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Contaminants in agricultural soils

Challenges and solutions

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Per- and polyfluoroalkyl substances (PFAS) as contaminants in agricultural soils

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1 Introduction

Per- and poly-fluoroalkyl substances (PFAS) are a diverse family of several thousand synthetic compounds, now coined 'forever chemicals' due to their high thermal and chemical stability and resistance to microbial mineralization (Schymanski et al., 2023). PFAS have been found in all environmental compartments globally due to their extensive use (Espartero et al., 2022; Panieri et al., 2022). The use and number of different compounds in this family have increased exponentially since they were first discovered (Gaines, 2022). Their stability, along with other unique properties including imparting grease-, water- and stain-repelling properties, have resulted in their use in a myriad of consumer products and manufacturing processes for several decades (Gaines, 2022; Glüge et al., 2020). Familiar uses of PFAS include grease and stain-repellant coatings on food contact materials, carpets, furniture, and other textiles, as well as on non-stick cookware and construction materials (Kotthoff et al., 2015). PFAS are present in numerous personal care products including

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lotions, shampoos, and cosmetics, as well as other cleaning products and automotive fluids (Whitehead et al., 2021). Since the 1960s, PFAS-containing aqueous-film forming foams (AFFFs) have been in use at airports and military sites because of their ability to extinguish hydrocarbon-based fires in a few seconds (Place & Field, 2012; Prevedouros et al., 2006). PFAS are also used in manufacturing electronics including semiconductor and computer chips, wire coatings, metal plating, medical devices, artificial turf, and automotive fluids among many others (Chen et al., 2024; Jacob et al., 2021; Lin et al., 2017). PFAS use in so many aspects of society, coupled to their persistence has resulted in their presence at varying scales in nearly all environmental compartments including drinking water, surface water, groundwater, soil, air, sediment, and household dust (Evich et al., 2022; Espartero et al., 2022; Panieri et al., 2022). Their ubiquitous presence and persistence, bioaccumulation potential, and linkages to numerous adverse effects in humans and animals, even at low concentrations (Fenton et al., 2021; Halldorsson et al., 2012; Dickman & Aga, 2022a; Lee et al., 2020; NASEM, 2022) have resulted in mounting concerns with regards to our water resources and food system.

PFAS presence in water used for agriculture, atmospheric deposition, and PFAS accumulation within the soil profile, where crops are grown and animals graze, are major challenges in agricultural systems. This chapter will focus on these challenges prefaced by briefly highlighting key PFAS characteristics followed by environmental entry modes and fate of PFAS, impacts on human and ecosystem health, and potential mitigation strategies in agriculture to keep PFAS-impacted farms viable and minimize PFAS impacts to our food system.

2 PFAS characteristics

PFAS occur in two broad classes, polymers and non-polymers, with the latter including the PFAS of greatest concern in terms of exposure to humans and biota. Unique to the chemical structure of most PFAS is the presence of one or more perfluorocarbons that are typically wihtin or as a linear or branched carbon chain of varying lengths (C_nF_{2n+1}-) bound to a polar functional group (e.g., carboxylic acid, sulfonate, alcohol, phosphate, amino group, or other group), which may be attached to a nonfluorinated hydrocarbon entity. The combination of chain lengths, number of perfluoroalkyl tails, polar functional groups and associated linkages govern their behavior, sub-classification, and nomenclature (Buck et al., 2011). Two classes within the PFAS family that have received the most attention are the perfluoroalkyl sulfonic acids (PFSAs) and the perfluoroalkyl carboxylic acids (PFCAs), grouped together as perfluoroalkyl acids (PFAAs), (Kwiatkowski et al., 2020; Wang et al., 2015). PFAAs are known as the terminal products of microbial degradation of larger PFAS known as precursor compounds. While PFAAs have garnered the most attention, in recent

years, we have found that the bulk of the PFAS-derived organofluorine present in our sludges, often converted to biosolids for land application, are precursors (Alukkal et al., 2024b; Lee et al., 2014a; Thompson et al., 2023). Therefore, although PFAS use in some processes and products is decreasing through phaseouts and stewardship programs (Gallen et al., 2022; Ulrich et al., 2016), alternative PFAS are being produced and the capacity to quantify a wider variety of PFAS is increasing (Dickman & Aga, 2022b). Moreover, legislative response to limit the production and use of PFAS varies globally. As early as 2009, the use of perfluorooctane sulfonate (PFOS) was partially restricted through the United Nations Stockholm Convention Persistent Organic Pollutants regulation, followed by a complete ban and addition of perfluorooctanoic acid (PFOA) in 2020 (Brennan et al., 2021). More recently in 2023, the European Chemicals Agency took legislative steps through the EU Registration, Evaluation, Authorization, and Restriction of Chemicals program to restrict all legacy and novel PFAS use except in firefighting foams starting in 2026 (Thomas et al., 2023). Other countries including Australia and Canada has taken steps to regulate the use of PFAS especially long-chain PFAAs. Legislation in the United State is less cohesive with federal and state governments considering different measures. The US EPA, which is responsible for PFAS regulation at the federal level in the USA, recently approved enforceable drinking water limits for six PFAS including PFOS, PFOA, perfluorohexanesulfonic acid (PFHxS), perfluorononanoic acid (PFNA), perfluorobutanoic acid (PFBS) and Gen-X (EPA, 2024a), although the 2025 administration have put a pause on the regulations for all but PFOS and PFOA. Meanwhile, individual states in the USA have pursued legislative actions, some of which are targeting banning PFAS in fire-fighting foam and consumer products, limiting land-application of biosolids and adopting independent drinking water standards (Langenback & Wilson, 2021; Hughes, 2023; BCLP, 2025a, 2025b). Despite some successes, enforcements of PFAS policies in the US remains a challenge at the national and state levels due to political and economic constraints, public awareness, feasibility of implementation, and inadequate scientific evidence. Ultimately, the nonuniformity in worldwide response to PFAS contamination poses difficulties to reducing PFAS emissions given the global nature of markets and economies.

PFAS are both hydrophobic (perfluoroalkyl tail) and lipophilic (polar functional group), which give them their surfactant behavior, and are proteinophilic. Within each PFAS class, they are grouped as short and long chains with the longer chains having higher sorption and bioaccumulation propensity. PFAS affinity for proteins has made some PFAS highly bioaccumulative, leading to their accumulation in blood, meat, liver, kidneys, and breast milk of humans throughout the world (Jian et al., 2018; Liu et al., 2019a; Poothong et al., 2020) and most recently in brains (Xie et al., 2024). The longer the perfluoroalkyl tail length, the higher their bioaccumulation and longer their half-lives within biota

(Khan et al., 2023; Shi et al., 2018). PFAS has been found in the blood samples of nearly all human beings (Calafat et al., 2019; Göckener et al., 2020a; Haines et al., 2017). In addition, PFAS are frequently found in wildlife, domesticated animal products, and foods (Brown et al., 2020; De Silva et al., 2021; Death et al., 2021). PFAS are not acutely toxic but have been associated with numerous adverse human health effects, including some types of cancers, thyroid and heart diseases, and immunosuppression (Fenton et al., 2021; Sinclair et al., 2020; Sunderland et al., 2019). Children's vulnerability to PFAS exposure was highlighted in a systematic review where positive links were observed between PFAS serum levels and dyslipidemia, renal function, and suppression of vaccine response (Rappazzo et al., 2017). Most toxicological studies have focused on PFOS and PFOA given their early detection in human serum and environmental prevalence; however, a recent study (Xie et al., 2023) reported a possible link between perfluorooctane sulfonamide (FOSA) and glioma (a type of brain tumor). Additionally, longitudinal studies of adverse health impact from PFAS exposure have mainly been on workers and communities impacted by PFAS manufacturing (Barry et al., 2013; Darrow et al., 2013; Lopez-Espinosa et al., 2012; Steenland et al., 2013). For example, plant workers' exposure to PFOA was connected to high cholesterol, kidney and testicular cancer, thyroid disease, and ulcerative colitis. Non-occupational exposure to PFAS in communities with contaminated drinking water sources has also been linked to altered thyroid and kidney function (Blake et al., 2018). The adverse effect of PFAS exposure in agricultural communities is currently unclear due to limited evidence. A study in the Veneto region of Italy reported high serum concentrations of PFOA and PFOS in farmers that were exposed to contaminated drinking water (Ingelido et al., 2020). However, the contamination was not from agricultural activities but from a PFAS manufacturing plant. Another study (Batzella et al., 2024) examined the impact of the exposure from the same cohort over a four-year period and found a positive relationship between PFAS concentration in serum and cholesterol levels, which corroborates with an earlier report from Uppsala, Sweden (Dunder et al., 2022). In another longitudinal study investigating lung functions of adolescents and young adults of an agricultural population, no evidence of negative effects from PFAS exposure was observed (Kornher et al., 2025). Similarly, exposure to background levels of eight PFAS mixtures among farmers and other rural residents did not increase the risk of coronary heart disease over the period of 10 years (Mattsson et al., 2015). Nonetheless, the risk of adverse health effects in agricultural workers might increase over time, especially in highly contaminated situations where PFAS exposure is comparable to occupational levels. In aquatic organisms and wildlife, PFAS at environmentally relevant concentrations adversely affected metabolic processes, neurological pathways, immune system function, and overall behavior (Dickman & Aga, 2022a; Lee et al., 2020).

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3 PFAS pathways into the environment and environmental concentrations

PFAS are released to the environment via a myriad of anthropogenic activities and multiple pathways, each with its own magnitude of impact, concentration, and suite of PFAS. Two of the most significant pathways of PFAS to air, land, and water include industrial discharges and aqueous fire-fighting foams (AFFF) releases during military and fire department use or training. Although these modes of entry have been dramatically reduced, they have left a long-term legacy of these persistent PFAS in the environment. Other frequent sources of PFAS into the environment are landfills (air emissions and unintended landfill leaching), wastewater discharge to surface water bodies, wastewater reuse (irrigation water), and land application of biosolids. The latter, along with AFFF-impacted surface and well water used for irrigation and livestock drinking water, are the two biggest sources directly impacting agriculture (Figure 1).

Direct PFAS sources include air emissions and liquid discharges from PFAS manufacturing, and release from PFAS-associated products with AFFFs dominating product sources of PFAS to the environment. PFAS in AFFFs enhance interfacial tension properties required to efficiently extinguish hydrocarbon fires. AFFFs have been used worldwide at military bases, airports, oil and gas refineries, chemical plants, and storage facilities, resulting in direct release of PFAS to soil during transport, storage, fire training activities, and disposal (Prevedouros et al., 2006). Concentrations up to 460 000 ng/g have been reported for PFOS and 50 000 ng/g for PFOA in soils collected from fire training sites (Brusseau et al., 2020).

Airborne-PFAS transport via wind beyond emission locations contributes to background levels of PFAS in soils, including remote areas. A global study analyzing soil samples from 62 locations with minimal to no human impact reported at least three quantifiable concentrations of PFAAs in all the sampling locations (Rankin et al., 2016). Total PFCAs and PFSAs concentrations ranged from 0.29 ng/g to 14.3 ng/g and 0.007 ng/g to 3.7 ng/g, respectively. Similarly, a study conducted in Maine, US reported background concentrations of PFCAs between 0.03 ng/g and 17.4 ng/g and PFSAs 0.16–4.35 ng/g (MDEP, 2022). Relatively higher maximum background concentrations were measured in Vermont, USA for PFCAs and PFSAs at 23 ng/g and 12 ng/g, respectively (Zhu et al., 2022).

4 PFAS routes into agriculture

Due to the extensive use of PFAS in consumer goods, waste disposal systems including landfills and wastewater treatment plants serve as the conduits for

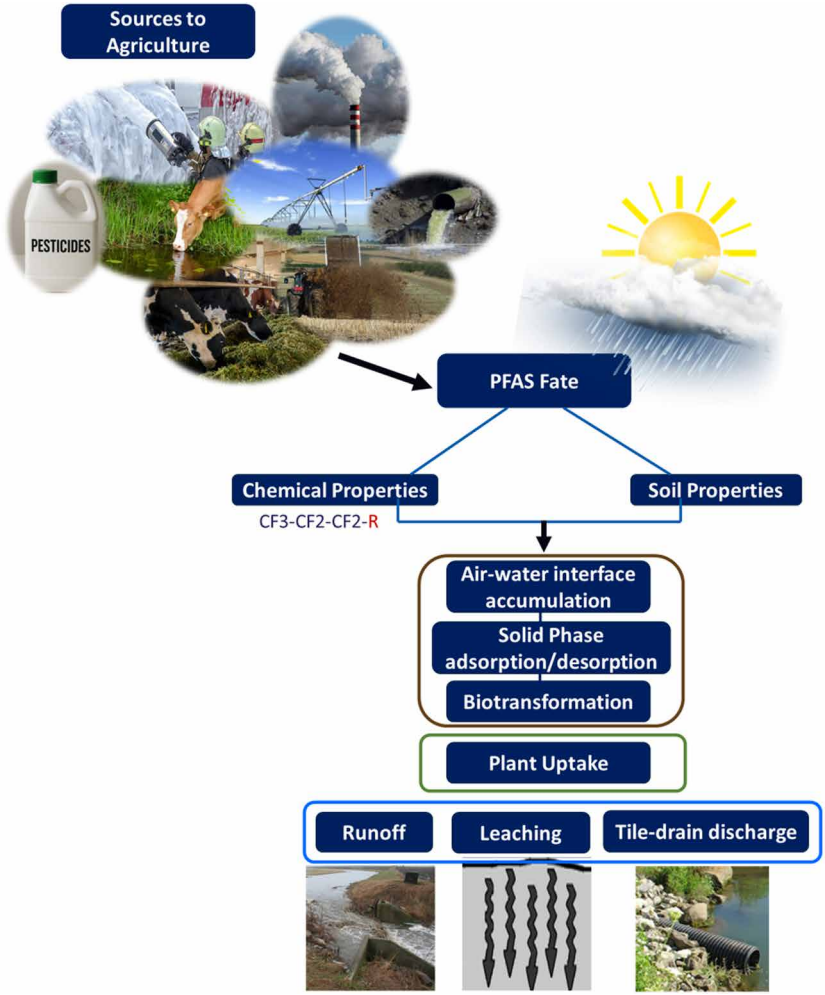

Figure 1 Schematic of primary pathways for PFAS entry into agriculture followed by subsequent fate and transport factors and processes.

PFAS release to agricultural soils in levels higher than prevailing background concentrations. The most documented PFAS sources associated with these systems are biosolids and irrigation water, which includes well water as well as surface water contaminated from a myriad of sources (e.g. wastewater effluent, AFFFs, industrial discharges) (Islam et al., 2025). Free-grazing livestock are particularly vulnerable to PFAS-contaminated surface water (Mikkonen et al., 2023). Additionally, atmospheric deposition, runoff or drainage onto reighboring properties, and agricultural inputs, e.g., pesticides are potential sources of PFAS in agricultural ecosystems.

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4.1 PFAS in land-applied biosolids

Biosolids are end-products of sludge generated from wastewater treatment plants that are rich in organic matter (OM), microbial biomass, and plant nutrients. Conversion of sludge to biosolids occurs through a variety of processes, including anaerobic/aerobic digestion and thermal hydrolysis targeted at reducing odor-producing compounds, vector attraction, and pathogens while improving beneficial reuse and commercial value, and in some cases, biogas production (Wang et al., 2009). The use of biosolids as a soil amendment in agriculture is a global practice. For example, an average of 44% of sewage sludge as biosolids is land-applied in the EU, 73% for Australia in 2021, and 60% in the USA in 2023 (Braine et al., 2024; USEPA, 2024b). Although treatment may alter PFAS present in biosolids through some precursor transformation, PFAS are not removed (Eriksson et al., 2017); Schaefer et al., 2022; Saliu and Sauvé, 2024; Alukkal et al., 2024a). Total PFAS loads reported for biosolids range from <100 to a few thousand µg/kg (Higgins et al., 2005; Sun et al., 2011; Moodie et al., 2021; Munoz et al., 2022; Schaefer et al., 2021), but this range is likely greatly underestimating the PFAS present. Precursor compounds (PFAS that break down to other PFAS) make up a substantial proportion of total PFAS measured in biosolids (Lee et al., 2014a; Thompson et al., 2023) and even for many of the known precursors, standards are not available. Alukkal et al. (2024b) targeted analysis of almost 60 PFAS (including many precursors), but the total calculated PFAS-associated organofluorine was less than 1%. Noteworthy is a recent study that revealed up to 75% of the organofluorine in biosolids originates from CF_n groups in pharmaceuticals, which by most definitions are not deemed PFAS (Ruyle et al., 2025), but that still leaves 25% or more unaccounted organofluorine likely associated with unquantified PFAS. Biosolids generated from industrial wastewater influent and paper fiber residuals typically have higher total PFAS loads but with fewer types of PFAS compared to biosolids derived from domestic wastewater (municipal biosolids) where PFAS concentrations are diffuse and originate from use of a variety of products. Therefore, beneficial reuse of biosolids can redistribute and elevate levels of PFAS in agricultural soils. The differences in PFAS types and concentrations in biosolids are reflected by the variable extent of contamination observed at impacted sites.

4.2 PFAS in irrigation water

In many parts of the world, irrigation with surface water, groundwater, and reclaimed wastewater is essential for crop production. Water resources impacted by landfill leachates, wastewater treatment effluent, AFFF use, and

industrial discharges are potential sources of PFAS to irrigation waters, thus a pathway to agricultural soils as reported in many countries (Pepper et al., 2021; Mroczko et al., 2022; Kim et al., 2019; Shigei et al., 2020).

4.3 PFAS in pesticides

Pesticides formulated with PFAS either as an active ingredient, present in adjuvants or the "inert" ingredients used, residual impurity, or PFAS leached from containers are direct sources of contamination. Fluorination serves to improve many characteristics of the pesticide including stereospecificity and enhanced effects on target pests particularly insects, lipophilicity and residual activity, facilitating application, and reducing pesticide drift (Alexandrino et al., 2022; Ogawa et al., 2020). The single largest contribution to PFAS in pesticides overall is as an active ingredient (Donley et al., 2024). Pesticides are highly diluted before application; however, continuous use contributes to PFAS accumulation in agricultural soils. For example, N-ethyl perfluorooctane sulfonamide (EtFOSA) is as an active ingredient in sulfluramid. EtFOSA degrades into PFOS through various intermediates. Application of this pesticide in Pine and Eucalyptus plantations in Brazil to control leaf cutter ants resulted in low concentrations of PFOS (0.03–5.4 ng/g) in plantation soils (Nascimento et al., 2018). The presence of PFOS has also been reported for insecticides with the following six active ingredients: abamectin, novaluron, mineral oil, imidacloprid, spiromesifen, and malathion (Lasee et al., 2022).

4.4 PFAS in atmospheric deposition

Neutrally charged PFAS are released into the atmosphere during manufacturing and industrial use. In addition, combustion of industrial wastes and landfill waste generate PFAS-containing particles or volatile PFAS compounds to the atmosphere, respectively (Baqar et al., 2024; Lin et al., 2024). PFAS that are emitted partition between particle and gas interfaces until they are deposited onto the biosphere including agricultural soils. Atmospheric deposition of PFAS can proceed via dry and wet deposition mechanisms. Dry deposition refers to release from the atmosphere as gases or particulate matter while wet deposition involves transfer of atmospheric-PFAS in aqueous form during precipitation. Precursors such as fluorotelomer alchohols (FTOHs) and perfluoroalkyl sulfonamides (FASAs) can undergo atmospheric oxidation to PFAAs prior to land deposition (Aschmann et al., 2000; D'Eon et al., 2006Ellis et al., 2004). Soils located near PFAS emission zones are susceptible to substantial levels of contamination. Total PFAS concentrations between 79.9 ng/g and 200 ng/g have been reported in agricultural soils located 300 m from a fluorochemical manufacturing plant (Liu et al., 2019b). Nevertheless, atmospheric deposition

is challenging to delineate because of long-range transport of PFAS across geopolitical boundaries.

5 PFAS fate in agricultural fields

PFAS behavior in the soil and transport outcomes depends on prevailing soil properties and PFAS class (Sharifan et al., 2021) as well as topography, climate, and management (Figure 1). Within a class, differences in perfluorocarbon chain length impact PFAS mobility and bioavailability. Short-chain PFAS are more mobile and bioavailable compared to their long-chain homologs. Both sorption of PFAS to soil surfaces as well as at the air-water interface (AWI) within the vadose zone directly affect their leaching behavior and bioavailability (Post et al., 2017; Lyu et al., 2018). For example, short-chain PFAS absorb less than longer-chain PFAS, leading them to a greater presence in porewater, thus leaching to a greater extent and being taken up more by plants than long-chain PFAS (Li et al., 2020). This leads to the inverse relationship between soil depth and the concentration of long-chain PFAS (Johnson, 2022; Pepper et al., 2023; Sepulvado et al., 2011; Washington et al., 2010; Álvarez-Ruiz et al., 2024). In addition, the presence of organic co-contaminants, nonaqueous phase liquids, or inorganic ions can influence PFAS sorption (Guelfo & Higgins, 2013; Le et al., 2022). For example, divalent cations increase sorption to soil surfaces as well as interfacial sorption of PFAS (Nickerson et al., 2023; Le et al., 2022). Once introduced into the soil, PFAS are subjected to multiple transport processes. Key processes in agricultural settings include sorption, accumulation at AWIs, transformation, plant uptake, surface runoff, and leaching.

5.1 PFAS sorption to soil surfaces

PFAS retention in the soil is controlled by a range of sorption mechanisms as well as air-water interfacial processes. The sorption (K_d, L/kg) of organic compounds to soils is determined by the properties of the compounds (chemical structure, ionic state, and surface-active charge) and the soil conditions (OM, pH, exchange capacity, and soil-solution ionic strength). In the environment, PFAS can exist as anions, cations, zwitterions, and as neutral molecules. The anionic form is more prevalent under typical environmental pH (4–9) ranges given that most PFAS, especially PFCAs and PFSAs, have relatively low acid dissociation constants (pK_a) (PFCAs, <1.6; PFSAs, <0.3) (Goss, 2008; Mei et al., 2021; Rayne & Forest, 2010; Vierke et al., 2013). The interactions between PFAS and soils include hydrophobic interactions, electrostatic interactions, and hydrogen bonding (Mei et al., 2021; Kookana et al., 2022).

Hydrophobic interaction can occur between the perfluoroalkyl chains and soil OM with longer chains having a higher affinity for OM. However, PFAS

sorption is complex and sorption trends derived from laboratory and field studies are not consistent. While several studies have shown organic carbon (OC) to be a driver, others have exemplified a weak relationship between soil OC and K_d values (Li et al., 2018; Kookana et al., 2022; Uhme et al., 2021; Evich et al., 2025). For short-chain PFAAs and some of the longer chain PFCAs, electrostatic and mineralogical features appear to play a greater role in retention, whereas for long-chain PFAS, particularly PFOS, air-water interfacial sorption and sorption to OC are larger drivers. Soil protein content has also been shown to impact PFAS sorption (Li et al., 2019; Longstaffe et al., 2012). Interactions between anionic PFAS and protonated amino and amide groups in proteins are one of the reasons hypothesized for such observations (Li et al., 2019). In addition, given that PFAS commonly occur as mixtures, competitive sorption further complicates sorption processes, especially when PFAS are present at high concentrations such as may be found at military, industrial, or other heavily impacted sites (Uhme et al., 2024), For example, at a dedicated land disposal site where PFAS-laden biosolids were applied annually for 40 years, PFOA along with several short chain PFAS leached to groundwater although most of the other long-chain PFAS mass, e.g., PFOS, was retained in the 0–30 cm depth (Alverez-Ruiz et al., 2024). Substantial PFOA concentrations found in the groundwater in the absence of any other long-chain PFAS reflect the likely role of competitive processes. Lastly, literature on the desorption behavior of PFAS is also inconsistent although data are limited. Desorption hysteresis is driven by slow desorption kinetics, steric hindrances, and/or thermodynamics. Uhme et al. (2024) reported nonlinear sorption isotherms for PFAS and complete reversibility based on initial 48-h batch equilibrations of freshly spiked soils at relatively high concentrations. Conversely, for AFFF historically contaminated soils, Schaefer et al. (2021) and Klamerus et al. (2025) observed linear desorption isotherms but with significant PFAS-dependent non-labile fractions.

For cationic and zwitterionic PFAS, electrostatic interactions dominate and can occur through divalent cation bridging between the negative charges on PFAS (e.g., RF-COO- in PFCAs and RF-SO3- in PFSAs) and soil organic matter (SOM) functional groups (Mejia-Avendaña et al., 2020; Xiao et al., 2019). Electrostatic interactions are pH dependent, with pH affecting both PFAS speciation and the speciation of soil functional groups. Soil cation exchange capacity and anion exchange capacity as well as Al- and Fe-oxides play important roles in electrostatic interactions (Gao & Chorover, 2012; Oliver et al., 2019, 2020). Increasing electrolyte concentration can also increase PFAS "hydrophobic" sorption in soils through the "salting-out" effect (Wang & Shih, 2011). Hydrogen bonds can occur between the hydroxyl groups in SOM and the oxygen atom in the carboxylic (–COOH) and sulfonic ($–SO_3H$) groups for PFCAs and PFSAs, respectively (Mei et al., 2021). Desorption hysteresis has

not been observed for cationic PFAS, whereas zwitterionic PFAS did display desorption hysteresis, which was attributed to entrapment mechanisms (Xiao et al., 2019).

5.2 Air-water interfacial sorption of PFAS

In addition to solid-phase sorption processes that impact most organic chemicals in the subsurface, PFAS with their combined hydrophilic polar head groups and hydrophobic fluorinated tails exhibit extraordinary surface-active properties. Therefore, PFAS can accumulate AWI and subsequently impact migration in the vadose zone (unsaturated) soils. Interfacial surfactant sorption is dependent on soil particle size, moisture content and PFAS affinity to the AWI, which is PFAS structure dependent. PFAS interfacial sorption increases with decreasing particle size and % moisture while increasing with increasing C-F chain length and ionic strength (Silva et al., 2019; Costanza et al., 2019; Fabregat-Palau et al., 2022; Liao et al., 2022; Li et al., 2023; Lyu & Brusseau, 2020). With the transient nature of saturation in the vadose zone, PFAS accumulation at the AWI can influence PFAS migration and reduces the surface tension of water, thus also affecting soil porewater drainage (Arshadi et al., 2024; Brusseau et al., 2021; Karagunduz et al., 2015; Silva et al., 2022). Air-water interfacial adsorption can account for over 50% of PFAS retention under unsaturated conditions (Costanza et al., 2019; Lyu et al., 2018).

PFAS sorption to the AWI relative to aqueous concentration is estimated using the air-water interfacial sorption coefficient (K_{aw}), which can be estimated indirectly through correlations with molar volume (Brusseau, 2019). The appropriate sorption model for fitting measured data to estimate K_{aw} is still debated; K_{aw} values estimated using Freundlich and Langmuir isotherm model fits vary considerably at low PFAS concentrations. Also, competitive sorption for the AWIs for PFAS mixtures has been observed further challenging predictions (Silva et al., 2021). Nevertheless, adsorption to the AWI has been integrated into models simulating steady unsaturated flow of PFAS through the vadose zone. Meanwhile, models for transient flow conditions are more challenging due to changes in the vadose zone interfacial area under natural dry and wet cycles (Guo et al., 2020). Incorporating air-water interfacial sorption into well-known transient flow models (HYDRUS-1D and LEACHM) has proven to reasonably capture observed PFOS and PFOA profiles in agricultural fields receiving wastewater effluent irrigation for four decades (Liao et al., 2024). Both PFAS sorption to soil surfaces as well as at the AWI within the vadose zone directly affect their leaching behavior and bioavailability (Post et al., 2017; Lyu et al. 2018) and sorption by both processes increases with increasing chain length. Therefore, short-chain PFAS tend to leach to a greater extent and get taken up by plants more readily than long-chain PFAS. This

leads to the inverse relationship between soil depth and the concentration of long-chain PFAS (Johnson, 2022; Pepper et al., 2023; Sepulvado et al., 2011; Washington et al., 2010).

5.3 Transformation of precursors

Both in the wastewater treatment processes and after land application, PFAS that are not PFAAs can undergo biotransformation to other PFAS. The chemical structure of the precursor, which is informed largely by the PFAS manufacturing process, plays a critical role in the transformation process. For example, n:2 fluorotelomer (FT) phosphate diesters (n:2 diPAPs), n:2 FT alcohols (n:2 FTOHs), FT sulfonates, and FT alkylbetaine degrade to several PFCAs through a series of intermediates (Mejia-Avendaño & Liu, 2015; Dasu et al., 2012, 2013, 2015; Royer et al., 2015; Benskin et al., 2013). Fluorotelomers break down first to yield n:2 FTOHs which then degrade through a hypothesized "one-carbon removal" pathway to yield PFCAs (Ruan et al., 2013; Shaw et al., 2019; N. Wang et al., 2012). Electrochemical fluorination (ECF)-derived sulfonamides such as FASAs will break down through a series of intermediates to the PFSA with the same perfluoroalkyl chain length as the precursor (Mejia-Avendaño & Liu, 2015; Zhang et al., 2016b). Transformation rates are much greater under aerobic conditions compared to almost negligible in typical anaerobic conditions (Zhang et al., 2013, 2016a). Also, FT-derived PFAS degrade much quicker than the ECF-derived sulfonamides (Guelfo et al., 2021). For example, in a recent study at a dedicated biosolids land disposal site on the west coast, most of the FT-PFAS were gone within one year after biosolids application leaving intermediate FTCAs and FT unsaturated acids and terminal PFCAs while changes in the ECF precursors were small (Álvarez-Ruiz et al., 2024). Root exudates and earthworm-excreted enzymes have been reported to increase PFAS degradation (Zabaleta et al., 2018) (Munoz et al., 2020). Enzymatic activities of glutathione S-transferase and cytochrome P450 in the earthworms are proposed to be the likely mechanisms for PFAS transformation. Failure to account for precursor loads in biosolids could lead to underestimating long-term mass release of intermediates and PFAAs in biosolids-amended soils.

5.4 Plant uptake

Plant activity in the vadose zone can moderate the biogeochemical processes that impact chemical fate including PFAS. Through the process of plant-mediated energy transfer (photosynthesis, mineralization, and evapotranspiration), organic molecules can be transported to their root surfaces where they may be transformed or retarded. Molecules that are restricted in the rhizosphere or within the root tissues are prevented from further transport, thus limiting

their leaching potential. In some instances, PFAS are translocated into other plant parts either through passive or active mechanisms. PFAS-specific plant uptake studies have evaluated the role of the root system in PFAS uptake and distribution. The general observation is that PFAS tends to accumulate passively in the root tissues, meaning that uptake is driven by diffusion from areas with high PFAS concentration (Blaine et al., 2014; Stahl et al., 2009; Zhao et al., 2017). The protein and lipid contents of roots can also impact PFAS accumulation in the root. Translocation from roots to shoots has also been observed especially for PFCAs in lab and greenhouse studies (Costello & Lee, 2020) primarily through transpiration-induced transport through the xylem. PFOS from biosolids-amended soils have been shown to be taken up into the leaves of forage crops (Simones et al., 2024). Plant uptake and translocation into its aerial part is inversely proportional to PFAS chain length (Wang et al., 2020; Costello & Lee, 2020). PFAS with shorter perfluorocarbon chain lengths are more mobile and available for plant uptake through transpiration. However, larger PFAS such as PFOS are also found in grasses and other plant parts. In some cases, offloading from the xylem to the phloem is required for PFAS to enter some crop parts such as the grain. Offloading may occur through non-selective substrate transporters – membrane proteins that can move chemically diverse molecules across the cell membrane (Yao et al., 2020; Gill et al., 2021). These transporters include those that transport amino acids, which have a polar group common to PFCAs (a carboxyl group). Detection of intermediate metabolites of PFAS precursors in plant tissues has suggested the possibility of PFAS biotransformation within plant cells. However, further investigation of PFAS metabolism in plants is required to understand transformation mechanisms.

5.5 Vertical transport (leaching)

PFAS transport through the soil profile to groundwater depends on multiple factors including PFAS sorption to soil and at AWIs (See section 3.3.2) as well as hydrological conditions and soil hydraulic properties. Natural cycles of weathering and drainage can release PFAS that were previously sorbed to the soil matrix. Therefore, the intensity of precipitation events such as rainfall and snowmelt as well as irrigation plays a key role in PFAS leaching. Additionally, release and vertical migration of soil colloids during infiltration can enhance PFAS leaching (Borthakur et al., 2021a, 2021b). Critical to PFAS transport are the rate and magnitude of water movement in the soil which is controlled by hydraulic properties such as texture, structure, and pore size distribution. As previously noted, short-chain PFAS (CF_2 < 6) tend to desorb easily from the soil, and thus susceptible to leaching compared to the longer chain homologs (Johnson, 2022; Peter et al., 2024; Sepulvado et al., 2011). This is reflected by the inverse relationship between concentration of long chain PFAS and soil

depth (Álvarez-Ruiz et al., 2024; Peter & Lee, 2024a; Peter, 2025; Peter and Lee, 2025).

The presence of artificial drainage networks can alter the movement of pore water down the soil profile, and thereby PFAS leaching. In the USA where poorly drained soils dominate such as in the Midwest, tile drains are installed in agricultural fields to lower the water table and promote aeration. Agrochemicals and fertilizers percolating through the soil are redirected into tile drains and ultimately surface water, thus short-circuiting direct transport to groundwater used for drinking water (Peter & Lee, 2025). Likewise, PFOS and PFOA were detected in tiles draining from a biosolids-amended field (Gottschall et al., 2017), but not in groundwater.

5.6 Surface runoff

During precipitation events, PFAS can be released from surface soils to become part of overland flow also known as runoff. The intensity, depth, and duration of rainfall typically influence the magnitude of chemical release. Soil properties that regulate leaching, such as micro and macropore distribution, texture, and bulk density also influence surface runoff. Due to their strong association with the solid phase, PFAS are likely subjected to "first flush" removal. This means that they are carried by runoff in the form of suspended solids early in a rainfall event and delivered to surface waters (Borthakur et al., 2021c). Peter and Lee (2024b) reported substantial PFAS in runoff, particularly of PFOS and PFOA at several hundred ppt, from an agricultural field that had not received biosolids for 6 years. Results are consistent with the longevity of long-chain PFAS in the upper soil profile keeping them available for continued runoff with intense storm events.

6 Human and ecological exposure routes and risks from agriculture

6.1 Humans

Although there are several ways that humans may be exposed to PFAS in both the indoor and outdoor environment, the key exposure route is through PFAS-contaminated water given the daily intake volumes of water (De Silva et al., 2021; DeLuca et al., 2022; Sunderland et al., 2019). Given the long half-lives of PFAS within the body, even low ppt levels of intake, especially long-chain PFAS, can lead to bioaccumulation in the body. PFAS may also be consumed through contaminated food sources including commercial stores, home gardens, and recreational fishing. PFAS presence in packaging used for food products can also lead to PFAS transfer into foods (Liu et al., 2023b; Phelps et al., 2024). PFAS are known to bioaccumulate in plants, milk,

meat, and particularly fish and shellfish. In addition, given the use of PFAS in so many products, it is not surprising that we also are exposed to PFAS through inhalation of airborne PFAS within our homes, cars, and workplace and dermally when using various PFAS-containing personal care products (e.g., cosmetics, shampoos, lotions, etc.).

The primary direct pathway linking PFAS exposure to agriculture includes plants and animals that have consumed PFAS-contaminated water or feed (De Silva et al., 2021; Jha et al., 2021). Animal product consumption, and particularly dairy, tends to be the largest exposure pathway of concern linked to agriculture because of PFAS bioaccumulation and biomagnification, PFAS affinity for proteins (high in meat and meal), and that children are the biggest direct consumers of milk. Cattle typically eat 25 pounds or more a day of plant matter; therefore, even low PFAS concentrations in the feed can lead to unacceptable levels in meat and milk (Fitzgerald et al., 2025). Currently, there are no formal maximum concentration levels set for PFAS in milk and meat; however, the State of Maine in the USA set action levels of <211 ppt in milk based on risk to 1–2 y old children (Maine CDC, 2017) and <3.4 ppb in meat based on risks to 1–< 6 y old children (Maine CDC, 2020). PFAS transfer from eggs via feed (Granby et al., 2024) and to laying hens free-ranging on PFAS-contaminated land including in home-produced eggs (Lasters et al., 2022; Göckener et al., 2020b). PFAS have been shown to accumulate in the yolk more than the high-protein albumen, which was attributed to PFAAs having higher binding affinities to low- and high-density lipoprotein and vitellin proteins in yolk compared to ovalbumin albumen proteins (F. Wang et al., 2019). Precursor transfer to eggs as well as in vitro precursor transformation with metabolites transferring to eggs has also been observed (Kolanczyk et al., 2020).

Indirect exposures include potential contamination of rural water supply from runoff of soil into surface waters from lands where PFAS-laden biosolids have been applied to grow crops as well as PFAS from the same source leaching to groundwater. Additionally, surface runoff into water bodies including farm ponds, streams/rivers where fishing, swimming, and other recreational activities occur can expose communities to PFAS intake. Aquatic species in closed water bodies versus streams are particularly vulnerable to bioaccumulation, although PFAS half-lives within aquatic species are relatively short compared to mammals. For example, early investigations of fish in private farm ponds located in biosolids-applied agricultural lands have shown, in some cases, PFOS at several hundred ppb in largemouth bass (Hoskins et al., 2023).

6.2 Ecological

The primary direct pathway linking PFAS exposure to wildlife with agriculture is primarily run-off or tile drain discharge from PFAS-contaminated agricultural

lands into the aquatic system, although terrestrial wildlife can also be exposed through direct plant consumption grown on PFAS-impacted lands (De Silva et al., 2021). Aquatic animals can be exposed through both their respiratory system (gills) as well as dietary exposure. Dermal exposure can also be important for frogs and salamanders given their permeable skin (Abercrombie et al., 2019, 2021). PFAS have been shown to adversely impact the physical development of aquatic organisms, the endocrine and immune systems, and lead to oxidative stress, endocrine disruption, immunotoxicity, and cognitive function impairment, among others (Ankley et al., 2021; Ma et al., 2023; Pandelides et al., 2023). Some studies have indicated that long-term exposure to PFAS can affect population dynamics in organisms by reducing reproductive success or survival rates and altering their general behavior (Ankley et al., 2021; Ma et al., 2023). While long-term ecological risk from PFAS had not been fully assessed due to data limitations, their potential to persist, bioaccumulate, and biomagnify in both terrestrial and aquatic wildlife could disrupt ecosystem functions.

6.3 Effects on soil health

PFAS impacts on soil health are limited, but high concentrations of PFAS (up to 1000 ppb) in laboratory studies have been shown to alter soil properties and microbial function. Xu et al. (2023) found a decrease in water-stable aggregates with PFOS exposure and soil respiration with both PFOS and PFBS exposure. However, Xu et al. (2023) also observed an increase in litter decomposition rates, which in turn increased pH possibly due to changes in microbial community composition. Impacts on soil microbial community composition, structure, enzymes, and gene expression have been observed in some studies with some communities and enzymes but not all, and observations are inconsistent (Qiao et al., 2018; Zhang et al., 2020; Xu et al., 2023). Changes in microbial diversity or function could adversely affect soil health and productivity given the crucial role they play in the biogeochemical cycling of nutrients.

7 Techniques in PFAS identification and tracking

Although scientific studies on PFAS have burgeoned over the past decade, the exact scope of global PFAS contamination is challenging to estimate with certainty. Studies focused on tracing and fingerprinting sources of PFAS are still in their nascent stage and have focused on characterizing sources based on distinct PFAS chemical profiles. For example, chemical fingerprinting with the aid of supervised machine learning was used to differentiate water sources impacted by AFFF from other sources based on the distribution of

PFAS in contaminated sites (Kibbey et al., 2021). Likewise, contaminated sites associated with landfill leachates have been reported to be enriched with FT carboxylic acids (Benskin et al., 2012). Characterizing PFAS chemical profiles depends heavily on advancements in analytical techniques. The number of targeted PFAS compounds in commercial labs has increased over time, from focusing only on PFOS and PFOA to 18 targeted compounds based on US EPA method 537.1, and currently 40 compounds in method 1633 (US EPA, 2024c USEPA, 2020). Research and private labs have developed more comprehensive methods using high-resolution mass spectrometry to quantify over 100 targeted and non-targeted PFAS compounds (Benotti et al., 2020). Furthermore, analysis that identifies PFAS isomer patterns has provided additional diagnostic insights. Branched and linear isomers typically denote ECF and FT manufacturing processes, respectively (Buck et al., 2011). ECF process was discontinued in the early 2000s. Therefore, analytical techniques that show the ratio of branched to linear isomers can provide additional source information with respect to manufacturers and the timeline of PFAS release (Benskin et al., 2010).

Despite the progress with PFAS analytical methods, tracking contamination from comingled sources remains a challenge. For example, the major PFAS sources to agricultural soils, including biosolids and contaminated irrigation water, are aggregates of multiple PFAS inputs, which further complicates forensic analysis. Further, tracking the contribution of such agricultural non-point sources to PFAS presence in connected hydrological systems is challenging for several reasons, including differences in environmental fate of individual compounds and biotransformation (Lee et al., 2010, 2014b; Schaefer et al., 2022; Silva et al., 2022). In such scenarios, source identification and tracking integrity could be enhanced by considering site-specific history and conditions, including surface application timelines and loading rates, soil properties, and hydrogeologic features (Charbonnet et al., 2021). Additionally, multivariate statistical analysis, such as hierarchical clustering, principal component, and multiple linear regression analysis on chemical and site data, can identify patterns of PFAS occurrence and map distinct chemical profiles to PFAS sources within a location of interest (Qi et al., 2016; Salim et al., 2019; Yao et al., 2014; Zhang et al., 2016b).

8 Remediation and mitigation options for PFAS-contaminated farms

Specific to agriculture, remediation strategies are needed for both water sources used for irrigation and livestock rearing, and soil that supports food crops and forage production. Reverse osmosis (RO) and sorbent technologies are the easiest and most acceptable to deploy for removing PFAS from well

water. Unfortunately, both RO and sorbent-based technologies result in PFAS-contaminated residuals (RO rejection water and spent sorbents) which will need to be managed (Tow et al., 2021; Didenko et al., 2024). Therefore, while treatment technologies that lead to PFAS destruction are becoming available, they are still far from being feasible for deployment at many farms.

The biggest challenge in agriculture is PFAS-contaminated soils given the longevity of PFAS in the soils and the scale of the problem. However, the development and testing of pragmatic approaches to minimize impacts of PFAS-contaminated farm soils on crops and livestock are underway to facilitate keeping farms viable. Approaches include the choice of crops grown, selection of crops used for feed, and sorbent addition in the till layer where crops are grown. For example, uptake of PFAS is minimal to negligible in the corn and cob (Bhattacharya et al., 2024) whereas PFAS accumulate in the corn silage (stalk and leaves) as well as grasses (Wang et al., 2020; Costello et al., 2020). Also, the long-chain and highly bioaccumulating PFAS like PFOS do not appear to be offloaded into the soybean (Lazo and Lee, 2024) similar to what has been observed for corn kernels (Bhattacharya et al., 2024). Therefore, shifting livestock feed away from silage and grasses facilitates PFAS reduction in milk and meat. Research is currently being done to optimize rotations and timing based on when the livestock commodity will go into the consumer market to take advantage of expected livestock excretion rates (PFAS half-life in the body). Also, there is some evidence that PFAS uptake into grasses changes throughout the season, thus the potential to use grass feeds when PFAS uptake is much lower. For farms in Maine, the Maine CDC (Simones et al., 2024) found PFOS uptake into grasses twice as high in late summer compared to earlier in the year. Similar trends may not be observed in other regions since PFAS uptake via transpiration will vary with function of temperature, moisture, and soil conditions. Intercropping has been proposed as a potential management strategy to alter PFAS uptake into a target crop, e.g., corn leaves and stalks used for feed, by interrow plant cover, but positive outcomes were not observed in greenhouse studies (Scearce et al., 2024) and potential reductions in yield due to nitrogen dynamics and competition for water and soil oxygen are likely (Qiu et al., 2025; Deines et al., 2022). Field studies to further examine interrow plant covers are underway (U.S.EPA, 2024d). However, such plant cover can also reduce runoff into surface water bodies used for livestock drinking or fishing as we have observed for other chemicals. Another potential PFAS management approach being investigated addresses enhancing PFAS excretion from livestock already impacted by PFAS in the feed or reducing PFAS uptake from PFAS-contaminated feed by using PFAS binders in the water or feed (U.S.EPA, 2024d).

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8.1 Sorbent technologies

Sorbent mitigation at the field scale has much promise to reduce PFAS availability to porewater, thus reducing leaching and plant uptake (Mahinroosta et al., 2020; Khair Biek et al., 2024). Both waste treatment residuals (WTRs) and biochars have been used as soil amendments in agricultural settings to improve soil health, increase crop yields, regulate the loss of applied nitrogen, and control phosphorus mobilization for decades across the globe (Bayley et al., 2008; Ibrahim et al., 2023; Li et al., 2024; Ippolito et al., 2009; 2011; Turner et al., 2019; Liu et al., 2013). Over the past decade, the potential for sorbents to alter PFAS availability has been explored at the lab-scale in spiked water, soil, and soil fractions, or from PFAS-containing biosolids (Sørmo et al., 2021; 2024; Openiyi et al., 2024; Holly et al., 2024). Generally, sorbent materials with a high specific surface area, small pore size distribution, and high exchange capacity have higher sorptive capacity (Lu et al., 2016) including for PFAS (Du et al., 2014). Solution factors observed at the lab-scale to affect sorbent potential for PFAS include soil-solution pH, ionic composition, and dissolved OM (Dasu et al., 2013; Zhang et al., 2021) Limited desorption from WTRs and biochars has also been demonstrated at the lab-scale (Openiyi et al., 2024; Gravesen et al., 2023). While lab-scale studies have shown the effectiveness of various sorbent materials to sorb PFAS and reduce PFAS leaching, applying these immobilization approaches to reduce both crop uptake and leaching of PFAS in agricultural fields and associated challenges are not yet well understood. Therefore, pilot-scale and field-scale data are needed to better understand the potential and limitations. Greenhouse studies using locally sourced sorbents mixed into PFAS-contaminated soil (e.g., 400 ppb PFOS) show great promise in reducing PFAS uptake by grasses, in which even 1.5 wt% high carbon wood ash reduced PFOS uptake by 80% and 6 wt% to >95% (Openiyi and Lee, 2024; Openiyi et al., 2025; Illango et al., 2024). Mitigating PFAS uptake into plants by sorbents will be dependent on PFAS type, soil properties, and crop species, as well as sorbent amendment type and rates (Ilaango et al., 2024). In addition, weather patterns may greatly alter expected mitigation in some cases. Research on effective strategies mitigate PFAS in farming systems is ongoing.

8.2 Phytoremediation

Exploiting plant uptake (i.e., phytoextraction) has been considered for phytoremediation of soils (Kavusi et al., 2023) but has several limitations. While low-cost and perhaps visually appealing, phytoextraction will be limited in terms of magnitude and number of PFAS as well as depth of soil explored. Also, prior to harvest, there is a high potential of posing a risk to wildlife, and then the

harvested plants will require subsequent disposal or destruction. In field trials with hemp at a former PFAS-contaminated Air Force Base, hemp was only able to extract 6 of 28 PFAS, mostly short-chain and PFCAs, and only at a maximum of 2% was removed (Nason et al., 2024). Harvested plants were subsequently treated using hydrothermal liquefaction, which reduced PFAS mass, but not all PFAS were removed. Huff et al. (2020) explored phytoremediation potential of eight herbaceous and seven woody plant species in a greenhouse study. They reported almost 3.8 mass % of PFOS and up to 43 mass % were taken up by *Festuca rubra* (red fescue) and *Schedonorus arundinaceus* (tall fescue), respectively. However, plant exposure to PFAS was done through weekly irrigation of PFAS-amended water, thus a vastly different scenario than trying to phytoextract PFAS adsorbed to soils that have been contaminated for a long time. In a greenhouse study , Illango et al. (2024) investigated the use of a grass-legume mix in combination with low doses of biochars (≤1%) to phytoextract PFAS from PFAS-contaminated soils amended with biosolids and additional PFAS. Phytoextraction varied with soil type and biochar type and rates, negatively correlated to PFAS chain length, and percent removal was higher for the PFCAs. Phytoextraction of PFOS, one of the PFAS of greatest concern due to its bioaccumulation and biomagnification potential, was less than 1%. It is plausible that genetic engineering techniques may aid in increasing the magnitude and number of PFAS that can be phytoextracted as well as the depth of soil that is explored by the roots (Mayakaduwage et al., 2022). Advancing endophyte-assisted phytoremediation for reducing PFAS concentrations has also been posed. Endophytes, such as bacteria and fungi, excrete enzymes that may promote organic chemical degradation (He et al., 2020); however, their value for degrading PFAS has not been explored (Mayakaduwage et al., 2022).

8.3 Bioremediation technology potential

Destruction of PFAS through biotransformation processes is highly desired due to being ecofriendly, likely cost-effective, and with high potential to be used in situ at a large scale like the agricultural landscape. PFAS degradation by microbes can occur through several reaction steps including reductive defluorination, decarboxylation, hydroxylation, and hydrogenation (Kumar et al., 2023); however, in most cases, fluorine removal has been limited. Multiple isolated bacterial species including *Pseudomonas*, *Acidimicrobium*, *Mycobacterium*, *Gordonia,* as well fungal species such as *Phanerochaete chrysosporium, Gloeophyllum trabeum,* and *Phanerochaete chrysosporium* have demonstrated the potential of transforming PFAS precursors, but only to terminal PFCAs (Kim et al., 2012; Merino et al., 2018; Shaw et al., 2019; Tseng et al., 2014; Van Hamme et al., 2013). Huang and Jaffé (2019) reported PFOS removal up to 33% with a pure *Acidimicrobium sp*. strain A6 and up

to 50% when used in a mixed culture (Ralstonia, Bacillus, Acidimicrobium, Aciditerrimonas, and Desulfosporosinus) in 100-d incubations in which Fe (III) and $NH4^+$ were supplied as the electron acceptor and donor, respectively. Similar studies were done for PFOA but included microcosms with biosolids (Huang et al. 2022). PFCAs were generated in some cases as well as fluoride, but the high concentrations and likely impurities in the PFAS material used, as well as fluorine mole balance, introduced uncertainties into the study outcomes (Liu et al., 2023a). Other researchers have isolated microorganisms from contaminated media that used PFOS and/or PFOA as sole carbon sources when amended at high ppm levels. For example, Beskoski et al. (2018) found chemoorganoheterotrophic bacteria and a mix of yeast and molds isolated from PFAS-contaminated sediments to remove up to 69% and 36% PFOS and PFOA, respectively, in the 30-d aerobic microcosms. However, through target and nontarget screening, they only found single F-containing fatty acids. Chetverikov et al. (2017) using *Pseudomonas plecoglossicida* 2.4-D isolated from soil near a petroleum plant, observed PFOS degradation to perfluoroheptanoic acid coupled to fluoride generation. Other researchers also isolated strains from contaminated media and claimed degradation, but the weight of evidence was lacking.

While PFAS do undergo biotransformation including defluorination at some level, degradation mechanisms and the functional genes responsible for PFAS transformation are currently not well understood (Kumar et al., 2023; Shahsavari et al., 2021). Future efforts in advancing PFAS bioremediation technology potential should do what is needed to provide fluorine mole balance and exploit technologies in microbial ecology (metagenomics, transcriptomics, and metabolomics). The latter coupled to advances in biotechnology may expedite the development of PFAS bioremediation technologies that can be scaled up to address PFAS-contaminated soils and contaminated groundwater in situ.

PFAS remediation technologies typically require significant investment during and post implementation. Depending on the level of technological advancement, the cost of remediating contaminated soils can range from $20 000 to $90 000 per hectare, with equipment costs accounting for 40%, labor 30%, energy 20%, and other factors 10% (Valalamontes & Adamopoulos, 2025). Advanced technologies such as plasma-based and thermal desorption treatment are more expensive compared to bioaugmentation and phytoremediation. Other options such as ex-situ soil washing and landfilling fall under medium cost at the rate of $100 to $200/ton according to (Quinnan et al., 2022). Unlike reports of combined treatment measures for aqueous matrices (Lu et al., 2020), there is currently no field-scale demonstration for contaminated soils. Integrated strategies can reduce treatment expenses by combining low-cost approaches such as the implementation of in-situ sorbent technologies combined with phytomanagement. Still, the cost of the least expensive

remediation option may be unfeasible for small-scale farms as treatment expenditures can exceed farm revenues. Nonetheless, the cost of inaction might surpass remediation costs in the long run. Contaminated farmlands can incur temporary or permanent losses from discarding impacted farm products or shutting down production. For example, farmers in several states in the USA had to dispose of milk and meat products and, in some instances, abandon the farm following incidences of high PFAS contamination (Cordner et al., 2021). Left untreated, non-productive farmlands may ultimately weaken food production and arguably pose obstacles to food security and local economies. Additionally, agricultural communities can incur an additional burden of health-related costs from exposure to PFAS, given that contaminated farmlands can become diffuse sources of PFAS to drinking water. Already, PFAS-related health costs for the United States are estimated at $37–$59 billion annually (Cordner et al., 2021), an amount that will likely increase if PFAS contamination is not contained. Other indirect costs derived from effects on the well-being of farmers and impacted communities, such as stress, depression, and low standards of living, are challenging to estimate but likely account for a sizable burden from PFAS contamination. Therefore, remediation of contaminated farmlands should be viewed as a societal issue, one that requires significant financial investment from the government and industry and cross-sector partnership between relevant stakeholders. Post remediation, long-term monitoring is required to assess the effectiveness of remediation efforts, especially for in-situ sorbent technologies. The availability of real-time sensors in the future can improve monitoring efficiency as readily available data can be coupled to predictive models to identify spatiotemporal trends of remediation outcomes.

9 Conclusion

PFAS contamination in agricultural systems is one of the biggest environmental issues of the twenty-first[t] century given the growing number of farmland contaminations and background levels in surface soil contamination and on a global scale. Sources of PFAS to the environment are diverse, but the primary sources in agricultural systems are the land application of biosolids and composted material and irrigation water contaminated by PFAS, typically from industrial, military, airport, or landfill sources. The properties of this family of several thousand compounds led to their use in multiple products and processes as well as their environmental presence, persistence, and complex retention and transport mechanisms.

Addressing PFAS contamination in agricultural systems is complex with multiple challenges. First, despite active research and ongoing monitoring efforts that have identified several contaminated farms, potential occurrence at most has not been assessed, especially in less-developed countries. There

is a need for rapid quantitative methods for PFAS that have been validated and standardized and can be done at a lower cost than current analysis. Such methods will expedite needed data collection for farms and communities to minimize risks to the food system and human health.

Product and process source controls are needed, including regulating out nonessential PFAS use in industrial processes and consumer goods, including use in food packaging. Such controls can minimize direct human exposure and PFAS presence in biosolids that can be beneficially reused and effluent-treated irrigation water that is often used for irrigation, especially in water-scarce regions. In addition, pre-treatment for PFAS capture or destruction in influent streams with high concentrations of legacy PFAS, such as those typical in many landfill leachates, is required. Actions to advance source control are occurring in some regions, but still quite limited.

While source control will eventually benefit the agricultural community as well as all systems impacted by PFAS, the PFAS legacy in the soils will remain a challenge for several decades. Currently, treatment of a PFAS-contaminated water source used for irrigation or livestock drinking water can be treated with capture technologies, but the soil associated with PFAS-contaminated landscapes with depth brings forth a particularly difficult challenge. While some PFAS do get taken up into vegetables and plants consumed directly by humans, the bigger concern is uptake into feed consumed by livestock, which then enters the human food web. PFAS's high affinity for proteins leads to high bioaccumulation in meat, milk, and eggs in livestock that typically consume a high ration of feedstock, particularly in cattle. Current mitigation approaches being used include selection of crop types and species that have lower PFAS uptake, the part of the crop used for feed, and when crops are harvested (times of high transpiration will lead to higher PFAS uptake). Other approaches being explored include the use of high carbon materials to reduce PFAS availability to the crop and leaching as well as seasonal rotation of feed and feed binders. For free grazing cattle, providing clean water for drinking and minimizing access to PFAS-contaminated streams can be implemented. Given the diverse nature of PFAS chemicals and their environmental behavior, the likelihood of a one-size-fits-all solution is not feasible, rather addressing contamination will require a combination of approaches. Mitigation strategies are costly, so funds to help farms stay in business while protecting the food system and developing PFAS thresholds and action limits are needed at the local, state, and federal levels. The latter requires more refined risk assessment and management plans and data to support the assessments, which are all lagging. PFAS exposure almost always occurs in mixtures, but quantification and understanding of chemical mixture toxicity remain elusive, and scaling from lab models to humans remains a challenge, although the rapid advancements in omics and other analytical

tools coupled with machine learning have promise to accelerate the gaps (Kleinstreuer & Hartung, 2024; Limbu et al., 2024; Cheng et al., 2024).

Furthermore, while progress to quantify and model the fate and transport of PFAS in agricultural systems and the environment has increased substantially in the past decade, there are still gaps in understanding. For example, the long-term impact of PFAS contamination on ecosystem health and function has not been adequately assessed. Filling these gaps will facilitate solutions for agricultural systems as well as aid in risk assessment, management, and regulatory guidance. Likewise, the development and integration of existing data networks and mapping tools are needed to aid in evaluating current and future areas of vulnerability and risk. The PFAS story is complex from occurrence to effects; therefore, transparent and targeted communication tools and educational programs for different and diverse stakeholders are needed to generate widespread awareness and public involvement. Communication and education need to surmise what is known about PFAS in layperson terms, address the growing uncertainty with regard to PFAS and food safety, reassure consumers, farm communities, and producers that PFAS in agricultural systems is being addressed, and generate assurance of alternative PFAS-safe paths.

10 Where to look for further information

Further reading: ITRC (Interstate Technology and Regulatory Council) offers free up to date information on PFAS with focused fact sheets as well as a full document capturing PFAS characteristics, occurrence, toxicity, fate, remediation, regulations, risk assessment, sampling and analysis, treatment technologies, stakeholders perspectives, and risk communication. https://pfas-1.itrcweb.org/-.

Perfluoroalkyl Substances in the Environment: Theory, Practice, and Innovation (2018). Editors: Kempisty, Racz, and Xing explores the challenges across the topical areas of regulation and management, toxicology, environmental remediation, and analytical sampling and analysis.

Per- and Polyfluorinated Alkyl Substances: Occurrence, Toxicity and Remediation of PFAS (2025). Editors: Naidu, Mallavarapu, Liu and Umeh captures PFAS dynamics in the terrestrial environment including many challenges relating to analytical science, environmental fate, toxicity, regulatory considerations, risk and remediation.

Key journals/conferences: Numerous journals include PFAS environmental fate and remediation articles including but not limited to *Environmental Science and Technology*, *Science of the Total Environment*, *Journal of Environmental Quality*, *Journal of Agriculture and Food Chemistry*, *Environment International*, *Environmental Toxicology and Chemistry*, *Chemosphere*, *Journal of Hazardous*

Materials, Water Research - American Society of Agronomy, Soil Science Society of America, and Crop Sciences of America (ASA/SSSA/CSA) annual meetings in the United States and Internationally. Society of Environmental Toxicology and Chemistry has annual meetings in multiple regions/continents around the world – American Chemical Society has annual meetings in North America – International Conference on Environmental Pollution and Remediation – PFAS International Congress: Management of Environmental & Health Risks, not annual but reoccurring, one planned for 2025 and again in 2027. National PFAS Conference, held even years and typically in North America, www.nationalpfasconference.org

Major research funding initiatives: SERDP/ESTCP – Strategic Environmental Research and Development Program (SERDP) Environmental Security Technology Certification Program (ESTCP) – funds numerous multi-year projects annually focused on environmental fate, detection, and remediation of PFAS. Project abstracts, reports, and webinars can all be found at https://serdp-estcp.mil/. The United States EPA funded ten new projects in 2024 focused on varied aspects of PFAS in agriculture, https://www.epa.gov/research-grants/research-grants-understanding-pfas-uptake-and-bioaccumulation-plants-and-animals

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