal, M., Grathwohl, P., 2025. Modeling PFAS Sorption in Soils Using Machine Learning. *Environ. Sci. Technol.* 59, 7678–7687. <https://doi.org/10.1021/acs.est.4c13284>
- Hofman-Caris, R., 2024. PFAS-verwijdering uit drinkwater; overzicht van literatuur en kooldata uit de praktijk (No. BTO 2024.037). KWR Water Research Institute.
- Hubert, M., Arp, H.P.H., Hansen, M.C., Castro, G., Meyn, T., Asimakopoulou, A.G., Hale, S.E., 2023. Influence of grain size, organic carbon and organic matter residue content on the sorption of per- and polyfluoroalkyl substances in aqueous film forming foam contaminated soils - Implications for remediation using soil washing. *Sci. Total Environ.* 875, 162668. <https://doi.org/10.1016/j.scitotenv.2023.162668>
- Li, Y., Han, C., Luo, J., Jones, K.C., Zhang, H., 2021. Use of the Dynamic Technique DGT to Determine the Labile Pool Size and Kinetic Resupply of Pesticides in Soils and Sediments. *Environ. Sci. Technol.* 55, 9591–9600. <https://doi.org/10.1021/acs.est.1c01354>
- Macoustra, G., Holland, A., Stauber, J., Jolley, D.F., 2019. Effect of Various Natural Dissolved Organic Carbon on Copper Lability and Toxicity to the Tropical Freshwater Microalga *Chlorella* sp. *Environ. Sci. Technol.* 53, 2768–2777. <https://doi.org/10.1021/acs.est.8b04737>
- Nguyen, T.M.H., Bräunig, J., Thompson, K., Thompson, J., Kabiri, S., Navarro, D.A., Kookana, R.S., Grimison, C., Barnes, C.M., Higgins, C.P., McLaughlin, M.J., Mueller, J.F., 2020. Influences of Chemical Properties, Soil Properties, and Solution pH on Soil–Water Partitioning Coefficients of Per- and

Polyfluoroalkyl Substances (PFASs). *Environ. Sci. Technol.* 54, 15883–15892.
<https://doi.org/10.1021/acs.est.0c05705>

Veggel, M. van, Hofman-Caris, R., 2024. PFAS-verwijdering met behulp van schuimfractionering (No. BTO 2024.041). KWR Water Research Institute.

I Appendix

I.I PFAS concentrations in particulate soil

Compound	Sandy1	Sandy2	Sandy3	Sandy4	Sandy5	Peat1	Peat2	Peat3	Silty1	Silty2
PFBA	5.3	7.3	7.4	9.0	11	5.2	4.9	3.7	4.5	2.0
4:2 FTS	8.9	11	11	11	13	10	7.5	8.3	8.0	3.6
6:2 FTS	8.1	11	11	9.2	9.6	8.3	3.7	7.5	8.4	3.2
8:2 FTS	8.7	10	12	8.8	9.3	11	6.8	8.2	9.7	2.9
FOSA	8.9	9.9	12	8.5	7.5	14	9.0	15	11	2.9
HFPODA	8.1	11	11	11	11	10	6.8	7.5	7.8	3.9
NEtFOSAA	8.8	7.0	12	8.3	7.9	14	8.3	8.6	10	2.4
NMeFOSAA	9.7	8.3	13	9.0	8.1	15	9.5	10	10	2.6
PFBS	7.9	9.7	10	10	12	9.8	7.1	7.8	7.8	3.5
PFDA	9.5	11	12	9.8	9.1	13	7.5	9.4	10	2.9
PFDoDA	8.9	6.4	12	8.5	7.4	13	8.2	10	9.6	2.5
PFDS	9.7	6.0	11	8.3	7.2	13	8.6	8.6	10	2.5
PFHpA	8.4	9.8	9.6	9.9	10	10	6.8	8.2	7.9	2.8
PFHpS	7.9	11	9.8	8.9	8.1	11	3.5	7.8	8.8	2.8
PFHxA	7.5	9.1	9.6	9.5	11	9.6	6.7	7.4	7.4	2.9
PFHxS	7.9	10	10	9.4	9.3	10	5.6	8.4	8.1	3.0
PFNA	9.3	12	12	9.7	9.5	12	5.7	9.7	10	2.8
PFNS	11	11	18	12	11	13	8.8	8.5	13	3.7
PFOA	9.2	11	10	9.3	9.3	10	4.3	8.5	8.6	2.8
PFOS	7.8	9.8	11	8.4	8.0	9.9	5.4	8.2	8.7	2.7
PFPeA	6.6	8.5	8.7	8.3	10	8.0	5.9	6.3	6.4	2.6
PFPeS	8.7	11	12	12	11	12	7.4	9.0	8.6	3.8
PFUnDA	9.3	8.1	12	9.1	8.5	13	8.4	8.7	11	2.7

Values are reported in µg/L in the 1 ml extract.

The reporting limit is 0.25 µg/L.

I.II PFAS concentrations in porewater

Compound	Sandy1	Sandy2	Sandy3	Sandy4	Sandy5	Peat1	Peat2	Peat3	Silty1	Silty2
PFBA	370	29	140	23	22	63	-	-	31	43
4:2 FTS	99	44	400	39	25	31	63	27	28	53
6:2 FTS	17	37	94	23	21	3.2	6.4	4.9	4.4	10
8:2 FTS	1.7	8.9	10	2.3	5.7	0.39	1.2	0.39	-	0.78
FOSA	0.74	38	5.5	0.33	1.1	<0.25	-	<0.25	-	-
HFPODA	95	41	230	41	26	41	72	30	-	-
NEtFOSAA	1.1	1.8	2.1	<0.25	0.56	<0.25	<0.25	<0.25	-	<0.25
NMeFOSAA	1.1	3.0	3.0	0.30	0.70	<0.25	<0.25	<0.25	-	<0.25
PFBS	79	42	240	42	24	34	70	28	22	47
PFDA	1.4	11	8.6	1.7	6.1	0.37	1.1	0.35	-	-
PFDoDA	0.80	1.3	0.83	<0.25	1.1	<0.25	0.28	<0.25	-	-
PFDS	0.74	1.7	1.1	0.31	1.5	<0.25	0.33	<0.25	-	0.90
PFHpA	40	40	170	31	23	11	28	14	-	13
PFHpS	3.2	27	63	8.9	14	1.6	2.9	2.3	0.58	2.2
PFHxA	77	40	200	36	22	26	56	25	11	30
PFHxS	13	38	120	24	21	5.1	13	7.1	2.5	7.0
PFNA	3.9	27	43	6.3	15	1.1	2.8	1.5	-	0.84
PFNS	1.1	8.2	4.1	1.1	5.1	0.30	0.92	<0.25	<0.25	<0.25
PFOA	13	37	100	17	20	3.4	6.5	5.2	-	7.6
PFOS	1.3	17	13	3.0	9.3	0.48	1.4	0.51	<0.25	0.55
PFPeA	97	37	180	35	20	44	71	29	23	43
PFPeS	39	38	220	36	22	17	42	20	9.1	19
PFUnDA	0.92	3.1	2.3	0.56	2.3	<0.25	0.49	<0.25	-	-

- : This parameter cannot be reported due to matrix effects, the internal standard does not meet the set criteria.

Shaded values indicate semi-quantitative results due concentrations falling outside the calibration range.

Values are reported in µg/L in the 1 ml extract.

The reporting limit is 0.25 µg/L.

I.III PFAS concentrations in passive sampling extracts

Compound	Sandy1	Sandy2	Sandy3	Sandy4	Sandy5	Peat1	Peat2	Peat3	Silty1	Silty2
PFBA	19	59	39	71	40	15	19	10	21	15
4:2 FTS	19	40	38	37	41	12	18	11	12	20
6:2 FTS	5.2	30	21	22	31	2.1	2.7	3.1	2.3	5.6
8:2 FTS	0.73	6.6	5.0	5.2	6.7	0.46	0.88	0.57	0.42	1.0
FOSA	0.39	13	7.2	4.3	3.6	<0.25	<0.25	<0.25	0.28	0.71
HFPODA	19	43	40	62	43	12	17	12	6.4	13
NEtFOSAA	<0.25	2.1	1.8	1.6	1.1	<0.25	<0.25	<0.25	<0.25	<0.25
NMeFOSAA	0.31	3.4	2.4	2.2	1.6	<0.25	<0.25	<0.25	<0.25	<0.25
PFBS	16	42	39	41	45	13	21	12	10	18
PFDA	0.60	6.9	4.8	7.3	7.0	0.48	0.82	0.65	<0.25	0.49
PFDoDA	<0.25	0.62	0.66	1.5	1.1	<0.25	0.25	<0.25	<0.25	<0.25
PFDS	<0.25	0.76	0.32	2.2	1.6	<0.25	<0.25	<0.25	<0.25	<0.25
PFHpA	9.4	34	31	43	38	5.1	9.4	6.7	2.3	5.0
PFHpS	1.7	26	11	14	24	1.4	1.5	1.7	0.90	1.9
PFHxA	16	38	37	53	39	9.6	16	9.8	5.1	10
PFHxS	4.4	35	22	25	36	3.1	5.2	4.3	2.0	4.5
PFNA	1.7	19	12	16	16	1.0	1.5	1.7	0.48	1.1
PFNS	0.34	4.4	2.0	4.8	5.7	0.34	0.82	0.48	<0.25	0.28
PFOA	4.2	31	21	28	30	2.3	2.7	3.4	1.0	2.2
PFOS	0.71	11	5.1	6.6	10	0.63	1.1	0.84	0.38	0.81
PFPeA	19	36	37	53	37	14	19	11	9.5	16
PFPeS	11	43	32	39	42	7.0	13	7.8	4.9	11
PFUnDA	<0.25	2.5	1.9	3.2	2.6	0.25	0.43	0.28	<0.25	<0.25

Shaded values indicate semi-quantitative results due concentrations falling outside the calibration range.

Values are reported in µg/L in the 1 ml extract.

The reporting limit is 0.25 µg/L.