

Review

Comprehensive strategies for the remediation of per- and polyfluoroalkyl substances (PFAS): Mechanisms, technologies, and future perspectives

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) are a large class of synthetic compounds extensively used in industrial and consumer products owing to their exceptional thermal stability, hydrophobicity, and resistance to chemical degradation. However, these same physicochemical properties that make PFAS valuable in manufacturing have resulted in their persistent accumulation across environmental matrices, including soil, sediments, water bodies, and biota. Elevated PFAS concentrations are commonly detected at aqueous film-forming foam application sites, fluoropolymer manufacturing facilities, landfills, and biosolid-amended agricultural lands. The removal of PFAS through conventional treatment methods is further constrained by the generation of toxic byproducts, degradation of treatment materials, and high operational costs. The pervasive presence of PFAS adversely affects human health, disrupts soil microbial diversity, impairs plant growth, and damages benthic ecosystems, while posing risks of bioaccumulation and trophic transfer. Detection and remediation of PFAS remain particularly challenging due to their chemical stability, complex mixtures, and typically low environmental concentrations. Moreover, microbial degradation of PFAS remains limited, although certain enzymes (like oxidoreductases, hydrolases) have shown potential to transform several PFAS and their precursors. Regulatory control is further complicated by the vast diversity of PFAS compounds, the scarcity of comprehensive toxicity data, and the lack of standardized guidelines for managing their environmental release and

Abbreviations: AFFF, Aqueous film-forming foam; BMFs, Biomagnification factors; DFA, Difluoroacetate; DWI, Drinking Water Inspectorate; ECF, Electrochemical Fluorination; EPA, Environmental Protection Agency; FTOH, Fluorotelomer Alcohol; FTCA, Fluorotelomer carboxylic acids; FTMAC, Fluorotelomer Methacrylate; FTSA, fluorotelomer sulfonic acids; GAC, Granular activated carbon; HFPO-DA, Hexafluoropropylene Oxide Dimer Acid; LC-MS/MS, Liquid Chromatography – Tandem Mass Spectrometry; MFA, Monofluoroacetate; N-MeFOSA, N-Methyl Perfluorooctane Sulfonamide; OECD, Organisation for Economic Co-operation and Development; PFPrA, Pentafluoropropionic Acid; PFAS, Per- and Polyfluoroalkyl Substances; PFAAs, Perfluoroalkyl Acids; PFBS, Perfluorobutane Sulfonate; PFBA, Perfluorobutanoic Acid; PFDS, Perfluorodecane Sulfonate; PFDA, Perfluorodecanoic Acid; PFHpA, Perfluoroheptanoic Acid; PFHxS, Perfluorohexane Sulfonate; PFHxA, Perfluorohexanoic Acid; PFNA, Perfluorononanoic Acid; FOSA, Perfluorooctane Sulfonamide; PFOS, Perfluorooctane Sulfonate; PFOA, Perfluorooctanoic Acid; PFPeA, Perfluoropentanoic Acid; PFUnDA, Perfluoroundecanoic Acid; PTFE, Polytetrafluoroethylene; TFA, Trifluoroacetate; WWTPs, Wastewater treatment plants; 6:2 FTS, 6:2 Fluorotelomer Sulfonate; 8:2 FTS, 8:2 Fluorotelomer Sulfonate.

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human exposure. Therefore, this review consolidates current knowledge on the environmental distribution, ecological impacts, and challenges associated with the PFAS remediation.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are globally known as a major class of organic pollutants due to their widespread occurrence across environmental domains, which leads to adverse health effects. The strong environmental persistence of PFAS is attributed to their robust carbon-fluorine (C-F) bonds, which provide significant thermal and chemical stability, making them resistant to natural breakdown processes. Consequently, PFAS can bioaccumulate within ecosystems and biomagnify through food chains, eventually toxifying both wildlife and human populations. The major toxicological effects include endocrine disruption, immunotoxicity, reproductive and developmental toxicity, and cancer-causing effects (Ehrlich et al., 2023).

Conventional treatment methods frequently used in wastewater treatment plants (WWTPs) and landfills, such as adsorption, sedimentation, and filtration, exhibit reduced efficacy in removing PFAS (Ahrens and Bundschuh, 2014; Cousins et al., 2020). In addition, the granular activated carbon (GAC), ion exchange resins, oxidation processes, and membrane technologies such as reverse osmosis and nanofiltration for instance are considered advanced treatment techniques that can remove some PFAS from water, but have serious drawbacks, including membrane fouling, regeneration issues, and inadequate efficacy against short-chain PFAS (Cousins et al., 2020). Owing to these scenarios there is strong need to explore alternative and sustainable approaches for treating PFAS. Many scientists across the globe are exploring different bioremediation approaches using microbial systems to treat PFAS. These systems offer multiple advantages, such as cost-effectiveness, environmental compatibility, and adaptability to a wider range of operational conditions (Harris et al., 2025). Different microbial enzymes involved in lignin and xenobiotic degradation such as laccases, peroxidases, and cutinases, are being studied for their potential to break down PFAS molecules (Harris et al., 2025; Shelke et al., 2023). Another approach involves the synthesis of green nanomaterials such as biochar-supported nanoparticles functionalized carbon composites, and metal-oxide catalysts (e.g., TiO_2 , ZnO , Fe_3O_4) that have shown promising performance for PFAS adsorption and degradation while reducing the secondary pollution (Nasrollahpour et al., 2025; Mousavi-Kamazani et al., 2025; Salehi et al., 2016; Restivo et al., 2024). Nevertheless, there is no proven evidence yet of complete or even partial degradation of PFAS using these systems.

Another challenge in PFAS contamination lies in the technical accuracy of detecting and characterizing these compounds despite growing regulatory and scientific attention. Considering the importance to health and safety, the United Kingdom's Drinking Water Inspectorate (DWI) recommends that the drinking water should contain less than 100 ng L^{-1} as the combined concentration of 48 PFAS (Drinking Water Inspectorate (DWI), 2025), while the European Union's Drinking Water Directive (DWD) sets a limit of less than 500 ng L^{-1} for total PFAS and less than 100 ng L^{-1} for a specific group of 20 compounds (Directive (EU), 2020). To meet these regulatory norms, advanced instrumentation and precise quality control procedures are required, but many laboratories still face challenges related to method validation, background contamination, and matrix interferences (Teymoorian et al., 2023; Han et al., 2021). Overcoming these challenges demands improved analytical approaches and broader monitoring strategies to support reliable risk assessments and regulatory enforcement.

Although extensive research has been conducted on PFAS chemistry, occurrence, and treatment, the comprehensive integration of emerging biotechnological innovations, degradation mechanisms, and regulatory perspectives remains limited. Existing literature frequently addresses these aspects in isolation, often focusing primarily on analytical

detection or remediation techniques without connecting them to practical applications. Therefore, comparative assessments of existing and emerging remediation strategies are essential to evaluate their scalability, environmental compatibility, and long-term viability. The present review contributes to addressing these gaps by (i) providing a comprehensive overview of on PFAS occurrence, distribution, and ecological consequences, (ii) critically analyzing the strengths and limitations of existing and emerging remediation techniques, and (iii) highlighting future research directions for developing sustainable PFAS management strategies.

In this review, we broadly summarize the primary sources of PFAS contamination, focusing on the contributions of wastewater treatment facilities and landfills. Further, we address the limitations of traditional and state-of-the-art treatment technologies in efficiently eliminating PFAS from polluted waters, highlighting the challenges posed by their chemical stability and structural diversity. Next, we explore novel biological treatment approaches, using microbial enzyme systems, as promising alternatives for sustainable degradation of PFAS, are explored. In addition, the analytical hurdles in detecting and quantifying PFAS are discussed. Finally, we identify existing knowledge gaps and suggest directions for future research to advance PFAS management, monitoring, and remediation.

2. Development and sources of PFAS

PFAS, first discovered in the 1940s, were developed to address increasing industrial and technological needs for materials with outstanding durability, chemical inertness, and heat resistance (Dewapriya et al., 2023). At the time of their development, the unique physicochemical characteristics, particularly the stability of the C-F bond, have given this group of chemicals an exceptional resistance to degradation, which was considered an asset for producing materials extremely resistant to heat, water, oils, and the majority of solvents (Organisation for Economic Co-operation and Development (OECD), 2018). PFAS became commercially known with the invention of polytetrafluoroethylene (PTFE), also known as Teflon, which found extensive application in nonstick cooking utensils and corrosive-resistant products (Glüge et al., 2020). Subsequently, industries sought materials for use in the aerospace, electronics, firefighting, textiles, and food packaging sectors (Buck et al., 2011) (Supplementary Table S1). PFAS had no equal as a surfactant, lubricant, insulator, and coating material with their low surface energy, thermal stability, and low surface tension capacity even at low concentrations. However, without fully understanding PFASs environmental persistence and bio-accumulative potential, their widespread adoption has now emerged as a major environmental issue and public health concern (Organisation for Economic Co-operation and Development (OECD), 2018; Buck et al., 2011; Wang et al., 2017) (Fig. 1).

2.1. Primary manufacturing of PFAS and industrial release mechanisms

Industrial facilities involved in the production or processing of PFAS, or those that utilize PFAS chemicals in their manufacturing operations, constitute major point sources of environmental release (Supplementary Table S1). PFAS manufacturers synthesize these compounds through processes such as electrochemical fluorination (ECF) and other re-olomerization processes (Dhore and Murthy, 2021; Gaines, 2023). Additionally, PFAS formulators blend PFAS feedstocks with other materials to create various commercial or intermediate products (Dhore and Murthy, 2021).

Releases from industrial facilities occur through multiple pathways,

including wastewater and stormwater discharges, on- and off-site disposal of solid wastes, accidental releases (such as leaks and spills), and stack and/or fugitive emissions (Cao et al., 2022; Allinson et al., 2019). Stack emissions can further lead to the aerial deposition of PFAS onto soil and surface waters, with subsequent leaching into groundwater. Industrial sites also contribute to PFAS release through fire training exercises or emergency response incidents where aqueous film-forming foam (AFFF) has been used (Jahura et al., 2024). The diverse mechanisms of release from primary manufacturing and processing facilities mean that industrial activity generates a broad environmental footprint of PFAS contamination, impacting water, soil, and air in surrounding areas and potentially extending much further through atmospheric transport (Fig. 1). The ability of some PFAS to undergo long-range environmental transport implies that industrial releases can have global ramifications, leading to the deposition of these chemical pollutants in even remote environments far from their point of origin.

2.2. Consumer products as widespread sources of PFAS

PFAS have been incorporated into a vast array of consumer products since the 1950s, primarily due to their desirable heat, grease, and water-resistant properties (Dewapriya et al., 2023; Organisation for Economic Co-operation and Development (OECD), 2021) (Supplementary Table S1). While historical compounds like perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) have been phased out of commercial products in some regions, other PFAS are now widely used as replacements (Wee and Aris, 2023). In consumer products, PFAS are frequently employed as surface-active agents and barrier materials. For instance, they are used in PTFE-coated nonstick cookware (e.g., Teflon) to prevent food adhesion and enhance heat resistance (Dewapriya et al., 2023). In the textile industry, PFAS-based finishes confer hydrophobic and oleophobic properties to fabrics used in rainwear, outdoor clothing, and umbrellas (Glüge et al., 2020). Similarly, stain-resistant treatments for carpets, upholstery, and other household textiles rely on PFAS to repel aqueous and oily substances (Supplementary Table S1) (Glüge

et al., 2020).

In the food industry, PFAS are incorporated into grease-resistant packaging materials, including fast food wrappers, microwave popcorn bags, pizza boxes, and candy wrappers, to prevent oil and moisture penetration (Supplementary Table S1) (Torres and De-la-Torre, 2023). In personal care formulations, PFAS function as emulsifiers, film formers, or conditioning agents in products such as shampoos, dental floss, nail polishes, and cosmetics (e.g., mascara, and foundation) (Torres and De-la-Torre, 2023). Firefighting foams, specifically AFFF, represent a significant source of PFAS contamination, particularly in military, aviation, and industrial fire suppression systems (Jahura et al., 2024). Some consumer-grade fire extinguishers may also contain fluorinated surfactants (Jahura et al., 2024). Additionally, PFAS are incorporated into specialized applications, including refrigerants, artificial turf, and construction materials such as insulation, wallboard, and ceiling tiles (Gaines, 2023). The widespread and persistent use of PFAS in these applications has resulted in their ubiquitous presence in the environment and had raised concerns regarding bioaccumulation, long-range transport, and potential adverse human health effects (Fig. 2).

2.3. Waste management facilities and other diffuse sources

Waste management facilities, particularly landfills, represent significant point sources of PFAS release into the environment. PFAS-containing products disposed of in landfills can leach chemicals like PFOA, PFOS, perfluorohexanesulfonic acid (PFHxS), and emerging compounds like GenX and fluorotelomer acids into the leachate, which can subsequently contaminate surrounding groundwater, surface water, and soil (Wei et al., 2019). Furthermore, wastewater treatment residuals and areas where biosolids are produced and applied also contribute to PFAS dissemination, with industrial wastewater discharges posing particularly significant environmental impacts (Buck et al., 2011; Wei et al., 2019). Firefighting foams (AFFF) that are stored, used, or released at various sites, including industrial facilities and fire training areas,

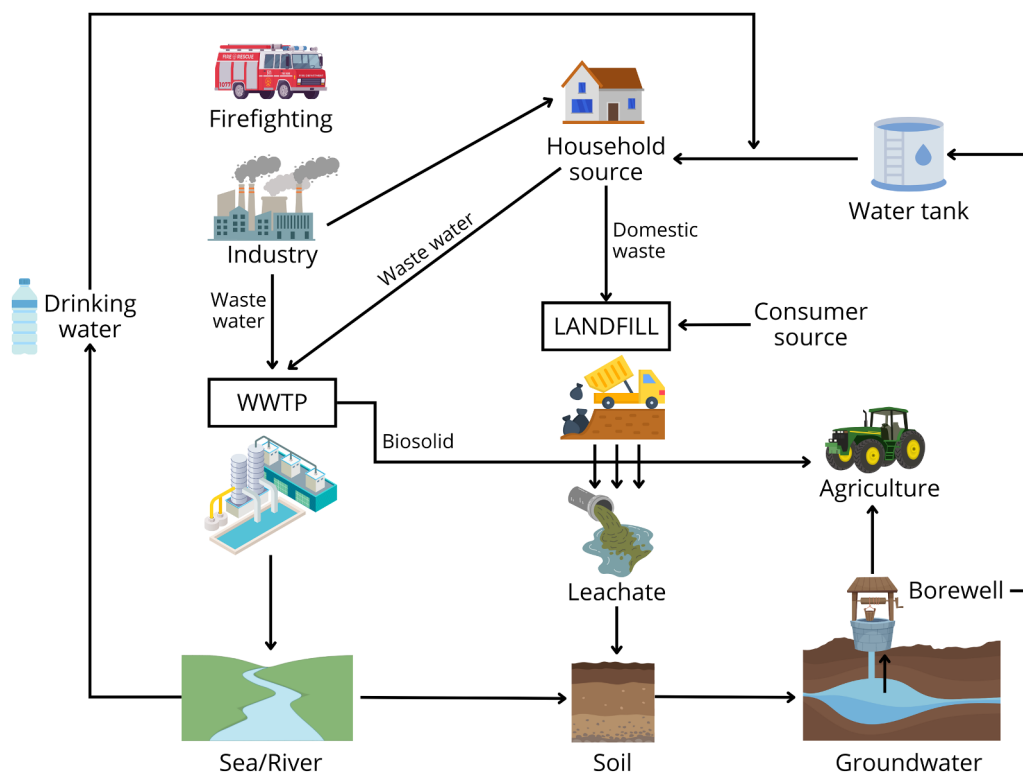


Fig. 1. Conceptual diagram depicting of PFAS contamination in the environment.

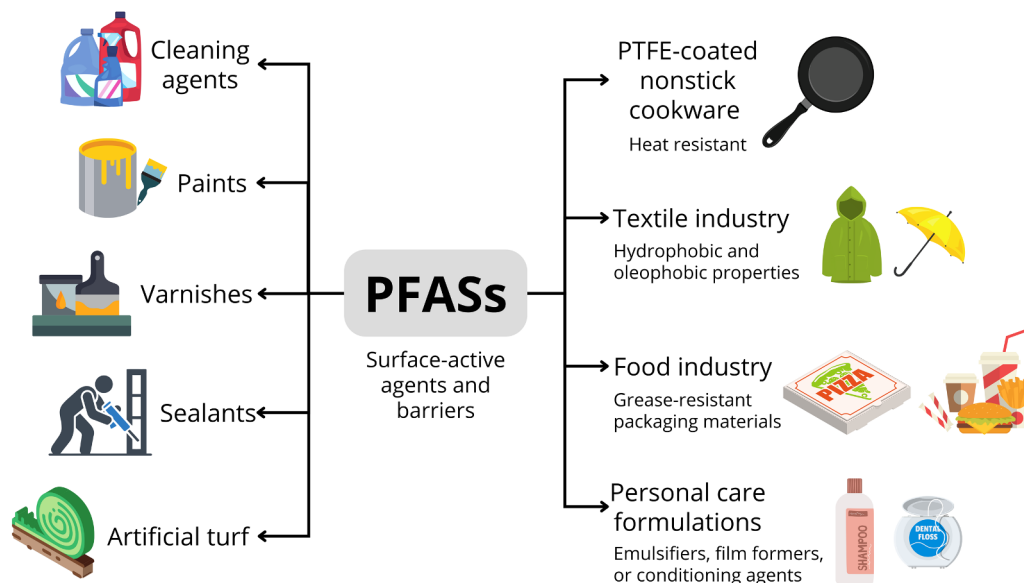


Fig. 2. PFAS in consumer products: Ubiquitous sources of exposure.

constitute another major source of environmental contamination (Jahura et al., 2024). The continued release of PFAS from "end-of-life" scenarios, such as landfills and wastewater treatment plants, indicates that even if primary manufacturing were to cease, the legacy of PFAS embedded in consumer products and industrial wastes would continue to serve as a source of environmental contamination.

3. Chemical classification of PFAS

PFAS represent a large and diverse group of synthetic chemicals, with over 4700 distinct compounds identified (United States Environmental Protection Agency (US EPA), 2025). PFAS can be broadly classified into two categories: polymer PFAS (fluoropolymers) and nonpolymer PFAS, which include both open-chain and cyclic molecules (Fig. 3) (Harris et al., 2025; Wang et al., 2017; Itumoh et al., 2024). Polymeric PFAS possess longer carbon backbones with multiple repeat

units (>10) and include three subclasses: fluoropolymers (e.g. PTFE, PVDF), side-chain fluorinated polymers (fluorinated polyurethane), and perfluoropolyethers (PFPE) (Wang et al., 2017; Itumoh et al., 2024). Nonpolymeric PFAS, on the other hand, contain carbon backbones with 2–10 fluorocarbon repeat units and incorporate polyfluoroalkyl substances (fluorotelomers), which contain at least one non-fluorinated carbon atom, and perfluoroalkyl substances, where fluorine substitutes all hydrogen atoms on the carbon chain (Williams et al., 2022; Ye et al., 2025) (Table 1). Fluorotelomers comprise of fluorotelomer alcohols (FTOHs), fluorotelomer carboxylic acids (FTCAs), and fluorotelomer sulfonic acids (FTSAs) (Fig. 3) (Harris et al., 2025). The broader class of perfluoroalkyl PFAS includes fluorinated compounds with different hydrophilic (acid) head groups attached to the fluorocarbon chains (PFAAs), which are commonly carboxylic acids (PFCAs and PFECAs), sulfonic acids (PFSAs), phosphonic (PFPA) or phosphinic (PFPIA) acids (Fig. 3) (Harris et al., 2025; Itumoh et al., 2024). These PFAS also

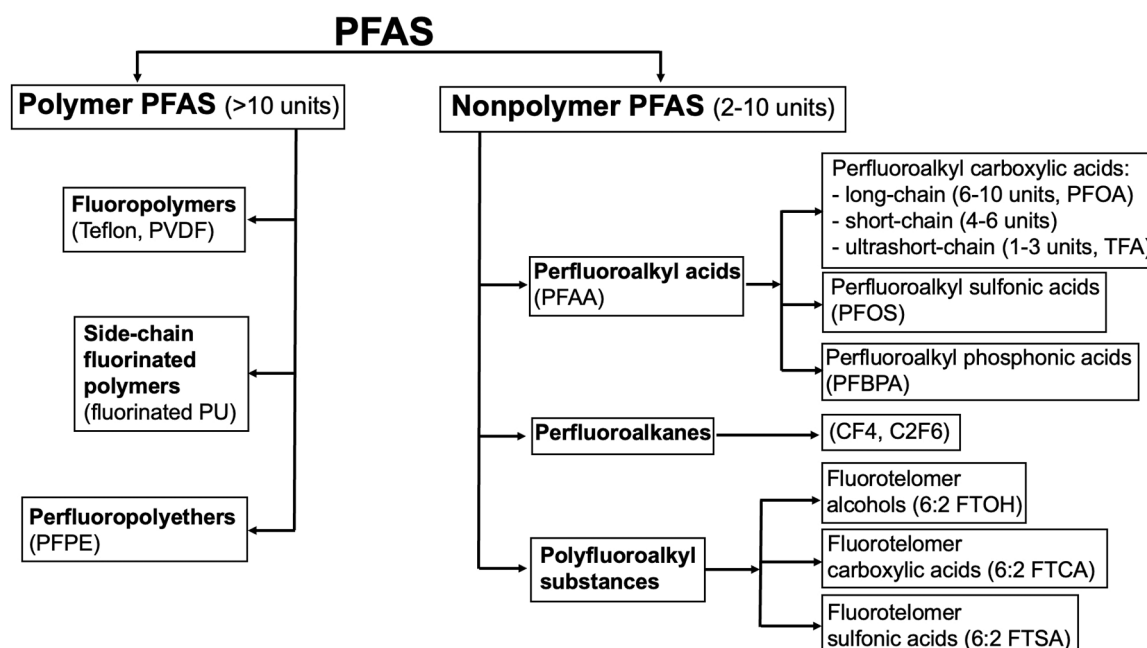


Fig. 3. Classification of per- and polyfluoroalkyl substances (PFAS).

Table 1

Classification of Per- and Polyfluoroalkyl Substances (PFAS).

Category	Subcategory	Description	Examples	Typical Uses/Applications	References
1. Structure-Based	Perfluoroalkyl Substances	Fully fluorinated carbon chains; high persistence	PFOA, PFOS, PFHxA	Non-stick cookware, fire-fighting foams (AFFF), stain-resistant fabrics	(Buck et al., 2011; Organisation for Economic Co-operation and Development (OECD), 2021)
	Polyfluoroalkyl Substances	Partially fluorinated; act as precursors	6:2 FTOH, PAPs	Paper coatings, food packaging, waterproof textiles	(Organisation for Economic Co-operation and Development (OECD), 2018; Wang et al., 2017; Organisation for Economic Co-operation and Development (OECD), 2023; Organisation for Economic Co-operation and Development (OECD), 2022)
2. Functional Group-Based	Carboxylic Acids (PFCAs)	Contain –COOH group; degradation products	PFOA, PFNA, PFHxA	Processing aids in fluoropolymer production	(Buck et al., 2011)
	Sulfonic Acids (PFSAs)	Contain –SO ₃ H group; persistent and bioaccumulative	PFOS, PFHxS, PFBS	Electroplating, firefighting foams, surface treatments	(Organisation for Economic Co-operation and Development (OECD), 2021; Giesy and Kannan, 2002)
	Sulfates	Sulfate esters; precursors to sulfonates	MeFOSAA	Surfactants, textile coatings	(Ullah et al., 2014)
	Amides	Amide-functionalized PFAS; precursors	FOSA, N-MeFOSA	Insecticides, surfactants	(Liu and Avendaño, 2013)
	Alcohols	Fluorotelomer alcohols; volatile precursors	6:2 FTOH, 8:2 FTOH	Water- and oil-repellent coatings, paper treatment	(Ellis et al., 2004)
	Phosphates	Fluorinated phosphate esters	PAPs, diPAPs	Food contact materials, grease-proof paper	(Organisation for Economic Co-operation and Development (OECD), 2022)
3. Chain Length-Based	Short-Chain PFAS	<C6 (sulfonates), <C8 (carboxylates); less bioaccumulative	PFBS, PFHxA	Substitutes for long-chain PFAS in cleaning products, textiles	(Wang et al., 2017; Organisation for Economic Co-operation and Development (OECD), 2021)
	Long-Chain PFAS	≥C6 (sulfonates), ≥C8 (carboxylates); highly persistent	PFOS, PFOA, PFNA	AFFF, fluoropolymer manufacturing, fabric treatments	(Yeardley et al., 2024; Ateia et al., 2023; Giesy and Kannan, 2002)
4. Use/Origin-Based	Legacy PFAS	Widely used historically; now phased out or restricted	PFOS, PFOA	Firefighting foams, industrial surfactants, coatings	(Wang et al., 2017; United Nations Environment Programme UNEP, 2022)
	Replacement PFAS	Developed to substitute legacy PFAS	GenX, ADONA, PFBS	Fluoropolymer synthesis, stain repellents	(Sun et al., 2016; Gomis et al., 2018)
	Precursor PFAS	Biotic/abiotic degradation forms terminal PFAS	FTOHs, PAPs, FOSA	Used in consumer goods that degrade over time	(Liu and Avendaño, 2013; Wang et al., 2013)
5. OECD-Based	Non-Polymeric PFAS	Discrete, well-defined molecular PFAS	PFCAs, PFSAs, FTOHs	Industrial and consumer chemical applications	(Organisation for Economic Co-operation and Development (OECD), 2021; Organisation for Economic Co-operation and Development (OECD), 2022; Organisation for Economic Co-operation and Development (OECD), 2025)
	Polymeric PFAS	Large molecules, may degrade to PFAS	Fluoropolymers, side-chain fluorinated polymers	Wire coatings, medical tubing, gaskets, membranes	(Organisation for Economic Co-operation and Development (OECD), 2021; Henry et al., 2018)

Abbreviations: Per- and Polyfluoroalkyl Substances (PFAS); Perfluorooctanoic Acid (PFOA); Perfluorooctane Sulfonate (PFOS); Perfluorohexanoic Acid (PFHxA); Aqueous Film-Forming Foam (AFFF); 6:2 Fluorotelomer Alcohol (6:2 FTOH); Polyfluoroalkyl Phosphoric Acid Diesters (PAPs); Perfluorononanoic Acid (PFNA); Perfluorohexane Sulfonate (PFHxS); Perfluorobutane Sulfonate (PFBS); N-Methyl Perfluorooctane Sulfonamidoacetic Acid (MeFOSAA); Perfluorooctane Sulfonamide (FOSA); N-Methyl Perfluorooctane Sulfonamide (N-MeFOSA); 8:2 Fluorotelomer Alcohol (8:2 FTOH); Polyfluoroalkyl Phosphate Diesters (diPAPs); Hexafluoropropylene Oxide Dimer Acid (GenX); 4,8-Dioxa-3H-perfluorononanoic Acid (ADONA)

encompass widely used PFOA, which is a highly persistent pollutant commonly known as legacy PFAS (Cousins et al., 2022). As a result of regulatory restrictions on the production of PFOA and PFOS (under Stockholm Convention on persistent organic pollutants), short-chain PFAS (<8 carbons in the backbone) were subsequently introduced as less persistent replacements (Lu et al., 2019; Zheng et al., 2023). Recently, ultrashort-chain (12–3 carbons) and short-chain (4–7 carbons) PFAS (Fig. 3) have been found to be ubiquitous in the environment, raising concerns about their potential negative effects (Zheng et al., 2023; Neuwald et al., 2022; Arp et al., 2024). Recent studies have shown that trifluoroacetate (TFA), an ultrashort-chain perfluoroalkyl acid (PFAA), is now the most abundant PFAS in the environment, accounting for more than 90 % of the total PFAS detected (Zheng et al., 2023; Neuwald et al., 2022; Arp et al., 2024).

Many PFAS share a general chemical structure represented as CnF2n+1 R, where n indicates the number of carbon atoms in the perfluorinated chain, 2 n + 1 denotes the number of fluorine atoms bonded to the carbon chain, and R represents a polar functional group (Dias et al., 2024). The variation in the fluorination, functional groups,

and chain length mainly accounts for their chemical diversity (Table 1). The perfluorinated "tail" is highly hydrophobic due to the strong C-F bonds, while the polar "head" confers hydrophilic properties. This dual character enables PFAS to interact with both aqueous and non-aqueous environments, making their widespread use in industrial and consumer applications. Additional functional groups commonly found in PFAS include carboxylates (–COO[–]/–COOH), sulfonates (–SO₃[–]/–SO₃H), phosphonates (–PO₃H₂), and hydroxyls (–OH) (Harris et al., 2025; Itumoh et al., 2024). These functional groups significantly influence the solubility, reactivity, and overall environmental behaviour of PFAS, with their specific impacts varying depending on factors such as pH (Williams et al., 2022; Dias et al., 2024).

In response to increasing awareness of the environmental persistence and health risks associated with these legacy PFAS, industries have shifted toward replacement compounds, typically short-chain PFAS containing fewer than six carbon atoms. However, these substitutes also exhibit high environmental persistence and are poorly removed by existing treatment technologies (Liu et al., 2022; Smaili and Ng, 2023). For example, GenX chemicals such as hexafluoropropylene oxide dimer

Table 2

Overview distribution of PFAS across various environmental domains.

PFAS Category	Example Compounds	Groundwater	Atmosphere	Other Matrices (Soil, Sediment, Biota)	Detection Frequency	Detection Examples	Typical Source/ Use	References
Long-chain PFASs	PFOS, PFHxS, PFDS	High	Moderate (aerosols)	Widespread in soil & wildlife	Very High	Military & airport sites (US, Sweden, Germany)	AFFF, textiles, plating	(Ahrens, 2011; Barzen-Hanson et al., 2017; McDonough et al., 2021)
Long-chain PFCAs	PFOA, PFNA, PFUnDA	High	Moderate (gas & particle)	Sediment, fish tissue	Very High	Ohio River Valley, Japan, Netherlands	PTFE production, waterproof coatings	(Agency for Toxic Substances Disease Registry (ATSDR), 2021; Hu et al., 2016; Wang et al., 2014)
Short-chain PFCAs	PFBA, PFPeA, PFHxA	Common	High (gas phase)	Rainwater, surface water	High	Rain and runoff in US, Europe & China	Fluorotelomer breakdown, food packaging	(Ahrens, 2011; Scott et al., 2006)
Fluorotelomer sulfonates (FTSAs)	6:2 FTS, 8:2 FTS	Near AFFF-contaminated sites	Rarely detected	Biosolids, soils	Moderate	US DoD sites, Canada, Germany	AFFF, coatings, surfactants	(Barzen-Hanson et al., 2017; Hu et al., 2016)
Perfluoroalkyl sulfonamides	FOSA, N-MeFOSA	Rare	Dominant airborne PFAS	Arctic mammals, seabirds	Moderate	Arctic & remote regions	Breakdown products of PFOS precursors	(Jahnke and Berger, 2009; Martin et al., 2003)
Perfluoroalkyl ether acids (PFEAs)	GenX (HFPO-DA), ADONA	High near sources	Not reported	Surface water, sediment	Moderate	Cape Fear River (USA), Netherlands	PFOA replacement in fluoropolymer plants	(Sun et al., 2016; Hopkins et al., 2018)
Ultra-short-chain PFAS	TFA, DFA	Rarely monitored	High (gas phase)	Cloudwater, snow	Emerging	High-altitude rainwater (Alps, Tibet)	Atmospheric oxidation of precursors	(Boiteux et al., 2017)
Chlorinated PFAS	Cl-PFNA, Cl-PFPeA	Localized detection	Rare	Wastewater sludge	Low	Industrial effluent zones (China)	Specialty fluorochemicals, AFFF additives	(Pan et al., 2021)

Abbreviations: Perfluorooctane Sulfonate (PFOS); Perfluorohexane Sulfonate (PFHxS); Perfluorodecane Sulfonate (PFDS); Perfluorooctanoic Acid (PFOA); Perfluorononanoic Acid (PFNA); Perfluoroundecanoic Acid (PFUnDA); Perfluorobutanoic Acid (PFBA); Perfluoropentanoic Acid (PFPeA); Perfluorohexanoic Acid (PFHxA); 6:2 Fluorotelomer Sulfonate (6:2 FTS); 8:2 Fluorotelomer Sulfonate (8:2 FTS); Perfluorooctane Sulfonamide (FOSA); N-Methyl Perfluorooctane Sulfonamide (N-MeFOSA); Hexafluoropropylene Oxide Dimer Acid (HFPO-DA); 4,8-Dioxo-3H-perfluorononanoic Acid (ADONA); Trifluoroacetic Acid (TFA); Difluoroacetic Acid (DFA); Chlorinated Perfluorononanoic Acid (Cl-PFNA); Chlorinated Perfluoropentanoic Acid (Cl-PFPeA); Aqueous Film-Forming Foam (AFFF); U.S. Department of Defense (US DoD); Polytetrafluoroethylene (PTFE)

acid (HFPO-DA) and its derivatives were introduced as alternatives to PFOA in fluoropolymer manufacturing, while PFBS and PFHpA have been used to replace PFOS in various applications (Kancharla et al., 2022). Other noteworthy PFAS include PFNA, PFDA, and PFHxS, all of which continue to attract scientific and regulatory attention due to their persistence and potential health impacts. These also include ultrashort-chain PFAS, such as trifluoroacetate (TFA) and perfluoropropanoic acid (PFPrA), which are known breakdown products of larger PFAS (Zheng et al., 2023; Arp et al., 2024; Sun et al., 2016). Recently, both TFA and PFPrA have been identified as the most abundant and rapidly increasing PFAS in the environmental matrices (Zheng et al., 2023; Neuwald et al., 2022). Furthermore, conventional treatment plants are largely ineffective at removing short-chain and ultrashort-chain PFAS (Ateia et al., 2019). Therefore, their structural characteristics explain the complexity of remediation efforts and account for their presence across various environmental domains.

4. Geographic distribution of PFAS contamination

PFAS were created for the very purpose to resist degradation under environmental conditions, now we are witnessing a widespread contamination on a global scale, with PFAS being detected across various environmental domains, including soil, water, air, food, wildlife, and humans (Hepburn et al., 2019). A recent meta study conducted by Cordner et al. (Cordner et al., 2024) identified 22,934 PFAS-contamination sites across Europe that included 13,745 sites presumably contaminated with fluorinated AFFF discharge, 2911 industrial facilities, and 4752 PFAS-containing waste sites. A multi-country study across Japan, China, and India analyzed seasonal and local variations of PFAS associated with atmospheric particulates

(PM₁₀–PM_{0.1}), revealing that ultrafine particles (PM_{0.1}) were enriched with PFOA, PFNA, and PFDA, while PFOS was more prevalent in coarser particles (Ge et al., 2017). Similarly, atmospheric monitoring across the Atlantic and Antarctic Peninsula detected 12 different PFAS compounds, ranging from 2.8 to 68.8 pg m⁻³ concentrations in the gas phase and 125–303 pg L⁻¹ in snow, with FTOHs being the major PFAS detected (Wang et al., 2015). Although global studies on PFAS contamination remain limited, several regional investigations have reported PFAS presence in various environmental domains. The following section summarizes these findings based on specific geographical locations.

4.1. Reported sites in aquatic environments

The dissemination of PFAS in aquatic ecosystems is generally led by their stable structure and solubility in water (Panieri et al., 2022). These properties promote their widespread occurrence across industrial, waste management, and urban systems, with frequent detection reports in WWTPs (Gobelius et al., 2019; Sinclair and Kannan, 2006) and municipal landfills (Busch et al., 2010). The presence of PFAS in surface and groundwater near firefighting sites has also been reported in several studies, suggesting that this represents a noteworthy route of human exposure (Awad et al., 2011; Kwadijk et al., 2014; Mussabek et al., 2022). PFAS-AFFF are widely used by firefighters at airfield sites worldwide. In Sweden, water samples around firefighting training sites showed average PFAS concentrations as high as 1.700 ng/L, with sediment levels up to 76 µg kg⁻¹ dw, primarily composed of PFHxS, and PFOS (Mussabek et al., 2019). Moreover, in southern Sweden, Mussabek et al. (Mussabek et al., 2022) evaluated groundwater contamination and found ΣPFAS concentrations ranging from 20 to 20,000 ng L⁻¹, predominantly comprising PFOS and PFHxS, which were attributed to AFFF

Table 3

Enzymes and microorganisms involved in PFAS degradation.

Enzyme Class	Specific Enzyme/System	Microbial Source	Target PFAS/ Organofluorine Compound	Key Mechanism	Observations	References
Hydrolase	fluoroacetate dehalogenases	<i>Burkholderia</i> , <i>Pseudomonas</i> , <i>Delftia</i> , <i>Rhodopseudomonas</i> , <i>Moraxella</i>	Fluoroacetate (MFA)	C-F bond cleavage (hydrolysis)	Model for C-F cleavage; activity decreases with increased fluorination.	(Kurihara et al., 2003; Chan et al., 2011; Camboim et al., 2012; Donnelly and Murphy, 2009)
	DAR3835	<i>Dechloromonas</i>	MFA, DFA	C-F bond cleavage	Complete	(Khusnutdinova et al., 2023)
	NOS0089	<i>aromatica</i>	MFA, DFA	(hydrolysis)	defluorination of DFA to glyoxylic acid	(Farajollahi et al., 2024)
	POL4478	<i>Nostoc</i> sp.	MFA, DFA	C-F bond cleavage	Active against MFA	
	RPA1163	<i>Polaromonas</i> sp.	MFA, DFA	(hydrolysis)	(higher rate than DeHa4) and DFA (less active than DeHa4); inactive against TFA, PFOA.	
Oxidoreductase	DeHa2 (haloacid dehalogenase)	<i>R. palustris</i>	MFA, DFA			
	DeHa4 (fluoroacetate dehalogenase)	<i>Delftia acidovorans</i> D4B				
	Peroxidases (HRP, Lignin Peroxidase, Manganese Peroxidase)	Various fungi (e.g., White rot fungi), engineered systems	Large PFAS molecules, PFOA, PFOS	Generation of hydroxyl radicals; fragmentation of large PFAS.	Efficiency limited by H ₂ O ₂ availability; can break large PFAS into smaller fragments.	(Colosi et al., 2009)
	Laccases (in LMS)	<i>Trametes versicolor</i> , <i>Pleurotus ostreatus</i>	PFAAs (e.g., PFOA, PFOS)	Oxidation via chemical mediators	Observed "removal" largely due to physical sorption onto enzyme, not true degradation.	(Luo et al., 2015)
	Oxygenases (e.g., 2- Fluorobenzoate–1,2- dioxygenase)	Various microorganisms	Aryl fluorides	Mono/ dioxygenation at C atoms bearing F or adjacent	Catalyzes aromatic cis- hydroxylation leading to decarboxylation and defluorination.	(Wackett and Robinson, 2024; Wackett, 2024)
Multi-enzyme System	Cytochrome P450s	Various microorganisms, engineered systems	PFAS (e.g., for C-F bond cleavage)	Oxidative defluorination; reductive dehalogenation (anaerobic).	Promising for direct C-F bond cleavage, especially in enzyme cocktails.	(Wackett, 2022)
	<i>Labrys portucalensis</i> F11	<i>Labrys portucalensis</i> F11 (bacterium)	PFOS, 5:3 FTCA, 6:2 fluorotelomer sulfonate	C-F bond cleavage, metabolism of carbon atoms	Metabolised > 90 % of PFOS in 100 days; removed fluorine from some toxic metabolites.	(Wijayahena et al., 2025)
	<i>Acetobacterium</i> species	<i>Acetobacterium</i> species (bacteria)	Unsaturated PFAS	Defluorination (C-F bond splitting)	Specialised channels pump out toxic fluoride, enabling survival and continued degradation.	(Yu et al., 2024)
Engineered Enzymes	Enz209	Engineered bacterial variants (Aether Biomachines)	PFOS-like substrates, Pentafluoropropionic acid (PFPrA)	Defluorination	Partial defluorination of PFOS (17 F to 14 F C7 alcohol); 5–30 % PFPrA degradation in 24 h.	(Saran et al., 2023)

Abbreviations: Monofluoroacetate (MFA); Difluoroacetate (DFA); Perfluorooctanoic Acid (PFOA); Perfluorooctane Sulfonate (PFOS); Perfluoroalkyl Acids (PFAAs); Pentafluoropropionic Acid (PFPrA); 5:3 Fluorotelomer Carboxylic Acid (5:3 FTCA); 6:2 Fluorotelomer Sulfonate (6:2 FTS)

usage during fire-training activities. Annual PFAS emissions were estimated between 11 and 1125 kg yr⁻¹, with the distribution influenced by PFAS chain length and the frequency of fire-training sessions (Mussabek et al., 2022). A study conducted in Canada, following a spill of PFOS-containing AFFF from Toronto International Airport in the year 2000, reported elevated levels of PFOS in Spring Creek Pond, indicating a localized but persistent contamination after nine years (Awad et al., 2011).

Evidence from regional monitoring studies also indicated that PFAS contamination is not restricted to industrial hotspots but has now been observed globally across aquatic environments, from urban rivers to remote oceanic and polar regions. A significant level of PFOA (0.4–96.4 ng L⁻¹) and PFOS (0.4–13.23 ng L⁻¹) were detected in river waters feeding into the Lake Victoria Gulf, while the lake itself exhibited slightly lower concentrations (PFOA: 0.4–11.65 ng L⁻¹; PFOS: 0.4–2.53 ng L⁻¹) (Orata et al., 2009), likely due to dilution effects. Similarly, in South Africa, analyses along the Vaal River revealed PFAS levels from <LOQ to 38.5 ng/L, indicating a potential risk from industrial and mining activities (Groffen et al., 2018). In China, multiple sites, such as Qing River in Hubei Province contained up to 207.59 ng L⁻¹ of 12 PFAS (Zhou et al., 2017), while surface waters of the Yangtze River in Jiangxi Province showed high concentrations of PFBS and PFOA ranging

from 146.2 to 586.2 ng L⁻¹ in Nanchang and 46.2–157.6 ng L⁻¹ in Jiujiang city (Tan et al., 2018). The surface seawater samples of the South China Sea, Hong Kong, and Macau ranged from 0.195 to 4.925 ng L⁻¹, highlighting socio-economic influences on PFAS levels (Kwok et al., 2015).

Furthermore, sediments in coastal and inland water bodies act as both repositories and secondary sources of PFAS. In the East China Sea, sediment samples showed PFOS as the dominant contaminant with 11.4 ng g⁻¹ dw concentration (Yan et al., 2015). In the Bohai Sea, seawater and sediment samples revealed PFOA as the primary contaminant (up to 99.4 ng L⁻¹ and 2.78 ng g⁻¹ dw, respectively) (Chen et al., 2016), with riverine discharges identified as major PFAS sources (Chen et al., 2017; Liu et al., 2019). The Pearl River Delta showed low but detectable levels of PFBA, PFOA, and PFOS in both water and sediment samples, along with emerging PFAS such as HFPO-DA and Cl-PFESAs (Wang et al., 2019). In the Red Sea near Saudi Arabia, up to 956 ng L⁻¹ of 12 PFAS were found, with PFHxA, PFHxS, and 6:2 FTS being the most prevalent compounds (Ali et al., 2021). In Spain's Ebro Delta, PFOA dominated freshwater samples (0.87–1.6 ng L⁻¹), while sediments were enriched in PFOS (up to 22.6 ng g⁻¹ dw) and showed seasonal fluctuations (Pignotti et al., 2017). Sediment analysis from the Rhone River in France recorded an increase in PFAS levels from 2 ng/g dw (pre-1984) to

51.4 ng/g dw (1994), followed by stabilization near 10 ng/g dw (Mourier et al., 2019), with more recent measurements (2017–2018) detecting PFHxA and PFOS in concentrations exceeding European EQS limits in over 80 % of samples (Schmidt et al., 2019). In Sweden's Lake Sänksjön, which is impacted by firefighting foam, PFOS and PFHxS were detected in water (95–100 ng/L) and sediments (3–61 $\mu\text{g kg}^{-1}$ dw) (Mussabek et al., 2020).

The most common PFAS found in sediments in North America, especially in the Great Lakes named Erie, Ontario, and St. Clair, were PFHxA and PFOS, with PFBA being the most prevalent (14.2–26.2 ng/g dw) (Codling et al., 2018). Along with causing immunological and liver damage, striped bass from North Carolina's Cape Fear River had an average PFOS level of 490 ng mL⁻¹, which is about 40 times greater than that of farmed controls (Guillette et al., 2020). According to Guo et al. (Guo et al., 2016), PFOS accounted for more than 80 % of the total PFAS load in Lake Ontario (13.1 ng/g) in sediment cores from the Great Lakes region. Concentrations of PFOS, PFBS, and PFOA in Port Phillip Bay, Australia, reached 75 ng L⁻¹, indicating the occurrence of PFAS even in sparsely populated areas (Allinson et al., 2019).

These studies demonstrate that PFAS contamination is not limited to urbanized or industrial areas; its detection in remote Arctic regions, including the Bering Sea and Western Arctic Ocean, also showed PFAS contamination in sediments (0.06–1.73 ng g⁻¹ dw), primarily from PFOS, PFNA, and PFBS, highlighting oceanic transport and AFFF discharge as likely sources (Lin et al., 2020). Likewise, in the Western Mediterranean Sea, surface waters showed uniform PFAS distribution (0.246–0.515 ng L⁻¹), with PFHxA, PFHpA, PFOA, and PFHxS being frequently detected, reflecting regional manufacturing practices and regulatory preferences for short-chain PFAS (Brumovsky et al., 2016).

4.2. Reported sites in air environments

Several studies have demonstrated the widespread presence of PFAS in the atmosphere, highlighting both their legacy persistence and growing recognition as emerging contaminants of concern. In Germany, variable levels of fluorotelomer alcohols (FTOHs), fluorotelomer acrylates (FTACs), and perfluorooctane sulfonamide derivatives (Et/MeFOSA and Et/MeFOSE) were detected in indoor air samples from schools and residences, indicating that inhalation may serve as a significant non-oral exposure route for vulnerable populations (Fromme et al., 2015). Over the northern South China Sea, FTOHs, FTAs, FOSAs, and FASEs were identified in the atmosphere at average concentrations of 54.5 pg m⁻³, with FTOHs being the most abundant subgroup (Lai et al., 2016). In Asian cities, hexafluoropropylene oxide dimer acid (HFPO-DA, commonly known as GenX) was detected in particulate matter (PM) samples at concentrations ranging from < 0.086–21.5 pg m⁻³, alongside legacy compounds such as PFOS, PFOA, and PFHxA (Lin et al., 2020). Additionally, a non-targeted analysis using cryogenic air sampling identified 38 PFAS classes and 117 homologues in both atmospheric particulate and gas phases, including novel chlorinated perfluoropolyether carboxylic acids and 12 hydro-substituted perfluoroalkyl carboxylates (H-PFCAs), underlining the environmental relevance of alternative PFAS and the need for further investigation (Yu et al., 2020).

4.3. Reported sites in soil environments

PFAS are widespread contaminants detected in soil and sediment across the globe. The extent and profile of PFAS contamination vary geographically, depending largely on the regional sources such as AFFF application, industrial production, biosolid usage, and atmospheric deposition. In China, substantial PFAS contamination has been reported in multiple regions. For example, Shanghai soils contaminated by AFFF showed $\Sigma 11$ PFAS concentrations between 3.4–531.7 ng g⁻¹ (Cao et al., 2022), while in Fujian, soils near POSF manufacturing facilities had extremely high $\Sigma 18$ PFAS levels of 26,300 ng g⁻¹ (Jia et al., 2023).

Similarly, sediments from the Xiaoqing River, influenced by a fluoropolymer production facility, exhibited $\Sigma 12$ PFAS concentrations ranging from 8.44 to 465.6 ng g⁻¹ (Sun et al., 2016). Shandong soils near industrial sources showed $\Sigma 18$ PFAS concentrations between 4.40–189 ng g⁻¹ (Xie et al., 2021).

In Europe, elevated contamination has been documented in several regions, particularly due to firefighting foam usage. In Norway, airport soils exhibited PFOS concentrations of 9–2600 ng g⁻¹ (Hale et al., 2017), with other sites reaching up to 6500 ng g⁻¹ ($\Sigma 12$ PFAS) (Høisæter et al., 2019). In Finland, soil concentrations were as high as 2800 ng g⁻¹ ($\Sigma 23$ PFAS) (Reinikainen et al., 2022). France and the Netherlands documented atmospheric deposition as a key source, with PFAS concentrations in soil ranging from 3.3 to 347 ng g⁻¹ (Martin et al., 2023; Galloway et al., 2020; Gerardu et al., 2023). In Australia, AFFF usage contributed to PFOS levels between 0.2 and 1400 ng g⁻¹ (Mesfin Tefera et al., 2022), while biosolids-amended soils contained $\Sigma 38$ PFAS concentrations of 10.4–104 ng g⁻¹ (Sivaram et al., 2023). In the United States, biosolid-based composts in soils reported $\Sigma 17$ PFAS levels ranging from 9.0 to 199 ng g⁻¹ (Kim Lazcano et al., 2020). And sediments in Pensacola Bay, influenced by military bases and carpet industry sources, exhibited $\Sigma 51$ PFAS from 0.04 to 0.48 ng g⁻¹ (Gan et al., 2022). Sediments from Michigan showed $\Sigma 10$ PFAS concentrations up to 3.25 ng g⁻¹, largely attributed to airports and industrial discharge (Choi et al., 2019). In South Korea, soils near industrial complexes and landfills recorded PFAS concentrations ranging from 0.346 to 11.7 ng g⁻¹ (Balgooyen and Remucal, 2022). Even remote regions are not exempt from contamination. In the Bering Sea, sediment samples showed PFAS concentrations of 0.06–1.73 ng g⁻¹, indicating long-range atmospheric transport and sedimentation (Lin et al., 2020). According to these statistics, PFAS contamination in soil and sediment is a global environmental issue, driven by both local anthropogenic activities and global atmospheric transport. Recent research has also highlighted the pervasive nature and differing environmental transport mechanisms of PFAS from manufacturing facilities (Schumacher et al., 2024). A study at a PFAS manufacturing facility in New Jersey, USA, investigated the distribution of PFAS across soil, soil gas, and groundwater samples (Schumacher et al., 2024). It revealed that fluorotelomer alcohols (FTOHs), highly volatile PFAS, were present in soil (0.26–36.15 ng g⁻¹) and in soil gas (160–12,000 E $\mu\text{g m}^{-3}$), while PFCAs were detected in soil (4.3–810 ng g⁻¹), soil gas (<0.10–180 $\mu\text{g m}^{-3}$), and groundwater (37–49 $\mu\text{g L}^{-1}$) (Schumacher et al., 2024). The decrease in FTOHs and PFCA concentrations with distance from the originating source, strongly suggested that FTOHs primarily moved in the vapor phase and PFCAs migration were through the aqueous phase (Schumacher et al., 2024). This study was the first to quantify volatile PFAS in soil gas at such a facility, providing crucial evidence for vapor intrusion as a potential exposure pathway for indoor air contamination (Schumacher et al., 2024).

In contrast to the findings at the manufacturing facility, Schumacher et al. (Schumacher et al., 2023) also investigated a closed municipal solid waste (MSW) landfill in Georgia, USA. At this site, the potential for PFAS contamination via the vapor intrusion pathway appeared significantly lower (Schumacher et al., 2023). PFAS concentrations were not detected in the methane vent wellhead gas samples, and the concentration of PFBA, PFOA, PFHpA and PFHxA were very low (<0.004 $\mu\text{g L}^{-1}$) in the groundwater samples collected at the Georgia landfill (Schumacher et al., 2023). This notable difference in PFAS distribution and concentration between the two sites suggests that the source type and its specific characteristics play a crucial role in determining the environmental mobility and potential for vapor intrusion of these pervasive chemicals.

Overall, while numerous studies have highlighted the impact of various PFAS across different countries, data on PFAS contamination remains limited, particularly in developing nations where regulatory frameworks are weak or absent. Baluyot et al. (Baluyot et al., 2021) demonstrated that PFAS levels in several Asian countries exceeded the

maximum exposure limits set by the National Food Agency of Sweden. The drive for industrial growth to compete in the global economy has led to unchecked PFAS emissions in many developing regions. Consequently, countries like China have reported a significant rise in PFAS levels, whereas data from other parts of Asia, especially South and Southeast Asia, remain sparse and largely unexamined (Baluyot et al., 2021). Given the persistence, mobility, and bioaccumulative nature of PFAS in terrestrial and aquatic ecosystems, there is a pressing need for coordinated global monitoring, regulation, and remediation efforts. Without such interventions, the long-term consequences of PFAS contamination will continue to escalate, threatening environmental integrity, wildlife, and human health.

5. Adverse effects of PFAS

5.1. Environmental impacts of PFAS

PFAS resistance to degradation and their high mobility rate pose a significant threat to soil ecosystems. Liu et al. (Liu et al., 2015), reported that soils accounted for over 40 % of total PFOS mass in a Chinese coastal region. The issue is further intensified by the use of PFAS-contaminated biosolids on agricultural fields, which contributes to soil accumulation, groundwater contamination, and eventual uptake by crops (Pepper et al., 2023). This disrupts the soil microbial communities, impacting the nutrient cycle and overall ecosystem (Xu et al., 2023). Cao et al. (Cao et al., 2022) investigated a fire-training areas of Shanghai, China and found substantial soil PFAS concentrations, ranging from 0.0034 to 0.5317 $\mu\text{g g}^{-1}$ dw, affecting bacterial diversity. PFAS were able to reduce the biodiversity and connectivity of soil bacteria at concentrations $> 0.1 \mu\text{g g}^{-1}$ dw (Cao et al., 2022). Similarly, Qiao et al. (Qiao et al., 2018) reported that low PFOS concentrations (0.01 $\mu\text{g/g}$ and 0.1 $\mu\text{g/g}$) enhanced bacterial richness, but higher levels (10 $\mu\text{g g}^{-1}$ soil) decreased the overall microbial diversity. Qiao et al. (Qiao et al., 2018) also confirmed that PFOS, 6:2FTS and 6:2F53B suppressed soil enzyme activity and destroyed gene expression, highlighting a profound disruption of crucial microbial functions. A six-week microcosm study conducted by Xu et al. (Xu et al., 2023) revealed that even environmentally relevant concentrations of PFAS (0.5 ng g^{-1} for PFBS) significantly impact soil health.

Expanding on the direct microbial impacts, further research on the long-term effects of PFOA at a field scale has revealed specific disruptions to nutrient cycling processes. Notably, PFOA exposure was found to inhibit alkaline phosphatase activity and reduce the abundance of phosphorus cycling-related functional genes, suggesting potential impairments to soil fertility (Huang et al., 2024). This collectively underscores PFAS as a potent driver of adverse environmental change in terrestrial ecosystems.

Further exploring PFAS trophic transfer in terrestrial systems, a study by Judy et al. (Judy et al., 2022) investigating the movement of seven distinct PFAS compounds from *Solanum lycopersicum* (tomato) plants to *Manduca sexta* (tobacco hornworms) revealed nuanced accumulation patterns. While biomagnification factors (BMFs) consistently remained below 1, indicating a lack of biomagnification within this specific transfer, PFOS exhibited significantly higher bioaccumulation in hornworms, with approximately five-fold greater BMFs and assimilation efficiencies compared to other tested PFAS. Importantly, this research also demonstrated a positive correlation between PFAS carbon chain length and both plant uptake and trophic transfer (assimilation efficiency and BMFs) for both sulfonates and carboxylic acids, suggesting that longer-chain PFAS may accumulate more efficiently within this particular food chain context.

A 119-day glasshouse study by Lal et al. (Lal et al., 2020) further illustrated the implications of PFAS accumulations in terrestrial biota, showing considerable PFOS uptake and accumulation in home-grown vegetables. While bioconcentration factors (BCFs) generally increased with soil PFOS concentration, a major finding was that daily intake rates

of PFOS via vegetable consumption exceeded the tolerable daily intake for young children, as per Food Standards Australia New Zealand (FSANZ). This underscores a significant human health risk through food chain transfer, aligning with broader concerns regarding PFAS persistence and bioaccumulation across environmental compartments. Beyond human exposure, PFAS accumulation also poses direct ecotoxicological risks to plants, including reduced seed germination, impaired root growth, and disruptions in nutrient assimilation (Li et al., 2022).

Earthworms, as critical components of soil ecosystems, serve as important bioindicators and vectors for PFAS contamination, demonstrating notable accumulation and transformation capabilities (Yeadley et al., 2024). Studies have shown that earthworms can take up polymeric FASA-based copolymers and subsequently biotransform them into lower molecular weight PFAS such as FOSAs, FOSA, FBSA, and PFBS, indicating that polymeric PFAS serve as indirect environmental sources (Fredriksson et al., 2024). This accumulation is particularly pronounced in sludge-amended soils, identifying sludge application as a major pathway for elevated PFAS levels in both soil and earthworms (Fredriksson et al., 2024). Further research that specifically evaluated the bioaccumulation and chronic toxicity of these short-chain compounds and precursors revealed that FBSA is significantly more bioaccumulative and toxic than PFBS, causing mortality and impaired behavioral impairments at even lower concentrations (Groffen et al., 2025). Given that earthworms are a common food source for various bird species and small mammals, the substantial PFCA concentrations detected in these worms suggest a potential pathway for PFAS chemicals to enter higher trophic levels of the food chain (Groffen et al., 2025).

Collectively, these findings highlights the pervasive nature of PFAS contamination in soil ecosystems, detailing their disruptive impact on microbial communities, trophic transfer potential through terrestrial food webs, and the resulting both direct and indirect risks to human health. Furthermore, the key systemic issues that challenge PFAS management include, but not limited to the following: 1) the significant data gaps pertaining to ecological effects and environmental behavior of newly-manufactured PFAS compounds; 2) unclear mechanisms and rates of precursor transformation; and 3) insufficient understanding of molecular-level interactions, like sorption and air–water interfacial adsorption. These uncertainties hinder accurate risk assessment and conceptual site model development. Regulatory frameworks lag behind the rapid introduction of novel PFAS analogues, resulting in fragmented policies. Additionally, increased costs and operational burdens, such as those linked to biosolid restrictions, add further strain to current waste management practices (Ehsan et al., 2024). Together, these challenges highlight the inadequacy of current approaches and underscore the need for a precautionary, class-based framework to enable proactive and sustainable PFAS regulation and control.

5.2. Impacts of PFAS on human health

Beyond ecological impacts, significant concerns exist regarding PFAS's influence on human health, particularly affecting the immune, reproductive, and endocrine systems. A meta-analysis evaluating prenatal and childhood PFAS exposure revealed suggestive evidence for associations with increased childhood infections, particularly for PFOS, PFOA, PFHxS, and PFNA, though evidence for reduced vaccine antibody titers remained moderate to inconclusive (Antoniou and Dekant, 2024). In the realm of reproductive health, systematic reviews and meta-analyses consistently link increased PFAS exposure to adverse outcomes. For instance, PFOA and PFOS exposure are negatively associated with female fecundability and PFOA has been linked to increased infertility (Wang et al., 2023). Similarly, reviews of male reproductive health highlight mounting evidence connecting PFAS exposure to testicular dysfunction and reduced semen quality, although certain epidemiological inconsistencies persist (Sun et al., 2023; Calvert et al., 2022).

Further elucidating their endocrine-disrupting potentials, a cross-

sectional study among an elderly Chinese population demonstrated widespread PFAS exposure, with PFOS and PFOA being the most prevalent compounds (Tan et al., 2024). This research identified significant correlations between specific PFAS (e.g., PFDA, PFOS, PFNA) and altered thyroid hormone levels (e.g., decreased TSH, and increased T4/T3), collectively indicating PFAS-induced thyroid disruption in older adults (Tan et al., 2024). Expanding on these observations, a large-scale analysis of 3954 adults in the United States National Health and Nutrition Examination Survey (NHANES; 2007–2012) corroborated the association between PFAS mixtures and an increased risk of thyroid hormone disorders (Zheng et al., 2023). Similarly, a study conducted on 151 boys from Spain's INMA-Granada birth cohort (2017–2019) reported that PFOA and PFUnDA were associated with increases in free thyroxine (T4) levels (0.72 pmol/L and 0.36 pmol/L, respectively), while PFAS mixture also elevated free T4 levels (6.47 pmol/L) (Freire et al., 2023). This research highlighted that thyroid alterations were more pronounced among boys with lower iodine intake, although the results remain uncertain due to the small sample size (Freire et al., 2023).

Furthermore, studies have also highlighted risks in pregnant individuals and adolescents. For instance, in pregnant individuals, exposure to PFOA, PFOS, and PFHxS was associated with significantly higher odds of gestational hypertension and increased diastolic blood pressure (Hu et al., 2023; Preston et al., 2022). Similarly, in adolescents, various PFAS (e.g., PFOS, PFNA, PFDA, PFOA, PFHxS, PFHpS) demonstrated positive associations with dyslipidemia (increased apolipoprotein B, total-, and LDL cholesterol), hypertension and obesity (Averina et al., 2021). Collectively, these findings establish PFAS as significant environmental contributors to the rising prevalence of cardiometabolic conditions. In terms of metabolic disorders, PFAS exposure is associated with increased risks of insulin resistance and type 2 diabetes (Jia et al., 2024). Apart from these systemic effects, direct human exposure through the ingestion of contaminated soil and dust also contributes significantly to overall PFAS body burden. Parker et al. (Parker et al., 2024) performed an *in vitro* assay with 14 PFAS in 33 sets of soils and dusts and detected that determined 78.0 %–88.9 % PFAS bio-accessibility. These findings collectively underscore the pervasive and multifaceted health risks posed by PFAS exposure, emphasizing the urgent need for stricter regulatory measures, continuous biomonitoring, and the development of effective mitigation strategies to protect human health.

5.3. Wildlife impacts of PFAS

PFAS contamination poses significant threats to wildlife across diverse food chain levels and habitats. Augustsson et al. (Augustsson et al., 2021) detected elevated levels of PFOS in freshwater fish, which were associated with liver toxicity, oxidative stress, and reduced reproductive success. Amphibians such as *Xenopus laevis* exposed to PFAS exhibited endocrine disruption and developmental delays (Degitz et al., 2024). PFAS were the predominant organohalogen contaminants detected across ten bird species in Korea, with PFOS reaching concentrations up to 11,300 ng g⁻¹ ww, the highest reported to date, while 49 % of individuals exceeded the predicted no-effect concentration, indicating potential ecotoxicological risk independent of trophic level (Barghi et al., 2018). Additionally, behavioral changes, including altered feeding and mating behaviors, have been observed in rodents exposed to PFAS, potentially affecting population dynamics in the wild.

6. Challenges in remediation, detection and environmental management

6.1. Remediation challenges

The remediation processes for PFAS treatment have been primarily designed for the removal of compounds from contaminated sites,

particularly water. However, the complete destruction of these chemicals remains challenging and is one of the main limitations in PFAS remediation (Ross et al., 2018). Several remediation technologies are currently available, focussing on either the separation or destruction of PFAS (Wanninayake, 2021). Detailed reviews by Ross et al. (Ross et al., 2018) and Wanninayake, (Wanninayake, 2021) have highlighted existing treatment technologies for PFAS, discussing their respective advantages and limitations. These technologies include incineration, flocculation, sedimentation, and various sorption techniques for separation (Lei et al., 2023), as well as advanced oxidation and reduction, biological remediation, and ultrasonication techniques for PFAS destruction (Ross et al., 2018). However, PFAS are chemically and thermally stable, necessitating careful consideration, as their persistence and resistance to degradation can lead to the formation of harmful by-products during treatment and disposal. For instance, incineration does not assure the complete breakdown of most PFAS and, under normal combustion conditions, can actually generate dangerous byproducts such as fluorinated dioxins, furans, and halogenated compounds (Stoiber et al., 2020). Likewise, WWTPs, which were not originally equipped to remove PFAS, commonly allow their partitioning into biosolids. The subsequent land application of these biosolids as fertilizers or soil conditioners re-introduces PFAS into the terrestrial environment (Schneider et al., 2017).

Conventional water treatment techniques of coagulation, sedimentation, and biological digestion are not very effective in combating PFAS (Supplementary Table S2). Although more advanced technologies, such as granular activated carbon (GAC), ion exchange resins, oxidation and membrane technologies (e.g., nanofiltration and reverse osmosis), can eliminate specific PFAS from water, they are plagued by limitations of breakthrough, membrane fouling, regeneration challenges, and ineffectiveness against short-chain PFAS (Cousins et al., 2020). Furthermore, these technologies are costly to install and operate, and can produce concentrated reject streams that create additional waste management issues. For instance, whilst PFAS removal using reverse osmosis yield high removal rates, this can pose economic and environmental challenges for disposing the concentrate (Ma et al., 2024). Similarly, oxidation methods also have high efficient rates of removal, they are burdened by high energy consumption and can lead to unwanted byproduct formation (Puttamreddy and Nippatlapalli, 2024). The gradual migration of PFAS in the vadose zone further complicates remediation, usually taking decades before natural attenuation or active remediation can show efficacy (Karimi Askarani et al., 2024).

6.2. Limitations of PFAS detection

Concurrent with the growing concern over PFAS contamination, the detection and characterization of these substances remain technically challenging. PFAS are typically present at very low concentrations in environmental matrices such as drinking water. As a result, in the United States, the Environmental Protection Agency (EPA) has set maximum contaminant levels of 4–10 ng L⁻¹ for six PFAS compounds (Cotruvo et al., 2023). In the United Kingdom, the DWI has provided guidance for England and Wales, stipulating that the combined concentrations of 48 PFAS compounds in drinking water should not exceed 100 ng L⁻¹ (Drinking Water Inspectorate (DWI), 2025). Similarly, the European Union's DWD sets a maximum allowable concentration of 0.5 µg L⁻¹ for total PFAS, and imposes a stricter limit of 100 ng L⁻¹ for the combined total of 20 specifically identified PFAS compounds (Directive (EU), 2020). While these legislative efforts primarily target the quality of drinking water, there are currently no established regulatory limits for PFAS discharges from WWTPs. Moreover, compliance with these stringent limits requires advanced analytical instrumentation, such as state-of-the-art LC-MS/MS systems, along with meticulous quality control measures. Yet, many laboratories face difficulties in minimizing background contamination and achieving consistent method validation (Teymoorian et al., 2023). Solid-phase extraction

(SPE), though widely employed for PFAS preconcentration, often fails to detect ultra-trace levels, particularly when non-targeted interferences are co-extracted (Han et al., 2021).

To overcome these analytical limitations, there is growing interest in adopting broader detection strategies. High-resolution mass spectrometry (HRMS) methods for screening both known and unknown PFAS have emerged as promising tools for identification, eliminating the dependence on commercial standards (Newton et al., 2018; Liu et al., 2019). Nonetheless, these approaches are resource-intensive, complicated by matrix interference, and require extensive data interpretation. Complementary techniques such as the total oxidizable precursor assay and total organic fluorine analysis offer more comprehensive assessments of PFAS and their precursors. Though, issues such as incomplete oxidation and interference from inorganic fluoride still remain problematic (Houtz and Sedlak, 2012; Ateia et al., 2023). Surrogate markers, including UV absorbance (at 254 nm) and total fluorescence, have also shown some utility in tracking PFAS breakthrough in filtration systems, but their lack of compound specificity limits broad application (Aung et al., 2022). An innovative and potentially transformative approach involves using exhausted point-of-use (POU) filters as integrated passive samplers (Newton et al., 2018; He et al., 2022). This method allows for sensitive analysis of long-term cumulative PFAS exposure while reducing the need for frequent sampling, as long as the filters consistently retain PFAS and allow for efficient extraction (Newton et al., 2018; He et al., 2022).

7. Microbial enzymes involved in PFAS degradation

Biocatalysis, especially through the strategic use of enzymes, is now proving to be a highly promising, eco-friendly, and energy-efficient alternative to conventional chemical and physical degradation processes for environmental contaminants (Ye et al., 2025). The high stability of C–F bond makes the microbial breakdown of PFAS an exciting but still developing field of study. Nonetheless, it has been demonstrated that some microbial enzymes possess the capability to degrade specific PFAS compounds, especially their precursor forms.

7.1. Factors contributing to PFAS degradation

The biodegradation of PFAS is influenced by several factors, in addition to the availability of proficient enzymes. Important among these are the intrinsic chemical nature of PFAS molecules and physico-chemical conditions. The chemical composition of PFAS significantly affects their enzymatic biodegradation, with perfluorinated chain length as a determining factor. Longer chain lengths, with a higher number of fluorine atoms, tend to be more recalcitrant and have longer biological half-lives, making them less amenable to biodegradation (Marciesky et al., 2023). For instance, the levels of fluoride release from *n*:2 fluorotelomer alcohols (FTOHs) decreases as chain length increases (Li et al., 2018). Structural branching, particularly trifluoromethyl branches at the α/β -positions of unsaturated fluorinated carboxylic acids, increases anaerobic defluorination and overall degradability (Yu et al., 2022). Headgroup chemistry also plays an important role in degradability. Chemicals like 8:2 FTCA, 8:2 FTSA, and 8:2 FTUCA are more easily converted by enzymes (Merino et al., 2023; Dong et al., 2023). In general, lower fluorination levels correspond to higher biodegradability, as heavily fluorinated compounds resist transformation and release less fluoride during microbial degradation (Yu et al., 2022; Skinner et al., 2025).

Similar to plastic pollutants, the biodegradability of polymeric PFAS (fluoropolymers) can also be influenced by their crystallinity (Yashwanth et al., 2025). For instance, amorphous areas of polyethylene terephthalate are more susceptible to enzymatic degradation than crystalline areas, and pretreatment for enhanced amorphous content can enhance PFAS degradation rates (Tournier et al., 2020; Santhanam et al., 2024). Surface area is another important factor since high surface

area provides greater enzymatic contact sites, thereby accelerating degradation (Sun et al., 2024). Optimum temperatures also enhance microbial activity and enzyme function, enhancing the biodegradation rates of certain PFAS compounds (Zhang et al., 2022). pH variation also influences enzyme function. Farajollahi et al. (Farajollahi et al., 2024) demonstrated that DeHa2 from *Delftia acidovorans* showed decreased monofluoroacetate (MFA) defluorination activity when pH was lowered from 7 to 6, while DeHa4 remained highly active at pH 5, highlighting the importance of optimizing conditions for each specific enzyme (Farajollahi et al., 2024). Oxygen availability further promotes oxidative degradation pathways and tends to promote microbial PFAS defluorination (Zhang et al., 2022; Che et al., 2021). Certain PFAS architectures such as unsaturated fluorinated carboxylic acids with α/β -double bonds can undergo anaerobic microbial defluorination, demonstrating structural specificity required in PFAS bioremediation (Yu et al., 2022; Che et al., 2021). High salinity environments generally suppress microbial activity, but, halotolerant microorganisms and their enzymes well studied in plastic waste degradation, could be harnessed for PFAS degradation under saline conditions. Nutrient availability is critical for enzyme synthesis and microbial growth, yet an excess of alternative carbon sources may divert microbial metabolism away from PFAS degradation (Yu et al., 2022). Exposure to UV light can trigger photo-oxidative cleavage of PFAS, generating reactive radicals that break C–C bonds and yield shorter, more degradable perfluoroalkyl products (e.g. PFHpA, PFHxA), particularly when paired with catalysts like Fe^{3+} or persulfate (Lee et al., 2024). Oxidative enzymatic systems (e.g. horseradish peroxidase with H_2O_2 and phenolic mediators) have achieved significant degradation of PFOA ($\approx 68\%$ removal in 6 h), accompanied by reduction in toxicity (Colosi et al., 2009). Metal cofactors are essential for facilitating oxidative degradation and enabling partial defluorination of highly persistent fluorinated pollutants such as PFOA and PFOS (Khan, 2025). Moreover, engineered enzymes or enzyme systems must withstand co-contaminant effects (e.g. plasticizers or pro-oxidant additives), as complex matrices can alter enzyme stability and PFAS degradability, as seen in analogous plastic degradation studies (Aguilar et al., 2025). Finally, while high PFAS concentrations may enhance reaction rates, they can also generate toxic fluoride intermediates and free fluoride that impair microbial function unless efficient fluoride efflux mechanisms are present to mitigate toxicity, mirroring the trade-offs in enzyme-mediated biodefluorination (Yu et al., 2022).

7.2. Microbial enzymes involved in PFAS degradation

The C–F bond is one of the most stable in organic compounds, with a bond dissociation energy of around 485 kJ/mol, rendering them very resistant to enzymatic cleavage. No matter the mechanism of cleavage, enzymatic defluorination necessarily ends in fluoride ion release as a product (Etz and Shukla, 2023). Oxidative defluorination occurs through redox reactions that enable C–F bond cleavage via redox enzymes. Peroxidases like horseradish peroxidase (HRP) form reactive hydroxyl radicals ($\bullet\text{OH}$), that initiate PFAS degradation, but their effectiveness is usually constrained by the availability of co-substrates such as hydrogen peroxide (Marciesky et al., 2023). Lignin peroxidases and manganese peroxidases, which are often secreted by white rot fungi, are also being investigated for their potential to cleave high-molecular-weight PFAS into lower-molecular-weight, and potentially more biodegradable units (Aguilar et al., 2025). Laccases (multi-copper oxidases), employed in laccase-mediator systems (LMS), have also been targeted for PFAS removal owing to their broad substrate range for the oxidation of recalcitrant organics (Aguilar et al., 2025). Furthermore, hydrolytic defluorination pathways involve hydrolytic dehalogenases, which break C–F bonds by hydrolysis (Wackett and Robinson, 2024). These enzymes belong to two major superfamilies: the α/β hydrolases (ABHs) and the haloacid dehalogenase-like hydrolases (HADs) (Kurihara and Esaki, 2008; Chan et al., 2010). The large ABH

superfamily encompasses a diverse array of enzymes, including lipases, proteases, carboxylesterases, haloalkane dehalogenases, and fluoroacetate dehalogenases (Holmquist, 2000; Ollis and Carr, 2009). Most characterized HAD-like hydrolases catalyze phosphoryl hydrolysis or transfer reactions, with several enzymes known to hydrolyze chlorinated, brominated, iodinated, and fluorinated 2-haloacids (Koonin and Tatusov, 1994; Kurihara et al., 2000). Defluorinating activity against MFA has been demonstrated in purified fluoroacetate dehalogenases FAc-DexFA1 from *Burkholderia* sp. FA1 and DehH1 from *Moraxella* sp. strain B (now *Delftia acidovorans*) (Kurihara et al., 2003; Liu et al., 1998). These enzymes are more active against MFA compared to chloroacetate, producing fluoride and glycolic acid as final products (Fig. 4). Later, screening of purified ABHs and HADs from various bacteria against MFA as a substrate identified the presence of MFA hydrolytic activity in other uncharacterized members of these superfamilies (Chan et al., 2010; Khusnutdinova et al., 2023). The crystal structures of the fluoroacetate dehalogenases FAcD from *Burkholderia* sp. FA1 and RPA1163 from *Rhodopseudomonas palustris* revealed their active sites and structural snapshots along the defluorination reaction (Jitsumori et al., 2009; Chan et al., 2011). Further studies with RPA1163 demonstrated that this enzyme can also accept bulkier substrates, including 2-fluoro-2-phenylacetic acid and 2-fluoro-2-phenylpropionic acid (Zhang et al., 2020; Wang et al., 2017). Recent research involving eight purified dehalogenases (four ABHs and four HADs) with reported MFA defluorination activity identified significant hydrolytic activity against difluoroacetate (DFA) in three ABH proteins: RPA1163, DAR3835 from *Dechloromonas aromatica*, and NOS0089 from *Nostoc* sp. (Khusnutdinova et al., 2023). LC-MS analysis of the reaction mixtures identified glyoxylic acid as the final organic product of DFA defluorination, indicating the occurrence of two sequential defluorination reactions (Fig. 4). In addition, purified haloacid and fluoroacetate dehalogenases DeHa2 and DeHa4 from *Delftia acidovorans* D4B have shown the capacity to defluorinate MFA and DFA, but not TFA and PFOA (Farajollahi et al., 2024).

Several microbial taxa have been identified as capable of degrading various PFAS compounds. Among the most notable examples is *Acidimicrobium* sp. strain A6, an autotrophic bacterium isolated from acid mine drainage, which has shown the capacity to anaerobically degrade both PFOA and PFOS by utilizing ferric iron [Fe(III)] as a terminal electron acceptor in a process known as Feammox (Huang et al., 2022). In laboratory experiments, this strain achieved over 60 % degradation of PFOA and PFOS in wastewater treatment plant sludge, generating fluoride ions and short-chain perfluorinated carboxylic acids such as PFBA, PFPeA, PFHxA, and PFHpA (Huang et al., 2022). To overcome the

reliance on iron as an electron acceptor, a recent innovation involved integrating *Acidimicrobiaceae* sp. strain A6 into microbial electrolysis cells (MECs), replacing Fe(III) with electrodes, which resulted in an even higher degradation efficiency of 77 % for PFOA (Ruiz-Urigüen et al., 2022). Furthermore, structure-reactivity studies have revealed that specific structural features, particularly α,β -unsaturation in fluorinated carboxylic acids are critical for microbial defluorination. Anaerobic enrichment cultures dominated by *Dehalococcoides* were able to achieve > 90 % transformation and up to 10 % defluorination of unsaturated short-chain PFAS such as PFMeUPA and FTMeUPA, whereas structurally similar saturated analogs like PFBA and GenX remained recalcitrant (Yu et al., 2020; Yu et al., 2022). These studies support the notion that introducing double bonds or trifluoromethyl branches at key positions may significantly enhance the biodegradability of PFAS, providing a molecular design strategy for developing environmentally degradable alternatives.

8. Future directions and research needs

The persistent and bioaccumulative nature of PFAS presents an enormous challenge in environmental remediation. This issue can be addressed by embracing a multi-stakeholder strategy that integrates innovative biotechnologies, catalytic upgrades, and policy interventions (Fig. 5).

• Engineering fluoride-resistant microbes

Among the top research needs is the design of microbial strains that can tolerate highly unnatural levels of fluoride. This approach involves the discovery or design of enzymes with strong fluoride-detoxifying activity. Designing effective fluoride exporters could actively remove fluoride ions from cells, thus preventing enzyme inhibition and cell toxicity.

• Enzyme discovery and protein engineering for improved catalytic efficiency and substrate scope

Future studies should also aim to enhance the catalytic activity and broaden the substrate range of enzymes with C-F cleavage capabilities. This could involve enzyme discovery, rational design and directed evolution of new defluorinases that can degrade a broader spectrum of PFAS at significantly higher rates (Marciesky et al., 2023).

• Development of synergistic multi-enzyme PFAS biodegradation systems

Considering the structural diversity and complexity of PFAS, it is

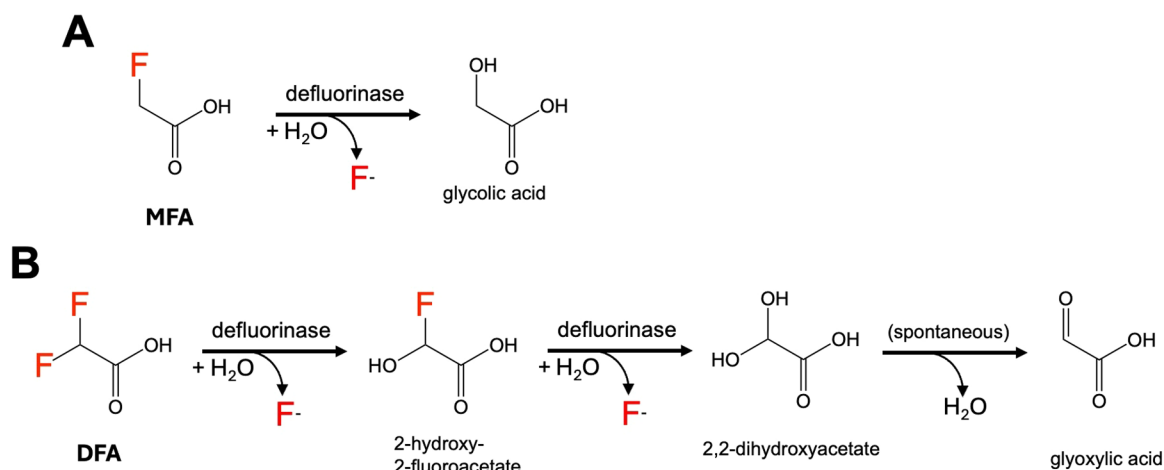


Fig. 4. Enzymatic defluorination pathways of monofluoroacetate (MFA) and difluoroacetate (DFA). (A) MFA undergoes a single enzymatic defluorination reaction catalyzed by defluorinase in the presence of water. (B) DFA undergoes two sequential enzymatic defluorination steps. The first step produces 2-hydroxy-2-fluoroacetate with the release of one fluoride ion (F^-). In the second step, another defluorinase-mediated reaction converts this intermediate into 2,2-dihydroxyacetate with the release of a second fluoride ion.

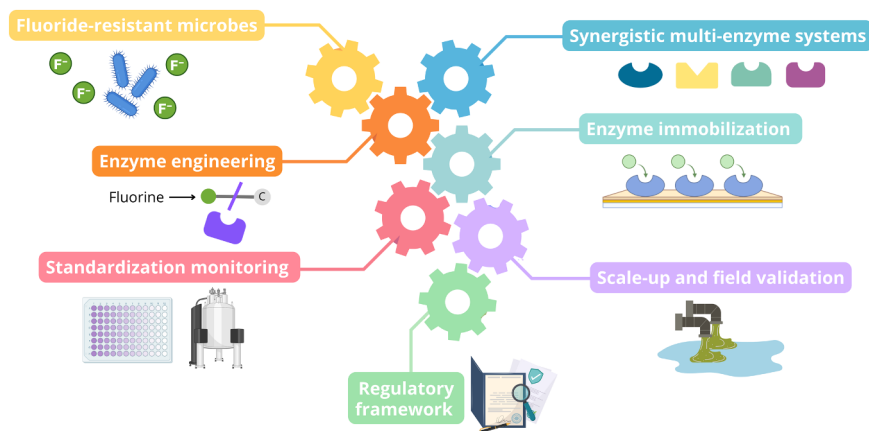


Fig. 5. Future directions for PFAS remediation and management.

unlikely that a single enzyme will be able to perform complete mineralization. Hence, research will turn towards multi-enzyme systems, in which different enzymatic reactions synergize *in vitro* and/or *in vivo* to ensure overall PFAS degradation.

• Enzyme immobilization and delivery for PFAS degradation

Enzyme immobilization and encapsulation technologies can improve enzyme stability, reusability, and usability in environmental conditions. For example, granular activated carbon immobilized laccase has exhibited good laboratory results (Mekureyaw et al., 2025). Mycoremediation methods, especially fungal systems combined with organic substrates, such as waste wood chips, are also being examined to utilize low-cost PFAS remediation in landfill leachate (Aguilar et al., 2025; Ayers and Zhang, 2025). Yet, vigorous analytical models need to be developed to fully describe all biotransformation products in order to guarantee complete mineralization and avoid the production of toxic intermediates.

• Standardization of monitoring and assay techniques

There is an urgent requirement for standardized, high-throughput, and real-time monitoring equipment to measure defluorination. The invention of new colorimetric assays and sophisticated methods like nuclear magnetic resonance (NMR) spectroscopy will greatly enhance research, allow process optimization, and make field-scale monitoring possible (Kumar et al., 2025).

• Scale-up and field validation

Scaling up from bench-scale experiments to pilot-scale demonstration projects is a critical step towards the development of enzymatic PFAS remediation technologies. Field-scale validation will give insight into the effectiveness and real-world practicalities of such technologies under natural environmental conditions.

• Regulatory framework and policy support

The establishment of standardized protocols, comprehensive toxicological databases, and strict control measures will be essential to constrain PFAS release and exposure. Such regulatory frameworks will also help facilitate the adoption and safe deployment of novel biodegradation technologies.

9. Conclusion

PFAS pollutants form a persistent and widespread environmental issue due to their remarkable chemical stability, widespread industrial and consumer uses, and resistance to traditional treatment processes. Despite the notable advances in high-resolution analytical detection techniques, including LC-MS/MS and HRMS, as well as in remediation technologies such as adsorption, ion exchange, and advanced oxidation processes, significant challenges remain in the precise monitoring of PFAS distribution, transformation products, and bioaccumulation

pathways. The efficient removal of both long- and short-chain PFAS compounds from complex environmental and drinking water matrices still remains a major challenge. Emerging biological treatment technologies, specifically enzymatic degradation mediated by oxidative enzymes (e.g., laccases, peroxidases) and microbial degradation employing PFAS-degrading bacteria or consortia, offer environmentally sustainable and energy-efficient alternatives to conventional methods. However, these technologies remain at an early stage of development and require extensive optimization in terms of enzyme specificity, microbial adaptation, and scalability for field applications. Therefore, to achieve effective and long-term PFAS management, it is quintessential to further refine analytical methodologies such as non-targeted screening, isotopic labelling, and real-time monitoring, and to combine them with robust and enforceable regulatory frameworks. This integration will enhance the understanding of PFAS fate and transport, support the development of reliable remediation strategies, and ultimately ensure the protection of environmental and public health.

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CRediT authorship contribution statement

Aman Thakur: Conceptualization; Data curation; Validation; Writing – original draft; **Anna N. Khusnutdinova:** Writing – original draft; Writing – review and editing; **Mireya Navarro-Márquez:** Visualization; Writing – review and editing; **Vinod Kumar:** Writing – original draft; Writing – review and editing; **Hairong Ma:** Writing – original draft; **Peter N. Golyshin:** Writing – original draft; **Tao Lyu:** Writing – original draft; **Alexander F. Yakunin:** Writing – original draft; **Gopalakrishnan Kumar:** Writing – review and editing; **K. Nithya:** Writing – original draft; **Guadalupe Gutiérrez Soto:** Writing – original draft; **Hafiz M.N. Iqbal:** Writing – review and editing; **Tony Gutierrez:** Conceptualization; Writing – original draft; Writing – review and editing; **Frederic Coulon:** Conceptualization; Project administration; Supervision; Validation; Writing – original draft; Writing – review and editing; **Roberto Parra-Saldívar:** Conceptualization; Project administration; Supervision; Validation; Writing – original draft; Writing – review and editing

Declaration of Competing Interest

On behalf of all the authors, I am writing to declare that we have no known conflicting financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

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Data availability

No new data were created or analysed during this study. Data sharing is not applicable to this article.

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