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REVIEW PAPER

Emerging contaminants of perfluoroalkyl carboxylic acids (PFCAs): a review of sources, occurrence, and accumulation in plants

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Abstract Perfluoroalkyl carboxylic acids (PFCAs) are emerging contaminants frequently detected in various environmental matrices, including plants. Their persistence, bioaccumulation potential, and toxicity pose substantial ecological and health risks. Despite their ubiquity, a comprehensive understanding of PFCAs behaviour in plants remains inadequate. This review systematically evaluates the sources, occurrence, and accumulation of PFCAs in plants, offering critical insights into their environmental behaviour and impacts. It outlines the distinctive physicochemical properties and bioaccumulation potential of PFCAs, and examines their environmental sources and plant uptake pathways. The occurrence of PFCAs across diverse plant species is explored, alongside the mechanisms driving their accumulation. Key factors influencing PFCAs accumulation, such as plant species, environmental conditions, and the physicochemical properties of PFCAs, are thoroughly analysed. Moreover, this review identifies key research needs, such as understanding foliar uptake and precursor transformation, strengthening urban vegetation monitoring, developing regulatory thresholds for plant contamination, and evaluating phytoremediation performance under complex environmental conditions.

Keywords Perfluoroalkyl carboxylic acids (PFCAs) · Occurrence · Accumulation · Plant uptake

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Introduction

Per- and polyfluoroalkyl substances (PFAS) constitute a large and diverse group of synthetic chemicals, with over 7 million compounds currently catalogued in the PubChem PFAS Tree (Schymanski et al., 2023). Since the 1940s, they have been extensively utilized in commercial and industrial applications due to their exceptional physicochemical properties, including resistance to heat, acids, and alkalis, water and oil repellency, and high surface activity (Dickman & Aga, 2022; Giesy & Kannan, 2002; Glüge et al., 2020). Among the PFAS family, perfluoroalkyl carboxylic acids (PFCAs) represent a subgroup characterized by the complete substitution of carbon-hydrogen bonds with carbon–fluorine bonds, terminating in a carboxyl group, as denoted by the chemical formula $C_nF_{2n+1}COOH$ (Buck et al., 2011). PFCAs exhibit exceptional environmental persistence due to their strong carbon–fluorine bonds, which confer high resistance to degradation under typical environmental conditions. Combined with their hydrophobic and lipophobic properties, PFCAs have been widely used across various industrial sectors and consumer products, including industrial and mining operations, food production, and firefighting foams (Glüge et al., 2020). This review specifically focuses on PFCAs, given their widespread detection in the environment and increasing attention to their behavior in plants. Compared with other PFAS, such as perfluoroalkyl sulfonic acids (PFSAs), PFCAs—particularly short- and medium-chain compounds—tend to exhibit greater mobility and translocation within plant tissues (Felizeter et al., 2014; Krippner et al., 2015), making them a growing focus in environmental fate and exposure research.

While PFCAs have been widely used in industrial and manufacturing processes, extensive animal studies have demonstrated that their exposure can induce various toxic effects, including neurotoxicity (Viberg & Mariussen, 2015), immunotoxicity (Keil, 2015; Liang et al., 2022), pulmonary toxicity (Ryu et al., 2014), hepatotoxicity (Tang et al., 2018), nephrotoxicity (Zhang et al., 2023), and developmental toxicity (Abbott, 2015; Zhou et al., 2022). Similarly, human studies have linked PFCAs exposure to an elevated risk of kidney and testicular cancer (Barry et al., 2013; Steenland & Winquist, 2021; Vieira et al., 2013). Recognizing the wide range of risks associated with PFCAs, international organizations have introduced several restrictions and regulatory measures. In 2019, perfluorooctanoic acid (PFOA), its salts, and PFOA-related compounds were listed in Annex A of the Stockholm Convention, mandating their global elimination (United Nations Environment, 2019). Additionally, C9–C14 PFCAs have been progressively included in the Candidate List of Substances of Very High Concern under the European chemicals regulation REACH (ECHA, 2023), and from February 2023, the European Commission formally restricted the production and use of these substances, including their salts and precursors, within the EU/EEA (European Commission, 2021). More recently, on April 10, 2024, the United States Environmental Protection Agency (EPA) finalized the National Primary Drinking Water Regulation (NPDWR) for six PFAS, setting enforceable maximum contaminant levels (MCLs) for two PFCAs—PFOA and PFNA—in drinking water, further reflecting growing global regulatory efforts (US EPA, 2024).

In recent years, the presence of PFCAs in the environment has raised global concerns about environmental health. Between 1951 and 2015, the estimated emissions of C4-C14 PFCAs ranged from 2,610 to 21,400 tons (Wang et al., 2014). These compounds have been widely released into water, soil, and the atmosphere, and are highly prone to bioaccumulate in living organisms. PFCAs have been frequently detected in various plant species, including vegetables, cereals, and aquatic plants, indicating that plants can act as important environmental sinks and potential entry points into the food chain (D'Hollander et al., 2015; Griffin et al., 2023; Herzke et al., 2013; Shen et al., 2024). Such accumulation raises concerns about indirect human exposure through the consumption of plant-based food products, particularly in agricultural areas impacted by PFAS-contaminated water or biosolid application (Liu et al., 2019; Yoo et al., 2011). Despite growing evidence of PFCAs occurrence in plants, the mechanisms governing their uptake, translocation, and accumulation remain poorly understood.

This review systematically analyses existing literature to comprehensively examine the sources, occurrence, and accumulation mechanisms of PFCAs in plants. By synthesizing current scientific evidence, it provides a detailed analysis of the distribution characteristics and behavioural patterns of PFCAs within plants, offering valuable insights and guidance for future research and policy development.

Nature of PFCAs

PFCAs are entirely synthetic fluorinated compounds, first developed in the mid-twentieth century. Their exceptional chemical and thermal stability drove widespread industrial use and large-scale production (Buck et al., 2011; Jessica, 2016; Sznajder-Katarzyńska et al., 2019). Industrial synthesis primarily occurs via two methods: electrochemical fluorination (ECF) and telomerization (De Silva & Mabury, 2006; Evich et al., 2022). ECF yields a mixture of linear and branched isomers, while telomerization produces predominantly linear fluorotelomer-based precursors. Critically, certain fluorotelomer compounds degrade under environmental or treatment conditions, forming PFCAs (De Silva & Mabury, 2006; Pickard et al., 2018; Prevedouros et al., 2006; Yu et al., 2018).

Beyond their industrial origins, the widespread environmental occurrence and behaviour of PFCAs are largely influenced by their unique physicochemical properties. Table 1 summarizes the chemical information of the primary straight-chain PFCAs. The carbon–fluorine (C–F) bond, with a bond energy of approximately 485 kJ/mol, is one of the strongest in organic chemistry and contributes significantly to the extreme stability and environmental persistence of PFCAs (Evich et al., 2022; Zhao et al., 2024). In addition, PFCAs possess amphiphilic characteristics: while the perfluorinated carbon backbone imparts hydrophobic and lipophobic properties, the terminal carboxylic acid group confers a degree of hydrophilicity (Higgins & Luthy, 2006; H. Li et al., 2023a, b; Rayne & Forest, 2009). This amphiphilic nature enables PFCAs to partition between aqueous and organic phases, facilitating their transport and distribution across various environmental compartments. The environmental fate of PFCAs is also influenced by their carbon chain length. Long-chain PFCAs (typically defined as those with ≥ 8 carbon atoms) exhibit higher hydrophobicity and tend to adsorb more strongly onto solid matrices such as soils and sediments. In contrast, short-chain PFCAs (with ≤ 7 carbon atoms) are more soluble and volatile, allowing them to migrate more readily through air and water (Brusseau, 2023; H. Li et al., 2023a, b). Due to these

Table 1 Chemical Information of C2-C21 PFCAs (Kim et al., 2022)

Compound	Abbreviation	Chain length	Formula	CAS*	Molecular weight (g/mol)
Trifluoroacetic acid	TFA	C2	CF3COOH	76–05-1	114.02
Perfluoropropanoic acid	PFPrA	C3	C2F5COOH	422–64-0	164.03
Perfluorobutanoic acid	PFBA	C4	C3F7COOH	375–22-4	214.04
Perfluoropentanoic acid	PFPeA	C5	C4F9COOH	2706–90-3	264.05
Perfluorohexanoic acid	PFHxA	C6	C5F11COOH	307–24-4	314.05
Perfluoroheptanoic acid	PFHpA	C7	C6F13COOH	375–85-9	364.06
Perfluorooctanoic acid	PFOA	C8	C7F15COOH	335–67-1	414.07
Perfluorononanoic acid	PFNA	C9	C8F17COOH	375–95-1	464.08
Perfluorodecanoic acid	PFDA	C10	C9F19COOH	335–76-2	514.08
Perfluoroundecanoic acid	PFUnDA	C11	C10F21COOH	2058–94-8	564.09
Perfluorododecanoic acid	PFDoDA	C12	C11F23COOH	307–55-1	614.1
Perfluorotridecanoic acid	PFTrDA	C13	C12F25COOH	72629–94-8	664.1
Perfluorotetradecanoic acid	PFTeDA	C14	C13F27COOH	376–06-7	714.11
Perfluoropentadecanoic acid	PFPeDA	C15	C14F29COOH	141074–63-7	764.12
Perfluorohexadecanoic acid	PFHxDA	C16	C15F31COOH	67905–19-5	814.13
Perfluoroheptadecanoic acid	PFHpDA	C17	C16F33COOH	57475–95-3	864.13
Perfluorooctadecanoic acid	PFODA	C18	C17F35COOH	16517–11-6	914.1
Heptatriacontafluorononadecanoic acid	PFNDA	C19	C18F37COOH	133921–38-7	964.1
Perfluoroeicosanoic acid	NA*	C20	C19F39COOH	68310–12-3	1014.2
Perfluorohenicosanoic acid	NA*	C21	C20F41COOH	NA*	1064.2

CAS* refers to the Chemical Abstracts Service registry number
NA* indicates that there is currently no abbreviation or CAS

characteristics, PFCAs have been detected globally in soil, water, and biota, including in remote regions far from known sources of contamination (Ahrens, 2011; Brusseau et al., 2020; Rankin et al., 2016).

PFCAs exhibit significant bioaccumulation and biomagnification effects due to their exceptional chemical stability and strong resistance to biodegradation (Stahl et al., 2011). Field studies in remote Arctic ecosystems have demonstrated that PFCAs can reach top predators such as polar bears through long-range transport and trophic magnification. For example, Boisvert et al. (2019) reported that PFCAs concentrations in polar bear liver (Σ_{13}PFCAs: 924 ± 71 ng/g ww) were over tenfold higher than those in their main prey, ringed seals (Σ_{13}PFCAs: 74 ± 6 ng/g ww), with even stronger biomagnification observed when comparing seal blubber to bear liver. The trophic magnification factor (TMF), a metric commonly used to evaluate the bioaccumulative potential of persistent pollutants in food webs, indicates biomagnification when exceeding 1 (Fisk et al., 2001). In subtropical and freshwater systems, Loi et al. (2011) observed TMFs above 1 for PFDA, PFUnDA, and PFDoDA in the Mai Po Marshes, and Xu et al. (2014) reported TMFs ranging from 2.1 to 3.7 for C9–C12 PFCAs in the Taihu Lake food web. These findings reflect consistent biomagnification from low trophic level organisms to top aquatic predators. As humans are terminal consumers in many food chains, the potential for dietary exposure to PFCAs has raised increasing concern. In humans, PFCAs accumulate in critical organs such as the brain, liver, lungs, bones, and kidney (Pérez et al., 2013), and exhibit prolonged biological half-lives, particularly for long-chain homologues ($C > 8$), with estimates ranging from 0.5 to 90 years (Y. Zhang et al., 2013a, b). Although the mechanisms underlying PFCAs bioaccumulation remain incompletely understood, two primary hypotheses have been proposed: one involving partitioning effects with membrane phospholipids and the other focusing on interactions between PFCAs and proteins (Ng & Hungerbühler, 2014).

Although the toxic effects of PFCAs and their precursors are well documented, these effects are closely associated with molecular characteristics such as carbon chain length. Studies have shown that the toxicity of PFCAs increases with longer carbon chains (Berntsen et al., 2017; Buhrke et al., 2013; Tang et al., 2023). To mitigate the associated environmental and health risks, long-chain PFCAs have gradually been replaced by short-chain homologues or other fluorinated and non-fluorinated alternatives in the manufacturing of products (Wang et al., 2013). For instance, perfluoroether carboxylic acids (PFECAs), which exhibit physicochemical properties similar to those of long-chain PFCAs, are now commonly used as fluoropolymer processing aids (Gomis et al., 2015). However, the hazard and risk assessments of these substitutes remain limited. Despite differences in chemical structure, many alternatives display comparable environmental persistence and mobility to long-chain PFCAs, suggesting potential for global dispersion and long-term environmental persistence (Li et al., 2020; Y. Wang et al., 2019a, b). Moreover, growing evidence shows that some fluorinated substitutes, including certain PFECAs, possess toxicological profiles of concern, thereby challenging the assumption that these replacements are inherently safer (J. Wang et al., 2020a, b, c; Y. Wang et al., 2019a, b; Wang et al., 2015).

Controlling pollution caused by PFCAs in the environment remains a significant challenge. Various physical, chemical, and biological treatment technologies have been explored to address PFCAs contamination in different water matrices, including groundwater, surface water, municipal wastewater, and industrial effluents (Qi et al., 2022). Among these, adsorption using granular activated carbon (GAC) and ion exchange (IX) resins is the most commonly employed approach. However, both methods present limitations: GAC is typically regenerated thermally off-site or replaced entirely, but shows poor removal efficiency for short-chain PFCAs and incurs high operational costs; IX resins, while effective, require frequent on-site regeneration, which produces highly concentrated PFCAs-laden waste solutions that require further treatment (Banayan Esfahani et al., 2023; Li et al., 2021; Nakazawa et al., 2023). In addition to adsorption, oxidative degradation methods such as UV-based advanced oxidation processes (UV-AOPs) and photocatalysis have gained attention. These approaches rely on reactive species (e.g., hydroxyl, sulfate, or superoxide radicals) to cleave C–F bonds and degrade PFCAs. Nonetheless, they often produce short-chain PFCAs or fluorinated intermediates as by-products, and are limited by catalyst inefficiency, interference from water matrix components, and the need for careful management of degradation residues (Liu et al., 2020; Wang et al., 2021). Biodegradation studies on PFAS have predominantly focused on precursor compounds such as FTOHs. In contrast, PFCAs are extremely stable and have long been considered difficult to biodegrade under environmental conditions (Liou et al., 2010; Sáez et al., 2008; Wackett, 2021). Nevertheless, a few microbial strains have demonstrated potential for PFCAs degradation. For instance, Huang and Jaffé (2019) reported that *Acidimicrobium* sp. strain A6 achieved a 60% removal rate of PFOA under laboratory enrichment culture conditions, indicating notable biodegradation capacity. However, PFCAs biodegradation remains highly challenging due to the limited number of effective microbial strains, the generally slow degradation rates, and the potential cytotoxicity of released fluoride ions, which can inhibit microbial activity and reduce overall treatment efficiency (Wackett, 2022). Phytoremediation may also offer a viable option. Reed plants (Phragmites australis) have been shown to accumulate various contaminants such as nutrients, heavy metals, and organic pollutants from water and soil (Rezania et al., 2019). Recent studies further demonstrated that P. australis can adsorb and accumulate PFASs, including PFCAs, with PFAS concentrations in plant tissues reaching up to 13 ng/g ww and removal efficiencies up to 50% in pilot-scale systems (Ferrario et al., 2022). However, the subsequent disposal of contaminated plant biomass typically requires high-temperature incineration, which raises concerns regarding both economic feasibility and potential environmental impacts.

Sources of PFCAs in plants

Soil and water are vital for plant growth, providing essential nutrients and moisture. Agricultural irrigation alone accounts for over 60% of global freshwater withdrawals and more than 80% of total consumptive water use (Siebert et al., 2010; Wu et al., 2022). Given that irrigation commonly relies on surface and groundwater, the contamination of these resources with PFCAs raises increasing concern. Owing to their strong stability and mobility, PFCAs are widely detected in the environment. For example, in a global survey, PFCAs (C6–C14) in soils ranged from 29 to 14,300 pg/g (Rankin et al., 2016), while in surface and groundwater from rural areas in eastern China, PFCAs (C4–C14) were detected at concentrations ranging from 3.4 to 612 ng/L (Chen et al., 2016). Plants can absorb PFCAs from these contaminated sources, thereby facilitating their entry into the food chain and raising concerns about potential dietary exposure (Qi et al., 2022). Details regarding PFCAs accumulation levels in plant tissues are further discussed in Sect. "Occurrence of PFCAs in plants". Therefore, identifying the sources and pathways of PFCAs in plants is crucial for evaluating their ecological impacts and implications for food safety and environmental sustainability. The potential sources of PFCAs in plants are illustrated in Fig. 1.

Sources of PFCAs in the environment are categorized into direct and indirect sources. The total historical global industry-wide emissions of PFCAs have been estimated at 3,200–7,300 tons (Ahrens & Bundschuh, 2014). Industrial discharges are one of the primary direct sources of PFCAs contamination, particularly in environments near industrial areas, where directly affect soil and water quality, and potentially impact plant growth (Lv et al., 2023). A study reported that the total concentrations of PFCAs (C4–C12) in crops grown near fluorochemical industrial parks were significantly higher than those in typical agricultural environments, with the highest level detected reaching 8082.81 ng/g in a specific plant species (Liu et al., 2019). Additionally, the use of firefighting foams is another point source of PFCAs in the environment. Multiple studies have reported elevated concentrations of PFCAs in water and soil samples collected near firefighting foam storage and application sites (Anderson et al., 2016; Barzen-Hanson et al., 2017; Dauchy et al., 2017; Zhu et al., 2022). A post-fire investigation of a large-scale fire revealed the presence of multiple PFCAs contaminants in surface water within the affected area (Aly et al., 2020).

Landfills, particularly informal or inadequately managed sites in developing countries, have been identified as significant sources of PFCAs in the environment (Hamid et al., 2018; Tian et al., 2018; Tolaymat et al., 2023). This is primarily due to the disposal of PFCAs-treated consumer products—such as textiles, carpets, and food packaging—as well as PFCAs-laden biosolids (Hamid et al., 2018). Upon landfilling, these materials are subjected to infiltrating precipitation and undergo both aerobic and anaerobic degradation, leading to the mobilization

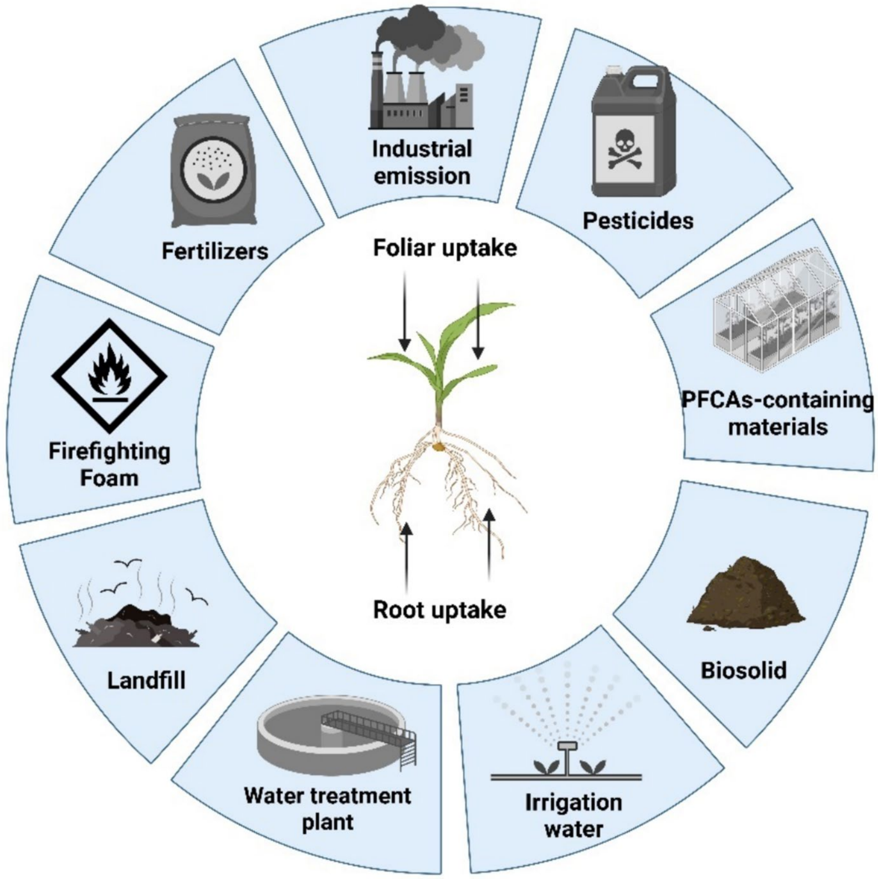

Fig. 1 Sources of PFCAs in plants (Created in BioRender)

of PFCAs into leachate and landfill gas (Allred et al., 2015; Yan et al., 2015). Water-soluble anionic PFCAs are frequently detected in landfill leachate and, in cases of liner failure or improper operational management, may pose a risk of contaminating adjacent soil, groundwater, and surface water systems (Benskin et al., 2012; Lang et al., 2017; J. Li et al., 2023a, b; Yan et al., 2015). Volatile precursors, including FTOHs, can partition into landfill gas and subsequently be released into the atmosphere in the absence of effective gas capture systems (Hamid et al., 2018). In the atmospheric compartment, PFCAs may undergo transformation and return to terrestrial environments through wet or dry deposition, with particulate matter acting as carriers for both ionizable and neutral species, including ultra-short-chain PFCAs (Tian et al., 2018; Tolaymat et al., 2023; Yao et al., 2017). Additionally, both abiotic (e.g., thermal and photolytic) and biotic (e.g., microbial) processes within landfill environments can convert PFCAs precursors into terminal PFCAs, particularly under prolonged anaerobic conditions (Allred et al., 2015; Tolaymat et al., 2023). These complex release, transformation, and transport pathways contribute to the elevated concentrations of PFCAs frequently observed in soils and vegetation near landfill sites (Tian et al., 2018; Xu et al., 2021). In some cases, landfills are located adjacent to agricultural fields. For instance, Xu et al. (2021) reported notable PFCAs accumulation in cabbage grown within 5 km of a landfill, with the highest concentrations observed within 1.5 km. The study indicated potential health risks associated with the consumption of such contaminated vegetables, thereby highlighting the role of landfills as not only environmental but also dietary sources of PFCAs exposure.

Wastewater treatment plants (WWTPs) are a major source of PFCAs in surface water due to the limitations of traditional treatment processes in effectively removing these compounds (Pan et al., 2016;

Tavasoli et al., 2021; W. Zhang et al., 2013a, b). In WWTPs, short-chain PFCAs dominate the aqueous phase, whereas long-chain PFCAs primarily accumulate in sludge (Pan et al., 2016; Tavasoli et al., 2021). Treated wastewater, widely used for agricultural irrigation, introduces PFCAs into soil and water systems, directly affecting agricultural ecosystems. Additionally, activated sludge, a byproduct of the wastewater treatment process, is further processed into biosolids. Rich in organic matter, trace nutrients, and microorganisms, biosolids are commonly applied in agriculture as soil amendments or fertilizers (Kumar et al., 2017). However, this practice has also become a significant source of PFCAs contamination in plants (Hamid & Li, 2016). For example, long-term application of biosolids derived from a wastewater treatment plant receiving fluorotelomer-based industrial waste in Decatur, Alabama, resulted in markedly elevated PFCAs concentrations in treated agricultural fields compared to nearby background soils, with levels of PFDA, PFDoA, and PFOA reaching up to 990, 530, and 320 ng/g, respectively (Washington et al., 2010).

The use of PFCAs-containing products and materials in agriculture has become a significant source of PFCAs contamination in plants. Some pesticides and other plant protection products (PPPs) containing PFCAs can enter ecosystems through direct application or residue migration (Joerss et al., 2024; Lasee et al., 2022; Nascimento et al., 2018). A study identified PPPs as a major source of TFA in groundwater across Europe, the United States, and China, despite regional differences in the types and application patterns of PPPs. In some areas, TFA emissions from PPPs were estimated to reach up to 83 kg per square kilometre (Joerss et al., 2024). In addition, the study by Zhou et al. (2024) highlighted that plastic films containing PFCAs, commonly used in greenhouse cultivation, represent a major source of PFCAs contamination in vegetables in certain regions. PFCAs such as PFHxA, PFOA, and PFDA were detected in greenhouse films at concentrations ranging from 0.12 to 12.2 μg/kg. These compounds can migrate via condensation water on the film surface into the soil at the corners of greenhouses, where they are subsequently taken up by plants and accumulate in edible tissues.

A variety of precursor compounds of PFCAs (e.g. FTOH) migrate and undergo transformation and degradation processes in various environmental media, becoming a significant indirect source of PFCAs generation (Pickard et al., 2018; Prevedouros et al., 2006; Yu et al., 2018). The transport properties of PFCAs suggest that even in areas without direct discharges, contamination may occur through long-range transport, indirectly impacting local plants (Ahrens, 2011; Prevedouros et al., 2006; Rankin et al., 2016).

The primary sources of PFCAs in plants are contaminated soils and water, while root uptake serves as the main pathway for their entry into plant tissues. This process is influenced by factors such as PFCAs chain length, plant species, and environmental conditions (Felizeter et al., 2012; Mei et al., 2021; T.-T. Wang et al., 2020a, b, c; Zhao et al., 2018). The absorption of solutes by roots occurs through apoplastic, symplastic, and transmembrane pathways, which collectively regulate the uptake and transport of external substances in plants (Mei et al., 2021; Miller et al., 2016). PFCAs can then be transported via the xylem or phloem from roots to stems, leaves, and fruits (Miller et al., 2016). Additionally, leaves can also serve as a pathway for PFCAs uptake by plants. Due to their low volatility and high water solubility, PFCAs are typically found in low concentrations in the air, limiting their contribution to foliar uptake (Prevedouros et al., 2006). However, a closed-chamber experiment with ryegrass demonstrated that the plant could absorb 8:2 FTOH from the air through its leaves and subsequently biotransform it into PFCAs (Yao et al., 2022). This finding underscores the potential for plants to absorb and transform PFCAs precursors via foliar pathways, thereby exacerbating PFCAs accumulation within plant tissues.

Occurrence of PFCAs in plants

PFCAs have garnered increasing attention due to their persistence and bioaccumulative potential (Ghisi et al., 2019; Qi et al., 2022). Plants, as a key interface between environmental media and the food chain, play an important role in the uptake and accumulation of these contaminants (W. Wang et al., 2020a, b, c). This section summarizes the reported occurrence of PFCAs in various plant species, providing a basis for understanding their environmental distribution.

As most investigations have focused on edible plants, Fig. 2 summarizes the occurrence of PFCAs in edible plant species reported from multiple countries. Due to substantial variability among

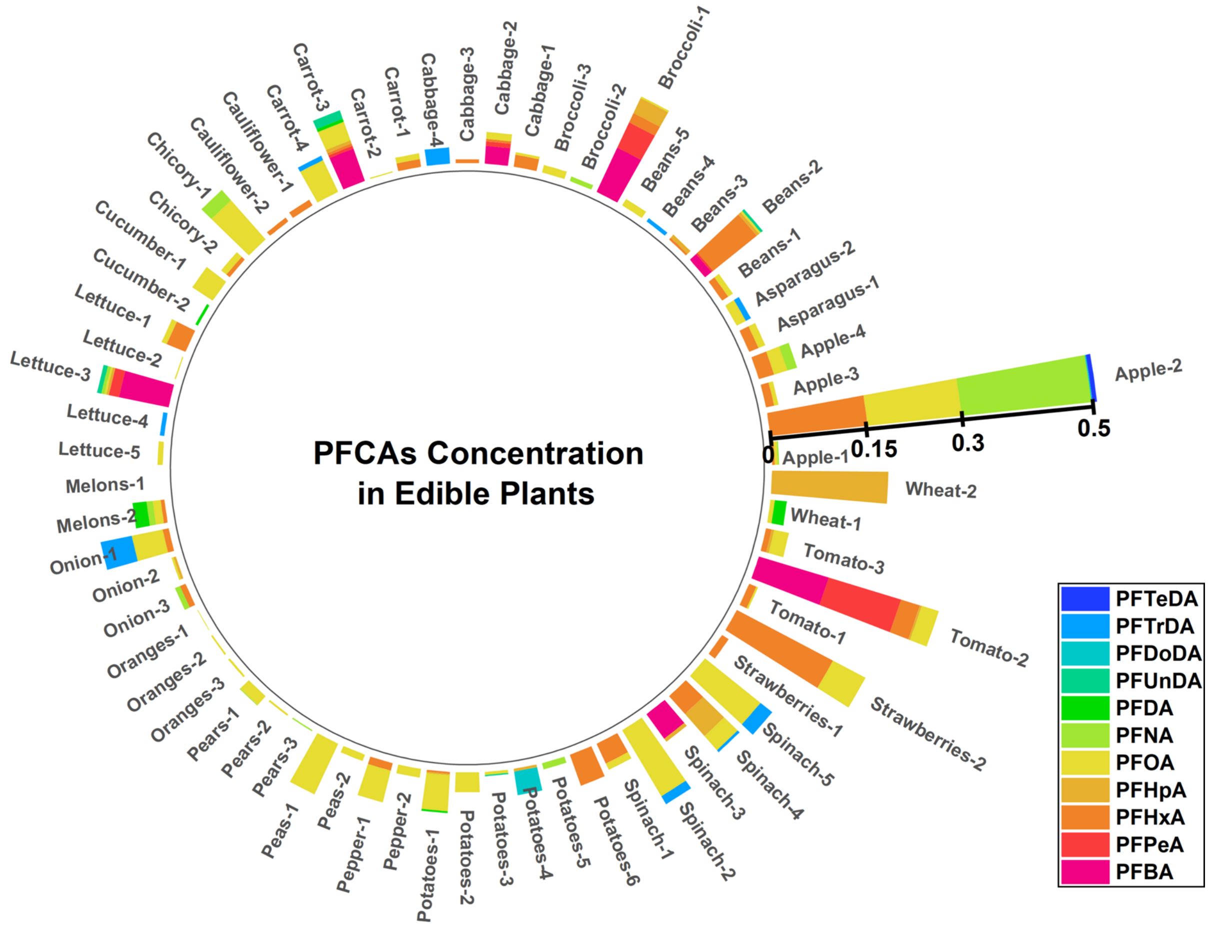

Fig. 2 Concentrations of PFCAs in edible plants in different regions (ng/g). This figure presents selected data on the occurrence of PFCAs in various edible plant species from multiple countries. The numbers 1, 2, 3, etc., correspond to different data sources, which are detailed in Appendix Table S1. The data are adapted from the following references: (D'Hollander et al., 2015; Haug et al., 2010; Herzke et al., 2013; Piva et al., 2023; Vestergren et al., 2012)

datasets—sometimes spanning several orders of magnitude—Fig. 2 presents a subset of representative plant species, including only those reported by two or more sources. A broader summary of PFCAs concentrations across all plant types and regions is provided in Table 2, while detailed concentration values supporting Fig. 2 are available in Supplementary Table S1.

Vegetables, as a direct link in the food chain, have garnered significant attention in research. A dietary study found that vegetables were the food group with the highest intake of PFHxA and PFOA, with vegetable consumption accounting for up to 69% of total intake (Klenow et al., 2013). In a study conducted by Herzke et al. (2013), 20 different types of vegetables from Belgium, the Czech Republic, Italy, and Norway were analysed for perfluoroalkyl substances. Among the detected PFCAs, PFOA exhibited the highest concentrations, ranging from 8 to 121 ng/kg fresh weight (fw), followed by PFHxA, with concentrations ranging from 4 to 52 ng/kg fw when present. In a study conducted in Shouguang City, the largest vegetable production region in China, Zhang et al. (2020) reported that PFBA was the predominant PFCAs detected in vegetables, with concentrations ranging from 0.98 to 17.85 ng/g dry weight (dw).

The European Food Safety Authority (EFSA) has established a tolerable weekly intake (TWI) of 4.4 ng/kg body weight for four major PFAS compounds: PFOA, PFNA, PFOS, and PFHxS. Additionally, the

Table 2 Concentrations of 11 PFCAs (C4–C14) in edible and non-edible plants from global studies (ng/g)

Species	Region	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	References
Vegetables	Catalonia, Spain	0.140	nd*	nd	0.360	0.370	nd	nd	nd	nd	nd	nd	Domingo et al. (2012)
	Sweden	na*	na	0.003	0.002	0.022	nd	0.003	nd	nd	nd	nd	Vestergren et al. (2012)
Fruit	Sweden	na	na	0.003	nd	0.015	nd	0.002	nd	nd	nd	nd	Vestergren et al. (2012)
	Catalonia, Spain	nd	nd	nd	nd	nd	nd	0.011	0.034	nd	0.016	0.019	Domingo et al. (2012)
Vegetable and fruit	Busan,Korea	0.030	0.019	0.039	0.002	0.001	nd	0.000	0.009	0.002	nd	nd	Heo et al. (2014)
Crop	Fen-wei Plain, China	0.435	0.272	0.324	0.241	1.631	0.313	0.315	0.508	0.287	na	na	Shen et al. (2024)
Apples	Czech Republic	na	na	0.004	nd	0.005	0.002	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Belgium	na	na	0.154	nd	0.149	0.205	nd	nd	0.002	nd	0.007	D'Hollander et al. (2015)
	Norway	na	na	0.013	nd	0.007	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Italy	na	na	0.025	nd	0.022	0.016	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Aquatic vegetation	Florida, US	3.510	na	8.205	0.283	0.339	0.261	0.210	0.176	0.078	na	na	Griffin et al. (2023)
Asparagus	Norway	na	na	0.016	nd	0.012	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	0.016	nd	nd	nd	nd	0.009	nd	Herzke et al. (2013)
Balsam pear	Shouguang, China	3.540	0.250	nd	nd	0.160	nd	nd	nd	0.100	na	na	Zhang et al. (2020)
Bananas	Italy	na	na	nd	nd	nd	0.003	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Beans	Italy	0.013	0.003	0.088	0.006	0.003	nd	nd	0.004	na	na	na	Piva et al. (2023)
	Italy	na	na	0.004	0.007	nd	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Czech Republic	na	na	nd	nd	nd	nd	nd	nd	nd	0.006	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	0.012	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Norway	na	na	0.012	nd	0.009	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Broccoli	Italy	0.077	0.044	0.016	0.026	0.003	nd	nd	nd	na	na	na	Piva et al. (2023)
	Belgium	na	na	nd	nd	0.012	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Czech Republic	na	na	nd	nd	nd	0.007	nd	nd	nd	nd	nd	Herzke et al. (2013)
Cabbage	Shouguang, China	5.640	0.880	0.680	0.880	2.430	0.130	0.060	0.060	0.030	na	na	Zhang et al. (2020)
	Norway	na	na	0.019	nd	0.004	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	0.028	0.007	0.004	nd	0.011	nd	nd	nd	na	na	na	Piva et al. (2023)
	Italy	na	na	0.006	nd	nd	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	nd	nd	nd	nd	nd	0.026	nd	Herzke et al. (2013)
Carrot	Norway	nd	nd	nd	nd	0.002	nd	nd	nd	nd	nd	nd	Haug et al. (2010)
	Norway	na	na	0.012	nd	0.009	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Shouguang, China	1.200	0.090	0.070	nd	0.180	nd	nd	nd	0.090	na	na	Zhang et al. (2020)
	Italy	0.054	0.005	0.004	0.006	0.031	nd	0.005	0.014	na	na	na	Piva et al. (2023)
	Italy	na	na	nd	nd	0.056	nd	nd	nd	nd	0.007	nd	Herzke et al. (2013)

Table 2 (continued)

Species	Region	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	References
Cauliflower	Norway	na	na	0.010	nd	nd	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Czech Republic	na	na	0.007	nd	nd	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Celery	Belgium	na	na	nd	nd	0.015	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Chicory	Norway	na	na	nd	nd	0.083	0.022	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	na	na	0.007	nd	0.010	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Chinese teas	China	na	1.189	15.809	na	0.235	0.146	0.161	0.070	0.013	0.025	na	R. Zhang et al., (2017a, b)
Courgette	Italy	0.092	nd	nd	0.004	0.003	nd	nd	0.005	na	na	na	Piva et al. (2023)
Cucumber	Shouguang, China	3.390	0.910	0.840	0.340	0.500	0.080	0.120	0.100	nd	na	na	Zhang et al. (2020)
	Czech Republic	na	na	nd	nd	nd	nd	0.004	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	na	na	nd	nd	0.038	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Eggplant	Shouguang, China	2.480	0.490	0.320	nd	0.370	0.210	0.180	0.120	0.170	na	na	Zhang et al. (2020)
	Italy	0.024	nd	nd	nd	nd	nd	nd	nd	na	na	na	Piva et al. (2023)
Fennel	Norway	na	na	nd	0.090	0.029	0.012	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Grapefruitalys	Norway	na	na	0.0188	nd	0.0138	0.0248	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Grapes	Belgium	na	na	nd	nd	0.007	0.007	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Lemons	Italy	na	na	nd	nd	0.002	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Lettuce	Norway	nd	nd	nd	nd	0.002	nd	nd	nd	nd	nd	nd	Haug et al. (2010)
	Norway	na	na	0.032	nd	0.009	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	0.008	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	0.080	0.016	nd	0.004	0.003	0.005	nd	0.007	na	na	na	Piva et al. (2023)
	Italy	na	na	nd	nd	nd	nd	nd	nd	nd	0.007	nd	Herzke et al. (2013)
Melons	Czech Republic	na	na	nd	nd	nd	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Norway	na	na	0.006	nd	0.0124	0.0099	0.022	nd	nd	nd	nd	D'Hollander et al. (2015)
Oats	Belgium	na	na	nd	nd	0.0301	nd	0.0242	nd	nd	nd	nd	D'Hollander et al. (2015)
Onion	Italy	nd	nd	nd	0.004	0.003	nd	nd	nd	na	na	na	Piva et al. (2023)
	Italy	na	na	0.010	nd	0.050	nd	nd	nd	nd	0.050	nd	Herzke et al. (2013)
	Czech Republic	na	na	0.010	nd	nd	0.009	nd	nd	nd	nd	nd	Herzke et al. (2013)
Oranges	Czech Republic	na	na	nd	nd	0.001	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Belgium	na	na	nd	nd	0.003	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Norway	na	na	nd	nd	0.003	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Peaches	Italy	na	na	0.018	nd	0.014	0.012	nd	nd	nd	nd	nd	D'Hollander et al. (2015)

Table 2 (continued)

Species	Region	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	References
Pears	Czech Republic	na	na	nd	nd	0.019	0.001	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Belgium	na	na	nd	nd	0.003	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Italy	na	na	nd	nd	nd	0.002	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Peas	Norway	na	na	nd	nd	0.088	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	0.011	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Pepper	Shouguang, China	1.580	0.680	0.400	nd	0.120	0.080	nd	nd	0.080	na	na	Zhang et al. (2020)
	Italy	nd	nd	nd	nd	0.014	nd	nd	nd	na	na	na	Piva et al. (2023)
	Italy	na	na	0.012	nd	0.052	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Plums	Belgium	na	na	nd	nd	0.002	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Potatoes	Norway	nd	nd	nd	nd	0.005	nd	nd	0.002	nd	nd	nd	Haug et al. (2010)
	Norway	na	na	nd	nd	0.032	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Czech Republic	na	na	nd	nd	nd	0.010	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Sweden	na	na	0.003	0.002	0.057	nd	0.003	nd	nd	nd	nd	Vestergren et al. (2012)
	Belgium	na	na	0.052	nd	nd	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	na	na	nd	0.004	nd	nd	nd	nd	0.038	nd	nd	Herzke et al. (2013)
Radish	Shouguang, China	1.660	0.780	0.800	0.130	0.270	0.090	0.030	0.140	0.110	na	na	Zhang et al. (2020)
	Italy	0.009	nd	nd	0.020	0.030	nd	nd	nd	na	na	na	Piva et al. (2023)
Rapeseed	Shouguang, China	3.530	0.790	0.740	0.120	1.690	0.050	0.050	0.050	0.030	na	na	Zhang et al. (2020)
Spinach	Shouguang, China	2.570	0.430	0.590	0.180	2.540	0.030	0.090	0.060	0.050	na	na	Zhang et al. (2020)
	Norway	na	na	0.034	nd	0.011	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Belgium	na	na	nd	nd	0.112	nd	nd	nd	nd	0.024	nd	Herzke et al. (2013)
	Italy	0.048	nd	nd	0.006	nd	nd	nd	nd	na	na	na	Piva et al. (2023)
	Italy	na	na	nd	nd	0.121	nd	nd	nd	nd	0.015	nd	Herzke et al. (2013)
	Czech Republic	na	na	0.038	0.046	0.030	nd	nd	nd	nd	0.004	nd	Herzke et al. (2013)
Sponge gourd	Shouguang, China	5.190	4.000	1.370	0.120	0.310	0.070	nd	nd	0.110	na	na	Zhang et al. (2020)
Strawberries	Czech Republic	na	na	0.012	nd	nd	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
	Belgium	na	na	0.171	nd	0.058	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Tangerines	Czech Republic	na	na	nd	nd	0.007	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Tomato	Shouguang, China	2.750	1.670	0.890	0.100	0.190	0.040	nd	0.050	0.100	na	na	Zhang et al. (2020)
	Norway	na	na	0.012	nd	0.003	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
	Italy	0.115	0.119	0.031	0.003	0.028	nd	nd	nd	na	na	na	Piva et al. (2023)
	Italy	na	na	0.009	0.004	0.024	nd	nd	nd	nd	nd	nd	Herzke et al. (2013)
Valerian	Italy	0.155	0.028	0.014	0.014	0.010	nd	nd	0.004	na	na	na	Piva et al. (2023)

Table 2 (continued)

Species	Region	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	References
Wheat	Belgium	na	na	nd	nd	0.0071	nd	0.019	nd	nd	nd	nd	D'Hollander et al. (2015)
	Italy	na	na	nd	0.183	nd	nd	nd	nd	nd	nd	nd	D'Hollander et al. (2015)
Zucchini	Shouguang, China	1.540	0.210	0.110	nd	0.640	0.040	nd	0.040	0.120	na	na	Zhang et al. (2020)

For data from multiple samples from the same location and species, mean values were calculated and selected

nd* means that the value is less than the minimum detection limit (MDL) and na* means that the target was not analysed

European Commission has proposed guideline threshold levels for PFOA and PFNA in vegetables and fruits, set at 0.01 ng/g and 0.005 ng/g, respectively (EFSA, 2020; European Commission, 2022). However, studies suggest that most edible crops already exceed these thresholds, posing potential health risks (EFSA, 2020). Furthermore, numerous studies have indicated that short-chain PFCAs tend to accumulate more readily in terrestrial plants compared to their long-chain counterparts (H. Zhang et al., 2017a, b, 2020).

Significant differences in PFCAs concentrations in plants have been observed across regions. For example, vegetables from Shouguang, China, exhibited PFCAs levels one to two orders of magnitude higher than those reported in other countries. According to Zhang et al. (2020), this elevated contamination may be attributed to the influence of a large fluorochemical industrial park (FIP) located approximately 50 km away. Pollutants released from the FIP were found to reach the agricultural area via atmospheric deposition and contaminated irrigation water, thereby exacerbating PFCAs accumulation in crops. Similarly, H. Zhang et al., (2017a, b) reported that vegetables from Hubei and Zhejiang provinces in China contained PFCAs concentrations far exceeding those in other countries, likely due to environmental contamination resulting from PFCAs production activities in China.

Specific studies have further elucidated the proximity effect of industrial areas, summarizing the occurrence of PFCAs in plants near various pollution sources, as illustrated in Table 3. For instance, Zhu and Kannan (2019) reported an average PFOA concentration of 174.33 ng/g in perennial grasses collected within a 1-mile radius of a fluorochemical industrial area. In a related study, Liu et al. (2019) found that short-chain PFCAs, particularly PFBA, were the predominant contaminants in wheat and corn grains cultivated near large fluorochemical industrial parks. These findings underscore the role of proximity to pollution sources in exacerbating PFCAs accumulation in plants. The variability in PFCAs concentrations among plants near pollution sources may be attributed to differences in the type and volume of emissions as well as the distance of plants from the source. For example, Liu et al. (2019) investigated PFCAs levels in plants located at different distances (0.3 km and 10 km) from a fluorochemical industrial park and found that plants closer to the pollution

Table 3 Concentrations of PFCAs (C4–C14) in plants collected near point sources (ng/g)

Locations	Source	Distance	Species	PFBA	PFPeA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	References
Shouguang, China	FIP	50 km	Vegetables	3.29	1.03	0.73	0.36	1.24	0.08	0.07	0.08	0.09	na*	na	Zhang et al. (2020)
Alabama, US	Biosolid	n*	Grasses	na	na	59.66	43.12	55.10	11.50	62.38	17.48	25.58	2.35	1.45	Yoo et al. (2011)
Changshu, China	FIP	< 30 km	Vegetables and rice	4.47	5.00	1.52	2.19	1.63	0.89	0.89	0.90	0.90	1.60	0.93	Gao et al. (2022)
Huantai, China	FIP	10 km	Vegetables	4.27	1.14	0.92	0.74	7.97	0.08	0.08	0.11	0.09	na	na	Liu et al. (2019)
		0.3 km	Vegetables	790.41	204.66	52.28	65.87	500.45	0.26	0.28	0.15	0.13	na	na	Liu et al. (2019)
Fuxin, China	FIP	0.2 km	Vegetables	69.00	3.10	0.28	nd*	3.20	nd	nd	na	na	na	na	Bao et al., 2019)
		1.5 km	Vegetables	1.30	nd	nd	nd	0.21	nd	nd	na	na	na	na	Bao et al. (2019)
Washington County, Ohio, US	FIP	< 1 mile	perennial grasses	na	na	na	2.80	174.33	1.39	1.94	3.74	3.05	na	na	Zhu and Kannan (2019)
Hangzhou, China	Landfill	< 5 km	Plants	23.13	9.56	1.30	0.42	0.48	0.16	0.35	0.06	0.04	na	na	Xu et al. (2021)
Veneto, Italy	Contaminated area	n	Tomatoes	34.05	11.87	1.83	nd	nd	nd	nd	nd	nd	na	nd	Battisti et al. (2024)
Luoma Lake, China	WWTPs*	n	Aquatic plants	42.37	45.15	39.52	35.98	33.34	10.56	36.05	17.36	9.06	16.99	25.23	Chu et al., 2022)
Daling River Basin, China	WWTPs	n	Aquatic plants	165.11	0.99	1.37	0.31	4.73	0.47	0.39	0.17	0.31	na	na	Wang et al., (2019a, b)

Mean values were used for all data

n* indicates cases where the distance from the pollution source cannot be defined or is not specified in the literature

WWTPs:* indicates that the plants were grown in water bodies receiving effluents contaminated with PFCAs from wastewater treatment plants

*nd** means the value was below the minimum detection limit (MDL), and *na** means the target compound was not analysed source exhibited significantly higher contamination levels—on average, PFCAs concentrations at 0.3 km were more than 100 times higher than those at 10 km.

Trees, as key biological indicators, sensitively reflect regional pollution levels and their potential environmental health impacts, making them invaluable in environmental monitoring studies. Research near fire training sites has demonstrated a consistent PFCAs accumulation pattern in trees such as birch (*Betula pendula*) and spruce (*Picea abies*), with concentrations following the order: leaves > twigs > roots > trunk/core, highlighting the varying adsorption properties among plant tissues (Gobelius et al., 2017). Similarly, studies on pine needles from ski resorts in Norway and Slovakia revealed distinct PFCAs distribution patterns, with PFOA dominating in Slovakia (8–93%) and PFBA prevailing in Norway (3–66%), suggesting significant regional variations in pollution sources (Chropeňová et al., 2016). In a fluorochemical industrial park in China, PFCAs concentrations in tree leaves and bark were markedly higher than in soil, primarily in the forms of PFOA and short-chain PFCAs, attributed to atmospheric deposition from gaseous pollutant diffusion (Shan et al., 2014). Furthermore, Jin et al. (2018) proposed tree bark as a potential indicator for monitoring atmospheric PFCAs pollution, emphasizing the expanding role of plants in environmental pollution monitoring.

Aquatic plants, as vital components of aquatic ecosystems, play a crucial role in maintaining water balance, purifying pollutants, and preserving biodiversity due to their unique physiological structures and ecological functions (Lesiv et al., 2020; Rezania et al., 2016). Numerous studies have confirmed their ability to absorb and accumulate significant amounts of PFCAs, with both short- and long-chain compounds being detected in plant tissues (Ferrario et al., 2021, 2022; Greger & Landberg, 2024; Griffin et al., 2023; Pi et al., 2017). For instance, Griffin et al. (2023) investigated PFAS occurrence in aquatic vegetation from eight locations in Florida, detecting 12 PFASs—including eight PFCAs—with concentrations ranging from 0.18 to 55 ng/g. The average bioconcentration factor (BCF) reached 1225, underscoring the high bioaccumulation potential of aquatic plants. Similarly, Ferrario et al. (2021) reported that PFCAs (C4–C12) were widely detected in reed tissues collected from a contaminated region in Northern Italy, with longer-chain PFCAs (C9–C12) mainly accumulating in roots and stems, and shorter-chain PFCAs like PFOA showing greater accumulation in stems and leaves. Pi et al. (2017) further observed that both submerged and free-floating aquatic macrophytes rapidly absorbed PFCAs within a few days of exposure, with bioconcentration factors increasing with carbon chain length. These findings collectively suggest that aquatic plants can serve as significant depots for PFCAs in aquatic ecosystems.

Beyond accumulation, aquatic plants also exhibit promising potential for PFCAs removal. In a pilot-scale study, Ferrario et al. (2022) found that *Phragmites australis* removed 30–65% of total PFASs from contaminated groundwater, with PFOA and PFBA concentrations in plant tissues increasing over time. Greger and Landberg (2024) evaluated three aquatic species under controlled exposure to PFAS-contaminated water (64,100 ng/L for 96 h), and reported the highest removal efficiency by *Carex rostrata* (25.4–83.8%), followed by *Eriophorum angustifolium* (14.0–61.7%) and *Elodea canadensis* (18.3–26.4%). These results confirm the effective phytoremediation capacity of aquatic macrophytes under both field and experimental conditions. In contrast, terrestrial phytoremediation studies targeting PFCAs remain relatively limited. Huff et al. (2020) conducted a greenhouse experiment exposing various woody and herbaceous species to six PFAS (1 mg/L each) over 14 weeks. Herbaceous plants such as *Festuca rubra* and *Equisetum arvense* demonstrated measurable uptake of PFCAs, with PFPeA recovery rates reaching up to 25%. Woody plants such as *Salix* and *Populus* also exhibited moderate accumulation, indicating the potential for terrestrial phytoremediation. However, compared to aquatic species, the overall removal efficiencies were generally lower. One reason for the limited research on terrestrial plants may be the intrinsic advantages of aquatic species in phytoremediation applications. Aquatic plants maintain constant contact with contaminated water, enabling more effective pollutant uptake. Additionally, they grow rapidly, are easier to harvest, and tolerate higher pollutant concentrations, making them particularly suitable for engineered treatment systems such as constructed wetlands. These advantages likely contribute to the greater research emphasis on aquatic plant-based PFCAs remediation.

Accumulation pattern of PFCAs in plants

Due to their long-range transport potential, PFCAs are widely distributed in various environmental media (Ahrens, 2011; Hu et al., 2016; Li et al., 2022). As plants serve as a key pathway for PFCAs transfer into the human food chain, elucidating their accumulation behaviour in plants is of critical importance. The accumulation of PFCAs in plants is influenced by factors such as molecular structure, plant species, and environmental conditions. Through migration and translocation processes, PFCAs exhibit varying levels of accumulation across different plant parts.

Overall accumulation patterns in plants

The accumulation of PFCAs in plants varies considerably depending on plant type, anatomical structure, and environmental exposure. In general, PFCAs tend to accumulate primarily in non-edible tissues such as roots, stems, and leaves, with significantly lower concentrations observed in edible portions like fruits or seeds (Battisti et al., 2024; Felizeter et al., 2014; Krippner et al., 2015; Zhu & Kannan, 2019). For instance, studies on tomatoes (Battisti et al., 2024) and corn (Krippner et al., 2015) consistently show that long-chain PFCAs are largely retained in vegetative tissues, while only short-chain compounds exhibit limited translocation into edible parts. Although PFCAs levels in edible tissues are generally low, the reuse of vegetative biomass (e.g., stalks or leaves) as fodder or compost introduces pathways for indirect human exposure through livestock or secondary crop contamination.

Aquatic plants generally exhibit greater potential for PFCAs accumulation than terrestrial plants, primarily due to their direct contact with contaminated water, larger surface area, and absence of soil-root barriers. Many studies have reported a chain-length-dependent bioaccumulation pattern, in which long-chain PFCAs exhibit stronger retention in aquatic vegetation. For example, in a mesocosm exposure study, Pi et al. (2017) found that the BCFs of long-chain compounds such as PFTeDA and PFTrDA reached 865 and 1280 L/kg respectively, while those of shorter-chain PFCAs were significantly lower (17.3–123 L/kg). Similarly, field investigations along the Qinghe River revealed that the bioaccumulation factors (BAFs) of submerged aquatic plants increased with increasing PFCAs chain length, confirming the preferential uptake of long-chain species under environmental conditions (Zhou et al., 2017). In addition, Greger and Landberg (2024) highlighted that plant biomass plays a crucial role in PFCAs removal, with higher biomass correlating with more efficient uptake. Their study also reported a time-dependent translocation of PFCAs from roots to stems and identified root-secreted enzymes such as peroxidase and laccase that may contribute to PFCAs degradation. Collectively, these findings indicate that aquatic plants possess morphological and biochemical advantages that facilitate preferential accumulation and potential transformation of long-chain PFCAs, setting them apart from terrestrial species in phytoremediation applications.

Mechanistic insights into the uptake and translocation of PFCAs in plants

The accumulation of PFCAs in plants is driven by a complex interplay of the physicochemical properties of the compounds, the structural and physiological characteristics of the plant root system, and environmental exposure conditions. PFCAs are generally absorbed through plant roots, where they may follow three distinct pathways: apoplastic transport, symplastic transport, and transmembrane transport (Liu et al., 2023; Miller et al., 2016; T.-T. Wang et al., 2020a, b, c). The apoplastic pathway involves passive diffusion through cell walls and intercellular spaces, bypassing cell membranes (Sattelmacher, 2001). The symplastic pathway allows solute movement between adjacent cells via plasmodesmata (Sevilem et al., 2013). The transmembrane pathway, by contrast, entails crossing cell membranes through transport proteins and may include both passive diffusion and energy-dependent mechanisms(T.-T. Wang et al., 2020a, b, c). These three uptake routes—along with the involvement of membrane transport proteins such as aquaporins, anion channels, and organic anion carrier proteins—are schematically illustrated in Fig. 3. The relative contribution of each pathway depends on PFCAs chain length and plant-specific root traits, including the presence of root barrier structures and the expression of membrane transporter genes.

Once PFCAs reach the rhizosphere, their uptake into plant roots involves a complex interplay of transport pathways, structural barriers, and membrane-associated mechanisms. Structural components in

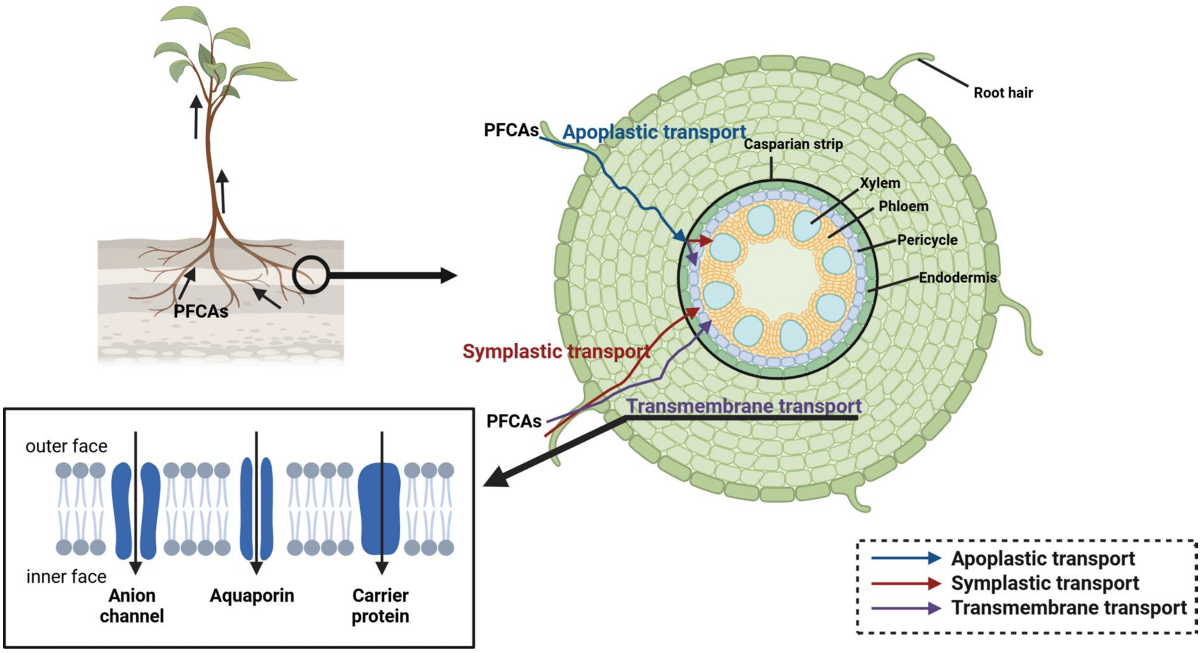

Fig. 3 Schematic diagram of PFCAs root uptake and transport mechanisms in plants (Created in BioRender)

roots—such as the Casparian strip—serve as selective barriers that restrict apoplastic flow, redirecting solutes into the symplastic pathway (Geldner, 2013). The physicochemical characteristics of PFCAs play a decisive role in their ability to traverse these barriers: short-chain PFCAs, due to their higher water solubility and lower sorptive affinity to root surfaces, are more likely to pass through and translocate into the vascular system, whereas long-chain PFCAs tend to accumulate in the root cortex owing to strong hydrophobic interactions and sorption to cell wall components and soil organic matter (Felizeter et al., 2014; Xu et al., 2022).

Beyond physical root barriers, cellular-level processes also contribute significantly to PFCAs uptake in plants. Membrane transport proteins—including both passive channels (e.g., aquaporins and anion channels) and active carrier proteins—are believed to mediate PFCAs movement across the plasma membrane (T.-T. Wang et al., 2020a, b, c; Wen et al., 2013; Zhang et al., 2019). In maize, Wen et al. (2013) demonstrated that the application of metabolic inhibitors reduced PFOA uptake by 83%, while anion channel blockers caused a 28% reduction, indicating the involvement of both energy-dependent processes and anion channels in uptake. Similarly, Zhang et al. (2019) observed that PFCAs absorption in wheat was significantly inhibited by metabolic blockers, suggesting active transport mechanisms. They further proposed that ultra-short-chain PFCAs (e.g., TFA and PFPrA) may pass through passive channels such as aquaporins and anion channels, while long-chain PFCAs, due to their larger molecular size and hydrophobicity, are less efficiently absorbed.

Once inside the vascular system, xylem transport is the predominant pathway for PFCAs translocation to aerial tissues (Felizeter et al., 2014). This movement is primarily driven by the transpiration stream, enabling water-soluble and low-molecular-weight compounds to migrate upward from roots to stems and leaves (Felizeter et al., 2014; Müller et al., 2016). Consequently, short-chain PFCAs such as PFBA and PFPeA, which exhibit higher hydrophilicity and lower sorptive affinity, demonstrate strong xylem mobility and are frequently detected at elevated concentrations in leaf tissues (Battisti et al., 2024; Krippner et al., 2015). In contrast, long-chain PFCAs like PFDA and PFTeDA show limited xylem translocation potential due to their greater hydrophobicity and stronger interactions with root and vascular tissues, resulting in preferential retention within root or stem compartments (Felizeter et al., 2014; Pi et al., 2017).

Factors influencing PFCAs accumulation

The accumulation of PFCAs in plants is governed by a complex interplay of factors, primarily including plant-specific traits, environmental parameters, and the intrinsic physicochemical properties of PFCAs themselves.

Influence of plant-specific traits

The accumulation patterns of PFCAs differ considerably among plant species due to inherent differences in morphology, physiology, and metabolic activity. A greenhouse study comparing PFCAs accumulation across eight herbaceous and seven woody plant species found that the herbaceous plant *Festuca rubra* exhibited significantly higher accumulation, particularly of PFPeA, with a BCF reaching 111.2 when normalized to soil concentrations (Huff et al., 2020). Further studies have identified a trend in PFCAs accumulation across plant types: leafy plants > root vegetables > fruits > seeds > flowers (Gao et al., 2022; Liu et al., 2019). This hierarchy may be attributed to differences in tissue exposure, translocation efficiency, and metabolic activity, which collectively influence the bioavailability and retention of PFCAs within plant organs. Notably, leafy vegetables with high transpiration rates and large surface areas tend to facilitate greater uptake and accumulation, raising concerns about their potential role as dietary exposure pathways.

In addition, physiological characteristics such as root protein and lipid content significantly affect PFCAs uptake. Wen et al. (2016) reported that plants with higher root protein content exhibited increased PFOA absorption, likely due to proteins providing adsorption sites and facilitating active transport. Conversely, plants with higher root lipid content showed decreased PFOA uptake, possibly because lipids impede membrane permeability or reduce interaction with water-soluble PFCAs. These results highlight how root biochemical composition mediates the initial sorption and subsequent translocation of PFCAs.

Moreover, studies have shown that aquatic plants differ substantially from terrestrial plants in their PFCAs accumulation behavior. Aquatic species can accumulate large amounts of PFCAs, and substantial interspecies variation has been observed (Chu et al., 2022; Davis et al., 2023; Griffin et al., 2023; Pi et al., 2017; P. Wang et al., 2019a, b), possibly due to differences in cuticle structure, water exposure surface area, and root aeration adaptations. Such physiological and structural traits may enhance the uptake efficiency of aquatic species, highlighting their promising potential for phytoremediation applications in PFAS-contaminated water bodies.

Influence of environmental conditions

Environmental parameters such as pH, salinity, temperature, and soil organic carbon content also modulate PFCAs accumulation in plants. Changes in environmental pH can alter PFCAs uptake by affecting their speciation and the chemical properties of soils, thereby impacting surface complexation and electrostatic interactions (Campos-Pereira et al., 2023; Nguyen et al., 2020). At the plant level, pH-dependent uptake has been observed in roots. For instance, in maize, PFDA uptake by roots decreased significantly with increasing pH, possibly due to enhanced membrane permeability under more protonated conditions at lower pH (Krippner et al., 2014). In wheat, however, PFCAs uptake exhibited a non-linear trend, with the root concentration factor (RCF) and shoot-to-root concentration factor (SRCF) reaching their highest values at neutral pH (7), while both lower (pH 4) and higher (pH 10) conditions suppressed accumulation (Zhao et al., 2018). These contrasting observations highlight the complexity of pH-dependent uptake, indicating that plant-specific physiological responses significantly mediate the effect of pH on PFCAs accumulation.

Salinity and temperature also play critical roles in PFCAs uptake and translocation. Zhao et al. (2016) demonstrated that increasing salinity and temperature significantly enhances PFCAs absorption in hydroponically grown wheat. Specifically, salinity increases from 0 to 0.4% led to 2.8- to 4.2-fold increases in PFCAs accumulation, with maximum concentrations in roots and shoots reaching up to 26,287.2 ng/g. Similarly, temperature elevations from 20 °C to 30 °C resulted in 1.5- to 2.3-fold increases in PFCAs uptake, with PFBA, PFHpA, PFOA, and PFDoA reaching concentrations of 858, 1,205, 1,098, and 14,576 ng/g, respectively. The enhanced uptake under elevated salinity may be attributed to increased water absorption for osmotic regulation and higher ionic strength, which promotes PFCAs adsorption to root surfaces. Rising temperatures can accelerate nutrient diffusion, reduce water viscosity, and stimulate plant metabolism, thereby facilitating greater contaminant uptake.

Soil organic carbon content further affects PFCAs bioaccumulation in plants. Higher organic carbon content reduces PFCAs bioavailability by enhancing their sorption onto soil particles, thus decreasing their dissolved fraction available for plant uptake (Blaine et al., 2014). For instance, the plant uptake factor for PFOA in lettuce significantly decreases with increasing soil organic carbon content (Lee et al., 2021). This behaviour is closely related to the organic carbon–water partition coefficient (Log Koc), a critical determinant of PFCAs mobility. Compounds with higher Log Koc values exhibit stronger affinity to soil organic matter (Pearson's $r = 0.82–0.85$), thereby reducing their bioavailability and resulting in lower BCFs in plant tissues (Pearson's $r = -0.72$ to -0.82) (Hilliard et al., 2023).

Influence of physicochemical properties of PFCAs

The chemical structure and physical properties of PFCAs are key determinants of their behaviour in plant systems. Numerous studies on terrestrial plants have shown that short-chain PFCAs primarily accumulate within plant tissues, whereas the detection levels of long-chain compounds are relatively low (Davis et al., 2023; Felizeter et al., 2014; Krippner et al., 2015; Liu et al., 2019; Piva et al., 2023; P. Wang et al., 2019a, b; Zhang et al., 2020).

Hydroponic exposure studies on tomatoes, cabbage, and zucchini confirmed that short-chain PFCAs preferentially translocate within plants, whereas long-chain compounds predominantly accumulate in roots (Felizeter et al., 2014). Additionally, a study using Arabidopsis revealed a competitive accumulation mechanism related to chain length, indicating that long-chain PFCAs primarily accumulate due to strong adsorption affinity, while short-chain compounds accumulate through efficient absorption and translocation mechanisms (Müller et al., 2016). Recent findings further indicate that molar volume also plays a role in plant uptake, as PFCAs with larger molar volumes exhibit lower translocation factors (TFs) ($r = -0.88$), suggesting smaller molecules are more readily transported from roots to aerial tissues (Hilliard et al., 2023).

Although long-chain PFCAs are generally considered more hazardous due to their strong bioaccumulative potential and long biological half-lives, recent studies have also raised concerns about short-chain PFCAs. Despite their lower bioaccumulation, short-chain PFCAs such as PFBA and PFHxA have demonstrated notable biological toxicity, including neurotoxicity and immunotoxicity (Ivantsova et al., 2024; Zhang et al., 2025). Their high environmental mobility and widespread occurrence in staple crops present additional health risks. Furthermore, many short-chain compounds are still not regulated under current international guidelines, posing challenges for effective health risk management.

While the concentrations of long-chain PFCAs in edible plant tissues are typically lower due to limited translocation, the concentrations reported by multiple monitoring studies summarized earlier in this review have exceeded the threshold values set by COMMISSION RECOMMENDATION (EU) 2022/1431 for PFOA and PFNA in fruits and vegetables (European Commission, 2022). The European Commission recommends that when such exceedances are observed, further investigation into the sources of contamination should be conducted to ensure food safety and prevent chronic human exposure. Therefore, both short- and long-chain PFCAs pose potential dietary risks through different mechanisms, underscoring the need for comprehensive risk assessments that consider compound-specific properties, accumulation patterns, and current regulatory gaps.

Future research needs

This review systematically examined the distribution, occurrence, and accumulation of common PFCAs in plants. However, it did not cover other PFAS subclasses such as PFSAs, which frequently co-occur with PFCAs in the environment and have also been shown to accumulate in plants. Due to differences in functional groups and physicochemical properties, PFSAs may exhibit distinct uptake pathways, translocation dynamics, and toxicities. Future research should conduct comparative studies on the plant accumulation behaviors of diverse PFAS groups to better evaluate their ecological risks and phytoremediation potential. Furthermore, with the growing use of PFAS alternatives, understanding their environmental fate and plant uptake characteristics has become increasingly urgent.

Regarding PFCAs regulation policies, existing frameworks primarily target environmental media such as water and soil, with limited attention to safety standards for plants. EFSA and the European Union are among the few institutions that have proposed plant-related guideline values, but these cover only two types of PFCAs: PFOA and PFNA. However, studies have shown that plants can accumulate high concentrations of PFCAs. Due to the lack of comprehensive policy support and regulatory guidelines, these contaminated plants may still enter the human food chain, directly or indirectly, posing potential health risks. Therefore, future research should advocate for the development of PFCAs safety standards specific to plants, particularly standards encompassing a broader range of PFCAs types, to ensure the safety of plant-based food and ecosystems.

While most current studies focus on edible crops in rural or peri-urban settings, research on PFCAs accumulation in urban vegetation remains limited. Urban areas often exhibit elevated PFAS concentrations due to dense populations, heavy traffic, and industrial point sources. However, long-term monitoring data on roadside trees, ornamental plants, and vegetation in recreational green spaces are scarce. Urban vegetation frequently interacts with multiple environmental media (air, dust, runoff) and may directly or indirectly expose the public through inhalation, dermal contact, or urban gardening. Therefore, future research should consider urban vegetation as a potential exposure vector as well as a promising bioindicator for spatial monitoring of PFAS contamination. Vegetation used in green infrastructure (e.g., bioretention systems) may further function both as interceptors of PFAS-contaminated runoff and as reservoirs of persistent pollutants, underscoring their dual role in urban risk assessment and management.

In terms of exposure pathways, research has primarily concentrated on root uptake, whereas foliar absorption remains relatively underexplored. Although airborne PFAS concentrations are generally lower than those in soil or water, volatile precursors such as FTOHs can be absorbed via leaves and subsequently biotransformed within plant tissues. This secondary source may significantly contribute to total PFCAs burdens. Future investigations should therefore examine the extent of foliar absorption and elucidate the mechanisms of precursor transformation within plant systems, to better understand multi-pathway exposure and internal accumulation dynamics.

Additionally, the dose–response relationship between environmental PFCAs concentrations and plant accumulation has not been well characterized. Most studies adopt a narrow concentration range, making it difficult to determine whether uptake follows linear, saturable, or threshold kinetics. Plant species may exhibit different dose–response behaviors depending on root structure, transport protein expression, and physiological stress tolerance. A more systematic understanding of these patterns is critical for predicting plant responses under varying exposure levels and for selecting species suitable for phytoremediation in diverse contamination scenarios.

Finally, future research should also address the lack of systematic comparisons between aquatic and terrestrial plants regarding their PFCAs accumulation patterns. Although both plant types exhibit phytoremediation potential, differences in exposure media, plant physiology, and environmental conditions make it challenging to draw direct conclusions. A clearer understanding of these differences is crucial for selecting appropriate species and optimizing phytoremediation strategies. In addition, the effectiveness of phytoremediation under complex water chemistries remains uncertain. Real-world contaminated waters—such as industrial wastewater, landfill leachate, and treated effluents—often contain elevated pH, salinity, and organic matter, all of which may influence PFCAs speciation and bioavailability, thereby impacting plant uptake. Therefore, future studies should evaluate phytoremediation performance under such challenging conditions to ensure its practical applicability in diverse contaminated environments.

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Declarations

Conflict of interest The authors declare no competing interests.

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