Environment International 215 (2026) 110439

Contents lists available at ScienceDirect

Environment International

journal homepage: www.elsevier.com/locate/envint

Full length article

Differential PFAS transfer through the terrestrial and aquatic food web to predatory spiders

Ioanna S. Gkika [a,*], J.Arie Vonk [a], Thomas L. ter Laak [a,b], Cornelis A.M. van Gestel [c], Jildou Dijkstra [a], Thimo Groffen [d], Lieven Bervoets [d], Michiel H.S. Kraak [a]

[a] Department of Freshwater and Marine Ecology (FAME), Institute for Biodiversity and Ecosystem Dynamics (IBED), University of Amsterdam, Sciencepark 904, 1098 XH Amsterdam, the Netherlands
[b] KWR Water Research Institute, P.O. Box 1072, 3430 BB Nieuwegein, the Netherlands
[c] Amsterdam Institute for Life and Environment (A-LIFE), Faculty of Science, Vrije Universiteit Amsterdam, De Boelelaan 1108, 1081 HZ Amsterdam, the Netherlands
[d] ECOSPHERE, Department of Biology, University of Antwerp, Groenenborgerlaan 171, 2020 Antwerpen, Belgium

ARTICLE INFO

Keywords:
Trophic transfer
Primary producers
Invertebrates
PFAS-contaminated ecosystem
Stable isotopes
Integrated approach

ABSTRACT

The environmental persistence of Per- and polyfluoroalkyl substances (PFAS) results in prolonged exposure of organisms, which in turn leads to bioaccumulation and trophic transfer. However, if trophic transfer is studied at all, different ecosystems are generally examined in isolation. The aim of this study was therefore to unravel the PFAS transfer through a contaminated terrestrial and aquatic food web, with spiders as top predators. To this end, organisms from the aquatic and terrestrial compartments of a PFAS-contaminated ecosystem were collected and screened for 44 PFAS. The trophic position of the organisms was investigated using stable isotopes. PFAS were present in organisms from all trophic levels (primary producers, herbivores, detritivores, predators), albeit with varying total concentrations and profiles. Several PFAS magnified towards the spiders through the terrestrial food web, adding to the growing body of evidence that some PFAS have the potential to magnify, while trophic transfer was less pronounced through the aquatic food web. There was no consistent pattern in which substances magnified and which got diluted across the food chain. TFA, the smallest and most polar PFAS, showed high food chain transfer. This emphasizes that we still lack insight into the drivers of environmental PFAS distribution, uptake by organisms and transfer through food webs.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a widely used group of synthetic chemicals, characterized by their great environmental persistence (Parsons et al., 2008). These properties make them suitable for numerous industrial and consumer applications, but also render them problematic environmental contaminants. Their widespread use in combination with their general high mobility results in the ubiquitous global detection of PFAS in many environmental compartments (Brunn et al., 2023; Wang et al., 2025). Once in the environment, their persistency results in prolonged exposure of organisms, which can eventually lead to bioaccumulation, trophic transfer, and consequent adverse effects on human and environmental health (Byns et al., 2022; Fiedler et al., 2019; Guo et al., 2019; Wang et al., 2025). The omnipresence of PFAS, in combination with their persistency and amphiphilic nature, has initiated extensive research on their occurrence, distribution, bioaccumulation and trophic transfer (Gkika et al., 2023). Yet, these studies still cover only a small fraction of the over 4,700 Chemical Abstracts Service (CAS)-registered PFAS that have been identified on the global market (OECD, 2018) and also cover only a limited number of organisms and ecosystems (Brunn et al., 2023; Gkika et al., 2025; Lewis et al., 2022; Wang et al., 2025). Nonetheless, the limited available information so far already shows varying PFAS bioaccumulation and trophic transfer between different ecosystems (Dai et al., 2025; Huang et al., 2022; van den Brink et al., 2016).

The complex and constant interplay between terrestrial and aquatic ecosystems and the potential transfer of PFAS from aquatic to terrestrial biota (Daley et al., 2011; Koch et al., 2020; Yu et al., 2013) highlight the need to study the fate of these contaminants in different ecosystems with an integrated holistic approach. Yet, if trophic transfer is studied at all, the different ecosystems are still examined in isolation, albeit the number of studies employing an integrated approach is increasing

* Corresponding author.
E-mail addresses: i.s.gkika@uva.nl, i.s.gkika@uu.nl (I.S. Gkika).

https://doi.org/10.1016/j.envint.2026.110439
Received 2 March 2026; Received in revised form 23 July 2026; Accepted 25 July 2026
Available online 26 July 2026

(Daley et al., 2011; Endicott et al., 2025; Gkika et al., 2025; Hopkins et al., 2023; Ibor et al., 2025; Koch et al., 2020; 2021). Here, we argue that riparian spiders offer the unique possibility to integrate the fate of PFAS in the aquatic and terrestrial environment, since they are among the few primarily terrestrial invertebrate predators that incorporate aquatic resources into their diets, thereby linking aquatic and terrestrial food webs (Fiolka et al., 2024, 2025; Middendorf et al., 2025; Paetzold et al., 2006).

The aim of this study was therefore to unravel the PFAS transfer through the terrestrial and aquatic food web in a contaminated ecosystem, ending up in riparian spiders. To this end, the structure of the integrated terrestrial-aquatic food web was unraveled using stable isotopes assessing the trophic position of each organism. Sampled organisms were grouped according to their trophic position into primary producers, consumers (including herbivores and detritivores), and riparian spiders as top predators. The concentrations and profiles of 44 PFAS were determined in all individual organisms. PFAS trophic transfer, expressed as either trophic magnification or trophic dilution factors, was examined by first excluding and then including the spiders as part of the food web. By doing so, the PFAS fluxes through the terrestrial and aquatic food web could initially be examined separately, while by adding the spiders the effect of a predator that feeds on both terrestrial and aquatic resources on the trophic transfer of PFAS was assessed.

2. Materials and methods

2.1. Field sampling

For a detailed description of the study area and sampling sites, we refer to Gkika et al. (2025), since the same samples were analyzed and exactly the same setup as for the current study was used. Briefly, the most common organisms from the terrestrial and aquatic ecosystem were sampled at five sites, between 200 and 300 m apart, along the western bank of Lake Blokkersdijk (Site 1; 51°13′46.2″N 4°20′29.6″E. Site 2; 51°13′56.0″N 4°20′28.9″E, Site 3; 51°14′05.4″N 4°20′40.1″E, Site 4; 51°14′09.3″N 4°20′43.0″E, Site 5; 51°14′09.9″N 4°20′51.8″E), nearby a fluorochemical plant close to Antwerp (Belgium). Sampling included primary producers, herbivores, detritivores and predators, all identified to the lowest possible taxonomic level.

For terrestrial primary producers, twigs with leaves were hand-picked from four plant species (*Rubus plicatus, Urtica dioica, Crataegus monogyna* and *Alnus glutinosa*). Upon collection, plant samples were rinsed with ambient water to remove any particles. Terrestrial invertebrates were manually collected from the soil surface and the surrounding vegetation and included the Gastropoda *Arion rufus* and *Cepaea* spp. (including both *Cepaea nemoralis* and *Cepaea hortensis*), Oligochaeta/Lumbricidae, the diplopod (Julidae) *Schizophyllum sabulosum,* and Isopoda/Oniscidea. Aquatic primary producers included phytoplankton (size class 41–335 μm; collected by filtering water through a 335 and 41 μm mesh plankton net, subsequently), periphyton scraped from reed stems, and whole plants of three submerged (*Potamogeton crispus, Elodea canadensis, Chara vulgaris),* one floating (*Lemna minor*), and young shoots (50 cm) of one emergent *(Phragmites australis*) macrophyte species. Upon collection, the macrophyte samples were rinsed with ambient water to remove any debris. Zooplankton was collected by filtering water through a 335 μm mesh plankton net and consisted almost exclusively of *Daphnia magna*. Aquatic macroinvertebrates were collected by sweeping a 0.5-mm mesh net through the aquatic vegetation as well as on top of the sediment layers and included *Corixa* spp. and *Lumbricus tubifex* and three taxa of (epi)benthic insect larvae (*Chironomus riparius, Cloeon dipterum, Mystacides longicornis*). Riparian spiders were collected from the littoral vegetation (mainly *P. australis*) and included two web building species (*Araneus diadematus, Tetragnatha* sp.) and two surface hunters (*Clubiona* sp., Lycosidae). For all taxa multiple samples were collected, except for Lycosidae and *M. longicornis* which were single replicates (details on the replication of field samples included in the Supporting Information Section S1, Table S1).

2.2. Construction of the terrestrial-aquatic food web by stable isotope analysis

In this study we focused on the basal levels of the food web and the selected taxa represent the main groups relevant for both the aquatic and the terrestrial ecosystem, including primary producers, herbivores and detritivores, both reliant on primary producers. The spiders were selected as the predators that link these basal levels of the aquatic and terrestrial food web. To construct the basal food web and assign a trophic position to each organism, stable isotope analysis was performed. Prior to the analysis all samples were freeze dried. Pooled *C. nemoralis/ C. hortensis* samples were acidified with sufficient HCl (3 mM) to remove any inorganic carbon and subsequently freeze-dried again. Samples were homogenized using a mortar and pestle or an herb grinder mill for the reed *P. australis*. When there was enough dry weight of the sampled organisms to allow for technical replicates, triplicate samples per taxon (2–7 mg of dry material) were weighed in tin capsules. Otherwise, one sample per taxon was analyzed. Total C, total N, $\delta^{13}C$ and $\delta^{15}N$ were measured using a Vario Isotope cube, Elementar GmbH (Langselbold, Germany) and sequentially a BioVision isotope ratio mass spectrometer (Elementar, UK Ltd., Manchester UK). Isotope ratios were defined following Equation (1), where R_{sample} is the ratio between the heavy and light isotopes ($^{13}C/^{12}C$ and $^{15}N/^{14}N$) in each sample and $R_{standard}$ is the same ratio in the reference material (^{13}C: IAEA-CH_6, sucrose and IAEA-600, caffeine; ^{15}N: USGS-25, ammonium sulfate and IAEA-N_2, ammonium sulfate). A higher $\delta^{13}C$ or $\delta^{15}N$ value (‰) means that the sample is more enriched with the heavy isotope, so with either ^{13}C or ^{15}N (van der Lee et al., 2021).

$$\delta_x = \left(\frac{R_{sample}}{R_{standard}} - 1\right) * 1,000 \tag{1}$$

The $\delta^{15}N$ values were used as an indication for the trophic position of the organisms. For all taxa, except for the Lumbricidae, the trophic position was determined based on the model proposed by Post (2002) (Equation (2)) for each consumer taxon sample separately ($\delta^{15}N_{consumer}$).

$$TP = 2 + \left(\frac{\delta^{15}N_{consumer} - \delta^{15}N_{base}}{\Delta_N}\right) \tag{2}$$

Following Borgå et al. (2012) and Post (2002), the $\delta^{15}N$ value of primary consumers, representing trophic position 2, was selected as the baseline ($\delta^{15}N_{base}$). For the terrestrial taxa, the $\delta^{15}N_{base}$ was the average $\delta^{15}N$ value of the Gastropoda. For the aquatic taxa $\delta^{15}N_{base}$ was the average $\delta^{15}N$ value of the microalgae consumers: zooplankton, *L. tubifex* and *Corixa* spp. (most likely the grazer *Corixa punctata,* the dominant species in the region). Although we had a limited number of primary consumer taxa, we decided against using primary producers as baseline since there is a temporal lag between resource and consumer isotopic signals (Post, 2002). The Δ_N represents the expected enrichment of $\delta^{15}N$ with each trophic position and was set to 2.55 for both terrestrial and aquatic taxa, based on the *meta*-analysis by Matthews and Mazumder (2008). The trophic positions of all primary producers, except for periphyton, were set to one, since periphytic biofilms include both autotrophs and heterotrophs, and so its overall $\delta^{15}N$ signature is expected to reflect mixed trophic contributions. For three out of the four riparian spider taxa (*Tetragnatha* sp., *A. diademantus, Clubiona* sp.), their isotopic signatures fell between those of terrestrial and aquatic resources. The diets of riparian spiders are known to consist of around equal contribution of both resources (Collier et al., 2002). Since these resources had quite distinct $\delta^{15}N_{base}$ values, we normalized the trophic position calculations of these riparian spiders based on the average of both baselines. The trophic position of Lycosidae was calculated based on only the terrestrial baseline, because the single sample we had for this taxon had a quite distinct isotope signature. For Lumbricidae, trophic position was calculated according to the regression proposed by Potapov et al. (2019) (Equation (3)), since both anecic and epigeic earthworm species were sampled and according to this recent study, earthworms tend to have different, more variable trophic enrichment factors. Trophic positions were calculated for each taxon individually as further elaborated in Section S2 (Text S1).

$$TP_{Lumbricidae} = 2.05 + 0.0658 * \delta^{15}N \tag{3}$$

2.3. PFAS extraction and analysis

All organisms were analyzed for 44 PFAS covering a wide range of structures, including six isomer pairs, (Section S3, Text S2). Individual branched isomers were not differentiated in the analysis and the term "branched isomers" refers therefore to the sum of all quantified branched isomers per compound. PFAS were divided into five subclasses: short and long perfluorosulfonic acids (PFSAs), short and long perfluorocarboxylic acids (PFCAs), and the category "other PFAS and precursors". Short-chain PFCAs included compounds with up to six fluorinated carbons, while short-chain PFSAs included compounds with up to seven fluorinated carbons (ITRC, 2023). When showing PFAS profiles, herbivores and detritivores were merged into the group of consumers and their concentrations were plotted together. The protocols used for PFAS extraction, analysis and quality assurance/quality control (QA/QC) assessment have been described previously (Gkika et al., 2024, 2025; Sadia et al., 2020, 2023). Briefly, solid–liquid extraction was performed followed by a weak anion exchange solid phase extraction and a clean-up step. A detailed description of the PFAS extraction for all matrices, the quantification method, and the quality assurance/quality control criteria can be found in Gkika et al. (2025), since results from the same samples were used in the present study. Some biota samples did not fulfill the QA/QC criteria and were therefore excluded from further analysis (Section S3, Text S2).

The concentrations of 44 PFAS were measured in the sampled terrestrial and aquatic organisms. In the aquatic biota, 42 PFAS were measured since two compounds (PFBS and 7:3 FTCA) did not fulfill the QA/QC criteria and were therefore excluded from further analysis. Since all sampling sites belonged to the same lake system and the variation between the sampling sites was relatively small compared to the PFAS concentrations measured in the environment (Gkika et al., 2025), the PFAS concentrations in the organisms from the five sampling sites were averaged per species for the whole study area. These average concentrations were then summed per PFAS subclass to determine the PFAS profiles in each of the taxa on a dry weight basis (ng/g dw). For species that were not present at all sampling sites, only the sites where the species were present were taken into account when calculating the average for the whole study area (Section S1, Table S1) (Gkika et al., 2025).

2.4. Trophic magnification factors

To evaluate the trophic transfer of PFAS through both the terrestrial and aquatic basal food web up to the riparian predatory spiders, a compound-specific regression analysis between the calculated trophic positions and the log-transformed PFAS concentrations of all terrestrial and all aquatic taxa separately was performed, similar to Jones et al. (2014) for metals. Trophic magnification was examined first by including only the primary producers, herbivores and detritivores of the aquatic and terrestrial basal food web separately, excluding the spiders. Next, trophic magnification was calculated again, now also including the spiders as predators, catching prey from both terrestrial and aquatic origin. Lastly, in the presence of spiders, trophic magnification was also examined by combining terrestrial and aquatic data points. Linear regressions to determine trophic magnification factors (TMFs) and trophic dilution factors (TDFs) of individual PFAS were performed only when more than 50% of the included organism data points had concentrations > LOQ (LOQ criterion). For these PFAS, the values for organisms with concentrations < LOQ were substituted by 0.5*LOQ (EFSA, 2018; Helsel, 2010; Ricolfi et al., 2025). For statistically significant regressions (Pearson correlation test; $\alpha = 0.05$) of the TMF (Equation (4), we distinguished three types of results based on the slope (b). A slope > 0 indicated trophic magnification and subsequently TMFs were calculated (Equation (4)). A slope < 0 pointed towards trophic dilution and subsequently, trophic dilution factors (TDFs) were calculated using the same equation. Non-significant regressions indicated that neither trophic magnification nor trophic dilution took place. For this analysis replicates were not averaged, and all individual data points were considered in the regressions.

$$\text{TMF or TDF} = 10^{b} \tag{4}$$

To evaluate how different approaches of dealing with censored (<LOQ) data would affect the calculated TMFs/TDFs we performed a sensitivity analysis. For each of the five examined scenarios (terrestrial without spiders; terrestrial with spiders; aquatic without spiders; aquatic with spiders; terrestrial-aquatic with spiders) the compound(s) with the most data points < LOQ were selected, as the most sensitive, and the < LOQ data were substituted with 1*LOQ.

3. Results

3.1. Aquatic-terrestrial food web structure

The food web structure of the joint terrestrial and aquatic ecosystem of Lake Blokkersdijk was evaluated using $\delta^{13}C$ and $\delta^{15}N$ stable isotope signatures (Fig. 1; Section S4, Table S2). In general, the aquatic primary producers had higher $\delta^{13}C$ (± SEM (Standard Error of the Mean)) values compared to the terrestrial ones, except for *Lemna minor*, which had among the lowest $\delta^{13}C$ signatures. The $\delta^{13}C$ values of terrestrial primary producers ranged from −31.8 ± 0.43‰ (*Urtica dioica*) to −28.7 ± 0.47‰ (*Alnus glutinosa*), while those of aquatic primary producers, excluding *L. minor*, ranged from −20.7‰ (*Chara vulgaris*; n = 1) to −20.0 ± 0.52‰ (*Potamogeton crispus*). Also, most of the terrestrial herbivores and detritivores showed more negative $\delta^{13}C$ values compared to the aquatic ones. Of the two terrestrial herbivores, *Arion rufus* had the lowest $\delta^{13}C$ value (−28.1 ± 0.69‰) and (pooled) *Cepaea nemoralis* and *Cepaea hortensis* the highest (−27.5 ± 0.04‰), while $\delta^{13}C$ values of aquatic herbivores ranged from −21.8 ± 1.8‰ (*Corixa* spp.) to −18.4 ± 0.56‰ (*Lumbriculus tubifex*). The three terrestrial detritivores had similar $\delta^{13}C$ values, ranging from −27.7 ± 0.41‰ (Lumbricidae) to −25.4 ± 0.30‰ (*Schizophyllum sabulosum*), while for aquatic detritivores they ranged from −26.4 ± 0.92‰ (*Cloeon dipterum*) to −21.3‰ (*Mystacides longicornis*; n = 1). All predatory spiders had $\delta^{13}C$ values similar to the aquatic organisms, ranging from –23.2 ± 1.0‰ (*Clubiona* sp.) to –22.0 ± 1.5‰ (*Tetragnatha* sp.), except for Lycosidae which appeared more depleted (−26.6‰; n = 1), with a value closer to that of the terrestrial organisms.

Average isotopic $\delta^{15}N$ signatures of terrestrial herbivores and detritivores (range −1.92 to 14.6‰) were generally lower than those of the aquatic ones (range 6.99 to 12.9‰). After calculating the trophic position of both food webs separately based on the base $\delta^{15}N$ signatures of the main herbivores in each system, the trophic position (± SEM) of the consumers showed similar ranges between the two parts of the food web. For the herbivores, the trophic position ranged from 1.4 ± 0.08 (*A. rufus*) to 2.6 ± 0.01 (pooled *C. nemoralis* and *C. hortensis*) for the terrestrial and from 1.5 ± 0.13 (*Corixa* spp.) to 2.4 ± 0.02 (zooplankton) for the aquatic organisms. The trophic position of the detritivores was also similar, except for the millipede *S. sabulosum*, which had the lowest trophic position of 1.1 ± 0.003. Trophic position values of the other two terrestrial detritivores were 2.7 ± 0.08 (Lumbricidae) and 2.9 ± 0.2 (Oniscidea), while for the aquatic detritivores they ranged from 2.6 ±

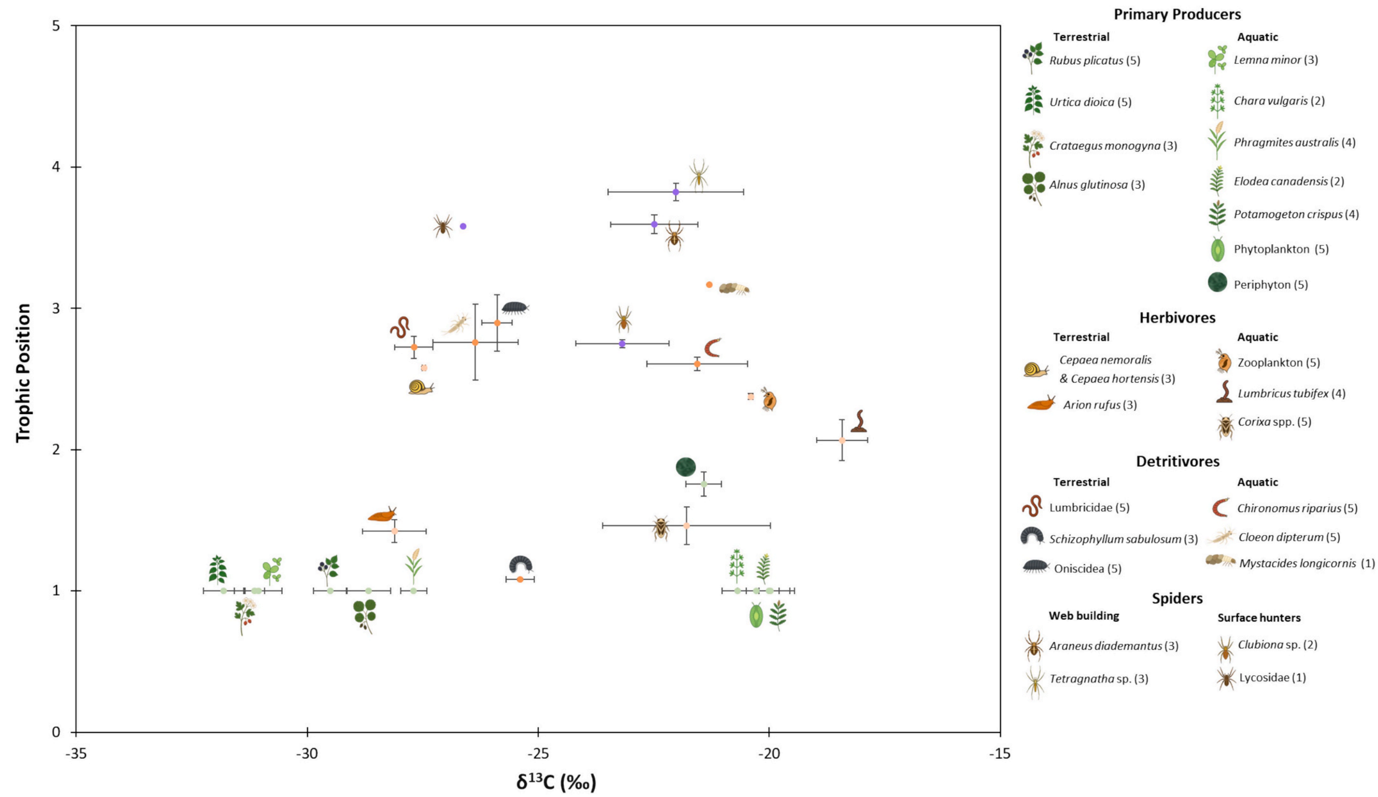

Fig. 1. Structure of the terrestrial and aquatic food web of the Lake Blokkersdijk ecosystem. Data points show average Trophic Position (TP) and δ^{13}C values and error bars represent the corresponding Standard Error of the Mean (SEM) for organisms from the five different sample sites in the area. TPs were calculated using the δ^{15}N stable isotope signatures, as explained in section 2.2. The different colors of the data points correspond to the different trophic categories of the organisms, including: primary producers (green), herbivores (light peach), detritivores (orange) and predatory spiders (purple). The legend on the right indicates the taxon that corresponds to each icon and the number of field replicates is shown in parenthesis. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

0.05 (*Chironomus riparius*) to 3.2 (*M. longicornis*; n = 1). The trophic position of all spiders, except for *Clubiona* sp., was higher compared to all other organisms, extending from 2.8 ± 0.03 (*Clubiona* sp.) to 3.8 ± 0.06 (*Tetragnatha* sp.).

3.2. PFAS profiles in the organisms

The numbers of unique PFAS quantified in each of the five organism groups ranked from highest to lowest were: 37 (aquatic primary producers) > 36 (terrestrial consumers) > 34 (aquatic consumers) > 26 (spiders) > 22 (terrestrial primary producers) (Section S5, Figs. S1-S5; Table S3). PFAS were then grouped into five subclasses, leading to the PFAS profiles described below. All PFAS concentrations were reported on a dry weight basis (ng/g dw), while the concentrations and profiles of PFAS in the abiotic environment were reported by Gkika et al. (2025).

3.2.1. Primary producers

All primary producers contained numerous PFAS, with varying profiles and individual PFAS concentrations (Fig. 2A-2B). In the terrestrial primary producers, the number of quantified PFAS ranged from 13 (*Crataegus monogyna;* 30% of quantified PFAS) to 20 (*U. dioica*; 46%). All terrestrial primary producers had very similar PFAS profiles, clearly dominated by short-chain PFCAs (> 92% in all cases) (Fig. 2A). The ∑PFAS concentrations in terrestrial primary producers were within the same order of magnitude, ranging from 319 (*C. monogyna*) to 901 ng/g dw (*Rubus plicatus*). TFA and PFBA always had the highest concentrations, ranging from 219 (*C. monogyna*) to 570 ng/g dw (*R. plicatus*) for TFA and from 79 (*C. monogyna*) to 281 ng/g dw (*R. plicatus*) for PFBA.

In the aquatic primary producers, the number of quantified PFAS ranged from 12 in the macrophyte *Elodea canadensis* (29%) to 31 in both periphyton and phytoplankton (74%). The PFAS profiles were more variable in the aquatic compared to the terrestrial primary producers (Fig. 2B). The PFAS profiles of *E. canadensis*, *L. minor* and *P. crispus* were dominated by long-chain PFSAs (88%, 75% and 57% respectively), with short-chain PFCAs also contributing considerably to the PFAS load in *P. crispus* (37%). In the remaining aquatic primary producers, short-chain PFCAs accounted for the majority of the total PFAS concentration (*Chara vulgaris*: 56%; *Phragmites australis*: 53%; periphyton: 44%; phytoplankton: 42%), but still with a considerable contribution of long-chain PFSAs (range 27% to 37%). A higher number of detected PFAS did not always translate into a higher ∑PFAS concentration. The ∑PFAS concentrations varied more than two orders of magnitude between the different aquatic primary producers, ranging from 59 (*P. australis*) up to 7,243 ng/g dw (*P. crispus*). The long-chain PFAS L-PFOS and PFBA were always among the top three compounds with the highest concentrations, ranging from 11 (*P. australis*) to 3,411 ng/g dw (*P. crispus*) for L-PFOS, and from 20 (*P. australis*) to 1,446 ng/g dw (*C. vulgaris*) for PFBA. The ultrashort-chain TFA was also among the compounds with the highest concentrations in some aquatic primary producers, reaching 1,985 ng/g dw in *C. vulgaris*.

3.2.2. Consumers

The number of quantified PFAS in the terrestrial consumers ranged from 12 in the pooled snails *C. nemoralis* and *C. hortensis* (27%) to 34 in the millipede *Schizophyllum sabulosum* (77%). Short-chain PFCAs was the most represented subclass in *C. nemoralis* and *C. hortensis* (60%), *A. rufus* (53%), and Lumbricidae (49%), followed by precursors (*A. rufus*; 25%) or long-chain PFCAs, which had a considerable contribution to the total PFAS load, especially in Lumbricidae (45%) (Fig. 3A). The PFAS profile in the millipede *S. sabulosum* was dominated by long-

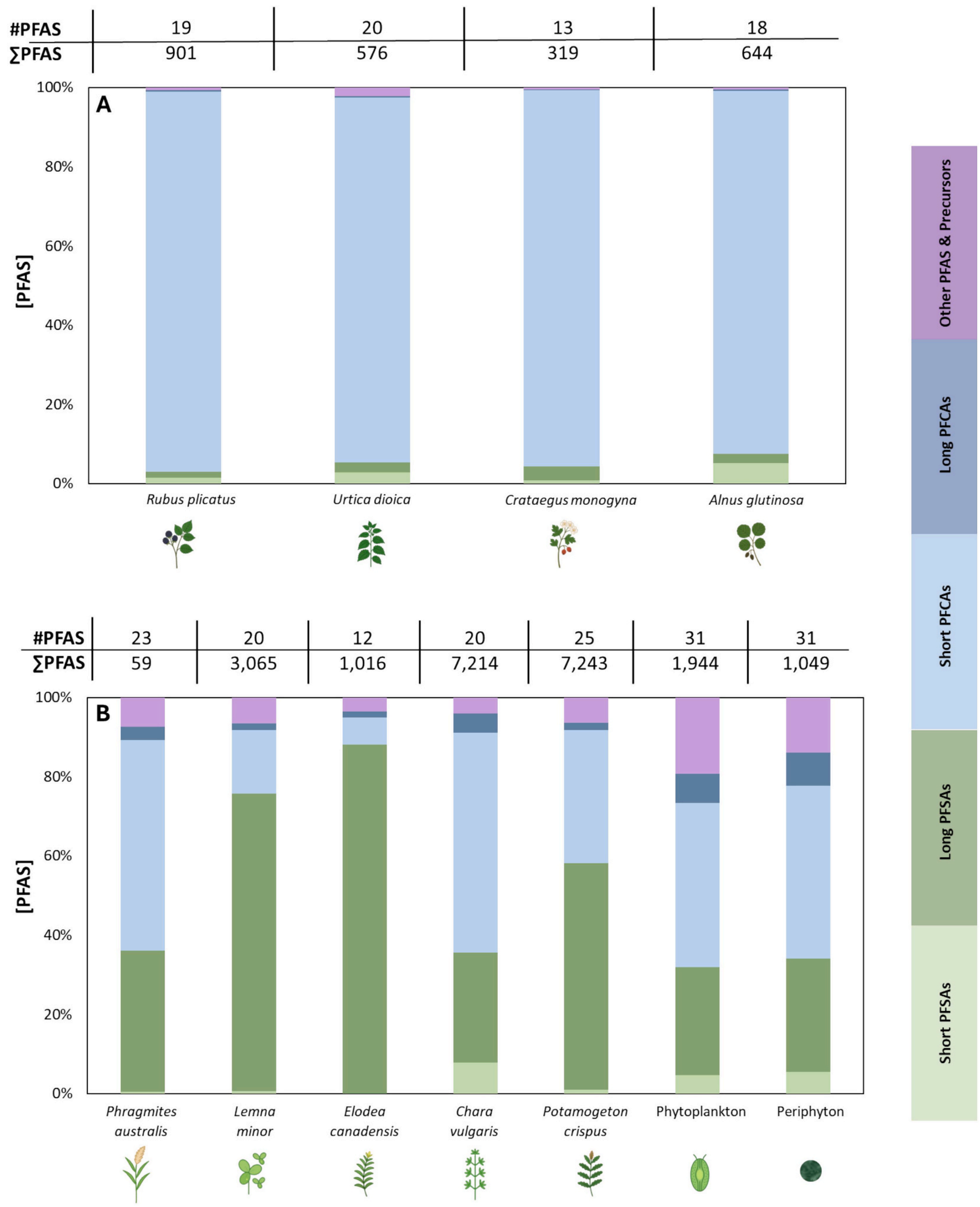

Fig. 2. Relative PFAS profiles per subclass (dry weight basis) in terrestrial (A) and aquatic (B) primary producers sampled from the Lake Blokkersdijk ecosystem. The different colors correspond to the five different PFAS subclasses defined in the legend on the right. Above each bar, the number of quantified PFAS (#PFAS) and the total average PFAS concentration (∑PFAS; in ng/g dry weight) are indicated.

chain PFSAs (41%), followed by short-chain PFCAs and precursors with comparable contributions (~20%). The ∑PFAS concentrations differed substantially between the terrestrial consumers and ranged from 306 (pooled *C. nemoralis* and *C. hortensis*) to 17,806 ng/g dw (Lumbricidae). TFA and L-PFOS were the two compounds with the highest concentrations, except in *A. rufus*, where TFA was followed by L-EtFOSAA. TFA concentrations ranged from 164 (pooled *C. nemoralis* and *C. hortensis*) to 7,898 ng/g dw (Lumbricidae), while L-PFOS concentrations ranged from 66 (pooled *C. nemoralis* and *C. hortensis*) to 5,839 ng/g dw (Lumbricidae).

The number of quantified PFAS in aquatic consumers ranged from 14 in *Mystacides longicornis* (33%) to 33 in *Corixa* spp. (79%). PFAS profiles were quite consistent between these taxa, with long-chain PFSAs making up more than 75% in all cases, except for *M. longicornis* (56%), followed by precursors, a subclass that had a considerable contribution to the total PFAS load in *M. longicornis* (36%) (Fig. 3B). Despite the rather homogenous profiles, ∑PFAS concentrations varied over an order of magnitude between the aquatic consumers, ranging from 941 (*M. longicornis*) to 23,951 ng/g dw (mayfly *C. dipterum*). L- and Br-PFOS together with some sulfonamide-based precursors (FBSA, FOSA, or L-EtFOSAA) were among the compounds with the highest concentrations in all aquatic consumers, except for *C. riparius*, where instead of Br-PFOS, PFBA dominated (102 ng/g dw). Concentrations of L- and Br-PFOS ranged from 310 (*M. longicornis*) to 7,696 ng/g dw in *C. dipterum* and from 148 in *M. longicornis* to 5,634 ng/g dw in *C. dipterum*.

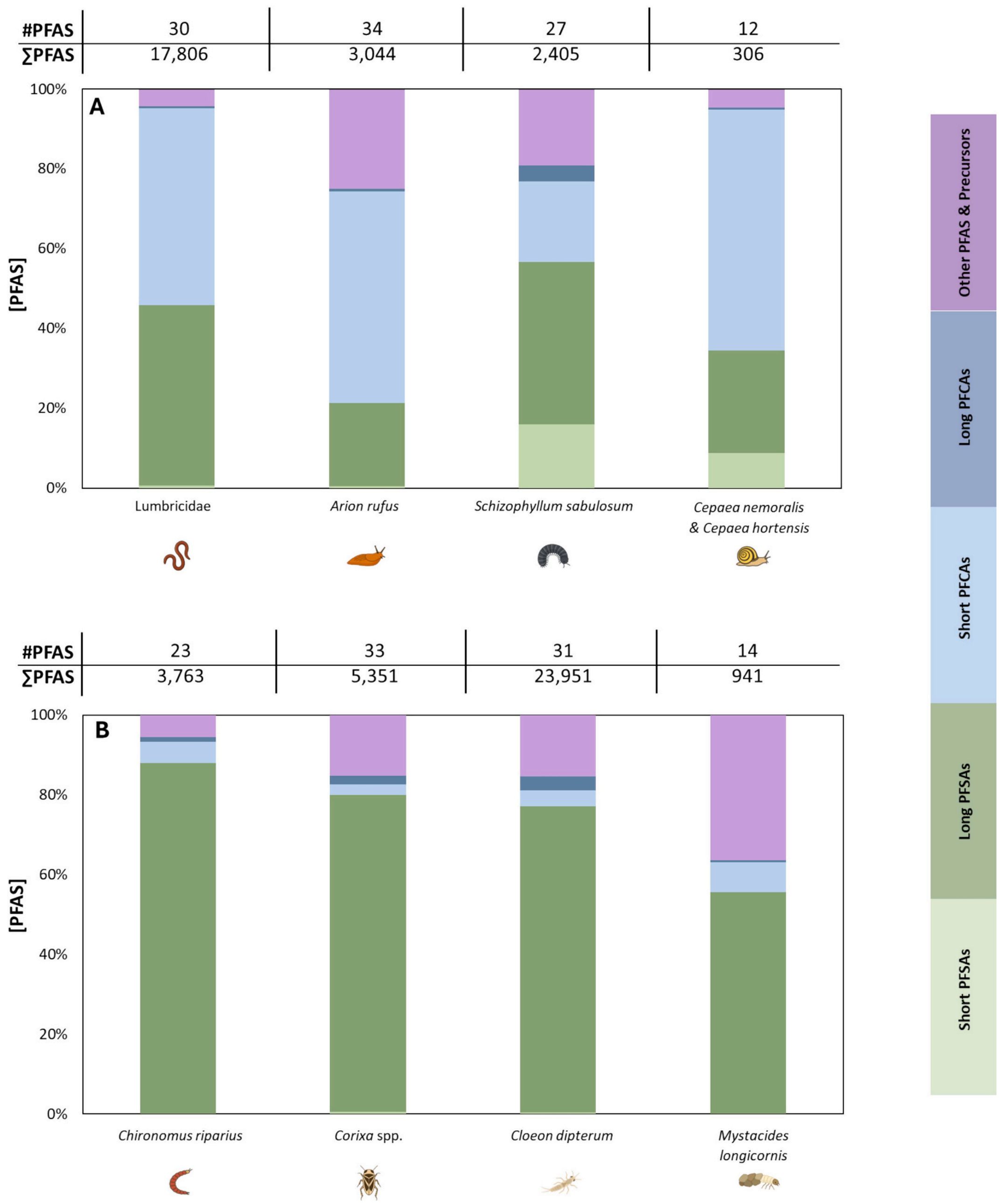

Fig. 3. Relative PFAS profiles per subclass (dry weight basis) in terrestrial (A) and aquatic (B) consumers (herbivores + detritivores) sampled from the Lake Blokkersdijk ecosystem. The different colors correspond to the five different PFAS subclasses defined in the legend on the right. Above each bar the number of quantified PFAS (#PFAS) and the total average PFAS concentration (∑PFAS; in ng/g dry weight) are indicated.

3.2.3. *Spiders*

In the predatory spiders the number of quantified PFAS ranged from 15 in *Clubiona* sp. (34%) to 22 in *Araneus diademantus* (50%). In all cases, profiles were dominated by short-chain PFCAs, reaching up to more than 80% in three out of the four spider taxa (Fig. 4). In Lycosidae, long-chain PFSAs contributed considerably to the PFAS load (32%), while the subclass of long-chain PFCAs was only present in the *Clubiona* sp.. The ∑PFAS concentrations ranged from 2,177 (*A. diademantus*) up to 18,272 ng/g dw (Lycosidae). TFA and L-PFOS showed the highest concentrations in all individual spider taxa, ranging from 1,810 to 14,453 ng/g dw for TFA and from 92 to 10,170 ng/g dw for L-PFOS.

3.3. *Trophic magnification*

All significant regressions for the two parts of the ecosystem in the absence and presence of the spiders and the combined terrestrial-aquatic ecosystem in the presence of spiders can be found in the SI (Section S6, Figs. S6-S8). Concerning the food web without the spiders, 16 and 20 PFAS met the LOQ criterion, but only four and two PFAS fulfilled the significance criterion, for the terrestrial and aquatic food web, respectively. When examining the complete food web, including the spiders, 14 and 20 PFAS fulfilled the LOQ criterion, while nine and three met the significance criterion for the terrestrial and aquatic food web, respectively. When examining the combined terrestrial-aquatic food web in the presence of spiders, 11 PFAS fulfilled the LOQ criterion, while six of them met the significance criterion (Fig. 5; Table 1). Statistical parameters for all regression models can be found in the SI (Figs. S6-8; Table S4). Results from the sensitivity analysis showed that substitution by either 0.5*LOQ or 1*LOQ had generally < 10% impact on the estimated TMF/TDF, with only two exceptions (38% difference for PFPrA;

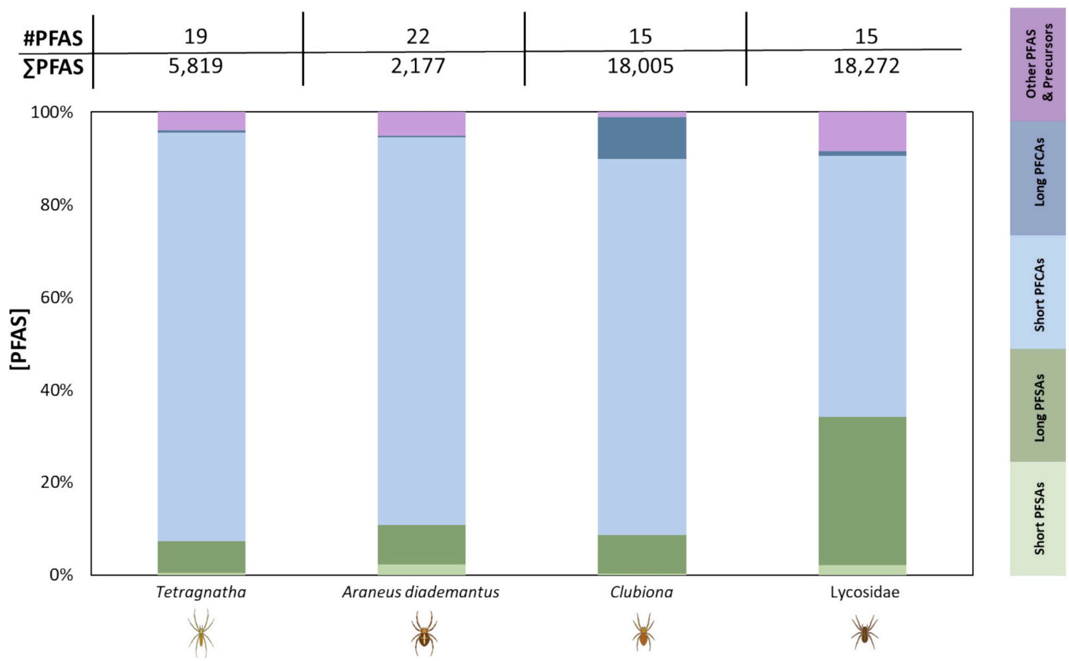

Fig. 4. Relative PFAS profiles per subclass (dry weight basis) in predatory spiders sampled on the banks of Lake Blokkersdijk. The different colors correspond to the five different PFAS subclasses defined in the legend on the right. Above each bar the number of quantified PFAS (#PFAS) and the total average PFAS concentration (∑PFAS; in ng/g dry weight) are indicated.

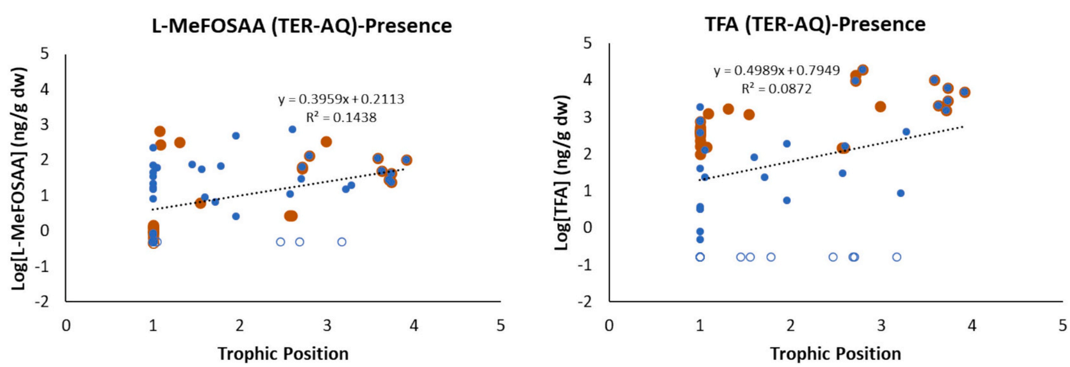

Fig. 5. Examples of regressions between the logarithmically transformed PFAS concentration and trophic position for the two compounds that showed trophic transfer in all scenarios including the spiders (terrestrial with spiders; aquatic with spiders; terrestrial-aquatic with spiders). Here the regressions for the terrestrial-aquatic scenario with spiders are shown. Terrestrial and aquatic data points are indicated in brown and blue, respectively. Hollow symbols indicate cases where PFAS concentrations were < LOQ and substituted with 0.5 * LOQ, while closed symbols are actual measured PFAS concentrations.

terrestrial without spiders, and −17% for FHxSA; aquatic without spiders) (Text S3, Table S5).

For the terrestrial food web in both scenarios, all PFAS with significant results showed trophic magnification, except for PFPrA in the absence of spiders that showed trophic dilution (TDF: 0.21). For the aquatic food web there were only five significant regressions, with all but one (PFPrA in the presence of spiders; TDF: 0.28) indicating trophic magnification. Moreover, little overlap was observed between the PFAS that magnified across the terrestrial and aquatic food webs, with only TFA and L-MeFOSAA showing consistent trophic magnification in both food webs in the presence of spiders with TMFs of 2.46 and 6.93 for TFA and 3.37 and 2.11 for L-MeFOSAA, for the terrestrial and aquatic ecosystem, respectively. Long-chain PFSAs and sulfonamide-based precursors, which were the subclasses with the highest TMFs in the terrestrial food web, showed no significant magnification in most cases in the aquatic food web. Nonetheless, results were consistent for individual compounds, meaning that PFAS that were magnifying in the terrestrial food web were not found to dilute in the aquatic food web and *vice versa*.

When including the spiders, more PFAS exhibited significant trophic magnification, but with lower TMF values. The highest TMFs were found in the terrestrial food web excluding the spiders for L-PFOS (11.1), Br-PFOS (6.27) and FBSA (5.35). The lowest TMFs were found, for PFPeA (1.74), Br-PFHxS (2.04) in the terrestrial food web including the spiders and for L-MeFOSAA (2.11) in the aquatic food web including the spiders, which all belong to different PFAS subclasses. The only compound that exhibited trophic dilution was PFPrA (TDF: 0.21 in the terrestrial food web in the absence of spiders; TDF: 0.28 in the aquatic food web in the presence of spiders). The combined regressions in the presence of spiders revealed six PFAS with significant trophic magnification. For the combined regressions the lowest TMFs were found for FBSA (2.13) and Br-PFOS (2.15) and the highest for TFA (3.16), followed by L-PFOS (2.50). These six PFAS were also found to magnify across the terrestrial food web in the presence of spiders, with slightly higher TMFs, while two out of these six compounds, TFA and L-MeFOSAA, also magnified across the terrestrial food web in the presence of spiders. Hence, these

Table 1
The trophic magnification (TMF; green) and trophic dilution factors (TDF; peach) expressed as 10 to the power of the slope of the log-transformed regression of PFAS concentration against trophic position, determined for different PFAS in the terrestrial (TER) and aquatic (AQ) food webs of the Lake Blokkersdijk ecosystem in two trophic transfer scenarios (absence and presence of spiders) and in the combined terrestrial-aquatic food web in the presence of spiders. Only PFAS that had more than 50% of data points above the LOQ in at least one of the five examined scenarios (terrestrial without spiders; terrestrial with spiders; aquatic without spiders; aquatic with spiders; terrestrial-aquatic with spiders) are shown. N.S. indicates that there was no significant relationship between PFAS concentration and trophic level. Hatched cells indicate that the 50% > LOQ criterion was not met. PFAS were grouped into four subclasses (PFCAs, PFSAs, sulfonamides and sulfonamidoacetic acids) and within each subclass ordered according to chain length from shortest to longest. The total number of carbons, number of fluorinated carbons and functional groups are indicated in columns two to four. The total number of PFAS with positive (trophic magnification), negative (trophic dilution) and non-significant values are summarized at the bottom of the table.

PFAS list	# C total	# C-F	PFAS Functional groups	TER Absence	TER Presence	AQ Absence	AQ Presence	TER-AQ Presence
PFBS	4	4	SO3H	N.S.	N.S.			
PFPeS	5	5	SO3H	N.S.		N.S.	N.S.	
L-PFHxS	6	6	SO3H			N.S.	N.S.	
Br-PFHxS	6	6	SO3H	N.S.	**2.04**	N.S.	N.S.	N.S.
6:2FTS	8	6	SO3H			N.S.		
L-PFHpS	7	7	SO3H			N.S.	N.S.	
L-PFOS	8	8	SO3H	**11.13**	**3.80**	N.S.	N.S.	**2.50**
Br-PFOS	8	8	SO3H	**6.27**	**3.62**		N.S.	**2.15**
PFECHS	8	8	SO3H; Cyclic			N.S.		
PFDS	10	10	SO3H			N.S.	N.S.	
TFA	2	1	COOH	N.S.	**2.46**	N.S.	**6.93**	**3.16**
PFPrA	3	2	COOH	**0.21**		N.S.	**0.28**	
PFBA	4	3	COOH	N.S.	N.S.	N.S.	N.S.	N.S.
PFPeA	5	4	COOH	N.S.	**1.74**	N.S.	N.S.	N.S.
PFHxA	6	5	COOH			N.S.	N.S.	
PFHpA	7	6	COOH	N.S.	N.S.	N.S.	N.S.	N.S.
L-PFOA	8	7	COOH	N.S.	**2.45**	N.S.	N.S.	N.S.
PFNA	9	8	COOH	N.S.	N.S.			
PFDA	10	9	COOH			**2.48**	N.S.	
FBSA	4	4	O=S=O; NH2	**5.35**	**2.47**	N.S.	N.S.	**2.13**
FHxSA	6	6	O=S=O; NH2			**3.11**	N.S.	
FOSA	8	8	O=S=O; NH2			N.S.	N.S.	
L-MeFOSAA	11	8	O=S=O; N-CH3; COOH	N.S.	**3.37**	N.S.	**2.11**	**2.49**
L-EtFOSAA	12	8	O=S=O; N-C2H5; COOH	N.S.	**3.60**		N.S.	**2.16**
Br-EtFOSAA	12	8	O=S=O; N-C2H5; COOH	N.S.	N.S.			
			# TMF	**3**	**9**	**2**	**2**	**6**
			# TDF	**1**	**0**	**0**	**1**	**0**
			# N.S.	**12**	**5**	**18**	**17**	**5**

two compounds showed trophic magnification in all three spider containing scenarios (terrestrial with spiders; aquatic with spiders; terrestrial-aquatic with spiders).

4. Discussion

4.1. PFAS profiles in the different trophic groups

Generally, very high total PFAS concentrations were measured in the organisms living in the studied contaminated ecosystem, with previously reported high bioaccumulation factors (Gkika et al., 2025). Among the terrestrial primary producers, despite some variation in the number of detected PFAS, levels and profiles were very similar, and by far dominated by short-chain PFCAs. This hints towards a common uptake mechanism in the plants, mainly via the soil pore water and the subsequent aqueous flow through the stems to the leaves (Adu et al., 2023; Felizeter et al., 2021; Groffen et al., 2023). Since plants were rinsed with ambient water upon collection, the contribution of particle-sorbed PFAS to the PFAS load of the plants can be considered negligible. For the aquatic primary producers, long-chain PFSAs and short-chain PFCAs were the dominant subclasses. The high variation in the number of detected PFAS, concentrations and profiles in the aquatic primary producers was even more prominent than previously reported (Chu et al., 2022; Griffin et al., 2023; Huang et al., 2022; Wang et al., 2025). Even in taxa with similar ∑PFAS concentrations, profiles could differ substantially (e.g. *Chara vulgaris* and *Potamogeton crispus*; or *Elodea canadensis* and periphyton). Moreover, the occurrence of a higher number of PFAS did not always translate into higher PFAS concentrations (e.g. *Phragmites australis* and *E. canadensis*). All primary producer groups containing algae (*C. vulgaris*, phytoplankton and periphyton) had a larger share of short PFSAs compared to vascular aquatic plants. The deviating PFAS profile in the floating *Lemna minor* may be due to its position at the water–air interface. Although PFAS uptake from the sediment by the rooting macrophytes may explain the quite distinct profiles between pelagic and benthic primary producer taxa, the large variation in both ∑PFAS concentration and profiles among the sediment rooting taxa (*P. australis*, *E. canadensis*, *P. crispus*, and *C. vulgaris*) indicates a much more complex interplay of uptake mechanisms compared to terrestrial plants (Colomer-Vidal et al., 2022; Gkika et al., 2025).

Among the terrestrial consumers, the numbers of detected PFAS, levels and profiles were quite different. This aligns with earlier findings from terrestrial field studies, where species-specific differences in PFAS concentrations and bioaccumulation were found (Buytaert et al., 2025; Groffen et al., 2023; Hopkins et al., 2023; Koch et al., 2020). In our case, and despite including a gut depuration step, the Lumbricidae had by far the highest ∑PFAS concentrations of all terrestrial consumers, which aligns with previously reported observations (Buytaert et al., 2025; Ibor et al., 2025). This could relate to the extensive dermal contact of this organism with the soil. In general, higher variation was expected for animals compared to primary producers, as they can move and adjust their diets according to the availability of resources (Szałkiewicz et al., 2022). Although clearly visible in the terrestrial ecosystem, this pattern was less clear in the aquatic ecosystem. In aquatic consumers, the PFAS profiles were quite consistent, but the number of detected PFAS and the total PFAS concentrations were not. The nymphs of *Cloeon dipterum* had by far the highest ∑PFAS concentration, which could relate to high exposure and uptake of PFAS through their extensive exterior gills. An additional explanation may be their distinct diet compared to the other aquatic organisms, as shown by their different $\delta^{13}C$ signature compared to the other aquatic invertebrates. The Trichoptera *Mystacides longicornis* (n = 1) had the lowest ∑PFAS concentration compared to all other consumers. Since this is a case-building species (Morse 2009), uptake of PFAS from the water could be limited compared to other aquatic invertebrates resulting in its lower PFAS concentrations (Higler 2008). Nonetheless, since only a single field replicate of *M. longicornis* was collected, results for this organism should be interpreted with some caution. Besides varying exposure to the different environmental compartments, the differences in the number of detected PFAS, levels and profiles observed between terrestrial and aquatic consumers could also relate to their body composition, as well as to their species-specific feeding strategies, which in turn may depend on their taxonomy and life-stage.

The riparian spiders contained overall higher ∑PFAS concentrations compared to the consumers, with only the ∑PFAS concentrations in *C. dipterum* and Lumbricidae exceeding that of some individual spider taxa. This is in contrast to patterns observed for riparian spiders along a stream, containing ∑PFAS concentrations that were around 75% of those in aquatic invertebrate consumers (Campbell et al., 2026). PFAS profiles were rather consistent among the different spider taxa, however, this was not the case for the ∑PFAS concentrations and the number of detected PFAS. Both surface hunter taxa (*Clubiona* sp. and Lycosidae) had much higher ∑PFAS concentrations than the web building spiders, while in the latter a higher number of PFAS were quantified. Since the collected spiders had comparable body sizes, these observed differences are expected to result from differences in exposure, activity, and food intake. The most obvious difference is that web building spiders are exposed via air and food and the hunting spiders also to the PFAS in the soil (Buytaert et al., 2025). Moreover, the larger food requirement of hunting spiders due to their higher energy turnover compared to web building spiders (Schmitz 2016), could have resulted in higher ∑PFAS concentrations. Additionally, the observed difference in PFAS profiles between the spiders could also relate to taxon-specific differences in diet reliance on emergent aquatic insects (Chumchal et al., 2022). The $\delta\ ^{13}C$ signatures of the Lycosidae were lower than those of the other spiders, indicating a higher reliance on terrestrial food sources. The $\delta\ ^{13}C$ signatures of *Clubiona* sp. and the web-building spiders overlapped, indicating a similar reliance on terrestrial and aquatic food sources.

4.2. Trophic transfer of PFAS

Examining the trophic transfer of PFAS at the base of the food web, i.e. plants and primary consumers only, revealed that a higher trophic position did not always translate into higher concentrations of individual PFAS. Some compounds did show statistically significant trophic magnification, with high TMFs especially in the terrestrial food web, highlighting that some bioaccumulative PFAS also have the potential to magnify across the food chain. To examine this in more detail, we evaluated for which of the six PFAS with significant trophic transfer in the combined terrestrial-aquatic regressions this was also the case in either the terrestrial or the aquatic food web in the presence of the spiders. This comparison revealed that for two out of the six PFAS, L-MeFOSAA and TFA, trophic magnification took place in all three scenarios. For the other four compounds, no significant trophic magnification was observed when examining the aquatic food web. Similarly to previous studies reporting TMFs (e.g. Li et al., 2026; Shaffer et al., 2025), we were not able to identify a common characteristic shared by the compounds with significant positive, significant negative or non-significant regressions between PFAS concentrations and trophic positions. The only exception was the terrestrial ecosystem in the absence of spiders, the scenario with most data points, where the membrane-to-water (K_{mw}) partition coefficient appeared to explain more than 80% of the variation in TMF (Fig. S9). These findings confirm the distinct, compound-specific PFAS trophic transfer trends, as reported by Ricolfi et al. (2025).

Distinguishing between environmental and dietary exposure was not possible with the current study design. Especially if the environment already contains high concentrations of PFAS, which was the case for some compounds here, it is even harder to distinguish between different exposure routes. Previous studies suggested that for higher trophic levels dietary uptake seems to play a bigger role than environmental assimilation (Arnot and Gobas, 2006; Chumchal et al., 2022; Koch et al., 2020; Mackay et al., 2016; Politi et al., 2021; Rosenberg and Ramus, 1984). However, for the lower trophic levels this perspective is different. Environmental exposure is expected to play a substantial role for these organisms (Campbell et al., 2026), due to their high surface area-to-volume ratio, high skin permeability and the short diffusion distances that contaminants need to cross. This argument is supported by a recent study, where partitioning resulted in the greatest PFAS enrichment for aquatic organisms (Campbell et al., 2026). The high bioaccumulation of PFAS in primary producers and primary consumers (Gkika et al., 2025), together with the limited trophic transfer along the basal levels of the food web observed in this study, emphasize this different perspective for lower trophic levels. The high variation of the individual PFAS concentrations in primary producers, all sharing the same trophic position, could have played a role in rendering the TMF estimations of many compounds not significant. Hence, the PFAS incorporation at the base of (aquatic) food webs requires further elucidation.

When introducing the spiders as joint predators in the combined terrestrial-aquatic food web, the trophic transfer patterns changed substantially. More PFAS exhibited significant trophic magnification in the terrestrial food web compared to the scenario without spiders, but the obtained TMF values were lower. Adding the spiders to the aquatic food web resulted in different compounds showing significant trophic transfer, but overall, there were only few significant regressions for the aquatic food web with or without the spiders. A possible reason for these observations could relate to the way spiders consume their prey, which is first pierced or crushed using their fangs and maxillae and digested externally to a liquid form, which is then sucked, discarding the hard parts (Foelix, 1996; Forster & Forster, 1999; Nyffeler et al., 1994). Hence, some PFAS could remain in the parts of the prey that are not consumed. The lack of significant trophic transfer may also be due to the emergent insects that comprise a considerable part of the diet of spiders (Burdon & Harding, 2008; Collier et al., 2002; Paetzold et al., 2006). PFAS concentrations tend to differ between the larvae and nymphs living in the water and the emerged adult insects. Adults may leave part of the contaminants behind in the pupal or nymphal skin (Koch et al., 2020; 2021), while for some PFAS bioamplification in adults was observed (Kotalik et al., 2025). Furthermore, the PFAS load in aquatic organisms was much higher compared to that in terrestrial organisms. Although the exact diet composition of the spiders could not be determined, all spider taxa were collected from the riparian zone, which implies that they consume both aquatic and terrestrial resources (Bollinger et al., 2023; Krell et al., 2015). Based on the isotopic signature of three spider taxa compared to these resources, we expected about equal contributions, as shown for both web building and surface hunting riparian spiders (Collier et al., 2002).

TFA concentrations were highest in the spiders, in which it contributed more than 80% to the total concentration in three out of the four taxa (*Tetragnatha* sp., *A. diademantus*, *Clubiona* sp.) and 56% in Lycosidae, although for the latter taxa this is based on a single field replicate. Moreover, TFA was the only short-chain compound that showed trophic transfer in both food webs in the presence of spiders, accompanied by a very high TMF in the aquatic food web. Short-chain PFAAs are commonly found in low trophic positioned organisms (Simmonet-Laprade et al., 2019; Du et al. 2021) and our study focused on the basal levels of the food web (i.e. no vertebrates included). Still, our results contradict the common opinion that the small, mobile and polar TFA is unlikely to bioaccumulate or magnify in biota (Hanson et al., 2024; Herzke et al., 2023).

The observed elevated absolute concentrations and TMFs of TFA at lower trophic levels are likely a result from synergy of (non-classical) magnification mechanisms. These processes may include non-specific protein binding, especially relevant in organisms with low adipose tissue like spiders, in combination with electrostatic interactions, particularly important for small PFAS (Bangma et al., 2023; Birchfield et al., 2025). Taking up the prey fluids in liquid form may also have contributed to the TFA-dominated PFAS profile in the spiders, as the polar TFA presumably accumulates in the tissue fluids of organisms, making it mobile between different parts of organisms (e.g. transpiration pathway in plants). Spiders seem to be selectively exposed to the dissolved, mobile PFAS fraction via fluid feeding (e.g. Nyffeler et al., 1994), while simultaneously eliminating or not accumulating other PFAS as efficiently, resulting in an apparent dominance of TFA. Our approach did not determine individual predator–prey enrichment, but system-wide trophic trends, which may mask selective processes. The unexpectedly high bioaccumulation of TFA indicates that better insights into mechanistic processes of resource acquisition by invertebrates are needed to understand trophic transfer of (ultra) short-chain PFAS.

We acknowledge that the TMFs calculated here for TFA may be overestimating the trophic transfer of this compound, since more processes may contribute to its high concentrations in the spiders. The experimental design, however, did not allow us to distinguish if the TFA in the spiders and the other biota resulted from the (bio)transformation of other PFAS or originated from dietary or environmental exposure. Nonetheless, the latter was expected to be limited for the spiders and the TFA concentrations in the environment were not substantially higher than those of the other PFAS. Moreover, given the long half-lives and environmental stability of most PFAS relative to the short life spans of most invertebrates, biotransformation is not expected to contribute substantially to the very high TFA concentrations in the spiders. The significant trophic transfer of the supposedly non-bioaccumulating TFA in both parts of the food web together with the unpredictability of which other PFAS do magnify or dilute, show that we still lack the insights into the exact drivers of PFAS bioaccumulation and trophic transfer (Arp et al., 2024; Herzke et al., 2023; Li et al., 2026).

Studying the aquatic and terrestrial compartments of contaminated ecosystems in isolation does not do justice to the life cycle of the many organisms that inhabit the aquatic environment as larvae or nymphs and the terrestrial environment as flying adults (Kraus et al., 2023), which is characteristic for a wide range of aquatic insect orders. Our findings further support this, clearly indicating that PFAS bioaccumulation and trophic transfer were affected in a compartment-specific, rather than a compound-specific way. The few studies that to the best of our knowledge have jointly examined PFAS trophic transfer in more than one ecosystem type acknowledge the challenge to understand PFAS exposure routes for the multiple species comprising these food webs (Campbell et al., 2025; Dai et al., 2025; Hopkins et al., 2023; Ibor et al., 2025), but also showed the added value of such studies. Hence, the current combined aquatic-terrestrial study on a variety of organisms and PFAS proved to be essential to unravel the complex environmental distribution and trophic transfer of these compounds. It should be noted though that the predators in the present study were invertebrate spiders and it is therefore expected that top predators would be even more impacted by the trophic transfer of PFAS, since adverse effects associated with PFAS exposure have been reported in terrestrial (Custer 2021; Sebastiano et al., 2023) and aquatic vertebrates (Ma et al., 2023).

5. Conclusions

The present study revealed the omnipresence and variable distribution of PFAS in terrestrial and aquatic organisms of a contaminated ecosystem. Some PFAS did transfer along the food chain, and specifically L-MeFOSAA and TFA showed trophic magnification in the combined terrestrial and aquatic basal food chains. Nonetheless, the mechanisms and drivers that determine which substances do show trophic magnification and which don't remain to be elucidated. This study emphasized the need for integrated terrestrial-aquatic approaches for PFAS environmental risk assessment.

CRediT authorship contribution statement

Ioanna S. Gkika: . **J.Arie Vonk:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Conceptualization. **Thomas L. ter Laak:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Cornelis A.M. van Gestel:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization. **Jildou Dijkstra:** Writing – review & editing, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Thimo Groffen:** Writing – review & editing, Methodology, Investigation. **Lieven Bervoets:** Writing – review & editing. **Michiel H.S. Kraak:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Thanks are due to Mohammad Sadia, Rick Helmus, Patricia Aguilar Alarcon, Samira Absalah, Eugenie Troia, Ingrida Bagdonaite, Rutger van Hall, Bram Ebben and Eva de Rijke for help in the practical work and data analysis. We would also like to thank Natuurpunt for granting us access and allowing us to perform the fieldwork at Blokkersdijk nature reserve. This research was financed by the Open Technology Program of The Netherlands Organization for Scientific Research (NWO), domain Applied and Engineering Sciences (TTW) under project number 18725. Thimo Groffen is a postdoctoral researcher (Research Foundation Flanders (FWO), grant nr: 1205724N).

Appendix A. Supplementary data

Additional methodological details, supporting figures, and tables containing the raw data and measured concentrations used in this study. Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2026.110439.

Data availability

All data are available in the Supporting Information.

References

Adu, O., Ma, X., Sharma, V.K., 2023. Bioavailability, phytotoxicity and plant uptake of per-and polyfluoroalkyl substances (PFAS): a review. J. Hazard. Mater. 447, 130805. https://doi.org/10.1016/j.jhazmat.2023.130805.

Arnot, J.A., Gobas, F.A.P.C., 2006. A review of bioconcentration factor (BCF) and bioaccumulation factor (BAF) assessments for organic chemicals in aquatic organisms. Environ. Rev. 14 (4), 257–297. https://doi.org/10.1139/a06-005.

Arp, H.P.H., Gredelj, A., Glüge, J., Scheringerlan, M., Cousins, I.T., 2024. The global threat from the irreversible accumulation of Trifluoroacetic Acid (TFA). Environ. Sci. Technol. 58 (45), 19925–19935. https://doi.org/10.1021/acs.est.4c06189.

Bangma, J., Guillette, T.C., Bommarito, P.A., Ng, C., Reiner, J.L., Lindstrom, A.B., Strynar, M.J., 2023. Understanding the dynamics of physiological changes, protein expression, and PFAS in wildlife. Environ. Int. 159, 107037. https://doi.org/10.1016/j.envint.2021.107037.

Birchfield, A.S., Musayev, F.N., Castillo, A.J., Zorn, G., Fuglestad, B., 2025. Broad PFAS binding with fatty acid binding protein 4 is enabled by variable binding modes. JACS Au 5 (6), 2469–2474. https://doi.org/10.1021/jacsau.5c00504.

Bollinger, E., Zubrod, J.P., Englert, D., Graf, N., Weisner, O., Kolb, S., Schäfer, R.B., Entling, M.H., Schulz, R., 2023. The influence of season, hunting mode, and habitat specialization on riparian spiders as key predators in the aquatic-terrestrial linkage. Sci. Rep. 13, 22950. https://doi.org/10.1038/s41598-023-50420-w.

Borgå, K., Kidd, K.A., Muir, D.C.G., Berglund, O., Conder, J.M., Gobas, F.A.P.C., Kucklick, J., Malm, O., Powell, D.E., 2012. Trophic magnification factors: considerations of ecology, ecosystems, and study design. Integr. Environ. Assess. Manag. 8 (1), 64–84. https://doi.org/10.1002/ieam.244.

Brunn, H., Arnold, G., Körner, W., Rippen, G., Steinhäuser, K.G., Valentin, I., 2023. PFAS: forever chemicals—persistent, bioaccumulative and mobile. reviewing the status and the need for their phase out and remediation of contaminated sites. Environ. Sci. Eur. 35, 20. https://doi.org/10.1186/s12302-023-00721-8.

Burdon, F.J., Harding, J., 2008. The linkage between riparian predators and aquatic insects across a stream-resource spectrum. Freshw. Biol. 53, 330–346. https://doi.org/10.1111/j.1365-2427.2007.01897.x.

Buytaert, J., Eens, M., Bervoets, L., Groffen, T., 2025. Distribution of legacy and emerging PFASs in a terrestrial ecosystem located near a fluorochemical manufacturing facility. Toxics 13 (8), 689. https://doi.org/10.3390/toxics13080689.

Byns, C., Teunen, L., Groffen, T., Lasters, R., Bervoets, L., 2022. Bioaccumulation and trophic transfer of perfluorinated alkyl substances (PFAS) in marine biota from the Belgian North Sea: distribution and human health risk implication. Environ. Pollut. 311, 119907. https://doi.org/10.1016/j.envpol.2022.119907.

Campbell, K.S., Wesner, J.S., Walters, D.M., Helton, A.M., Baranovic, A.M., Agrawal, A., Stricker, C.A., Provatas, A.A., Krausm, J.M., Briggs, M.A., Brandt, J.E., 2026. An ecosystem-scale model of PFAS dynamics in stream-to-riparian food webs. Environ. Sci. Technol. 60 (14), 11022–11035. https://doi.org/10.1021/acs.est.5c16942.

Chu, K., Lu, Y., Zulin Hua, Z., Liu, Y., Ma, Y., Gu, L., Gao, C., Yu, L., Wang, Y., 2022. Perfluoroalkyl acids (PFAAs) in the aquatic food web of a temperate urban lake in East China: Bioaccumulation, biomagnification, and probabilistic human health risk. Environ. Pollut. 296, 118748. https://doi.org/10.1016/j.envpol.2021.118748.

Chumchal, M.M., Beaubien, G.B., Drenner, R.W., Hannappel, M.P., Mills, M.A., Olson, C. I., Otter, R.R., Todd, A.C., Walters, D.M., 2022. Use of riparian spiders as sentinels of persistent and bioavailable chemical contaminants in aquatic ecosystems: a review. Environ. Toxicol. Chem. 41 (3), 499–514. https://doi.org/10.1002/etc.5267.

Collier, K.J., Bury, S., Gibbs, M., 2002. A stable isotope study of linkages between stream and terrestrial food webs through spider predation. Freshw. Biol. 47, 1651–1659. https://doi.org/10.1046/j.1365-2427.2002.00903.x.

Colomer-Vidal, P., Jiang, L., Mei, W., Luo, C., Lacorte, S., Rigol, A., Zhang, G., 2022. Plant uptake of perfluoroalkyl substances in freshwater environments (Dongzhulong and Xiaoqing Rivers, China). J. Hazard. Mater. 421, 126768. https://doi.org/10.1016/j.jhazmat.2021.126768.

Custer, C.M., 2021. Linking field and laboratory studies: reproductive effects of perfluorinated substances on avian populations. Integr. Environ. Assess. Manag. 17 (4), 690–696. https://doi.org/10.1002/ieam.4394.

Dai, S., Zhang, G., Dong, C., Yang, R., Pei, Z., Li, Y., Li, A., Zhang, Q., Jiang, G., 2025. Occurrence, bioaccumulation and trophodynamics of per- and polyfluoroalkyl substances (PFAS) in terrestrial and marine ecosystems of Svalbard. Arctic. Water Resour. 271, 122979. https://doi.org/10.1016/j.watres.2024.122979.

Daley, J.M., Corkum, L.D., Drouillard, K.G., 2011. Aquatic to terrestrial transfer of sediment associated persistent organic pollutants is enhanced by bioamplification processes. Environ. Toxicol. Chem. 30 (9), 2167–2174. https://doi.org/10.1002/etc.608.

Du, D., Lu, Y., Zhou, Y., Li, Q., Zhang, M., Han, G., Cui, H., Jeppesen, E., 2021. Bioaccumulation, trophic transfer and biomagnification of perfluoroalkyl acids (PFAAs) in the marine food web of the South China Sea. J. Hazard. Mater. 405, 124681. https://doi.org/10.1016/j.jhazmat.2020.124681.

Endicott, D., Silva-Wilkinson, R., McCauley, D., Armstrong, B., 2025. Per- and Polyfluoroalkyl Substances (PFAS) in sediment: a source of PFAS to the food web? Integr. Environ. Assess. Manag. 21 (4), 810–822. https://doi.org/10.1093/inteam/vjaf010.

European Food Safety Authority (EFSA), Arcella, D. & Gómez, Ruiz, J.A. (2018). Technical report on use of cut-off values on the limits of quantification reported in datasets used to estimate dietary exposure to chemical contaminants. EFSA supporting publication 2018:15(7):EN-1452. 11 pp. 10.2903/sp.efsa.2018.EN-1452.

Felizeter, S., Jürling, H., Kotthoff, M., de Voogt;, P., McLachlan, M.S., 2021. Uptake of perfluorinated alkyl acids by crops: results from a field study. Environ. Sci. Processes Impacts 23, 1158–1170. https://doi.org/10.1039/D1EM00166C.

Fiedler, H., Kallenborn, R., Boer, J., Sydnes, L., 2019. The Stockholm Convention: a tool for the global regulation of persistent organic pollutants. Chem. Int. 41, 4–11. https://doi.org/10.1515/ci-2019-0202.

Fiolka, F., Fuchs, T., Roodt, A.P., Manfrin, A., Schulz, R., 2025. Flood-borne pesticides are transferred from riparian soil via plants to phytophagous aphids. Chemosphere 377, 144355. https://doi.org/10.1016/j.chemosphere.2025.144355.

Fiolka, F., Roodt, A.P., Manfrin, A., Schulz, R., 2024. Flooding as a Vector for the transport of pesticides from streams to riparian plants. Environ. Sci. Technol. Water 4 (11), 4970–4977. https://doi.org/10.1021/acsestwater.4c00571.

Foelix, R.F., 1996. Biology of Spiders, 2nd ed. Oxford University Press, New York. https://archive.org/details/biologyofspiders00foel_0.

Forster, R., Forster, L., 1999. Spiders of New Zealand and Their Worldwide Kin. University of Otago Press, Dunedin.

Gkika, I.S., Kraak, M.H.S., van Gestel, C.A.M., ter Laak, T.L., van Wezel, A.P., Robert Hardy, R., Sadia, M., Vonk, J.A., 2024. Bioturbation affects bioaccumulation: PFAS uptake from sediments by a rooting macrophyte and a benthic invertebrate. Environ. Sci. Technol. 58 (46), 20607–20618. https://doi.org/10.1021/acs.est.4c03868.

Gkika, I.S., Vonk, J.A., ter Laak, T.L., van Gestel, C.A.M., Dijkstra, J., Groffen, T., Bervoets, L., Kraak, M.H.S., 2025. Strong bioaccumulation of a wide variety of PFAS in a contaminated terrestrial and aquatic ecosystem. Environ. Int. 202, 109629. https://doi.org/10.1016/j.envint.2025.109629.

Gkika, I.S., Xie, G., van Gestel, C.A.M., Vonk, J.A., van Wezel, A.P., Kraak, M.H.S., 2023. Research priorities for the environmental risk assessment of per- and polyfluorinated substances. Environ. Toxicol. Chem. 42, 2302–2316. https://doi.org/10.1002/etc.5729.

Griffin, E.K., Hall, L.M., Brown, M.A., Taylor-Manges, A., Green, T., Suchanec, K., Furman, B.T., Congdon, V.M., Wilson, S.S., Osborne, T.Z., Martin, S., Schultz, E.A., Holden, M.M., Lukacsa, D.T., Greenberg, J.A., Quiñones, K.Y.D., Lin, E.Z., Camacho, C., Bowden, J.A., 2023. Aquatic vegetation, an understudied depot for PFAS. J. Am. Soc. Mass Spectrom. 34 (9), 1826–1836. https://doi.org/10.1021/jasms.3c00018.

Groffen, T., Prinsen, E., Stoffels, D.-O.-A., Layla Maas, L., Pieter Vincke, P., Lasters, R., Eens, M., Bervoets, L., 2023. PFAS accumulation in several terrestrial plant and invertebrate species reveals species-specific differences. Environ. Sci. Pollut. Res. 30, 23820–23835. https://doi.org/10.1007/s11356-022-23799-8.

Guo, W., Pan, B., Sakkiah, S., Yavas, G., Ge, W., Zou, W., Tong, W., Hong, H., 2019. Persistent organic pollutants in food: contamination sources, health effects and detection methods. Int. J. Environ. Health Res. 16 (22), 4361. https://doi.org/10.3390/ijerph16224361.

Hanson, M.L., Madronich, S., Solomon, K., Andersen, M.P.S., Wallington, T.J., 2024. Trifluoroacetic Acid in the environment: consensus, gaps, and next steps. Environ. Toxicol. Chem. 43 (10), 2091–2093. https://doi.org/10.1002/etc.5963.

Helsel, D.R., 2010. Summing nondetects: incorporating low-level contaminants in risk assessment. Integr. Environ. Assess. Manag. 6, 361–366. https://doi.org/10.1002/ieam.31.

Herzke, D., Nikiforov, V., Yeung, L.W.Y., Moe, B., Routti, H., Nygård, T., Gabrielsen, G. W., Hanssen, L., 2023. Targeted PFAS analyses and extractable organofluorine – Enhancing our understanding of the presence of unknown PFAS in Norwegian wildlife. Environ. Int. 171, 107640. https://doi.org/10.1016/j.envint.2022.107640.

Higler, L.W.G., 2008. 1 Verspreidingsatlas Nederlandse kokerjuffers (Trichoptera) [distribution Atlas of Caddisflies (Trichoptera) of the Netherlands], 1st ed. EIS-Nederland, Leiden.

Hopkins, K.E., McKinney, M.A., Saini, A., Letcher, R.J., Karouna-Renier, N.K., Fernie, K. J., 2023. Characterizing the movement of Per- and Polyfluoroalkyl Substances in an avian aquatic–terrestrial food web. Environ. Sci. Technol. 57 (48), 20249–20260. https://doi.org/10.1021/acs.est.3c06944.

Huang, K., Li, Y., Bu, D., Fu, J., Wang, M., Zhou, W., Gu, L., Fu, Y., Cong, Z., Hu, B., Fu, J., Zhang, A., Jiang, G., 2022. Trophic magnification of short-chain Per- and Polyfluoroalkyl Substances in a terrestrial food chain from the Tibetan Plateau. Environ. Sci. Technol. Lett. 9 (2), 147–152. https://doi.org/10.1021/acs.estlett.1c01009.

Ibor, O.R., Adeogun, A.O., Chukwuka, A.V., Dami Omogbemi, D., Oluwale, F.V., Oni, A. A., Arukwe, A., 2025. Biomagnification of per-and polyfluoroalkyl substances in aquatic food species from a tropical lagoon ecosystem. Mar. Environ. Res. 208, 107117. https://doi.org/10.1016/j.marenvres.2025.107117.

ITRC (Interstate Technology & Regulatory Council). (2023) PFAS Technical and Regulatory Guidance Document and Fact Sheets PFAS-1. Washington, D.C.: Interstate Technology and Regulatory Council, PFAS Team. .

Jones, H.J., Swadling, K.M., Butler, E.C.V., Barry, L.A., Macleod, C.K., 2014. Application of stable isotope mixing models for defining trophic biomagnification pathways of mercury and selenium. Limnol. Oceanogr. 59, 1181–1192. https://doi.org/10.4319/lo.2014.59.4.1181.

Kotalik, C.J., Hubbard, L.E., Perrotta, B.G., Walters, D.M., Kolpin, D.W., Gray, J.L., Zachritz, A.M., Kraus, J.M., Givens, C.E., Lamberti, G.A., Kidd, K.A., 2025. Bioaccumulation and transfer of Per- and Polyfluoroalkyl Substances (PFAS) in a stream and riparian food web contaminated by food processing wastewater. Environ. Sci. Technol. 59 (36), 19444–19456. https://doi.org/10.1021/acs.est.5c04867.

Koch, A., Jonsson, M., Yeung, L.W.Y., Kärrman, A., Ahrens, L., Ekblad, A., Wang, T., 2020. Per- and Polyfluoroalkyl-contaminated freshwater impacts adjacent riparian food webs. Environ. Sci. Technol. 54 (19), 11951–11960. https://doi.org/10.1021/acs.est.0c01640.

Koch, A., Jonsson, M., Yeung, L.W.Y., Kärrman, A., Ahrens, L., Ekblad, A., Wang, T., 2021. Quantification of biodriven transfer of Per- and Polyfluoroalkyl substances from the aquatic to the terrestrial environment via emergent insects. Environ. Sci. Technol. 55 (12), 7900–7909. https://doi.org/10.1021/acs.est.0c07129.

Kraus, J.M., Skrabis, K., Ciparis, S., Isanhart, J., Kenney, A., Hinck, J.E., 2023. Ecological harm and economic damages of chemical contamination to linked aquatic-terrestrial food webs: a study-design tool for practitioners. Environ. Toxicol. Chem. 42 (9), 2029–2039. https://doi.org/10.1002/etc.5609.

Krell, B., Röder, N., Link, M., Gergs, R., Entling, M.H., Schäfer, R.B., 2015. Aquatic prey subsidies to riparian spiders in a stream with different land use types. Limnologica 51, 1–7. https://doi.org/10.1016/j.limno.2014.10.001.

Li, F., Zhou, S., Han, D., Guo, H., Huang, K., Yang, T., Yang, L., Yao, L., Ren, Z., Chen, L., Shi, J., 2026. Quantifying the trophic transfer process and dietary source of legacy and emerging Per- and Polyfluoroalkyl Substances in the food web in Bohai Sea. Environ. Sci. Technol. 60 (10), 8169–8179. https://doi.org/10.1021/acs.est.6c00335.

Lewis, A.J., Yun, X., Spooner, D.E., Kurz, M.J., McKenzie, E.R., Sales, C.M., 2022. Exposure pathways and bioaccumulation of per- and polyfluoroalkyl substances in freshwater aquatic ecosystems: key considerations. Sci. Total Environ. 822, 153561. https://doi.org/10.1016/j.scitotenv.2022.153561.

Ma, T., Wu, P., Wang, L., Quanguo Li, Q., Li, X., Luo, Y., 2023. Toxicity of per- and polyfluoroalkyl substances to aquatic vertebrates. Front. Environ. Sci. 11. https://doi.org/10.3389/fenvs.2023.1101100.

Mackay, D., Celsie, A.K.D., Arnot, J.A., Powell, D.E., 2016. Processes influencing chemical biomagnification and trophic magnification factors in aquatic ecosystems:

Implications for chemical hazard and risk assessment. Chemosphere 154, 99–108. https://doi.org/10.1016/j.chemosphere.2016.03.048.
Matthews, B., Mazumder, A., 2008. Detecting trophic-level variation in consumer assemblages. Freshw. Biol. 53 (10), 1942–1953. https://doi.org/10.1111/j.1365-2427.2008.02018.x.
Middendorf, F., Bundschuh, M., Eitzinger, B., Entling, M.H., Schirmel, J., 2025. Review of the importance of aquatic prey for riparian arthropod predators. Basic Appl. Ecol. 86, 1–10. https://doi.org/10.1016/j.baae.2025.04.007.
Morse, J.C., 2009. Chapter 257 - Trichoptera (Caddisflies). In: Resh, V.H., Cardé, R.T. (Eds.), Encyclopedia of Insects (second Edition). Academic Press, San Diego, pp. 1015–1020. https://doi.org/10.1016/B978-0-12-374144-8.00266-6.
Nyffeler, M., Sterling, W.L., Dean, D.A., 1994. How spiders make a living. Environ. Entomol. 23 (6), 1357–1367. https://doi.org/10.1093/ee/23.6.1357.
Organization for Economic Co-operation and Development (OECD). (2018). Environmental Directorate Joint Meeting of the Chemicals Committee and the Working Party on Chemicals, Pesticides and Biotechnology, toward a New Comprehensive Global Database on Per- and Polyfluoroalkyl Substances (PFASs): Summary Report on Updating the OECD 2007 List of Per- and Polyfluoroalkyl Substances (PFASs). Series on Risk Management No. 39. https://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV-JM-MONO(2018)7&doclanguage=en.
Paetzold, A., Bernet, J.F., Tockner, K., 2006. Consumer-specific responses to riverine subsidy pulses in a riparian arthropod assemblage. Freshw. Biol. 51, 1103–1115. https://doi.org/10.1111/j.1365-2427.2006.01559.x.
Parsons, J.R., Sáez, M., Dolfing, J., & de Voogt, P. (2008). Biodegradation of Perfluorinated Compounds. In: Whitacre, D. (eds) Reviews of Environmental Contamination and Toxicology Vol 196. Reviews of Environmental Contamination and Toxicology, vol 196. Springer, New York, NY. 10.1007/978-0-387-78444-1_2.
Politi, Y., Bertinetti, L., Fratzl, P., Barth, F.G., 2021. The spider cuticle: a remarkable material toolbox for functional diversity. Phil. Trans. R. Soc. A 379 (2206). https://doi.org/10.1098/rsta.2020.0332.
Post, D.M., 2002. Using stable isotopes to estimate trophic position: models, methods, and assumptions. Ecology 83, 703–718. https://doi.org/10.1890/0012-9658(2002)083[0703:USITET]2.0.CO;2.
Potapov, A.M., Tiunov, A.V., Scheu, S., Larsen, T., Pollierer, M.m., 2019. Combining bulk and amino acid stable isotope analyses to quantify trophic level and basal resources of detritivores: a case study on earthworms. Oecologia 189, 447–460. https://doi.org/10.1007/s00442-018-04335-3.
Ricolfi, L., Yang, Y., Pottier, P., Morrison, K., Williams, C., Pollo, P., Hesselson, D., Neely, G.G., Taylor, M.D., Nakagawa, S., Lagisz, M., 2025. Unravelling the magnitude and drivers of PFAS trophic magnification: a meta-analysis. Nat. Commun. 16, 10720. https://doi.org/10.1038/s41467-025-65746-4.
Rosenberg, G., Ramus, J., 1984. Uptake of inorganic nitrogen and seaweed surface area: volume ratios. Aquat. Bot. 19 (1–2), 65–72. https://doi.org/10.1016/0304-3770(84)90008-1.
Sadia, M., Nollen, I., Helmus, R., ter Laak, T.L., Béen, F., Praetorius, A., van Wezel, A.P., 2023. Occurrence, fate, and related health risks of PFAS in raw and produced drinking water. Environ. Sci. Tech. 57 (8), 3062–3074. https://doi.org/10.1021/acs.est.2c06015.
Sadia, M., Yeung, L.W.Y., Fiedler, H., 2020. Trace level analyses of selected perfluoroalkyl acids in food: method development and data generation. Environ. Pollut. 263 (Pt A), 113721. https://doi.org/10.1016/j.envpol.2019.113721.
Shaffer, K.W., Ye, X., Lee, C.S., Shipley, O.N., McDonough, C.A., Venkatesan, A.K., Gobler, C.J., 2025. Accumulation and trophic transfer of per- and polyfluoroalkyl substances (PFAS) in estuarine organisms determined via stable isotopes. Sci. Total Environ. 967, 178742. https://doi.org/10.1016/j.scitotenv.2025.178742.
Sebastiano, M., Jouanneau, W., Blévin, P., Angelier, F., Parenteau, C., Pallud, M., Ribout, C., Gernigon, J., Lemesle, J.C., Robin, F., Pardon, P., Budzinski, H., Labadie, P., Chastel, O., 2023. Physiological effects of PFAS exposure in seabird chicks: a multi-species study of thyroid hormone triiodothyronine, body condition and telomere length in South Western France. Sci. Total Environ. 901, 165920. https://doi.org/10.1016/j.scitotenv.2023.165920.
Simmonet-Laprade, C., Hélène Budzinski, H., Babut, M., Le Menach, K., Munoz, G., Lauzent, M., Ferrari, B.J.D., Labadie, P., 2019. Investigation of the spatial variability of poly- and perfluoroalkyl substance trophic magnification in selected riverine ecosystems. Sci. Total Environ. 686, 393–401. https://doi.org/10.1016/j.scitotenv.2019.05.461.
Szałkiewicz, E., Kałuża, T., Grygoruk, M., 2022. Detailed analysis of habitat suitability curves for macroinvertebrates and functional feeding groups. Sci. Rep. 12, 10757. https://doi.org/10.1038/s41598-022-15096-8.
van den Brink, N.W., Arblaster, J.A., Bowman, S.R., Conder, J.M., Elliott, J.E., Johnson, M.S., Muir, D.C.G., Natal-da-Luz, T., Rattner, B.A., Sample, B.E., Shore, R.F., 2016. Use of terrestrial field studies in the derivation of bioaccumulation potential of chemicals. Integr. Environ. Assess. Manag. 12 (1), 135–145. https://doi.org/10.1002/ieam.1717.
van der Lee, G.H., Vonk, J.A., Verdonschot, R.C.M., Kraak, M.H.S., Verdonschot, P.F.M., Huisman, J., 2021. Eutrophication induces shifts in the trophic position of invertebrates in aquatic food webs. Ecology 102 (3), e03275. https://doi.org/10.1002/ecy.3275.
Wang, C., Magnuson, J.T., Zheng, C., Qiu, W., 2025. Incidence of pollution, bioaccumulation, biomagnification, and toxic effects of per- and polyfluoroalkyl substances (PFAS) in aquatic ecosystems: a review. Aquat. Toxicol. 286, 107469. https://doi.org/10.1016/j.aquatox.2025.107469.
Yu, J., Wang, T., Han, S., Wang, P., Zhang, Q., Jiang, G., 2013. Distribution of polychlorinated biphenyls in an urban riparian zone affected by wastewater treatment plant effluent and the transfer to terrestrial compartment by invertebrates. Sci. Total Environ. 463–464, 252–257. https://doi.org/10.1016/j.scitotenv.2013.06.006.