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Review

Contamination of per- and poly-fluoroalkyl substances in agricultural soils: A review

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ABSTRACT

Numerous reviews have focused on the chemistry, fate and transport, and remediation of per- and poly-fluoroalkyl substances (PFAS) across various environmental media. However, there remains a significant gap in the literature regarding a comprehensive review specifically addressing PFAS contamination within agricultural soils. Recognizing the threat PFAS pose to ecosystems and human health, this review critically examines the sources of PFAS in agricultural environments, their uptake and translocation within plant systems, and recent advancements in soil remediation techniques. PFAS ingress into agricultural soils primarily occurs through the application of biowastes, wastewater, and pesticides, necessitating a thorough examination of their pathways and impacts. Factors such as carbon chain length, salinity, temperature, and pH levels affect PFAS uptake and distribution within plants, ultimately influencing their transfer through the food web. Moreover, this review explores a range of physical, chemical, and biological strategies currently employed for the remediation of PFAS-contaminated agricultural soils.

1. Introduction

Per- and poly-fluoroalkyl substances (PFAS) are a large group of synthetic fluorinated chemicals characterized by strong carbon-fluorine (C-F) bonds (Buck et al., 2011). Due to the resilience of these bonds, PFAS persist in the environment for decades, earning the moniker "forever chemicals" (Bell and Tachovsky, 2022). PFAS are resistant to breakdown and can easily migrate between environmental matrices (Mejia Avendaño and Liu, 2015; Rahman et al., 2014). Over the years, PFAS have been widely utilized in consumer and industrial products due to their unique properties, such as reducing surface tension, repelling oil, dirt, and water, and resisting acids and high temperatures (Death et al., 2021; Whitehead et al., 2021). Despite efforts to phase out PFAS, they are still being regularly detected in water bodies, soils, plants, wildlife, humans, and other environmental matrices (Hansen et al., 2001; Jian et al., 2018; Schulz et al., 2020).

Three PFAS compounds: perfluorooctanesulfonate (PFOS), perfluorohexanesulfonate (PFHxS) and perfluorooctanoic acid (PFOA) have been classified as persistent organic pollutants (POPs) under the Stockholm Convention due to their persistence, toxicity, bioaccumulation potential, and ability to travel long distances from the discharge point (UNEP, 2023). Exposure to PFOS has been suggested to have a "probable link" to various health effects, including kidney and testicular cancers, thyroid diseases, ulcerative colitis, hypercholesterolemia, and pregnancy-induced hypertension (Olsen, 2015). Countries participating in the Convention are required to reduce or eliminate the release of POPs into the environment. Consequently, there has been a global shift in manufacturing from long-chain PFAS ($CF_2 \geq 7$ for carboxylates and $CF_2 \geq 6$ for sulfonates) to short-chain PFAS ($CF_2 \leq 6$ for carboxylates and $CF_2 \leq 5$ for sulfonates) (Gewurtz et al., 2019; Mumtaz

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et al., 2019).

Elevated PFAS concentrations have been documented in agricultural soils and crops proximal to contaminated areas, including fire-training zones, landfills, wastewater treatment plants (WWTP), and fluorochemical manufacturing facilities (Cheng et al., 2023; Gan et al., 2021; Liu et al., 2014). Also, the application of compost and biosolids in agricultural fields has resulted in the contamination of soil along with adjacent water bodies (Padhye et al., 2023). A notable instance of PFAS-contaminated farmland has been reported in Decatur, Alabama, USA, where the application of WWTP sludges to agricultural fields over a decade led to the contamination of both groundwater and surface water (Lindstrom et al., 2011; Washington et al., 2010; Yoo et al., 2010).

Various PFAS, such as PFOS and perfluoro decanoic acid (PFDA), and precursors such as fluorotelomer alcohols (FTOHs), which can transform into perfluoroalkyl acids (PFAAs) under certain environmental conditions, were detected in contaminated agricultural soils. Long-chain PFAS were predominantly found in topsoil, while short-chain PFAS were more prevalent in deeper soil layers (Washington et al., 2010; Yoo et al., 2010). The effects of long- and short-term municipal biosolids application on PFAS leaching through soil layers have been investigated (Röhler et al., 2023; Sepulvado et al., 2011). Research findings indicate that while the leaching of PFAS to groundwater remains uncertain, the transport of PFAS in soil from the biosolids application zone to depths of up to 120 cm is possible. Moreover, longer-chain PFAS exhibit substantially lower mobility compared to their shorter chain counterparts (Sepulvado et al., 2011).

The average half-life of most PFAS in soil can extend for years, increasing the likelihood of their absorption by plants and eventual entry into the human food chain. Plant uptake of PFAS can significantly influence their environmental fate and transport (Houde et al., 2011). Many of these PFAS compounds are persistent and can bioaccumulate in animals and humans (Brown et al., 2020; Houde et al., 2011; Li et al., 2019; Liu et al., 2019; Reiner et al., 2015; Xiao, 2017; Zabaleta et al., 2018; Zhang et al., 2020). Therefore, a clear understanding of PFAS interactions with plants, including their accumulation, phytotoxicity, and bioavailability, is crucial.

PFAS compounds are readily absorbed and transferred into plant tissues due to their high water solubility. However, different PFAS have different water solubility, affecting their transport and plant uptake (Bolan et al., 2021). PFAS primarily enter plants through root uptake from soil and water, moving along concentration and water potential gradients created by plant transpiration (Lee et al., 2014; Stahl et al., 2009). Their low sorption potential in soils, especially for short-chain homologues, facilitates migration towards roots (Buck et al., 2011; Zhao et al., 2012).

Agriculture relies heavily on chemical products such as fertilizers, pesticides, and insecticides; thus, understanding how and when PFAS contaminate soil, water, air, and food is crucial. Despite extensive research on PFAS, significant gaps remain in our understanding of their prevalence and long-term effects on agricultural soils and food safety. Therefore, a comprehensive assessment of PFAS contamination in agricultural soils is essential to evaluate and mitigate the risks associated with PFAS exposure, safeguarding both agricultural productivity and human health. This review aims to provide an in-depth analysis of:

a) Sources of PFAS in agricultural soils, including pesticides/insecticides, biosolids, wastewater irrigation, and airborne emissions;
b) Uptake of PFAS by plants and crops, including the mechanisms of plant uptake and the subsequent transfer of PFAS into various plant parts; and
c) Methods for remediating PFAS-contaminated agricultural soils, evaluating their effectiveness, feasibility and limitations.

By addressing these objectives, this review seeks to enhance our understanding of PFAS dynamics in agricultural soils and inform strategies for managing their risks.

2. Search methodology

A comprehensive search strategy was employed to identify published papers, including original research articles and reviews, on PFAS in agricultural soils. Databases such as Scopus, Google Scholar, and Web of Science were utilized with the keywords “PFAS,” “perfluoro,” “polyfluoro,” and “agriculture.” This search yielded 1118 documents, from which 163 research articles, reviews, and reports published between 2008 and 2023 were included in this review.

The search results from all databases were merged, and duplicates were removed using the Bibliometrix R-Tool (Aria and Cuccurullo, 2017). Works with missing information were excluded from the results. For example, the studies that did not include proper methodology, had no supporting documents to prove their claim, or reported results were not conclusive enough to be considered for the review. A two-stage screening process was then conducted. Initially, the titles and abstracts were screened based on predetermined inclusion and exclusion criteria. Articles that met the exclusion criteria were eliminated from the review. Fig. 1 illustrates the PRISMA search and screening process including the detailed exclusion criteria. The keywords and terms used to determine articles fitting the exclusion criteria included: pollen food allergy (abbreviated as PFA), clinical studies, in vitro studies, machine learning models, aqueous film-forming foam, cookware, utensils, adsorption, sorption, removal of PFAS, and drinking water.

Publications included in the review were those written in English and focused on PFAS contamination related to agricultural activities. The screening was categorized into four groups: sources of PFAS through agricultural activities, presence of PFAS in crops, impact of PFAS across different trophic levels, and methods for PFAS remediation. The second stage of screening involved a full-text review to determine inclusion based on the same criteria. Publications that passed this second screening were included in the review.

3. Sources of PFAS in agriculture

PFAS contamination has been identified in various environmental matrices, including soil, surface water, groundwater, biowastes, and water used for irrigation (Ghisi et al., 2019; Key et al., 1997; Pan et al., 2018; Schroeder et al., 2021; Shigei et al., 2020; Szabo et al., 2018). The sources of PFAS contamination in agriculture are listed in Table 1. The most extensively examined sources of PFAS in agricultural lands are biosolids and irrigation water (Costello and Lee, 2020; D’Ambro et al., 2021; Key et al., 1997; Pan et al., 2018). Fig. 2 illustrates the potential origins of PFAS in agricultural lands and the food chain.

PFAS are prevalent in numerous water resources worldwide. When these contaminated water sources are used for irrigation, they can significantly increase the PFAS load in soil (Ogawa et al., 2020). Treated wastewater, often reclaimed for irrigation purposes, contains PFAS compounds that can easily infiltrate agricultural soil (Pepper et al., 2021). Additionally, waste-derived soil amendments, which are frequently added to land to enhance soil structure and nutrient- and water-holding capacity, also contribute to PFAS contamination (Levine et al., 2023). Understanding the diverse pathways of PFAS entry into agricultural soils is essential for assessing their environmental impact. Empirical studies worldwide have provided critical insights into the extent and sources of PFAS contamination in agricultural settings.

For instance, the occurrence of 21 PFAS in irrigation water and agricultural soil was investigated in the Changshu City region of China (Li et al., 2019). PFOA and other perfluoroalkyl carboxylic acids (PFCA) were predominant in agricultural soil samples, indicating a strong correlation between PFAS in agricultural soil and irrigation water. The concentration of PFAS ranged from 10.5 to 22.4 ng g^{-1} (Li et al., 2019). PFAS concentrations in the leaves and soil of agricultural land in Brazil reached up to 1 ng g^{-1} and 5.4 ng g^{-1}, respectively (Nascimento et al., 2018). PFOS and perfluorooctane sulfonamide were the most frequently detected and dominant compounds. The authors identified sulfuramid

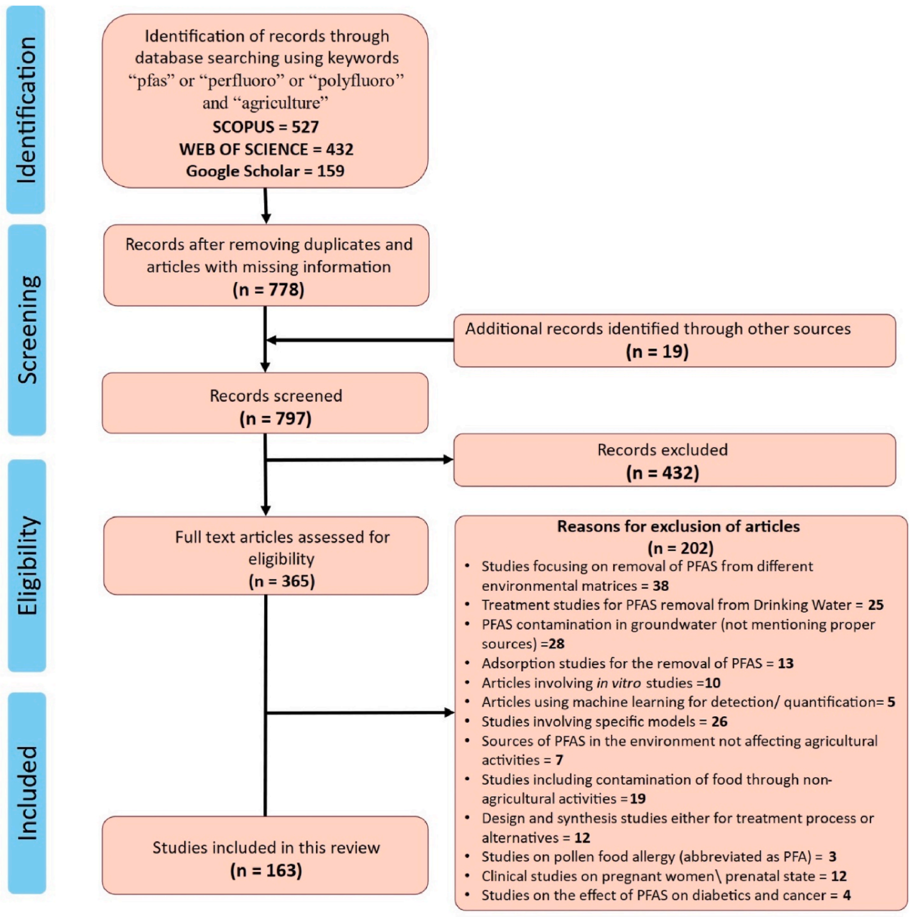

Fig. 1. PRISMA flowchart of the review.

pesticide as a possible source of PFAS contamination (Nascimento et al., 2018). Large-scale contamination of agricultural soils in Germany was identified due to the application of PFAS-contaminated compost materials (Röhler et al., 2021). At one of the German sites, 6300 μg kg^{-1} of PFOS and PFOA were detected in the topsoil, contributing to the contamination of approximately 1802 ha of farmland (Röhler et al., 2021). The following are the major sources of PFAS in agricultural soils.

3.1. Pesticides/Insecticides

The presence of PFAS in pesticides occurs when trifluoromethyl (-CF3) groups are intentionally incorporated into their molecular structure. This chemical modification increases hydrophobicity and lipophilicity, enhancing both effectiveness and persistence (Pesticide Action Network Europe, 2023). The use of PFAS-containing pesticides and insecticides in agriculture can lead to their unintentional accumulation in soil and plants, negatively impacting crop yield (Nascimento et al., 2018). While the judicious use of pesticides supports larger food production, excessive application and emerging resistance pose threats to the demand and supply chain (Picó et al., 2021). Consequently, producers prefer novel agrochemicals that are highly selective, minimally toxic and more efficient.

In many developing countries, pesticides such as sulfluramid, flursulamid, and lithium perfluorooctane sulfonate (LPFOS) which contain perfluoroalkyl chains, are widely used to control pests like ants and cockroaches (Ogawa et al., 2020). Sulfluramid, a pesticide containing N-ethyl perfluorooctane sulfonamide (EtFOSA) as its active ingredient at a concentration of 3000 mg L^{-1}, has been banned in several countries due to its role as a precursor of PFOS. However, it remains commonly used across most of Latin America to control leaf-cutting ants (Vinha et al., 2020).

A Brazilian study detected ∑PFAS concentrations of up to 1020 pg L^{-1} in coastal water, 979 pg g^{-1} in eucalyptus leaves, 5400 pg g^{-1} in soil, and 198 pg g^{-1} in coastal sediment, with PFOS and perfluorooctane sulfonamide (FOSA) as the primary PFAS contributors(Nascimento et al., 2018). The transformation products of EtFOSA, including PFOS and FOSA, were identified in Brazilian leaves, soil, marine waters, and coastal sediments, suggesting that coastal sediments serve as a significant PFAS sink (Nascimento et al., 2018). The study’s authors identified pesticide application as the primary PFAS source in the region.

Both biotic and abiotic processes can facilitate the transformation of EtFOSA into perfluorooctane sulfonamido acetic acid (FOSAA), PFOA, FOSA, PFOS, and other PFAS in the environment (Letcher et al., 2014; Nascimento et al., 2018; Rhoads et al., 2008; Zhang et al., 2017). Due to substantial evidence of their toxicity, environmental durability, long-range transport, and potential for bioaccumulation and biomagnification, the Stockholm Convention on POPs added perfluorooctane sulfonyl fluoride (PFOSF), PFOS, and its salts to its global restriction list in 2009 (Wang et al., 2009). However, Brazil, a leading global agricultural producer, exempted PFOSF for sulfluramid production (Nascimento et al., 2018).

Between 2004 and 2015, Brazil consumed an estimated 30 tons of EtFOSA annually, making it the world’s third-largest user of PFOS and PFOS-related substances from 2003 to 2008 (Nascimento et al., 2018). From 2010 to 2018, EtFOSA may have resulted in PFOS emissions up to 616 tons (Barbosa Machado Torres et al., 2022).

In a report published by the Maryland Pesticide Education Network and Public Employees for Environmental Responsibility (PEER), it was found that the insecticide commonly used by the Maryland Department of Agriculture (USA) for the state’s annual mosquito control program

Table 1
Sources of PFAS contamination in agriculture.

Source Classification	Source Description	Usage	Types of PFAS	Country	Concentration of PFAS detected	Reference
Pesticide	Sulfuramid	Pest Control (ants and cockroaches)	PFOS, FOSA, EtFOSA	Brazil	EtFOSA: 3000 mg L^{-1} in the pesticide	Vinha et al. (2020).
	Premanone 30-30	Mosquito control	PFOA, HFPO-DA (GenX)	USA	PFOA: 3500 ng L^{-1}, HFPO-DA: 630 ng L^{-1} in the pesticide	Bennett and Berlin (2021)
	Anvil 10 + 10	Mosquito control	PFOA, HFPO-DA	USA	PFOA: 250 g L^{-1}, HFPO-DA: 260–500 g L^{-1} in the pesticide	Abel (2020)
	Intrepid 2F, Malathion 5 EC, Oberon 2SC	Agricultural Pest Control	PFBS, PFOA, PFHpS, PFBA	USA	Intrepid 2F: PFBS: 350 ng L^{-1}, Malathion 5 EC: PFOA: 510 ng L^{-1}, Oberon 2SC: PFBA: 1500 ng L^{-1} in the pesticide	Donley (2023)
Biowaste	Biosolids	Soil amendment/ fertilizer	Long-chain PFAS, short-chain PFAS	Germany	15.3 kg of PFAAs in the amendment	Stahl et al. (2018)
	Composted biosolids	Soil amendment	PFHxA, PFBS, PFOA	USA	PFBS: 0.66–0.45 μg L-1, PFHxA: 0.98–0.31 μg L^{-1}; PFOA: 0.14 μg L^{-1} in the leachate samples	Levine et al. (2023)
	Sewage sludge	Soil amendment	PFOS, PFOA	Australia	PFOS: 253 ng g^{-1}, PFOA: 7.90 ng g^{-1} in the biosolids	Kumar et al. (2023).
	Biosolids from agricultural industrial and urban waste	Soil amendment	PFOS, short-chain PFAS	France	agricultural biosolids: ∑46 PFAS: 0.66 μg kg^{-1} dryweight industrial and urban waste: ∑46 PFAS: 220 μg kg^{-1}	Munoz et al. (2021)
	Commercial compost containing food packaging	Soil amendment	PFOA, PFOS, short-chain PFAAs	USA	PFOA and PFOS in all samples; short-chain PFAA >64 % in compost	Choi et al. (2019)
	Sewage sludge from WWTP	Agricultural soil application	PFOA, short-chain PFAS	China	∑PFAS: 126–809 ng g−1 dw; PFOA: 23.2–298 ng g−1 dry weight in sludge	Yan et al. (2012)
	Commercial composts and garden soils	Agricultural soil amendment	Short-chain PFAS	USA	1.26–11.84 μg kg^{-1} dry weight in sludge	Sivaram et al. (2022)
	Sewage sludge from 43 WWTPs	Soil amendment	ΣPFAS and PFOS	Czech Republic	ΣPFAS: 5.6–963.2 ng g^{-1} and PFOS: 932.9 ng g^{-1} in biosolids	Semerád et al. (2020)
	limed biosolids WWTP	Land application	12 PFAS, mostly PFNA, PFOA, PFOS	USA	PFNA: 25.1 ng g^{-1} dry weight, PFOA: 23.5 ng g^{-1} dry weight, PFOS: 22.5 ng g^{-1} dry weight in biosolids	Armstrong et al. (2016)
Water and wastewater	Wastewater from 19 WWTPs	Treated wastewater for irrigation	PFOA, PFNA, PFDA, PFHpA, PFPeA, PFHxA	Australia	ΣPFAS: 9.3–520 ng L^{-1}; average 110 ng L^{-1} in water samples	Coggan et al. (2019)
	Recycled water from WWTP	Irrigation for agricultural crops	20 PFAS, mostly PFOS, PFBS, PFOA, PFBA	Australia	PFOS: <0.03–34 ng L^{-1}; PFOA: 2.2 ng L^{-1}; PFBS: 4.4 ng L^{-1}; PFBA: 6.1 ng L^{-1} in groundwater	Szabo et al. (2018)
	Reclaimed wastewater for irrigation	Irrigation for crops	PFOS, PFOA	USA	ΣPFAS 115 ng L^{-1}; PFOS: 22 ng L^{-1}; PFOA: 30 ng L^{-1}; in water from monitoring wells	Mroczko et al. (2022)
	Wastewater effluent enters River, River water used for irrigation	Irrigation for crops	∑PFAS, mostly PFOA and PFPeA in effluent, PFPeA, PFOA and PFDA downstream of the river	Jordan	∑PFAS in WWTP effluent: 14–24 ng L^{-1}; ∑PFAS in downstream of river: 16–27 ng L^{-1}	Scher et al. (2018)
	Contaminated groundwater	Soil irrigation, gardening	PFBA, PFPeA	USA	PFBA: 2.5 μg L^{-1} (max), PFPeA: 0.18 μg L^{-1} (max.)	(Brandsma et al., 2019)
Airborne emissions	Fluoropolymer production facility, operational from 1951 to 2013	Deposition on surface water and agricultural soils	PFOA, HFPO-DA	USA	PFOA at 143 ng L^{-1} and HFPO-DA at 42 ng L^{-1} in surface water 28 km from the plant	(Galloway et al., 2020)
	Fluoropolymer manufacturing plant	Deposition on surface water and leaves	GenX, PFOA	Netherlands	PFOA in leaves: 0.7–11 ng g^{-1} wet weight; GenX in water: 1.4–8.0 ng L^{-1} PFOA in water: 1.9–7.1 ng L^{-1}	(Brandsma et al., 2019)
	Airborne emissions from Teflon production	Atmospheric deposition	PFOA	USA	Significant accumulation downwind of factories	Schroeder et al. (2021)

contains potentially harmful PFAS compounds (Bennett and Berlin, 2021). The insecticide, Premanone 30-30, was found to have approximately 3500 ng L^{-1} of PFOA and around 630 ng L^{-1} of hexafluoropropylene oxide dimer acid (HFPO-DA, also called GenX) (Bennett and Berlin, 2021). The report also notes that testing detected only 36 PFAS compounds, leaving open the possibility that other PFAS may be present in Premanone 30-30. Additionally, the source of contamination remains unidentified (Bennett and Berlin, 2021). According to the PEER, the pesticide Anvil 10 + 10 has been sprayed over millions of acres in Florida, New York, Massachusetts, and at least 23 other states in the USA to control mosquitoes (Abel, 2020) contributing 250 g L^{-1} of PFOA and 260–500 g L^{-1} of HFPO-DA to overall PFAS contamination.

Similarly, findings from the Center for Biological Diversity and PEER indicate that California's most widely used insecticide, Intrepid 2F, along with Malathion 5 EC and Oberon 2SC, contains potentially harmful levels of PFAS (Donley, 2023). Intrepid 2F is extensively used in California (USA), with approximately 1.7 million pounds applied to 1.3 million acres of agricultural land in 2021. It was found to contain 350 ng L^{-1} of perfluorobutane sulfonate (PFBS) (Donley, 2023). Malathion 50 EC contained 510 ng L^{-1} of PFOA and 680 ng L^{-1} of perfluoroheptane sulfonate (PFHpS), with PFOA levels exceeding drinking water safety threshold (4 ng L^{-1}) by more than 100,000 times (Brusseau, 2023). Oberon 2SC, was found to contain 1500 ng L^{-1} of perfluorooctanoic acid (PFOA) (Donley, 2023).

The US EPA identifies fluorinated containers as a major source of PFAS contamination in pesticides, as PFAS compounds can migrate from

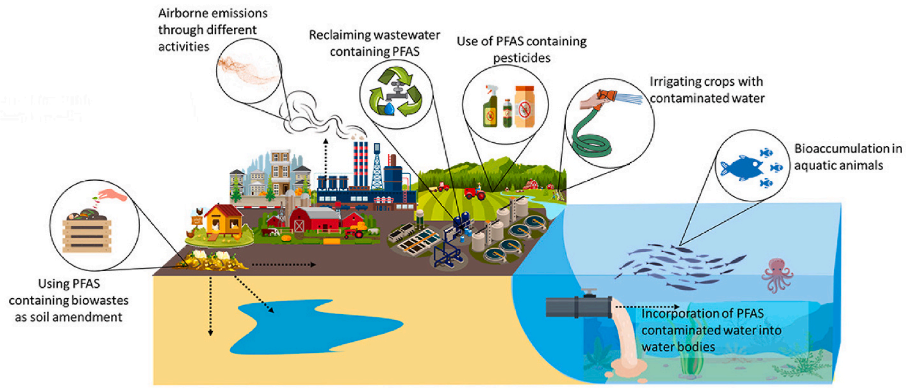

Fig. 2. Sources of PFAS in agricultural land and food chain.

container liners into pesticide formulations. These containers, often treated with fluorinated coatings to enhance chemical resistance, may leach PFOA, PFOS, and other perfluoroalkyl compounds under certain conditions. However, recent investigations highlight an important distinction: PFHpS and PFBS, two commonly detected PFAS, were not found to leach from fluorinated containers during testing (US EPA, 2023).

A targeted analysis of PFAS in insecticide formulations used in the United States detected PFOS in 6 out of 10 formulations, with concentrations ranging from 3.92 to 19.2 mg kg^{-1} (Lasee et al., 2022). This raised concerns about the presence of PFAS in commonly used insecticides and their potential environmental and health risks. However, further US EPA testing found no traces of PFAS in these formulations (US EPA, 2023). Recently, researchers from the Environmental Working Group identified more than 1400 pesticides containing PFAS in Maine, USA (Environmental Working Group, 2023). This alarming finding underscores the potential widespread presence of PFAS in agricultural chemicals, whether due to intentional inclusion during manufacturing, contamination in production, or leaching from fluorinated packaging materials.

3.2. *Biowastes*

Biowastes are classified as wastes that consist of biodegradable components. They can originate from agricultural and farming activities, municipal solid waste, forest residues, and food industries, among others (Hoang et al., 2022). These wastes are commonly applied as fertilizers and soil amendments to improve soil health and enhance crop yield due to their high nutrient and carbon content. However, if contaminated with PFAS, biowaste application in agricultural fields can serve as a pathway for human and animal PFAS exposure (Bao et al., 2018; Hoang et al., 2022).

The partitioning of PFAS between the soil matrix and plants is influenced by PFAS chain length (Brusseau, 2023). Long-chain PFAS strongly sorb to organic matter in soil, reducing mobility but increasing persistence in topsoil. In contrast, short-chain PFAS exhibit greater solubility and mobility, enabling leaching into deeper soil layers and enhanced plant root absorption. This partitioning behavior directly impacts PFAS accumulation in plants (Brusseau, 2023). Due to their higher mobility, short-chain PFAS are more likely to translocate within plants and accumulate in edible parts. A study in Germany estimated that approximately 15.3 kg of perfluoroalkyl acids (PFAAs) are introduced into agricultural lands solely through biowaste applications (Stahl et al., 2018). It further demonstrated that sewage sludge acts as a reservoir for long-chain PFAAs, while plants readily absorb short-chain PFAAs. These findings highlight the distinct environmental behavior and risks associated with PFAS of varying chain lengths in agricultural systems.

A 12-year study on agricultural land investigated the impact of seasonal variations on PFAS bioaccumulation and mobility following the use of contaminated compost materials (Röhler et al., 2021). Results indicated that short-chain PFAS generation and accumulation of short-chain PFAS peaked in spring and summer, whereas their mobilization occurred primarily in the fall and rainy seasons. In contrast, long-chain PFAS remained stable due to their strong soil sorption. Short-chain PFAS migrated into deeper soil layers before eventually stabilizing, due to the generation of precursors. Meanwhile, long-chain PFAS remained concentrated in topsoil due to their lower mobility (Röhler et al., 2021).

In the USA, a WWTP was authorized to apply its produced sludge to approximately 2000 ha of agricultural fields (Washington et al., 2010). The WWTP received industrial discharges from sectors involved in fluorotelomerization, electrochemical fluorination, and perfluoroalkylate and fluorotelomer compound processing. As a result, the soil amended with sludge exhibited elevated levels of perfluorinated compounds, including PFDA, perfluorododecanoic acid (PFDoA), PFOA and PFOS (Washington et al., 2010). Studies indicate that PFOS and PFOA present in the top 90 cm of soil can migrate to depths of up to 9 m and have been detected in groundwater as deep as 17 m below ground level (Johnson, 2022).

A study in Arizona, USA, identified irrigation water and biosolids as major sources of PFAS contamination across 72 agricultural sites (Pepper et al., 2021). Soil, biosolid, and irrigation water samples were collected and analyzed across various depths. The results showed that none of the PFAS detected in the biosolids were present in agricultural soils. Long-chain PFAS, although present in biosolids, were largely absent from soil, suggesting that biosolids acted as major reservoirs for these compounds. Additionally, N-methyl fluorooctane sulfonamido acetic acid (N-MeFOSAA), a perfluorinated precursor, was present in biosolids but undetectable in soil, indicating potential transformation. PFAS concentrations in soil also decreased by approximately 70 % with increasing depth (Pepper et al., 2021).

In northern Queensland, Australia, PFAS were detected in six biosolid samples, collected from seven regional councils. The most prevalent contaminants were PFOS and PFOA, with the highest recorded PFOS concentration at 253 ng g^{-1} and PFOA at 7.90 ng g^{-1} in the smallest regional council (Kumar et al., 2023).

Biosolids, nutrient-rich organic materials derived from treated sewage sludge, are frequently used as soil amendments to improve soil properties (Stahl et al., 2018). However, they often contain persistent emerging contaminants, such as PFAS. Studies have shown that perfluorohexanoic acid (PFHxA), PFBS and PFOA leach from biosolid-based compost into the soil, with concentrations declining over time (Levine et al., 2023). For instance, the mean concentration of PFBS in leachate decreased from 0.66 to 0.45 μg L^{-1} within the first week and dropped below detection limits by week four. Similarly, PFHxA concentrations fell from 0.98 to 0.31 μg L^{-1}, while PFOA levels in leachate samples remained at 0.14 μg L^{-1}, a concentration comparable to that of PFOA found in most surface waters in the USA (Levine et al., 2023).

In the Czech Republic, the concentration of PFAS in sewage sludge samples from 43 WWTPs ranged from 5.6 to 963.2 ng g^{-1}. PFOS was the most prevalent compound, detected in 60 % of samples with a maximum concentration of 932.9 ng g^{-1} (Semerád et al., 2020). Similarly, another study detected 12 PFAS in municipal WWTP biosolids before land application (Armstrong et al., 2016). Seven PFAS remained stable in sewage sludge over eight years, with the highest mean concentration of perfluorononanoic acid (PFNA), PFOA, and PFOS recorded at 23.5 ng g^{-1} dw (Armstrong et al., 2016). These compounds exhibited concentrations 2.5–5 times higher than other detected PFAS.

In Shanghai, China, an analysis of sewage sludge from 25 WWTPs revealed that PFOA was the predominant PFAS, with short-chain PFAS occurring more frequently than long-chain PFAS (Yan et al., 2012). The authors suggested that PFAS undergo multiple transformations in soil, posing potential risks to both ecological and human health when sewage sludge is applied to agricultural fields (Yan et al., 2012).

A study in France screened 160 PFAS compounds across 47 organic waste materials used in agriculture and found that the target PFAS levels were low in poultry manure and pig slurry (mean $\sum$46 PFAS: 0.66 μg kg^{-1} dw) but significantly higher in industrial and urban wastes, including sewage sludge, household compost, and paper mill sludge (mean $\sum$46 PFAS: 220 μg kg^{-1}) (Munoz et al., 2021). Another study measured 38 PFAS in 19 commercially available composts, garden soils, and potting mixtures, with concentrations ranging from 1.26 to 11.84 μg kg^{-1} dw (Sivaram et al., 2022). Using a total oxidizable precursor (TOP) assay, researchers observed an increase in short-chain PFAS concentrations (0.48–7.63 μg kg^{-1}) suggesting precursor transformation into short-chain PFAS in soil (Sivaram et al., 2022).

The leachability of 17 PFAAs was analyzed in commercial compost samples (Choi et al., 2019). PFAA levels ranged from 28.7 to 75.9 μg kg^{-1} in compost containing food packaging and from 2.38 to 7.60 μg kg^{-1} in compost without food packaging. Although PFOA and PFOS were detected in all samples; short-chain PFAA dominated (>64 %) in municipal solid waste composts (Choi et al., 2019).

3.3. *Wastewater for irrigation*

Wastewater is widely recognized as a major source of PFAS in agriculture (O'Connor et al., 2022). When treated wastewater is partially or inadequately used for irrigation, persistent PFAS can unintentionally accumulate in agricultural products because conventional wastewater treatment methods are largely ineffective at removing PFAS from water.

A study in Australia analyzed water samples from 19 WWTPs and found 21 distinct PFAS compounds, with concentrations ranging from 9.3 to 520 ng L^{-1}, averaging 110 ng L^{-1} (Coggan et al., 2019). Notably, PFCAs such as PFOA, PFNA, PFDA, perfluoroheptanoic acid (PFHpA), perfluoropentanoic acid (PFPeA), and PFHxA were more prevalent in the effluent than in the influent, as their concentration gradually increased throughout the treatment process until final discharge (Coggan et al., 2019). In contrast, alternatives to PFOS, such as 6:2 fluorotelomer sulfonates (6:2 FTS) and F-53B (a mixture of 6:2 chlorinated polyfluoroalkyl ether sulfonate [6:2 Cl-PFAES] and 8:2 Cl-PFAES), were detected at greater concentrations in the influent. Approximately 99 % of the aqueous samples contained 6:2 FTS.

Another Australian study suggested that using treated wastewater for irrigation could contribute to PFAS contamination in groundwater and soil (Szabo et al., 2018). This study analyzed groundwater in an area irrigated with recycled water from a WWTP facility. Among the 20 PFAS compounds detected (including perfluoropentane sulfonate [PFPeS], PFOS, PFBS, PFOA, PFHxA, PFNA, PFDoDS, PFDA, PFHpA, PFBA, PFPeA, PFHxS, PFDS, PFTeDA, 6:2 FTS, PFHpS, PFUnDA, PFNS, PFDoDA, and PFTrDA), PFOA, PFBS, and PFBA were the most prevalent in groundwater. However, the study could not determine the fate of PFAS compounds during treatment, including the transformation of PFAS precursors (Szabo et al., 2018).

Even reclaimed wastewater used for irrigation can serve as a significant source of PFAS compounds. In the USA, researchers found that total PFAS concentrations in water samples from monitoring wells and wastewater discharge in a mixed-use forested and agricultural area subjected to wastewater spray irrigation reached 115 ng L^{-1} (Mroczko et al., 2022). Similarly, in Jordan, the Zarqa River, which receives wastewater from a major WWTP, is also used for irrigation (Shigei et al., 2020). The study revealed that reclaimed wastewater used for irrigation introduced more than 20 PFAS compounds into soil and crops (Shigei et al., 2020).

A study conducted in Minnesota (USA) analyzed soil, groundwater, and produce from several gardens in a region where four disposal sites were used by the 3M Corporation for PFAS-containing waste over decades. The study found that PFOA, PFOS, and PFBA were present in all soil samples, while groundwater contained elevated levels of PFBA, followed by PFPeA. The produce from these gardens also contained high concentrations of PFBA. The study inferred that short-chain PFAS compounds are highly soluble and mobile, facilitating their translocation and bioaccumulation in garden produce (Scher et al., 2018).

3.4. *Airborne emissions*

The evidence on airborne emissions of PFAS compounds is limited, but research confirms that they can be transported through the atmosphere and deposited (Galloway et al., 2020). Once deposited, these compounds can settle on agricultural soils, potentially affecting plants and crops. Recent studies specifically link soil and water contamination to atmospheric deposition of two key PFAS compounds: PFOA and HFPO-DA. For instance, HFPO-DA has been found in atmospheric deposits near industrial sites in West Virginia (USA) and the Netherlands (Brandsma et al., 2019; Galloway et al., 2020).

Additionally, PFAS compounds have been widely detected in settled dust, often originating from various building materials such as insulation, flooring products, and wall coverings. One notable source is wooden flooring treated with sealants and adhesives, which can release multiple PFAS compounds, including PFPeA, PFOA, PFHxA, PFHpA, PFBS, PFHpS, and PFOS.

In regions of New York and Vermont (USA), where Teflon coating processes have been used for over 50 years, PFOA has significantly accumulated in local and downwind areas due to emissions from PFAS-producing (Schroeder et al., 2021). This accumulation provides clear evidence that airborne PFAS emissions can contaminate groundwater over considerable distances downstream from emission sites (Schroeder et al., 2021).

The atmospheric deposition of PFAS compounds onto agricultural land near emission sources may further contribute to soil and water contamination. If groundwater contaminated by these deposits is used in agriculture, it can serve as a persistent source of PFAS exposure.

4. Interaction of PFAS with soil

PFAS interact with environmental matrices based on their alkyl chain length and functional groups (Bräunig et al., 2019). Their sorption process is influenced by electrostatic interactions with soil minerals and organic compounds, as well as hydrophobic interactions with soil organic carbon (Bolan et al., 2021; Bräunig et al., 2019). Additional factors such as clay content, soil pH, salinity, and texture also affect PFAS sorption (Bolan et al., 2021; Cai et al., 2023). Among PFAS compounds, sorption increases with chain lengths, which in turn reduces leaching and bioavailability (Navarro et al., 2024; Srivastava et al., 2024 Within the same chain length, PFSAs exhibit higher sorption than PFCAs (Campos Pereira et al., 2018; McLachlan et al., 2019). At higher pH levels, the sorption of anionic PFAAs decreases due to increased anionic charge. Moreover, longer-chain PFAAs preferentially sorb to humin soil organic matter, while shorter-chain PFAAs interact more with fulvic and humic domains (Li et al., 2018). Variations in organic matter sources from different crops and management practices further influence PFAS mobility, retention, and plant uptake potential (Campos Pereira et al., 2018). Inorganic anions, such as phosphate, also impact PFAS sorption by competing for binding sites. For instance, phosphate reduces PFOS sorption, particularly in soils rich in ferric oxide and low in organic matter (Qian et al., 2017).

Understanding soil toxicology is crucial because PFAS interactions affect toxicity, environmental persistence, and bioavailability, while also shaping soil health, plant uptake, and broader ecological consequences. When multiple contaminants co-occur in soil, their interactions can modify overall toxicity. For example, PFAS may exhibit synergistic effects when combined with heavy metals or organic contaminants, resulting in enhanced toxicity (Cai et al., 2023). Conversely, certain soil amendments or environmental factors may reduce PFAS mobility or toxicity leading to antagonistic effects. For instance, organic matter can bind with PFAS, limiting their bioavailability to plants or microorganisms and mitigating toxic effects (Li et al., 2018).

Additionally, the presence of multiple PFAS compounds may lead to additive effects, particularly among chemically similar compounds such as long-chain PFAAs. These additive effects become more significant when contaminants persist in the environment for extended periods. For example, the simultaneous presence of PFOA and PFOS in soil has been shown to produce additive effects, where their combined impact on soil organisms and plant uptake corresponds to the total PFAS load (Stahl et al., 2009).

5. Uptake of PFAS by crops

The uptake of PFAS by various crops and plants can be understood through both occurrence surveys and systematic plant uptake studies using known PFAS concentrations. One of the earliest studies on PFAS uptake examined the effects of PFOS and PFOA on multiple plant species, including potatoes, oats, spring wheat, perennial ryegrass, and maize (Stahl et al., 2009). Results indicated that PFOA and PFOS were present in different plant parts, with greater accumulation observed at higher doses. In most species (except potatoes), PFOA concentrations exceeded those of PFOS and were generally higher in vegetative tissues compared to storage organs.

In a separate study conducted in Saudi Arabia, natural vegetation and edible crops in wastewater-impacted areas were analyzed for 21 PFAS compounds (Picó et al., 2021). PFAS concentrations in surface water and sediment were 29.7 ng g^{-1} and 5.66 ng g^{-1}, respectively. Measured concentrations in crops varied, with values of 0.46 ng g^{-1} in human food crops, 3.2 ng g^{-1} in farm crops, and 1.88 ng g^{-1} in wild plants.

5.1. Factors affecting plant update of PFAS

PFAS uptake by plants varies widely across species due to three primary factors: (i) plant protein content, (ii) root systems, and (iii) biomass accumulation (Ghisi et al., 2019). Species with higher protein content tend to accumulate more PFAS, as these compounds bind with protein-rich tissues (Wen et al., 2016). For instance, leafy vegetables such as spinach and kale, which contain relatively higher protein levels, often accumulate greater PFAS concentrations.

Root system structure also significantly influences uptake. Fibrous root systems (e.g., grasses and cereals) provide a greater surface area for PFAS absorption compared to taproot systems (e.g., carrots and radishes). Additionally, species with larger biomass tend to accumulate more PFAS due to their increased tissue capacity to sequester contaminants.

Similar to other contaminants, PFAS uptake is governed by plant water and nutrient demands until a saturation threshold is reached (Gobelius et al., 2017). Uptake and distribution are further influenced by carbon chain length, salinity, temperature, and, in some cases, pH. For example, maize root uptake of PFCAs decreases as chain length increases from 4 to 7 carbons, but uptake rises again for chains exceeding 7 carbons (Table 2). pH also affects uptake; in acidic environments, PFDA uptake is 1.6 times greater than at neutral pH (pH 5 = 2.51 μg g^{-1} vs. pH 7 = 1.52 μg g^{-1} dry root weight), while other PFCAs (PFPeA, PFHpA, PFNA, PFOA) and PFSAs (PFBS, PFHxS, PFOS) do not exhibit clear pH-dependent uptake patterns (Krippner et al., 2014). Conversely, wheat plants accumulate more PFOS at neutral pH than at low or high pH (Zhao et al., 2016).

Salinity has been shown to enhance PFCA uptake, with a strong positive correlation ($R^2 > 0.84$). For example, wheat plants exhibit a 2.9-fold, 3.3-fold, 4.2-fold, and 2.8-fold increase in uptake of PFBA, PFHpA, PFOA, and PFDoA, respectively, as salinity rises from 0 % to 0.4 % (Zhao et al., 2016). Similarly, temperature plays a key role, with higher temperatures (30 °C) leading to 1.5- to 2.3-fold greater PFCA uptake than at 20 °C (Zhao et al., 2016).

Roots serve as the primary pathway for PFAS uptake (Qi et al., 2022), but several processes may limit absorption, including PFAS sorption to soil particles and root surfaces, as well as abiotic or biotic transformations in the soil. Although PFAS exhibit relatively low sorption to soil, these interactions reduce dissolved PFAS availability for root uptake.

Beyond root uptake, PFAS can also enter plants through foliar absorption. Vapor-phase or particle-bound PFAS can be absorbed by aboveground tissues, as demonstrated by the detection of PFHxA (17–559 μg kg^{-1}) in tree bark near a fluorochemical plant—an environment with minimal water flow through bark (Jin et al., 2018).

5.2. Mechanism of plant uptake of PFAS

Studies suggest both active and passive uptake mechanisms. For instance, PFOA uptake in maize appears energy-dependent, as it significantly decreases with metabolic inhibitors, whereas PFOS uptake is less affected, suggesting a passive process (Wen et al., 2013). Similarly, PFAS uptake in wheat exhibits dose-dependent inhibition with Na_3VO_4, further supporting active transport for certain PFAS (L. Zhang et al., 2019a). While initial research has identified cellular transport pathways, many aspects remain unclear, particularly species-specific uptake differences and the role of membrane channels and carriers (Blaine et al., 2014; Wen et al., 2013).

PFAS, characterized by a hydrophilic head group and a hydrophobic -CF_2 chain tail, exhibit relatively high water solubility and can be transported to plant roots due to the water potential gradient created by transpiration (Wang et al., 2020). While the movement of PFAS from soil solution to plant roots is well-documented, the mechanisms underlying root uptake require further investigation (Mei et al., 2021; Wang et al., 2020, 2020b). Studies on the wetland plant *Alisma orientale* indicate that PFOA and PFOS were found in intercellular spaces, cell walls, and various organelles, including mitochondria, cytoplasm, and the nucleus. This suggests that PFAS compounds can adhere to cell walls and move both within and between cells (Wang et al., 2020a).

Chemicals in the soil solution can be absorbed through the epidermis and subsequently enter the vascular tissue of plant roots via three uptake pathways: apoplastic (through adjacent cell walls), symplastic (through plasmodesmata), or transmembrane (through water channels in the plasma membrane) (Fig. 3). In *A. orientale*, both PFOA and PFOS were detected in these spaces (Wang et al., 2020a). Furthermore, the cell and tissue distribution of PFOA and PFOS in eight wetland plants (*Thalia dealbata, Phragmites australis, Cyperus alternifolius, Canna indica, Cyperus papyrus, Arundo donax, Pontederia cordata,* and *Alisma orientale*) was analyzed using Desorption Electrospray Ionization Mass Spectrometry, which mapped their presence based on ion intensity. This study revealed that transport from the hydroponic solution to the root cortex occurred via both the apoplastic and symplastic pathways in four of these plants (*Phragmites australis, Canna indica, Cyperus alternifolius,* and *Thalia dealbata*) (Wang et al., 2020b). After penetrating root cells, PFAS can enter the root vascular cylinder and ascend to the shoot system through

Table 2
PFAS uptake in various plants or crops and associated uptake parameters.

Name and part(s) of the plant/crop	PFAS compound(s)	Treatment concentration [a]	Concentration measured in plant part	Calculated uptake or distribution parameter [b]	Reference
Maize - root	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS.	100 μg L^{-1}	PFBA: 2.46 μg g^{-1} PFPeA: 0.87 μg g^{-1} PFHxA: 0.35 μg g^{-1} PFHpA: 0.12 μg g^{-1} PFOA: 0.58 μg g^{-1} PFNA: 1.22 μg g^{-1} PFDA: 1.95 μg g^{-1} PFBS: 1.04 μg g^{-1} PFHxS: 0.70 μg g^{-1} PFOS: 3.63 μg g^{-1}	–	Krippner et al. (2014)
Wheat – root and shoot	PFBA, PFHpA, PFDoA, PFOA	1 μg mL^{-1}	PFBA: 200–500 ng g^{-1} (root), 450–1300 ng g^{-1} (shoot) PFHpA: 580–1450 ng g^{-1} (root), 200–1050 ng g^{-1} (shoot) PFDoA: 0–1000 ng g^{-1} (root), 9050–25,000 ng g^{-1} (shoot) PFOA: 550–2300 ng g^{-1} (root), 250–750 ng g^{-1} (shoot).	Transfer factor[c] for PFBA: 0.85 PFHpA: 0.46 PFDoA: 0.33 PFOA: 0.43	(H. Zhao et al., 2016)
Birch, spruce, bird cherry, ground elder, and mountain ash trees – foliage and twigs.	26 PFAS compounds were analyzed. 17 were detected in soil and 10 were detected in plants.	Occurrence study – Stockholm Arlanda airport. Total PFAS concentration in soil = 16–160 ng g^{-1}	Total PFAS concentration in foliage of birch = 12–97 ng g^{-1}, spruce = 14–94 ng g^{-1}, bird cherry = 4.3–21 ng g^{-1}, ground elder = 0.89–23 ng g^{-1}, mountain as = 2.1–3.1 ng g^{-1} and in twigs of birch = 5.3–40 ng g^{-1}, spruce = 4.1–4.2 ng g^{-1}, bird cherry = 1.2–1.3 ng g^{-1}, ground elder = 0.87–2.2 ng g^{-1}, mountain ash = 0.15 0.78 ng g^{-1}.	Total tree burden[d] for birch = 1.5–11 mg) and for spruce = 0.26–1.8 mg)	Gobelius et al. (2017)
Spinach, tomato, and corn	PFBS, PFOS, PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUdA, PFDoA.	Spinach was exposed to field soil with ΣPFAS = 106.87 ng g^{-1}, tomato was exposed to ΣPFAS = 7.82 ng g^{-1}, and corn was exposed to 50 mg kg^{-1} of PFOS dose.	Spinach ΣPFAS = 0.99–5.33 ng g^{-1}. Tomato ΣPFAS = 61.3 ± 8.04 ng g^{-1} (fruit), 96.50 ± 41.35 ng g^{-1} (leaves), 26.84 ± 4.17 ng g^{-1} (stem), 9.17 ± 0.90 ng g^{-1} (root). Corn PFOS = 254 ± 72.3 μg g^{-1} (root), 23.1 ± 6.13 μg g^{-1} (leaf).	Transfer factor for PFOS = 4.76 (spinach) and 3.44 (tomato root), Bioaccumulation factor[e] for PFOS = 5.48 (tomato stem), 22.2 (tomato leaf), 12.6 (tomato fruit).	Navarro et al. (2017)
Yam roots, maize cobs, and sugarcane stems.	26 PFAS compounds including 13 PFCAs, 4 PFSAs, 3 FOSAs, 2FOSEs, 3 FOSAAs, and FTSA. However, only PFHpA, PFOA, PFNA, PFBS and FOSA were identified in plants.	Occurrence study – Nakivubo wetland and Lake Victoria at Kampala, Uganda. ΣPFAS in soil = 1700–7900 pg g^{-1}.	ΣPFAS = 200 ± 64 pg g^{-1} (maize cobs) 350 ± 64 pg g^{-1} (sugarcane stem) 360 ± 170 pg g^{-1} (yam roots).	Concentration ratio = yam root/soil - 0.53–0.59 (PFBS), maize cob/soil = 0.56 (PFBS), Sugarcane stem/soil = 0.65–0.67 (PFHpA).	(Dalahmeh et al., 2018)
Rice – roots, straw, unhulled rice, white rice, and bran.	5 PFSAs (PFBS, PFHxS, PFHpS, PFOS, and PFDS), 11 PFCAs (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, and PFTeDA), FOSA, N-EtFOSA, and N-EtFOSAA.	Not mentioned	(In the order of root, straw, unhulled rice, white rice, and bran) PFBS: 4, <4, <6, <18, <44 pg g^{-1} PFHxS: 3865, 365, 164, <4, 30 pg g^{-1} PFHpS: 3375, 509, 90, <19, <48 pg g^{-1} PFOS: 55,485, 11,586, 364, <19, 102 pg g^{-1} PFDS: 117, 63, <27, <19, <48 pg g^{-1} FOSA: 14,217, 3644, 50, 9, 41 pg g^{-1} N-EtFOSA: 9017, 816, 79, 33, <50 pg g^{-1} N-EtFOSAA: 3079, 348, <28, <80, <200 pg g^{-1} PFBA: 45, 94, 277, 7, 55 pg g^{-1} PFPeA: 72, 196, 110, <20, <50 pg g^{-1} PFHxA: 160, 701, 37, 23, 54 pg g^{-1}	Transfer factors for ΣPFAS: 0.001–4 (white rice), 0.005–10 (bran), 0.002–10 (unhulled rice).	(Yamazaki et al., 2019)

(*continued on next page*)

Table 2 (*continued*)

Name and part(s) of the plant/crop	PFAS compound(s)	Treatment concentration [a]	Concentration measured in plant part	Calculated uptake or distribution parameter [b]	Reference
			PFHpA: 148, 281, 87, <20, <10 pg g^{-1} PFOA: 1214, 1882, 21, <20, <50 pg g^{-1} PFNA: 384, 232, 25, 7, 19 pg g^{-1} PFDA: 170, 145, <6, <20, <50 pg g^{-1} PFUnDA: 532, 269, <28, <20, <50 pg g^{-1} PFDoDA: 66, 55, <6, <20, <50 pg g^{-1} PFTrDA: 108, 160, 9, <20, <50 pg g^{-1} PFTeDA: 35, 36, 10, <20, <50 pg g^{-1}		
Grains: wheat, corn, and soybean. Vegetables: Radish root and shoot, carrot root and shoot, Chinese chives, Chinese cabbage, lettuce, Welsh onion, pepper, and cauliflower.	9 PFCAs and 3 PFSAs.	Occurrence study – open-air fields near a mega fluorochemical industrial park. ΣPFAS in soil = 79.9 ng g^{-1} – 200 ng g^{-1} and 2.09 ng g^{-1} to 3.75 ng g^{-1} in fields which are 0.3 km and 10 km away from the plant.	ΣPFAS = 2355 ng g^{-1} (shoot vegetables), 1115 ng g^{-1} (fruit vegetables), 410 ng g^{-1} (flower vegetable), 333 ng g^{-1} (root vegetables), and 580 ng g^{-1} (grain crops).	Bioaccumulation factor for ΣPFAS = 24.3 (shoot vegetables), 6.63 (fruit vegetables), 4.23 (flower vegetables), 4.05 (grain crops), and 3.58 (root vegetables).	(Liu et al., 2019)
Wheat grain, maize grain and leaves, Jerusalem artichoke, and ryegrass.	16 perfluoroalkane sulfonamidoethanol (FASE)-based transformation products and 18 PFAAs.	Wheat grain, maize grain and leaves, and Jerusalem artichoke were grown on PFAS contaminated and PFAS free fields.	ΣPFAS = 0.1931 μg kg^{-1} (wheat grain), 0.0083 μg kg^{-1} (maize grain), 0.969 μg kg^{-1} (maize leaves), 0.1645 μg kg^{-1} (Jerusalem artichoke), and 0.4601 μg kg^{-1} (ryegrass)	-	Muschket et al. (2020)
Red chicory	9 PFAAs including PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, and PFOS.	Red chicory was grown in nutrient solution spiked with different concentrations of PFAAs mixture (62.5, 125, and 250 μg L^{-1}).	ΣPFAS in root = 10,990.3 ng g^{-1} (for 62.5 μg L^{-1}), 29,035.7 ng g^{-1} (for 125 μg L^{-1}), and 48,674.5 ng g^{-1} (for 250 μg L^{-1}). ΣPFAS in shoot = 2765.5 ng g^{-1} (for 62.5 μg L^{-1}), 4157 ng g^{-1} (for 125 μg L^{-1}), and 8982.8 ng g^{-1} (for 250 μg L^{-1}).	Translocation factor[f] for PFBA = 3.9, PFPeA = 4.3, PFHxA = 2.3, PFHpA = 0.91, PFOA = 0.28, PFNA = 0.08, PFDA = 0.02, PFBS = 1.6, PFOS = 0.03.	Gredelj et al. (2020)
Natural vegetation and crops	21 PFAS	Occurrence study – seven locations at South Riyadh and six locations at Al-Jubail industrial city (Saudi Arabia).	ΣPFAS = 20 ng g^{-1} (crop for consumption, n = 22), 20.5 ng g^{-1} (crop in farm, n = 25), and 9.12 ng g^{-1} (natural vegetation, n = 8).	-	Picó et al. (2021)

[a] The concentration of PFAS used in the experiment. For occurrence studies, soil concentration is considered the treatment concentration. The unit follows the methodology of each respective study: w/w if plants are grown in soil, and w/v if plants are grown in nutrient solution.

[b] Different studies use different uptake parameters. Empty cells indicate no parameter was calculated in the respective study.

[c] Transfer factor also known as the root concentration factor is the ratio of the PFAS concentration in the plant part to the concentration in soil or nutrient solution. Refer to Section 5.3.

[d] Sum of PFAS concentration in root, trunk, twigs, and foliage. Refer to Section 5.3.

[e] It is the ratio of PFAS concentration measured on a dry weight basis, between the different crop organs and the corresponding soils. Refer to Section 5.3.

[f] The translocation factor (TF) is an indicator of PFAS translocation from the roots to the aboveground parts. A TF greater than one suggests that the chemical bioaccumulates more in shoots compared to roots. Refer to Section 5.3.

the xylem or phloem.

5.3. Distribution and accumulation of PFAS in plants

The uptake, translocation, and bioaccumulation of PFAS in plants can be quantified using various parameters, such as the root concentration factor (RCF), translocation factor (TF), transpiration stream concentration factor (TSCF), total tree burden (TB), and bioaccumulation factor (BAF) (Table 2) (Dalahmeh et al., 2018; Yamazaki et al., 2019).

- RCF represents the ratio of PFAS concentration in roots to that in soil or nutrient solution (Equation (1)) and reflects root uptake efficiency.
- TF indicates the movement of PFAS from roots to aboveground parts (Equation (2)).
- TSCF provides a more precise translocation measurement by comparing PFAS concentrations in xylem sap (which transports chemicals via transpiration water) with those in soil (Equation (3)).
- TB quantifies PFAS accumulation in larger plants by considering concentration across multiple plant tissues (Equation (4)).
- BAF measures the accumulation of PFAS in specific plant parts, such as fruits and leaves, which may enter the food chain (Equation (5)).

$$RCF = \frac{C_{roots}}{C_{soil}} \tag{1}$$

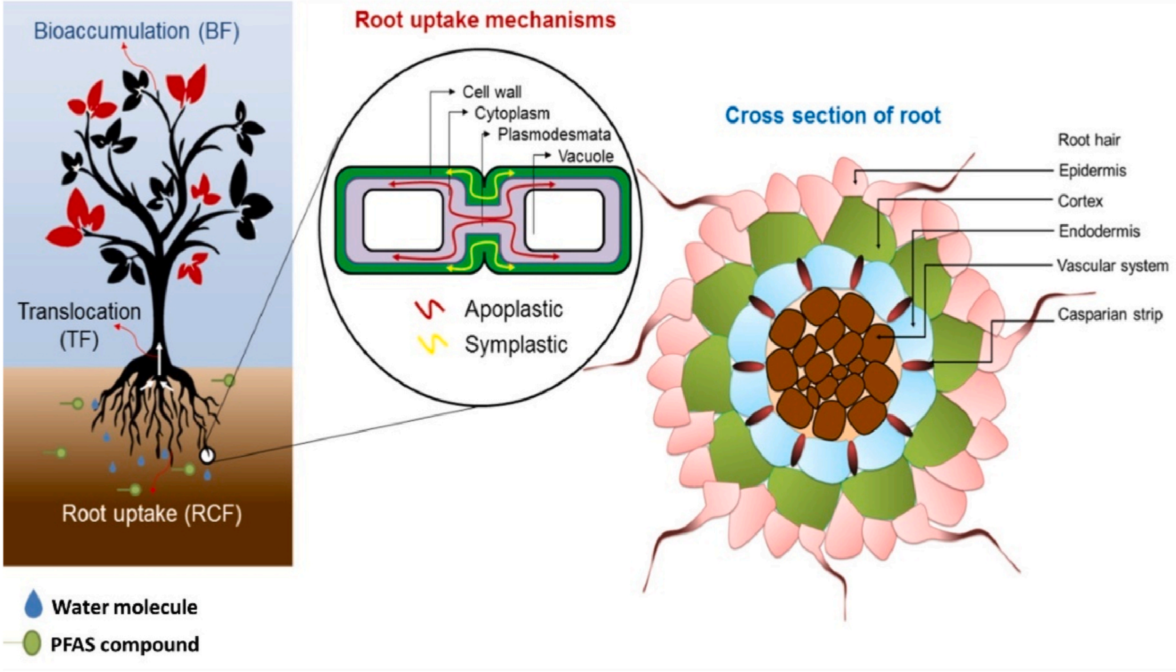

Fig. 3. Plant root uptake pathways and related uptake parameters (RCF: Root Concentration Factor; TF: Translocation Factor; BF: Bioaccumulation Factor).

$$TF = \frac{C_{shoots}}{C_{roots}} \tag{2}$$

$$TSCF = \frac{C_{shoots} \times m_{shoots}}{C_{soil} \times V_{trans}} \tag{3}$$

$$TB = \sum PFASs_{root} + \sum PFASs_{trunk} + \sum PFASs_{twig} + \sum PFASs_{foilage} \tag{4}$$

$$BAF = \frac{C_{shoot}}{C_{soil}} \tag{5}$$

where C_{roots} C_{soil}, and C_{shoot} are PFAS concentrations (ng g^{-1}) in roots, soil (or nutrient solution), and shoot, respectively. m_{shoots} is the dry shoot mass (g), and V_{trans} is transpired water volume (L).

Higher vascular plants possess a specialized barrier in the root endodermis, known as Casparian strips, which partially restricts apoplastic entry into the inner vascular system (Fig. 3) (Grebe, 2011). These strips, composed of hydrophobic lignin and suberin, function similarly to tight junctions in animal cells, preventing the unregulated movement of substances. In plants with Casparian strips, apoplastic PFAS uptake is limited, thereby reducing its entry into the vascular system (Grebe, 2011; Mei et al., 2021; Wang et al., 2020, 2020a, 2020b; Xu et al., 2022). However, short-chain PFAS can penetrate Casparian strips more easily, leading to higher uptake and translocation. Consequently, long-chain PFAS tend to accumulate in roots, whereas short-chain PFAS are more readily transported (Krippner et al., 2015). Fig. 4 illustrates this phenomenon, showing that long-chain PFAS exhibit higher RCF values (indicating greater root accumulation) and lower TF values (indicating restricted translocation to shoots).

Bioaccumulation of PFAS increases with carbon chain length (Houde et al., 2011). In terrestrial food webs, PFAS with 9–11 carbon atoms exhibit the highest trophic magnification factors, whereas marine environments show twice the magnification rates in terrestrial ones (Müller et al., 2011). The half-life of PFAS varies significantly among species and chemical structures (Kudo, 2015).

Due to their strong affinity for proteins, PFAS are primarily detected in liver and blood (Brown et al., 2020). However, PFAS concentrations also vary across different tissues, including kidney, bile, lung, skin, fat, muscle, and liver (Kudo, 2015; Pizzurro et al., 2019). Some PFAS compounds, due to their proteinophilicity, can transfer via eggs (Wilson et al., 2021), milk, and the placenta providing direct pathways for human exposure (Kato et al., 2015; Pizzurro et al., 2019). A study identified contaminated water and food as the primary sources of human PFAS exposure (Jian et al., 2017). In the USA, the total PFAS concentration in blood samples from the exposed population was 11 μg L^{-1} in 2013–2014 (Graber et al., 2019).

In a study on wheat plants, the concentration of trifluoroacetate in the shoots (indicative of translocation) was found to be approximately 2.5 times higher than that of PFOS and about 5.5 times higher than that of PFOA (Zhang et al., 2019b). Fig. 4a and b illustrate the average values of RCF and TF of PFAS for various crops, respectively, based on previous studies, while Fig. 5 presents the BAF of four abundant PFAS. Notably, PFAS accumulation in terms of concentration was higher in root and shoot vegetables compared to fruit or flower vegetables, likely due to the role of water uptake and transpiration in PFAS translocation within plants (Meng et al., 2024). Since PFAS move with water molecules, they translocate more rapidly to plant parts with higher water demand, which may explain the lower BAF values observed in grains and cereals. Moreover, leaves exhibited greater PFAS accumulation than roots, fruits, or other shoot organs in a multi-crop study involving two root vegetables (carrot and radish), four shoot/leaf vegetables (Chinese cabbage, Welsh onion, Chinese chives, and lettuce), one fruit (pepper), one flower (cauliflower), and two grains (corn and wheat). Interestingly, leaf blades of root crops like carrots and radishes had the highest PFAS concentrations, even among root vegetables. (Liu et al., 2019).

6. A brief overview of remediation of PFAS-Contaminated soil

Soil contamination with PFAS has become a significant environmental and public health concern due to their persistent and toxic nature. The limited availability of effective remediation techniques stems from the unique physicochemical properties of PFAS, which influence their behavior in soil systems. The efficacy of these treatment methods largely depends on soil characteristics such as mineralogy, clay and organic content, pH, and salinity, all of which affect adsorption of PFAS (Naidu et al., 2020). This section provides an overview of various techniques proposed for remediating PFAS-contaminated soil.

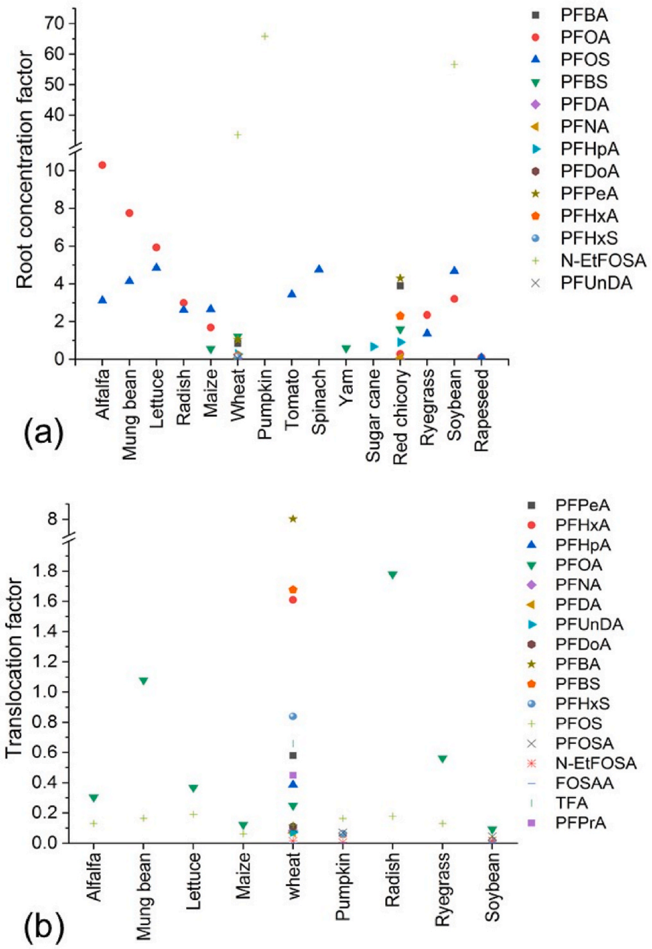

Fig. 4. (a) Root concentration factor and (b) Translocation factor of various PFAS in crops (Blaine et al., 2014; Dalahmeh et al., 2018; Felizeter et al., 2012; Gredelj et al., 2020; Lan et al., 2018; Navarro et al., 2017; Xiang et al., 2018; Yamazaki et al., 2019; Yu et al., 2018; L. Zhang et al., 2019a; Zhao et al., 2016, 2018, 2017, 2014).Click or tap here to enter text.

6.1. *Chemical remediation methods*

6.1.1. *Chemical oxidation*

Chemical oxidation is a widely studied method for PFAS degradation and can be applied either in situ or ex situ. This process involves the use of highly reactive oxidants such as ozone, hydrogen peroxide, sodium persulfate, and potassium permanganate to break down organic pollutants into CO_2 or other more biodegradable compounds (Bolan et al., 2021).

In situ chemical oxidation involves direct injection of oxidants into contaminated soil and groundwater. Typically, vertical injection wells introduce oxidants under pressure, facilitating contaminant degradation (Dombrowski et al., 2018). However, if potable water sources are nearby, extraction wells must be installed to prevent further groundwater contamination. This method is particularly effective in permeable soil, where oxidants can efficiently interact with contaminants. The success of oxidation-based remediation depends on multiple factors, including soil composition, contaminant concentration, and temperature. Oxidants may be applied individually or in combination to enhance treatment efficiency (Umeh et al., 2023).

Ozone is a highly reactive oxidant, is 12 times more soluble than oxygen and readily transitions into the aqueous phase, generating hydroxyl radicals. Peroxone (a combination of ozone and hydrogen peroxide) and Fenton's reagent also produce hydroxyl radicals at neutral or low pH. However, these radicals alone are ineffective in breaking the C-F bond in PFAS. Due to the high electronegativity of fluorine atoms along the carbon chain, PFAS oxidation is slow and requires significant energy input (Vecitis et al., 2009).

Studies have demonstrated that oxidation at an alkaline pH (11) results in 80–100 % degradation of PFOS and PFOA, whereas acidic conditions (pH 4–5) yield no measurable degradation. Alkaline environments promote the formation of superoxide radicals ($\bullet O_2^-$), which enhance PFAS breakdown (Lin et al., 2012). Permanganate (MnO_4^-) is another commonly used oxidant due to its stability. Research indicates that PFOA degradation reaches 46.7 % and 71.7 % after 18 days at pH 2, when heated to 65 °C and 85 °C, respectively. Additionally, reducing pH and adding MnO_2 have been shown to improve PFOS degradation rates (Liu et al., 2012).

Persulphate ($S_2O_8^{2-}$), a strong oxidant, is widely employed for treating environmental pollutants. It can directly react with PFAS or be activated photochemically or thermally at elevated temperatures and pH levels (Dombrowski et al., 2018). Activated persulfates have also been investigated for their efficacy in degrading PFOA and other PFCAs (Lee et al., 2009).

To maximize the efficiency of chemical oxidation, oxidants should be applied in saturated or semi-saturated soil, allowing them to move freely through the medium and interact with the pollutants (Mukhopadhyay et al., 2021). Monitoring the transport, distribution, and degradation of oxidants and pollutants throughout the treatment process is essential for optimizing remediation outcomes. However, long-chain PFAS compounds exhibit strong resistance to oxidation due to their strong C-F bonds. Incomplete oxidation can result in toxic byproducts, potentially exacerbating environmental hazards.

While strong oxidants such as ozone, hydrogen peroxide, and persulfates effectively degrade PFAS, their application can significantly alter soil chemistry and disrupt microbial communities. For instance, highly oxidative conditions often increase soil pH beyond 10, negatively impacting microbial diversity, nutrient cycling, and enzymatic activity (Mukhopadhyay et al., 2021). Because most natural soils maintain near-neutral pH, the risk of oxidative stress must be carefully assessed. Mitigation strategies, including the use of buffering agents and controlled oxidant delivery, may help minimize adverse impacts while maintaining treatment effectiveness.

6.1.2. *Chemical reduction*

The chemical reduction methods are used for PFAS degradation, which can be applied ex situ or in situ depending on the site-specific requirements. This approach utilizes both reactive free radicals and direct electron transfer to degrade PFAS compounds. Typically, reductants such as ferrous iron, zero-valent iron, and sodium dithionite have been tested for PFAS degradation. These reductants either generate reducing radicals like hydrated electrons (e^- aq) and hydrogen radicals ($\bullet$H), or transfer electrons directly (Liu et al., 2016). The efficiency of chemical reduction generally depends on several factors including soil pH, contaminant concentration, reductant concentration, and the presence of ions in the soil (Nzeribe et al., 2019).

Despite its potential, this approach faces several limitations that hinder its effectiveness and broader application. These challenges arise primarily from the intrinsic chemical properties of PFAS and complexities of the reduction process itself. Chemical reduction often leads to incomplete degradation, producing intermediate products that may still be environmentally persistent or toxic. This issue is common in many chemical treatment methods, where the degradation process does not fully mineralize PFAS into non-toxic end products (Nzeribe et al., 2018). Moreover, the fate and toxicity of PFAS degradation products remain poorly understood, raising concerns about the potential formation of unknown or harmful byproducts (Thapa et al., 2024).

The effectiveness of chemical reduction varies significantly based on soil type, organic content, and other environmental factors. For instance, high organic content in soils can hinder the reduction process, as hydrophobic PFAS are difficult to remove from organic-rich

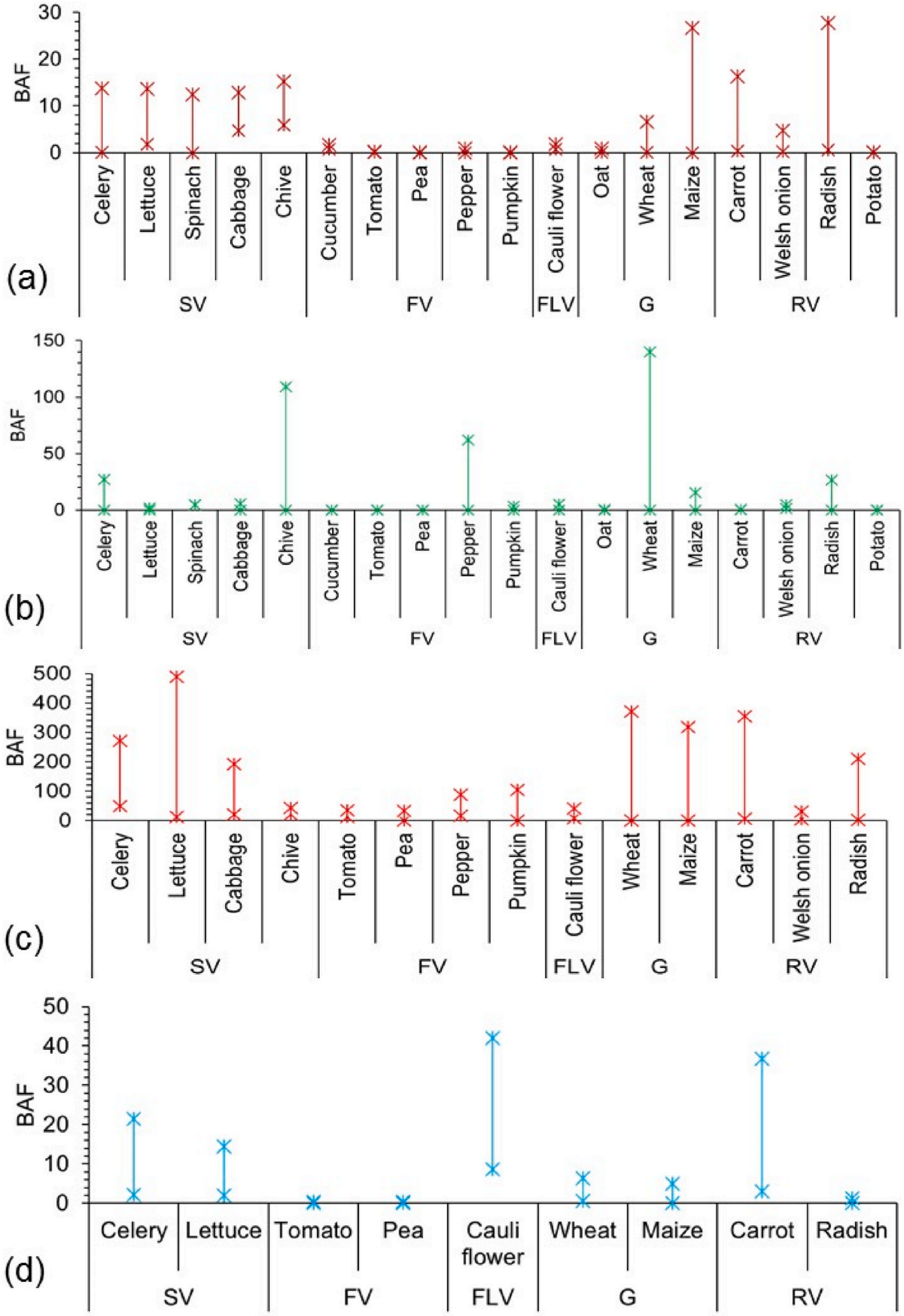

Fig. 5. Range of bioaccumulation factor (BAF) of (a) PFOA; (b) PFOS; (c) PFBA; and (d) PFBS in various shoot vegetables (SV), fruit vegetables (FV), flower vegetables (FLV), grains (G), and root vegetables (RV). Line represents the range and points mark minimum and maximum BAF (Bizkarguenaga et al., 2016; Krippner et al., 2014, 2015; Lechner and Knapp, 2011; Liu et al., 2019; Stahl et al., 2009; Xu et al., 2022).

sediments (Nguyen et al., 2023). Similarly, co-contaminants present in soil may also interfere with the reduction process, leading to competitive reactions that diminish PFAS degradation efficiency (Eberle et al., 2017).

Additionally, chemical reduction often requires tightly controlled conditions, such as specific pH and redox environments, which are challenging to maintain in situ. As a result, its application is often limited to controlled environments, such as laboratory or pilot-scale studies (Eberle et al., 2017). The cost and energy demands of chemical reduction can also be significant, particularly for large-scale remediation efforts. Furthermore, additional treatment steps are often necessary to manage byproducts and ensure complete PFAS removal, further increasing operational costs (King and Chaplin, 2024).

6.1.3. Mobilization

The mobilization technique for PFAS remediation in soil involves the use of agents to enhance PFAS desorption and subsequent extraction from contaminated soils. This approach relies on processes such as solubilization, complexation reactions, and desorption. Commonly employed methods include soil flushing and plant uptake, which facilitate contaminant removal. In soil flushing, a flushing solution is injected into the ground to extract contaminants (Hale et al., 2017). This technique enables in situ treatment of large soil volumes without the need for excavation or transportation (Jawitz et al., 2000). The efficacy of soil flushing in removing POPs relies on surfactants with both hydrophobic and hydrophilic structural groups (Bolan et al., 2021). Additionally, various soil flushing additives, including organic and inorganic acids, bases, and solvents such as ethanol or methanol, can enhance PFAS removal. PFAS extraction from solid and aqueous matrices has also been achieved using supercritical fluid and reverse osmosis-assisted methods (Bolan et al., 2021).

While mobilization techniques facilitate PFAS removal, they pose a significant risk of spreading contamination to groundwater and surface water. This risk is particularly high for short-chain PFAS, which are highly mobile and prone to desorption, potentially introducing more toxic, bioavailable forms into adjacent water sources. Moreover, PFAS mobilization efficiency varies based on chain length and chemical composition; as shorter-chain PFAS are generally easier to mobilize, whereas long-chain PFAS exhibit stronger hydrophobic interactions, reducing their mobility (Nguyen et al., 2022; Usman et al., 2024). The diverse physicochemical properties of PFAS compounds necessitate tailored remediation approaches for optimal effectiveness (Wang et al., 2018). Additionally, soil characteristics – such as pH, clay content, organic matter, and saturation conditions – significantly influence mobilization success, necessitating site-specific considerations (Sleep and Juhasz, 2021).

Selecting effective mobilizing agents, such as methanol or hydroxypropyl-β-cyclodextrin (HPCD) is critical, as their performance depends on both PFAS types and soil conditions (Usman et al., 2024). However, these agents may also pose environmental risks, including secondary contamination or adverse effects on soil microbiota, highlighting the importance of careful selection (D'Annibale et al., 2013). Moreover, the feasibility of large-scale mobilization is limited by cost considerations and the need for site-specific modifications (Bierbaum et al., 2022). Ensuring the long-term removal of PFAS remains challenging, as incomplete extraction or subsequent re-sorption may necessitate ongoing monitoring and supplementary treatments to maintain stability and protect environmental health (Khair Biek et al., 2023).

6.1.4. Immobilization (Sorption and Stabilization)

The immobilization approach involves introducing various sorbents such as activated carbon, carbon nanotubes, and biomaterials, into contaminated soil to bind PFAS and reduce their mobility. This technique prevents PFAS leaching into groundwater (CRCCARE, 2017). By binding PFAS within the soil matrix, sorbents restrict their movement, making immobilization a widely employed remediation strategy.

For instance, at an airfield in Norway, activated carbon was successfully used to eliminate PFOS, PFOA, and PFHxS completely from the aqueous phase (Kupryianchyk et al., 2016). Similarly, a study at a Norwegian airport site demonstrated that incorporating montmorillonite and activated carbon into sandy soil resulted in a 99 % and 33 % reduction in PFOS leaching, respectively (Hale et al., 2017). Additionally, biochar produced from waste timber at 900 °C has shown remarkable efficacy, reducing PFAS leaching by 98–100 % in heavily contaminated soil (Sørmo et al., 2021).

Although activated carbon is highly effective for PFAS removal, its efficiency declines when PFAS concentrations are high or when organic co-contaminants are present. Alternative sorbents, such as resins and biomaterials, can also be effective but may be cost-prohibitive and degrade over time. Meanwhile, minerals such as silica, clays, iron oxides, and zeolites have demonstrated potential for contaminant removal from groundwater and soil (ITRC, 2018). Notably, organoclays with modified surfaces exhibit enhanced PFOS and PFOA sorption. Additionally, an amine-modified palygorskite clay sorbent has been patented for PFOS and PFOA treatment due to its effectiveness (Rusmin et al., 2016).

The adsorption efficiency of PFAS depends on multiple factors. For example, adsorption decreases in alkaline conditions (Das et al., 2013) and is reduced in soils with low clay and organic content (Higgins and Luthy, 2006). Furthermore, PFAS chain length also plays a key role, with longer-chain PFAS exhibiting greater adsorption capacity than their shorter-chain counterparts (Darlington et al., 2018).

Despite their promise, PFAS immobilization faces several challenges that impact its effectiveness. Soil variability significantly influences remediation outcomes, as factors like pH, clay, and organic matter content directly affect sorption efficiency (Esmaeilian et al., 2023; Sleep and Juhasz, 2021). Additionally, in real-world applications, soil heterogeneity can lead to uneven distribution of immobilization agents, resulting in inconsistent PFAS retention (Illangasekare et al., 2023).

The chemical diversity of PFAS compounds further complicates immobilization efforts. As discussed earlier, long-chain PFAS bind more readily to sorbents, while short-chain PFAS are more mobile and less likely to remain immobilized in the soil matrix (Zhang and Liang, 2022). Moreover, long-term stability and field-scale applications present additional hurdles. Environmental factors, such as weathering and microbial activity can gradually reduce sorbent efficacy, increasing the risk of PFAS re-release (Zhang and Liang, 2022). On a larger scale, challenges such as soil disturbance and water infiltration further complicate efforts to maintain uniform PFAS immobilization (Illangasekare et al., 2023).

Despite these limitations, immobilization remains a valuable tool for PFAS remediation. However, continued research and site-specific adaptations are essential to enhance its consistency and long-term performance.

6.2. Biological remediation methods

Biological remediation is considered the most efficient, cost-effective, and environmentally friendly approach compared to other remediation strategies. Bioremediation involves breaking down, transforming, removing, or immobilizing contaminants through the metabolic activity of living organisms. Its primary goal is to degrade or convert pollutants into less toxic forms. This technique has been successfully applied to eliminate organic pollutants such as polycyclic aromatic hydrocarbons, gasoline, and pesticides. However, significant knowledge gaps remain regarding biological remediation of PFAS (Biswas and Goel, 2022; Kucharzyk et al., 2017).

One of the primary challenges in PFAS biodegradation is the strength of the carbon-fluorine (C-F) bond, which makes these compounds highly resistant to microbial breakdown, requiring high energy to initiate degradation. Microorganisms are unable to utilize the fluorine-saturated carbon chain as an energy source due to the formation of a hydrophobic barrier around carbon-carbon (C-C) bonds and the absence of hydrogen atoms in the PFAS alkyl chain (Rayne and Forest, 2009). Some bacterial species, such as *Pseudomonas* sp., have demonstrated significant PFAS bioaccumulation under aerobic conditions, though to a much lesser extent in anaerobic environments. However, to date, no verified reports confirm the microbial removal of fluorine atoms from PFAS.

Phytoremediation, a plant-based remediation technique, involves the uptake and sequestration of contaminants, thereby reducing their bioavailability in soil (Bolan et al., 2022). For instance, one study reported that individual PFAS were removed by up to 96 % through plant uptake in a full-scale tropical constructed wetland (Yin et al., 2017). Another study examined the uptake and accumulation of seven PFAS by the wetland species *Juncus effusus*, reporting a removal efficiency of 11.4 % (mass basis) from solution for spiked PFAS (W. Zhang et al., 2019c). Similarly, plants at a firefighting site near Stockholm, Sweden, were found to have accumulated 26 PFAS compounds (Gobelius et al., 2017). Herbaceous species such as *Amaranthus tricolor*, *Equisetum hyemale*, and *Schedonorus arundinaceus* exhibited above-average accumulation of multiple PFAS substances. Additionally, tree species like *Fraxinus pennsylvanica*, *Liquidambar styraciflua*, and *Betula nigra* have shown some capacity for PFAS accumulation in woody tissues. However, field studies are essential to verify the practicality of phytoremediation in remediating PFAS-contaminated soil, particularly regarding the excretion and potential release of PFAS from plants.

Despite its environmental and economic advantages, biological remediation faces notable challenges in treating PFAS-contaminated soils. The chemical stability of the C-F bond significantly limits microbial degradation, hindering complete removal (Beškoski et al., 2024;

Shahsavari et al., 2021). Although phytoremediation facilitates PFAS uptake, improper disposal of contaminated plants poses a risk of introducing PFAS into the food chain. Furthermore, PFAS's bioaccumulative nature adds complexity, as sequestered compounds may lead to secondary contamination, necessitating careful management to prevent unintended environmental consequences (Beškoski et al., 2024).

The effectiveness of bioremediation also depends on site-specific factors, including soil pH, organic matter content, and co-contaminants, which influence PFAS bioavailability and microbial activity (Gu, 2024; Sleep and Juhasz, 2021). Additionally, over time, PFAS sequestration in soil reduces microbial accessibility, further complicating degradation (Gu, 2024). Limited availability of effective PFAS-degrading microorganisms and discrepancies between laboratory and field conditions present additional hurdles (Romantschuk et al., 2023). To overcome these challenges, hybrid approaches integrating biological, chemical, or physical treatments may be necessary. Advances in microbial engineering could enhance degradation rates and minimize the formation of incomplete breakdown by-products (Nguyen et al., 2023).

7. Environmental significance

The limited understanding of PFAS accumulation in agricultural soils raises concerns about their potential uptake by livestock, crops, and food sources, which could pose health risks to consumers. The presence of PFAS in agricultural soils, crops, water sources, and food products necessitates a comprehensive understanding of their sources, uptake mechanisms, ecological impacts, and remediation strategies. Effectively addressing PFAS contamination is essential to safeguarding food safety, ecosystem integrity, and public health. This review serves as a guideline for future research aimed at mitigating PFAS risks in agricultural systems.

8. Conclusions

The continued presence of PFAS in various environmental matrices suggests that these chemicals remain in use despite efforts to phase them out. This persistent contamination has negatively impacted agricultural soils, posing risks to the food chain. Human activities have contributed to the increasing prevalence of PFAS in agroecosystems, resulting in adverse effects on animal and human health. Short-chain PFAS exhibit higher bioaccumulation potential than long-chain counterparts due to their greater mobility and shorter retention time in plants.

Treatment technologies such as chemical oxidation, stabilization, and bioremediation offer promising approaches for mitigating PFAS contamination in soil. However, their effectiveness depends on soil characteristics, including mineralogy, clay and organic content, pH, and salinity. To validate these techniques, pilot-scale field studies should be conducted.

This review provides a foundation for understanding PFAS sources in agricultural soils, their plant uptake, and remediation strategies. Future research should focus on identifying PFAS sources in agricultural ecosystems, including industrial waste, paper mill effluent, pesticides, and recycled water. Additionally, studies should investigate the impact of PFAS on irrigation water quality and explore the effects of land application of treated effluent or waste-derived fertilizers. Expanding PFAS quantification efforts to include a broader range of compounds will help align research with emerging scientific developments. Moreover, advanced analytical techniques should be employed to detect additional PFAS, including precursors that may serve as long-term reservoirs for mobile PFAS. A deeper understanding of plant-specific uptake pathways and physiological responses to PFAS exposure is also critical, particularly under realistic environmental conditions.

CRediT authorship contribution statement

Bishwatma Biswas: Writing – review & editing, Writing – original draft, Visualization, Formal analysis. **Anuja Joseph:** Writing – original draft, Visualization, Formal analysis. **Naseeba Parveen:** Writing – original draft, Visualization. **Ved Prakash Ranjan:** Writing – original draft. **Sudha Goel:** Writing – review & editing, Supervision, Conceptualization. **Jajati Mandal:** Writing – original draft. **Prashant Srivastava:** Writing – review & editing, Writing – original draft, Supervision, 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.

Data availability

No data was used for the research described in the article.

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