


 Cite this: *RSC Adv.*, 2025, 15, 50963

Sustainable remediation strategies for emerging PFAS contaminants in water

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At present, the decline in drinking water quality may pose risks to human health. Various types of organic as well as inorganic contaminants in drinking water are reported globally. Among emerging organic contaminants, poly- and perfluoroalkyl substances (PFAS) have been identified in water as major concerns for humans due to their toxic health effects. Anthropogenic activities are considered the dominant source of PFAS addition in the hydrosphere. PFAS toxicities are largely chronic in nature, as they may be carcinogenic and can lead to central nervous system disorders and reproductive problems. This review provides a critical discussion of the different remediation techniques for PFAS, such as chemical redox reaction, adsorption, electrochemical treatment, bioremediation, membrane filtration, and the treatment-train approach. Despite these techniques, extensive future research is required to develop better techniques for PFAS remediation.

 Received 30th July 2025
 Accepted 27th November 2025

DOI: 10.1039/d5ra05521k

rsc.li/rsc-advances

1. Introduction

Since the 1950s, per- and polyfluoroalkyl substances (PFAS) have been employed as synthetic materials in industry throughout the world. They are persistent in nature, and therefore have numerous applications in industrial as well as consumer settings as surfactants and external protectors, and are being extensively used throughout the world.^{1–4} According to Buck *et al.*⁵ highly fluorinated aliphatic substances are a subset of fluorinated substances. These substances may contain one or more C atoms in which all H substituents of the corresponding non-fluorinated analogues are replaced by F atoms, resulting in the perfluoroalkyl moiety C_nF_{2n+1}. As these substances possess higher stability as well as hydrophilic and lipophilic characteristics, they have numerous industrial applications, such as in fire-resistant fabric, non-stick cookware, waterproofing, food packaging, and firefighting foams.^{6–8}

Owing to their stability, these substances are not readily degraded and can accumulate in the atmosphere, leading to

their global distribution. Nearly every individual in the world has been in contact with PFAS and has traces of PFAS in their blood.^{9,10} Some of these substances enter the ecosystem and food chain, where they become bioaccumulated in the organs of both humans and animals. Due to the persistent and bioaccumulative nature of perfluorooctanesulfonic acid (PFOS), its salts, and perfluorooctane sulfonyl fluoride (PFOSF), these substances were added to the persistent organic pollutants (POPs) list under the Stockholm Convention on Persistent Organic Pollutants in 2009. In addition, C₈–C₁₄ perfluoroalkyl carboxylates (PFCAs) and perfluorobutane sulfonate (PFH₂S), and their sodium and ammonium salts have been included in the candidate listing of Substances of Very High Concern (SVHC) in the EU. Meanwhile, the production of AFFFs (aqueous film-forming foams) consisting of long-chain PFAAs is banned in Australia.^{11–14}

The regular use of AFFFs is a crucial legacy problem. AFFF has been broadly applied for fire protection personnel, particularly where petroleum hydrocarbons are stored and used, at airfields and in the petroleum industry.^{15–17} During regular training of firefighters and testing of fire protection systems, large amounts of AFFF have been released into the environment, and it has been observed at many sites, such as air force bases, metropolitan and rural fire training grounds, and in the petroleum industry at refineries and oil terminals.

The ingredients of PFAS in the foam are persistent in nature and can be adsorbed in soil and groundwater, ultimately moving several kilometers away from the source.^{18,19} There are some sites where many PFAS are present at higher concentrations in groundwater as well as surface water.²⁰ As fluoro-alkyl compounds are stable under usual atmospheric conditions,

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they persist in the environment for several years once released.²¹ Long-term exposure to PFAS in an individual leads to their accumulation in organs, resulting in severe toxicity.^{22,23}

The term PFAS represents a broad group that contains about 4700 compounds, and their characteristics correspond to the multiple fluorine atoms attached to alkyl chains of variable length. They are categorized as short-chain and long-chain PFAS. Short-chain PFAS, the most persistent of the two groups, are highly stable. Long-chain PFAS are also stable but can degrade into short-chain PFAS.^{24–26} Initially, PFAS were believed to be nonreactive and non-toxic, and little attention was paid to their environmental fate or their effects on the health of humans as well as the ecosystem. However, PFAS have become a global concern owing to their omnipresence, high stability, and increasing reports of toxicity in both animals and humans. PFAS comprise thousands of compounds, with perfluorooctanoic acid (PFOA) and perfluoro-octane sulfonic acid (PFOS) being the most extensively studied.^{26,27}

PFAS are a group of organofluorine substances with a hydrophilic functional group (carboxylic acid or sulfonic acid) head along with a hydrophobic fluorinated carbon chain. The main families of PFAS comprise perfluoroalkyl sulfonic acids (PFSA), perfluoroalkyl carboxylic acids (PFCA), and the potential PFCA and PFSA precursor compounds. PFCA has a perfluorinated carbon chain, typically containing up to 16 carbon atoms, along with functional groups such as carboxylic acids. PFSA is a perfluorinated compound with a sulfonic acid functional group. Table 1 compiles the chemical structures of PFAS that are commonly used worldwide.

Since the 1960s, PFAS have been used globally in a broad range of products due to their water- and oil-repellent properties. They are generally employed as stain protectors for products in the textile, upholstery and carpet industries.^{29,30} Additionally, they are used as specific surfactants in the fluoropolymer industry. In addition to this, they have been used in the production of important firefighting foams. Moreover, PFAS are also used to make precise extinguishing materials as well as to store large amounts of flammable chemicals.^{12–31} Due to their widespread use, traces of PFAS have been detected in industry facilities, living organisms, wastewater, commercial household products and food items.^{32,33} The exposure of humans to PFAS (at small concentrations) may result in accumulation in the blood of individuals.³⁴ In this review, the authors discuss the detrimental effect of PFAS contamination in water on human health. Furthermore, diagnosis and remediation techniques are also extensively reviewed.

2. Transport of PFAS in an aqueous environment

Lower PFAS concentrations have been found in aqueous solutions between pg L^{-1} and $\mu\text{g L}^{-1}$ levels.^{35,36} Long-chain compounds, such as perfluorooctanoic acid (PFOA), as well as perfluorooctane sulfonic acid (PFOS), are the widely assessed PFAS.³⁷ Numerous studies have confirmed the large presence of PFAS in aqueous environments (river water, storm water, lake

water, groundwater, wastewater, and drinking water).^{38–42} A portion of PFAS remains in wastewater treatment plants and is subsequently released into the receiving environment, contributing to their presence in aqueous systems. A recent study suggests that the potential impact of PFAS on groundwater is linked to the use of treated wastewater for irrigation.⁴³ Insufficient data are available for the removal efficiency of conventional water treatment plants and the transformation of PFAS precursor compounds; therefore, their occurrence and distribution are only partially understood.^{38,42} Applications of PFOS and PFOA have decreased because of their detrimental effects on the health of the public, but there has been no decrease in the total quantity of PFAS released into the environment, due to the replacement of long-chain compounds by short and ultra-short PFAS.⁴¹ Short and ultra-short PFAS have lower bioaccumulation as well as bioconcentration *via* trophic levels in contrast to long-chain species; nevertheless, they are environmentally persistent.^{44,45}

3. Sources of PFAS

Recently, approximately 80% of electronic waste (e-waste) has been disposed of directly in landfills or indirectly ended up there due to inadequate recycling practices.⁴⁶ Regardless of the alarming consequences, e-waste substances as sources of PFAS in the environment around processing sites and landfills are continuously overlooked.^{47–49} Because of limitations in analytical methods for detecting PFAS, relatively low concentrations in mixed substances, a lack of specific regulatory policy measures, non-specific e-waste processing, and the high costs for treating e-waste, PFAS compounds are often not properly detected. Fig. 1 shows the major sources of PFAS in aqueous environments along with their transport routes.

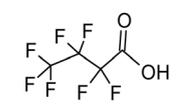
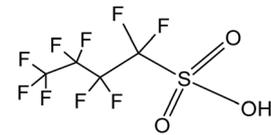
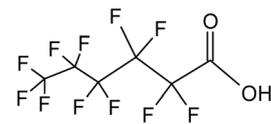
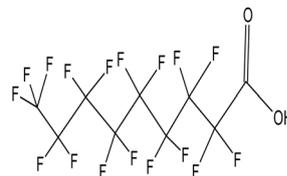
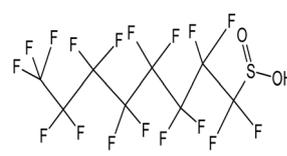
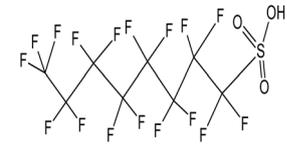
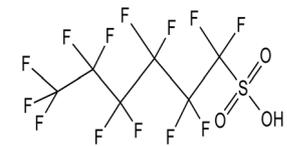
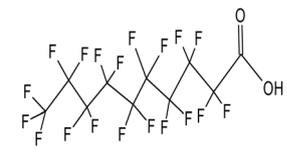
E-waste leachates have been reported as sources of hazardous waste because of the occurrence of alkanes, halogenated plasticizers, heavy metals, and dioxins; nevertheless, the occurrence of PFAS is overlooked.^{50,51} The leachate generated by treatment at landfills is either treated at on-site treatment plants or transported to wastewater treatment plants, and none of the treatment trains put in place are adequate to remediate or destroy PFAS compounds. Therefore, e-waste treatment plants, landfills, and wastewater-treatment plants are considered the point sources of PFAS pollution in an aqueous environment.^{52,53} Additionally, PFAS manufacturing and processing facilities, airports, and military installations that use firefighting foams are some of the main sources of PFAS.^{54,55}

4. Harmful effects of PFAS

Proliferated peroxisome expression is the primary mechanism by which PFAS are known to exhibit their toxicity. Mammals, such as rodents, have been used in experiments to try and understand the harmful consequences of PFAS. However, because these organisms differ in how peroxisome proliferation is expressed, it is not fully acceptable to extrapolate the findings of this research to humans.⁵⁶ A possible correlation has been suggested by numerous case studies examining the negative



Table 1 Structure of some important PFAS (modified from Ahrens *et al.*²⁸)

Name	General formula	IUPAC name	Chemical structure
Perfluorobutanoic acid (PFBA)	C ₄ HF ₇ O ₂	Heptafluorobutanoic acid hydrate (1 : 1)	
Perfluorobutanesulfonic acid (PFBS)	C ₄ HF ₉ O ₃ S	1,1,2,2,3,3,4,4,4-Nonafluorobutane-1-sulfonic acid	
Perfluorohexanoic acid (PFHxA)	C ₆ HF ₁₁ O ₂	Undecafluorohexanoic acid	
Perfluorononanoic acid (PFNA)	C ₉ HF ₁₇ O	Heptadecafluorononanoic acid	
Perfluorooctanoic acid (PFOA)	C ₈ HF ₁₅ O ₂	1,1,2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-Heptadecafluorooctane-1-sulfonic acid	
Perfluorooctanesulfonic acid (PFOS)	C ₈ HF ₁₇ O ₃ S	1,1,2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-Heptadecafluorooctane-1-sulfonic acid	
Perfluorohexanesulfonic acid (PFHxS)	C ₆ HF ₁₃ O ₃ S	1,1,2,2,3,3,4,4,5,5,6,6,6-Tridecafluorohexane-1-sulfonic acid	
Perfluorodecanoic acid (PFDA)	C ₁₀ HF ₁₉ O ₂	Nonadecafluorodecanoic acid	

effects of PFAS exposure in humans, between PFAS compound exposure and the emergence of diseases such as hypertension, cancer, thyroid disease, ulcerative colitis, *etc.*^{57,58} Children are more vulnerable to PFAS exposure than adults, according to a study.⁵⁹ This may be related to children's faster metabolisms and constantly changing and developing physiological systems. Epidemiologic data linking prenatal and/or childhood exposure to polyfluoroalkylamides with immunity, infection, asthma, cardio-metabolic problems, changes in neurodevelopment, thyroid issues, renal issues, and puberty onset were recently reviewed.⁵⁹ Food and drinking water are the primary pathways

that expose human beings to PFAS, among several other routes.⁶⁰ A group of Chinese women's plasma PFAS concentrations were positively correlated with fish and crustacean consumption; however, their plasma PFAS concentrations were negatively correlated with the consumption of drinking water and soy products.⁶¹

4.1 Cancer

PFAA carcinogenic effects have mostly been investigated in human subjects. Communities consuming PFAS-contaminated water and chemical industry personnel exposed to PFOA and



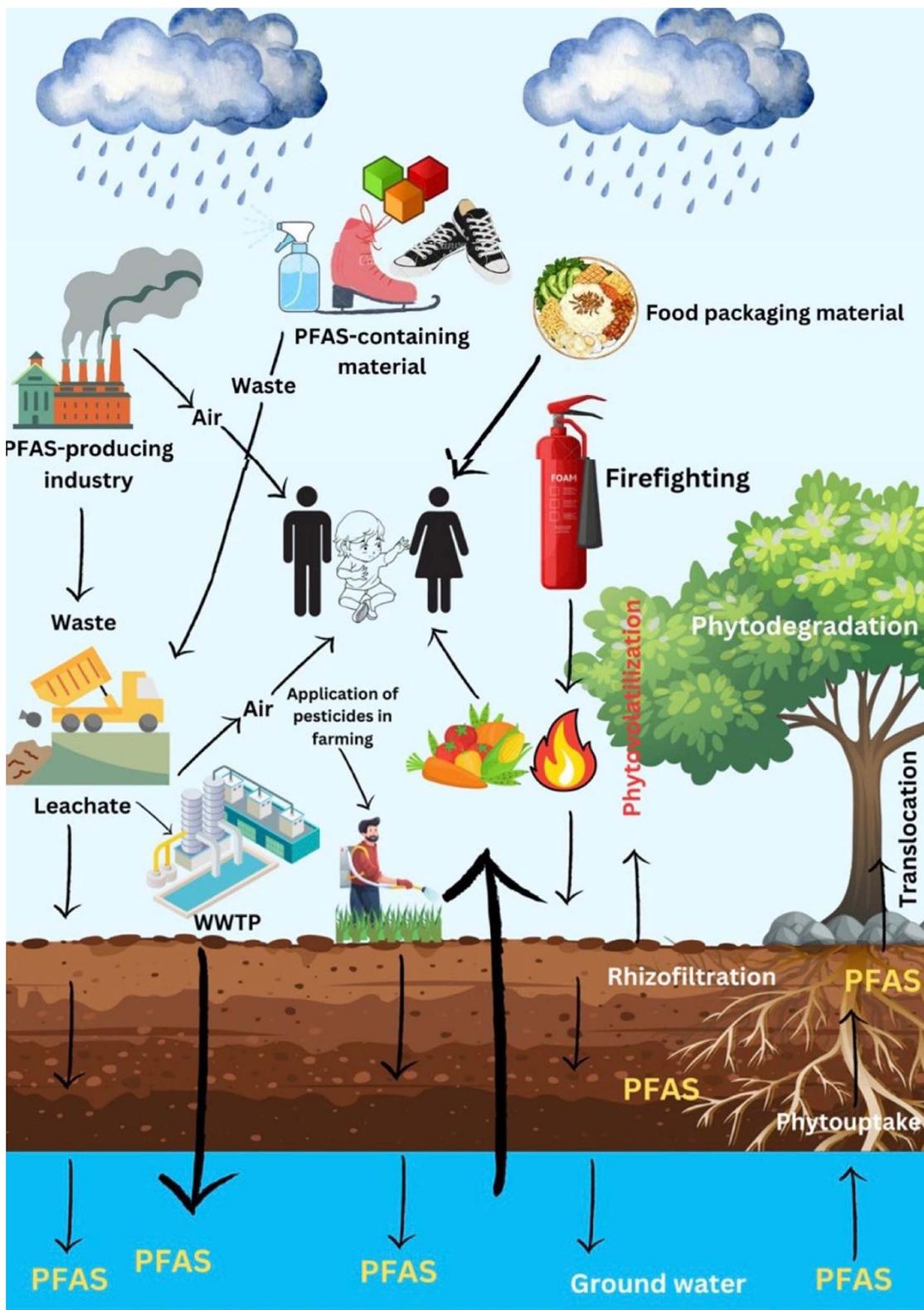


Fig. 1 Sources and routes of PFAS in the environment.

PFOS were the primary subjects of this investigation. According to the International Agency for Research on Cancer (IARC), PFOA may be carcinogenic to humans. Male employees at PFOA-producing plants had a 3.3-fold increase in prostate

cancer mortality during the past ten years.⁶² A follow-up investigation using a group for occupational exposure failed to find any evidence of a significant correlation between cancer mortality and ammonium perfluorooctanoate exposure; among



exposed workers, no cases of kidney, prostate, or breast cancer were noted.⁶³ Two studies that focused on towns in the Mid-Ohio Valley and West Virginia located near chemical plants examined PFOA-contaminated drinking water, and found a link between PFOA exposure and cancer.^{57,58} Two studies that focused on communities that had access to PFOA-contaminated drinking water due to their proximity to chemical facilities in the Mid-Ohio Valley and West Virginia reported a link between PFOA exposure and cancer. The research examines PFOA serum levels and medical history in a cohort of adult C8 Health Project participants, revealing a strong correlation between kidney and testicular cancer and serum PFOA concentrations.^{57,58} To assess the cancer risk associated with exposure to PFOA and PFOS, a case study including the general Danish population was conducted.⁶⁴ The plasma concentrations of participants of both chemicals were evaluated, and the report revealed no correlation between the plasma concentrations and the risk of hepatic, pancreatic, bladder, or prostate cancer. Strong evidence of the carcinogenic effects of exposure to PFOA and PFOS is currently lacking and weak, despite research on the link between PFAS and cancer.

4.2 Immune system disorders

PFAS, in particular PFOS and PFOA, have been the subject of significant research on the immunotoxic effects on animals. It has been observed that these compounds modify both innate and adaptive immunity while also causing variations in inflammatory responses and cytokine production. Previous research on animals has demonstrated the immunotoxic effects of PFOA and PFOS,⁶⁵ prompting subsequent studies to investigate their impact on immunity-related disorders in humans.⁶⁶ Reports show the relationship between immunosuppression and PFAS exposure; the results are evaluated at the molecular and organ/system levels.⁶⁷ A two-fold increase in PFAS concentration was linked to a -49% (95% confidence interval [CI], -67% to -23%) decrease in serum antibody concentration for tetanus and diphtheria during the study of the impact of PFAS exposure on immunity in children.⁶⁸ The extent to which PFAS exposure affected the humoral immune response of the participating children may have varied depending on exposure levels, vaccine strains, and the time elapsed since immunization. Grandjean *et al.*⁶⁸ conducted a study on adolescents aged 7 to 13 years, which revealed a decrease in diphtheria antibody concentration in response to increased serum concentration of PFAS. The results indicated that doubling PFAS exposure at 7 years was linked to a decrease in diphtheria antibody level at 13 years.

4.3 Metabolic disorders

Studies on PFAS exposure and its effects on metabolism were conducted on a range of population groups, including adults, adolescents, children, pregnant women, and newborns. Clinical trials involving workers and individuals with obesity and diabetes have also been supported. Gilliland and Mandel⁶² conducted one of the earliest studies in this regard, examining the relationship between PFOA serum levels and hepatic enzymes, lipoproteins, and cholesterol in workers exposed in the

workplace. The predicted PFOA levels did not appear to be related to any major liver damage, even though the investigators measured the exposure using total serum fluorine. Studies on PFAS exposure and its effects on metabolism have been conducted on a range of population groups, including adults, adolescents, children, pregnant women, and newborns. Serum PFAS were positively correlated with glucose homeostasis in a cohort of adults and adolescents participating in the National Health and Nutrition Examination Survey (NHANES), indicating metabolic syndrome. The concentration of serum perfluorononanoic acid (PFNA) is primarily correlated with serum HDL-C and hyperglycemia (odds ratio 3.16). PFAS exposure has also been associated with increased blood insulin levels and β -cell activity, suggesting effects on glucose metabolism.⁶⁹ It was believed that the continuous exposure to the PFAS lead to the variation in the impact spectrum. The effects of PFOS and PFOA on humans have been extensively studied, and the literature reports the most persistent or long-term effects. In a community-based study of children in the C8 health project, increases in serum PFOA and PFOS between the first and fifth quintiles were linked to increases in total cholesterol of 4.6 mg dL⁻¹ and 8.5 mg dL⁻¹ and low-density lipoprotein cholesterol of 3.8 mg dL⁻¹ and 5.8 mg dL⁻¹.⁷⁰ Similarly, among patients from NHANES 2003–2004, total cholesterol levels 9.8 mg dL⁻¹ and 13.4 mg dL⁻¹ were linked to an increase in PFOA and PFOS levels in serum from the lowest to highest quartiles.⁷¹ Prenatal exposure is found to have effects on both mother and child later in life, according to studies conducted among a small cohort of pregnant women in the US.⁷²

The chemical properties of different PFAS influence their biological effects, such as long half-lives in humans and other species, and their environmental persistence, which can be effectively irreversible. Adipose tissue could not support the bioaccumulation of PFAS.⁷³ Certain PFAS have structures that are similar to those of natural fatty acids.⁷⁴ According to Pérez *et al.*,⁷³ PFAS mostly accumulate in organs such as the brain, lungs, liver, serum, and kidney. The long-term exposure to human beings would increase the risk of cancer and has long-term impacts on immunological, endocrine, metabolic, and reproductive systems.⁷⁵ Research is being carried out to determine the significance of the data supporting all these interaction modifications through both outcomes and specific PFAS.⁷⁶

PFAS exposure in pregnant women may lead to reproductive or pregnancy-related complications, such as birth of underweight infants,^{77–80} high blood pressure,⁸¹ preeclampsia,⁸² increased time to pregnancy,^{83,84} and gestational diabetes.⁸⁵

According to a thorough assessment of the literature, 69 epidemiologic studies examined the connection between PFAS exposure and a variety of metabolic problems, including obesity, cardiovascular disease, and lipid homeostasis. According to Sunderland *et al.*⁶⁷ there is a positive relationship between blood lipids, such as triglycerides and total cholesterol, and PFAS exposure. The adverse effects of PFAS exposure on hypertension, stroke, diabetes, vascular disease, and insulin resistance were also examined by the authors through an



analysis of epidemiologic research. Fig. 2 shows the negative health effects caused by PFAS ingestion *via* drinking water.

According to Boas *et al.*⁸⁶ PFAS impacts the thyroid through a variety of biological pathways related to thyroid homeostasis. These mechanisms include transport, metabolism, interference with thyroid receptors in target tissues, and thyroid hormone production. Possible mechanisms for thyroid disorder caused by exposure to perfluorocarbons (PFAS) include decreased circulating levels of thyroxine (T4) owing to the competitive binding of thyroid hormone transporter (THT) proteins,⁸⁷ elevated thyroid levels and hepatic T4 metabolism decreasing the thyroid fabrication of T4,⁸⁸ and decreased thyroid peroxidase (TPO) activity.⁸⁹ It was believed that the kidney primary affected organ by exposure of PFAS. This view was backed by epidemiologic data from an increasing number of individuals, animal studies, and *in vitro* models. Long-term kidney effects include kidney cancer, chronic kidney disease, and reduced kidney function over time.^{90,91}

5. Determination techniques for PFAS in water

5.1 Ion mobility spectrometry (IMS)

IMS is a newly developed separation method that makes ion distinctions in the gas phase according to their size, shape, and charge state.⁹² Whereas IMS separates ions based on variations in gas phase electrophoretic mobility, gas chromatography and

liquid chromatography separate analytes based on changes in boiling point and polarity.⁹³ After ionization, inert gas (also known as the buffer gas) is introduced into a designated mobility separation area, where IMS separations take place.⁹⁴ An applied electric field causes ions to move through this region, and the length of time it takes for them to migrate through the gas (known as the drift time) is directly related to the size of the ions. The ion's collision cross section (CCS, usually expressed in units of \AA^2) is a standard term used to characterize the measured size. Smaller ions migrate more quickly and encounter fewer collisions with the buffer gas in the IMS studies than bigger ions, whose migration is inhibited.⁹⁵ Because IMS separations take place in the gas phase, mobility analysis happens quickly, typically in less than 100 milliseconds per spectrum. Because of this, IMS may be easily integrated with current untargeted LC-MS/MS suspect screening techniques,⁹⁶⁻⁹⁸ which allow for the addition of a molecular descriptor and additional separation mechanism for analyte characterization without requiring more time for analysis.⁹⁹ Furthermore, across a range of molecular classes, including carbohydrates, lipids, peptides, carbohydrates, and environmental pollutants like polyaromatic hydrocarbons (PAHs) and polybrominated diphenyl ethers (PBDEs), prior IMS research has reported intrinsic connections between CCS and mass.¹⁰⁰⁻¹⁰² Based on the variations in gas phase packing efficiency (compactness) of the various chemical classes, distinct trend-lines linking CCS to m/z have been found in all of the

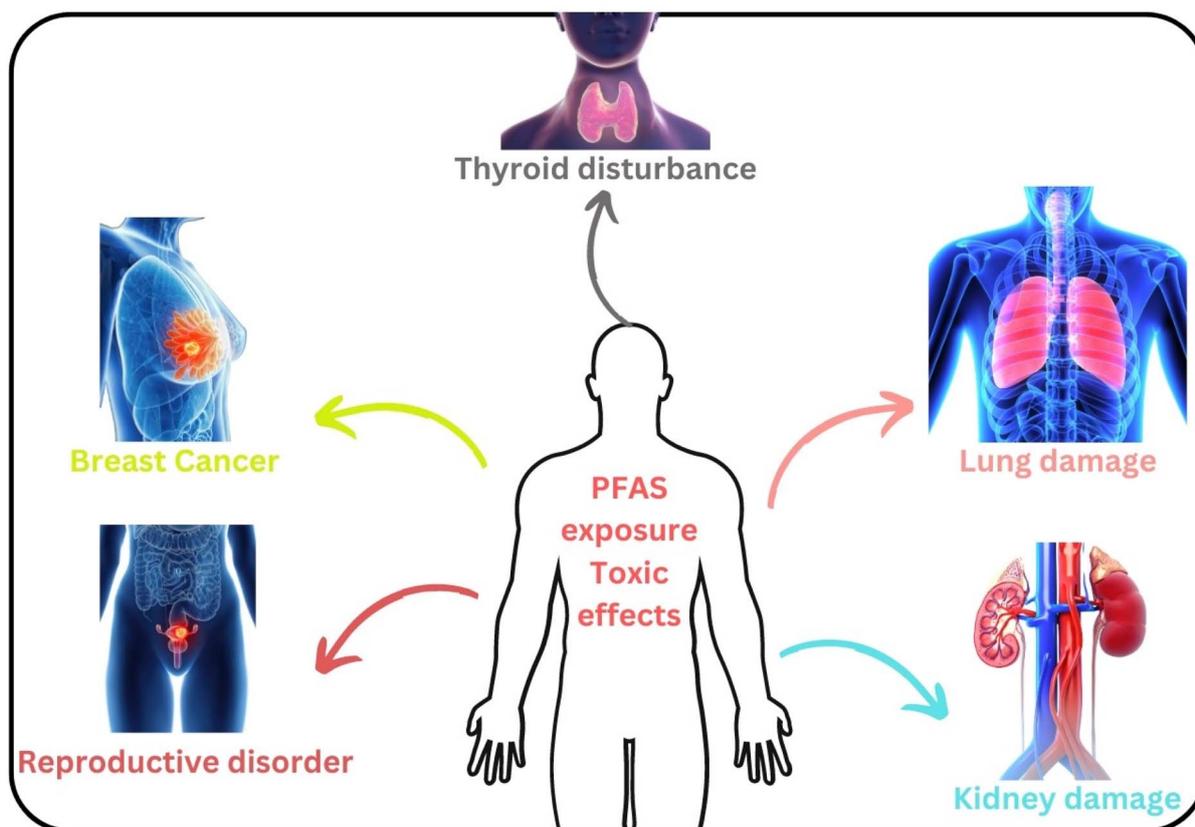


Fig. 2 Toxic effects caused by contamination by PFAS in aqueous solutions.



aforementioned situations. Hines *et al.*¹⁰³ reported that subclasses of vitamins, antibiotics, and other tiny biomolecules have unique mass-CCS trendlines. This information is helpful for untargeted research or natural product discovery, as it offers extra molecular details for unannotated properties.¹⁰⁴ IMS methodologies are increasing in popularity and are being used by several manufacturers for integrating IMS separations into untargeted MS processes owing to their precise selectivity and rapid actions.^{105–109}

Yukioka *et al.*¹¹⁰ carried out a study to determine an association between fragment ions and their molecule ions using the drift time obtained by ion mobility spectrometry. Standard solutions containing PFAS were examined using liquid chromatography/ion mobility quadrupole time-of-flight mass spectrometry (LC/IM-QTOF-MS) in order to validate the procedure. Even though co-eluting compounds were observed at similar retention times, all fragmentation flags for the PFAS were detected within a specified drift time range, and the molecular ion matched that of the PFAS in the spiked standard solution. Nine molecular ions were discovered when this technique was used on a domestic fire extinguisher liquid. As a result, the new linking technique used LC/IM-QTOF-MS to quickly search for potential molecular ions.

Luo *et al.*¹¹¹ investigated the application of coupled liquid chromatography, ion mobility spectrometry, and mass spectrometry (LC-IMS-MS) for the quick characterization of targeted as well as non-targeted PFAS in AFFF. Both the negative and positive ion modes of LC-IMS-MS were employed for the analysis of 10 formulations of AFFF from seven brands. Untargeted analysis of the formulations was followed by identification of PFAS-like features utilizing database matching, mass defect and homologous series evaluation, and MS/MS. The findings of this study revealed that 33 homologous series were identified in the tested AFFF formulation.

In contrast to data-independent acquisition, mass spectrometry has the potential to analyse target as well as non-target species. Nevertheless, it is taught to correlate the retention time with fragment ions of target precursor ions, due to the presence of co-eluting species in the environmental samples. As the conventional DIA approach is not sufficient to separate the ions, this technique, associated with ion mobility mass spectrometry, makes it easy to separate the target ion from the co-eluting ions by drift time. Initially, 32–96 PFASs were detected per sample during suspect screening using an internal database in groundwater samples impacted by firefighting foam ($n = 8$). It was observed that of all the pairs of respective precursor ions and fragment ions of the PFAS suspects, 5–19% (4–9 PFAS) of them were associated without considering the drift time of IMS, while 37–49% (15–43 PFAS) of them were associated with considering the drift time. The consideration of the drift time increased the association ratios in all samples.¹¹²

Mu *et al.*¹¹³ also investigated the coupling ability of ion mobility spectrometry (IMS) with HRMS and machine learning techniques to accomplish the rapid and precise target and non-target screening of PFAS in wastewater. It was seen that only a few interfering peaks and a clearer spectrum were observed compared to traditional HRMS. Approximately 63% of

misleadingly optimistic results were eliminated through the application of the collision cross section (CCS) technique in PFAS homologous series.

5.2 Liquid and gas chromatography

The analysis and determination of PFAS is frequently complicated due to inadequate information and a heterogeneous array of PFAS in the environment. Ultra-high performance liquid chromatography-quadrupole Orbitrap high-resolution mass spectrometry (UHPLC-Q-Orbitrap HRMS) was used to develop and validate a highly precise, accurate, and selective procedure for the qualitative and quantitative assessment of PFAS (24 perfluorocarbons and 30 precursors).¹¹⁴ PFAS detection limits range from 0.6 to 5.4 ng L⁻¹ (Shimadzu LC-MS-8060), with 84.13% recovery and a calibration range of 5–200 ng L⁻¹ attained for 94% of the compounds examined, which includes all those specified in ASTM D7979.¹¹⁵ With a total concentration ranging from 0.3 to 32.9 ng L⁻¹, 20 PFA, mostly PFBS, PFBA, PFOA, and PFPeA were successfully detected using the optimized approach in groundwater samples taken in China. Using 3-bromo-acetyl-7-methoxycoumarin (BrAMC), a fluorescence-based compound, HPLC with a UV detector was used to separate the long and short-chain PFAS in rodent liver samples.¹¹⁶ The application of LC-MS/MS can help to check the presence or absence of PFOA, PFOS, and PFHxS in water.¹¹⁷ Since these studies, a great deal of progress has been accomplished, including the introduction of various sample preparation methods, the application of various inner isotope guidelines (IS) for quantitative analysis, and tactics to increase accuracy while maintaining the best possible explanatory outcome. Although the collected samples are solid, they should all be dried, sieved, homogenized, extracted, and concentrated before being subjected to HPLC-MS analysis. It is necessary to employ a variety of pretreatment techniques to remove foundation interference and increase recovery. Bao *et al.* (2010)¹¹⁸ found eight PFAS in river sediments using HPLC-MS/MS with negative electrospray ionization and a 1:5 ratio of sodium carbonate (Na₂CO₃) to tetrabutylammonium hydrogen sulfate for sample extraction.

In a separate study, Naile *et al.*¹¹⁹ employed a method for recognizing multiple PFAS, and twelve PFAS were discovered in samples of soil and sediment collected from various sources. The material was homogenized, sonicated, and then HPLC-MS/MS analysis was carried out. However, because the solvents in some of the compounds were hydrophobic, the recoveries were only 20% or less. The detection of PFOS and PFOA in sewage sediment was carried out using a high sampling volume of methanol:acetic acid (9:1), followed by cleanup to extract the chemicals and analyze them using HPLC-MS/MS.¹³ The samples were taken from landfills using passive sampling methods with sorbent-impregnated polyurethane foam (SIP) discs.¹²⁰

According to Nakayama *et al.*¹²¹ GC-MS is primarily used to identify PFAS that are present in the air as volatile, semivolatile, neutral, and nonionic compounds. These compounds include perfluoroalkane sulfonamides, fluorotelomer alcohols (FTOHs), and perfluoroalkane sulfonamide ethanols.¹²² However,



compared to liquid chromatography, this technique's restricted selection for PFAS measurement stems from its reliance on the type of analytes, which limits its application. Martin *et al.*¹²³ used GC with chemical ionization mass spectrometry (GC/CIMS) for the analytical isolation and identification of PFOS precursors and fluorotelomer alcohols (FTOHs) present in ambient air. An analytical approach involving PFAS enrichment on glass-fiber filters (GFFs), polyurethane foams (PUFs), and XAD-2, followed by chromatographic measurement, was devised, optimized, and confirmed. Gas-chromatography electron impact mass spectrometry (GC-EIMS) has been used to analyze the concentration of perfluoroalkyl sulphonamides, such as *N*-methylperfluorooctanesulfonamidoethanol (MeFOSE), among others, to ascertain their sources, degree of occurrence, and partitioning in the air.¹²⁴ Alzaga *et al.*¹²⁵ established an analytical method for measuring perfluoro carboxylic acids (also known as PFC7-10A) in harbor sediments. Along with GC-MS-based determination, the process incorporated head-space solid-phase micro-extraction (SPME) and pressurized fluid extraction (PFE). PFCAs were found in the harbor sediments at low ppb quantities of 8 to 11 ng g⁻¹ after the extraction parameters were optimized to produce recoveries of more than 95%, a limit of detection of 0.5–0.8 ng g⁻¹, and relative standard deviations from 15.5% to 16.8%. Fluorotelomer olefins (FTOs), fluorotelomer acrylates (FTAs), fluorotelomer alcohols (FTOHs), sulfonamides, and sulfonamidoethanol were among the perfluorinated compounds that were found to be most common in a spatial survey conducted throughout Asia as part of another study that used GC-MS analysis. Solvent extraction coupled with gas chromatography-positive chemical ionization mass spectrometry (SE-GC-PCIMS), a technique, was utilized to extract and analyze three neutral hydrophobic metabolic precursors of PFOS found in fish, fast food, and liver samples from Arctic marine mammals. Over 90% recoveries were achieved using this strategy. For the assessment and characterization of PFAS, a brand-new and effective analytical technique based on head-space solid-phase micro-extraction (HS-SPME) in conjunction with gas chromatography-atmospheric pressure photoionization high-resolution mass spectrometry (GC-APPI-HRMS) was presented. According to Ayala-Cabrera *et al.*¹²⁶ the approach demonstrated efficiency even at low concentration levels of PFAS, with low detection limits (0.02–15.2 ng L⁻¹), strong repeatability (RSD% < 11), and trueness (RE% < 12).

5.3 Electrochemical sensors

To date, several types of electrochemical sensors have been used to measure impedance (Z) over a range of frequencies (Hz). An ion-selective membrane measures the potential (mV), voltammetric sensors measure the change in current (pA), impedimetric sensors measure changes in the impedance (Z) over a range of frequencies (Hz), and conductometric sensors measure changes in the conductance (G , Ω^{-1}), which take place due to electrochemical reaction originating from an applied voltage (mV). Usually, electrochemical sensors such as voltammetric as well as potentiometric sensors are used for analysing PFAS. However, before the application of these electrodes, the

surface should be functionalized so that they directly interrelate to the target analyte either by complexation or ion exchange. Molecularly imprinted polymers may be applied to surface functionalization, offering a polymeric matrix over the surface of the electrode with voids or detection sites, which correspond to the functional groups of the target analyte, shape, and size.¹²⁷

Chen *et al.*¹²⁸ investigated potentiometric detection of PFO⁻ and PFOS in water, even for low concentrations ($\sim 0.07 \mu\text{g L}^{-1}$), using ion-selective electrodes (ISEs), which were equipped with fluorine anion liquid exchange membranes. It was observed that the selectivity of ion-selective electrodes was affected by the occurrence of other perfluorinated compounds, which are only different in the number of carbon atoms. Karimian *et al.*¹²⁹ applied ferrocenecarboxylic acid (FcCOOH) to overcome the drawback (difficult detection of PFOS with poor electrical conductivity) and ferrocenecarboxylic acid (FcCOOH) works as an electroactive correspondent molecule contending with PFOS (non-electroactive) for molecularly imprinted polymers. The voltammetric signal is reduced, and the association between the concentration of PFOS and the signal is developed if PFOS is in water. The findings indicate that in this case, the voltammetric signal of the electroactive correspondent molecule (FcCOOH) is inversely proportional to the level of PFOS in the aqueous solution.

5.4 Biosensors

Currently, biosensors are predominantly utilized for the detection of PFAS in aqueous solutions. In this context, Zhang *et al.*¹³⁰ utilized electrochemical biosensors for the analysis of PFO that depends upon the inhibition of the biocatalysis process in enzymatic biofuel cells by the PFOS. Cennamo *et al.*¹³¹ introduced a new method for biosensors by developing a configuration that involved a platform functionalized with a bio-receptor. This biosensor is characterized by a surface plasmon resonance (SPR) platform based on plastic optical fibers (POFs), together with a bio-receptor for the detection of PFOA and PFOS.

6. Remediation techniques

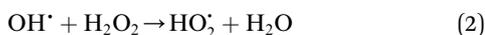
6.1 Chemical redox reactions

A DFT-based kinetic model was applied by Blotvogel *et al.*¹³² to investigate the reductive defluorination of perfluorooctanoic acid through nanoscale zero-valent iron (nZVI) and zinc, and it was observed that the first step reductive PFOA defluorination has half-lives of about 8 years (Zn⁰) and for Fe⁰ about 500 000 years at metal-to-water ratios usually employed in permeable reactive barriers. These authors summarized that for the removal of PFOA from the water, the utilization of dehalogenation by zero-valent metals is not an optimal technique if appropriate catalysts are recognized to increase the rate of defluorination. Yin *et al.*¹³³ applied activated persulfate under acidic conditions and achieved PFOA degradation at approximately 89.9%. The findings indicate that the production of SO₄^{•-} played a prime role in the degradation of PFOA. Eqn (1)–(4) represent the prime mechanism behind the degradation of the PFOA. Both processes of HF removal as well as



decarboxylation continue until the PFOA is changed into CO₂ and fluoride.

A specific type of CHP (catalyzed hydrogen peroxide propagation) called modified Fenton's reagent (MFR) has a long history of being used for ISCO of polluted soil and groundwater.¹³⁴ To start the propagation processes, MFR/CHP uses hydrogen peroxide and initiators like divalent minerals, soluble iron(II) or iron(III) chelates. A variety of reactive oxygen species and free radicals, such as the hydroxyl radical (OH[•]), perhydroxyl radical (HO₂[•]), superoxide radical anion (O₂^{•-}), and hydroperoxide anion (HO₂⁻), are produced by these catalyzed reactions in circumneutral pH systems. O₂^{•-} and HO₂⁻ are both reductants and weak and strong nucleophiles, respectively:



In CHP systems, significant concentrations of superoxide radical anion are only seen at pH ranges close to 4.8 and higher (pK_a in reaction (2)). To investigate the oxidative treatment of PFOA, Mitchell *et al.*¹³⁵ used CHP reactors with neutral pH and hydrogen peroxide concentrations ranging from 0.25 to 2 moles per liter (M). In comparison to PFOA loss of 15% in a control sample, PFOA concentrations (baseline 100 μg L⁻¹) decreased to 68, 85, and 89% after 150 minutes at starting H₂O₂ concentrations of 0.25, 0.5, and 1 M, respectively. These experiments showed that MFR/CHP can quickly decrease PFOA concentrations. Heat-activated persulfate oxidation of per- and poly-fluoroalkyl (PFOA) was further explored for groundwater cleanup by Park *et al.*¹³⁶ and Bruton and Sedlak.¹³⁷ For example, the kinetics of PFOA degradation increased five times at 50 °C for every five times increase in S₂O₈²⁻.

6.2 Membrane filtration

The developed membrane technology can be easily customized to specific needs, which makes it a good candidate for the treatment of aqueous solutions. This method is typically applied for the separation and determination of the level of PFAS.¹³⁸ This is a superior method for the remediation of PFAS (with very high removal efficiency <99%) that can discard highly loaded PFAS when the concentration of the PFAS is very high 1000 mg L⁻¹.¹³⁹⁻¹⁴⁵ Size exclusion is the predominant mechanism responsible for the elimination of the PFAS *via* the membrane process. Furthermore, electrostatic repulsion, diffusion, and cake layer filtration are the other mechanisms responsible for the removal of PFAS. Nanofiltration and reverse osmosis membranes are types of membranes that perform well in the rejection of PFAS. Nanofiltration is economical and more suitable for the remediation of PFAS as compared to reverse osmosis. NF270 nanofiltration can eliminate up to 99% of PFAS; however, some PFAS compounds,

such as short-chain PFAS, have molecular masses below 270 Da and may not be fully removed.¹⁴¹ NF270 might eliminate 96.6–99.4% PFAS for specific compounds like PFHxA.¹⁴⁰ The concentration of the PFHxA in the retentate was up to 870 mg L⁻¹. The shortcoming of the membrane method is that the concentration has to deal with a destructive method, such as AOP (advanced oxidation process).¹⁴⁰ In this situation, electro-oxidation is used and decreases PFHxA to 8–21 mg L⁻¹. Appleman *et al.*⁴² found in their investigation that NF270 might be applied for the treatment of highly saline groundwater. Furthermore, Boonya-chart *et al.*¹⁴⁶ applied nanofiltration for eliminating PFOA from groundwater and showed 99.49 to 99.54% rejection of PFOA. The membrane can eliminate both neutral soluble and charged organic compounds having a molecular weight less than 200 Da. Therefore, nanofiltration could eliminate PFOA with a molecular weight of 414 g mol⁻¹. The membrane approach is an efficient method for the remediation of PFAS from aqueous solutions, although membrane fouling and low water levels would eventually reduce the work of the membrane. This will happen due to the presence of inorganic ions as well as humic acid in the groundwater, which leads to a decrease in the rejection efficiency of the membrane.¹⁴⁷ The PFAS rejection efficiency of the membrane can be enhanced by high-pressure PFOA concentration, but this will increase the operational cost.^{146,148,149}

The membrane approach is highly developed and easily customizable, expanding its applicability and versatility across a range of applications, including wastewater treatment and desalination.^{150,151} Boonya-atichart *et al.*¹⁴⁶ found a 99.49 to 99.54% removal efficiency of PFOA in groundwater using a nanofiltration (NF) membrane. The membrane can discard soluble neutral and charged organic molecules with molecular weights greater than 200 Da because of the pore size constraint. Therefore, PFOA with a molecular weight of 414 g mol⁻¹ can be eliminated by NF. Furthermore, a study by Tang *et al.*¹⁵² found that NF membranes may reject up to 90–99% PFOS. Fig. 3 shows an illustration of efficient techniques used for the remediation of PFAS from aqueous solutions.

6.3 Adsorption

6.3.1. Activated carbon. Activated carbon is emerging as an effective adsorbent for the elimination of pollutants from aqueous solutions due to its high sorption capacity as well as low cost.¹⁵³⁻¹⁵⁷ Geometrical (for variation in size and shape of pores) and chemical heterogeneity (for different functional groups) are used for the characterization of activated carbon.¹⁵⁸ Several studies reported the use of granular activated carbon for the remediation of the PFAS from aqueous solutions *via* adsorption processes, including applications at large scales. Due to a large size, volume, and surface area, activated carbon has shown higher adsorption capacity.¹⁵⁹⁻¹⁶² Granular activated carbon successfully eliminated long-chain PFAS through hydrophobic interactions, although it did not eliminate short-chain PFAS successfully (achieving equilibrium in a shorter time). Powder-activated carbon is effectively used for the



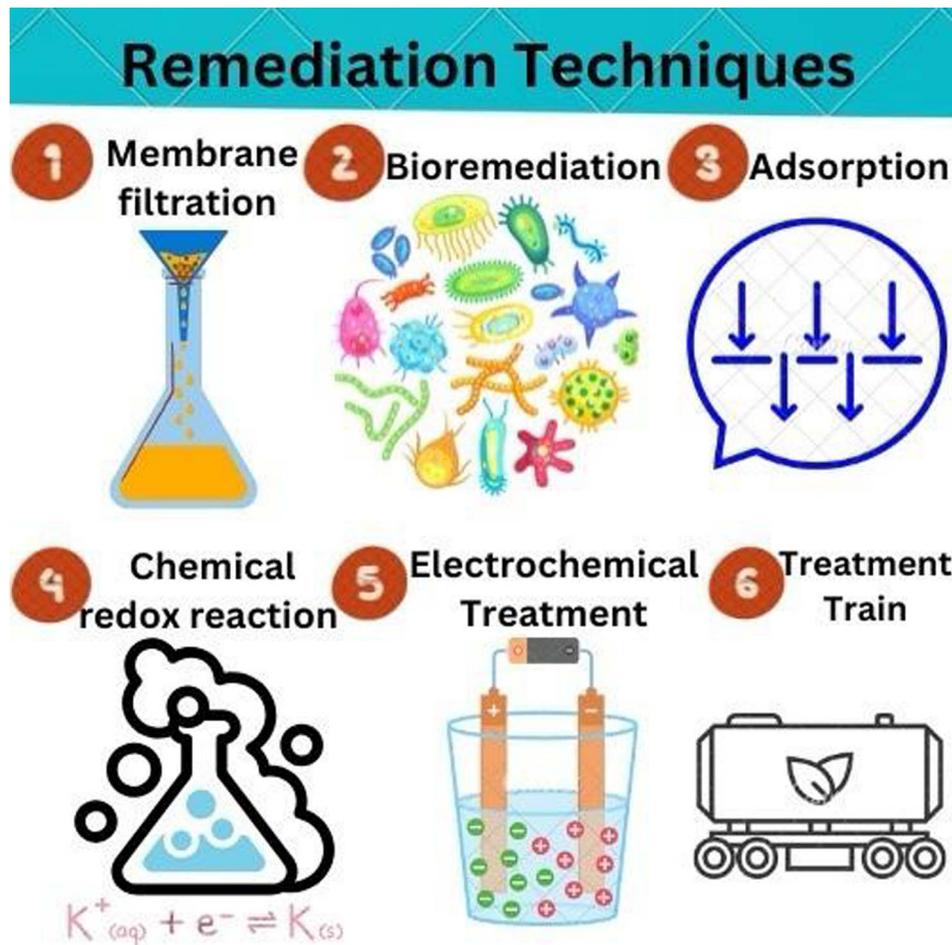


Fig. 3 Techniques used for the remediation of PFAS from aqueous solutions.

elimination of short-chain PFAS owing to its small size in comparison with granular activated carbon.¹⁶⁰

Powdered activated carbon shows a comparatively higher adsorption capacity for PFOA as well as PFOS than granular activated carbon.¹⁶⁰ Du *et al.*¹⁶¹ investigated the adsorption behavior of bamboo-derived activated carbon for PFOA, and the findings indicate that an equilibrium was reached in 34 h (a long time). Table 2 shows the efficiency of various adsorbents used for the removal of PFAS from water.

6.3.2. Biochar. Biochar is a porous, stable, carbonaceous, and environmentally benign substance. The main raw material used to make biochar is the abundant, cost-effective, and easily available biowastes from the food and agriculture sectors.^{168,169} Biochar possesses various functional groups, primarily heterocyclic and aromatic, such as hydroxyl and carbonyl, which enhance PFAS adsorption.^{170,171} PFAS removal with biochar is mostly achieved through the effects of hydrophobicity and electrostatic interactions.¹⁶⁴ It has been demonstrated that biochar is useful for eliminating heavy metals and pharmaceuticals from water.^{172,173} Furthermore, its remarkable physical characteristics, such as pore size and specific surface area, can be tuned to meet the unique requirements for the removal of short-chain PFAS.

Patel *et al.*¹⁶⁴ fabricated an affordable biochar for the removal of PFAS from wastewater. The biochar was obtained from the anaerobic digestion of organic pollutants in the wastewater treatment process. Anaerobic digestion of organic pollutants produces biogas, which is subsequently used to sequester carbon *via* catalytic breakdown to produce hydrogen and carbon nanomaterials (CNMs), and then the biochar is loaded with carbon nanomaterials, and recognized as CNM-laden biochar. CNM-laden biochar showed a superior removal efficiency for PFAS (79%) as compared to biochar modified with ilmenite (54%). In order to remove PFAS, biochar was prepared by pyrolyzing hardwood and softwood shavings at different temperatures of 300, 400, and 600 °C. These biochars were post-modified *via* thermal oxidation for 30 minutes at 400 °C, which increased the pore size and surface area. After PFAS were adsorbed on biochar, heat activation at 500 °C with nitrogen or air destroys them. After heat activation, the biochar exhibits improved PFAS elimination efficiency in subsequent cycles.¹⁷⁴ In addition, biochar was obtained from the hardwood and softwood shavings at various pyrolysis temperatures from 300 to 600 °C. The material undergoes hot air oxidation after pyrolysis, which increases the material's surface area and pore size, leading to improvement in its absorption capacity. Compared to untreated biochar, this procedure triples the reactivity of



Table 2 Summary of the adsorption efficiency of various adsorbents for the removal of PFAS

Adsorbent	Type of PFAS	Adsorbent dosage	Initial concentration	Removal efficiency	References
DUT-5-2	PFOA	5 mg	30 mg L ⁻¹	60.9	Hu <i>et al.</i> ¹⁶³
Carbon nanomaterial-coated biochar (CNM-biochar)	PFOS	1 g	310 µg L ⁻¹	95	Patel <i>et al.</i> ¹⁶⁴
Sewage sludge-activated carbon (ZnCl ₂ -impregnated)	PFBA	5 mg	50 µg L ⁻¹	87	Mohamed <i>et al.</i> ¹⁶⁵
Biochar-alginate composite beads	PFOS	1.5 g L ⁻¹	—	99	Militao <i>et al.</i> ¹⁶⁶
Classical powder-activated carbon	PFOS	0.004 g	2 ppm	93	Pala <i>et al.</i> ¹⁶⁷
MIL-101 (Cr)@AC MOF	PFOS	100 ppm	2 ppm	80	

biochar derived from softwood for certain PFAS. The materials were subjected to a solution comprising PFCAs, PFSAs, and GenX. They were then allowed to dry, and the PFAS were removed by heating the dried materials for 30 min at 500 °C in air. Following treatment, additional sorption tests were conducted on the biochars and granular activated carbon (GAC), using a mixture of PFBA, PFPeA, GenX, PFHpA, and PFOA.

Krahn *et al.*¹⁷⁵ prepared a biochar from sewage sludge and wood chips by pyrolyzing at 700 °C for the elimination of long-chain PFAS-like PFOA, PFNA, and PFDA, and short-chain PFAS-like PFPeA, PFHxA, and PFHpA. All of the biochar's adsorption onto sewage sludge and wood chip biochar, except PFPeA and PFHxA, followed a strong log-linear relationship ($r^2 > 0.9$). These poor fits may be explained by the fact that ionic interactions are more significant for the adsorption of these short-chain PFCAs compared to hydrophobic ones for the adsorption of the longer-chain PFCAs.

In other studies, leftover coffee grounds were used to prepare biochar *via* two-step pyrolysis, followed by post-treatment to enhance its properties. In order to functionalize the biochar surface with nitrogen-containing groups such that radical-initiated polymerization may occur, two distinct methods were looked into. The first method was electrophilic aromatic substitution, which involved reducing the nitro group to -NH_x with acetic acid, NH₄OH, and Na₂S₂O₄, then combining precisely measured amounts of pristine biochar with HNO₃ and H₂SO₄. In the second method, the biochar-water mixture was combined with melamine (BC-M) or ammonium chloride (BC-N), and the nitrogen modification was initiated by heating the mixture. Using thermally accelerated radical-initiated polymerization, a layer of molecularly imprinted polymers (MIP) was applied to the surface of nitrogen-modified biochar to increase the adsorbent's selectivity for PFAS. In comparison to unmodified biochar, the BC-M-MIP showed higher K-selectivity for PFBS (4.52) and PFOA (3.76) at 0.043 and 0.039 mg PFAA/g × g m⁻², respectively.¹⁷⁶ In a different experiment, different PFAS chain lengths (4 to 11 fluorinated carbons) were eliminated using eucalyptus charcoal and tree bark. The fact that the sorption increased with the increase in chain length indicates that hydrophobic interactions play a key role in the adsorption process.¹⁷⁷ To create biochar, Wu *et al.*¹⁷⁸ employed a variety of feedstocks, including switchgrass (SG), water oak leaves (WO), and biosolids (BS), and altered them by adding carbon nanotubes (CNTs) and FeCl₃ to increase their porosity and surface area. The composite made of oak leaves and biochar rapidly attained the adsorption equilibrium. In

comparison to the other two biochars, the modified biosolid-biochars showed increased PFOA adsorption. BS-Fe demonstrated the highest adsorption capacity for PFOA (469.65 µmol g⁻¹), while WO-CNT demonstrated the lowest adsorption potential of 39.54 µmol g⁻¹.

In a different study, biochar was made from Douglas fir by gasification at 900–1000 °C, with syngas as a byproduct. Iron(III) chloride (18 g) and iron(II) sulfate (36.6 g) were added to a solution (2 L) containing biochar (50 g) to change the biochar. Fe₃O₄ particles were then created and deposited onto biochar following a 24-hour period during which the pH was maintained at 10, and a gentle introduction of 10 M NaOH was carried out. The Fe₂O₃-modified biochar was ready following the steps of vacuum filtration, three ethanol washes, and 50 °C drying. In comparison to PFOS (82%), the Fe₂O₃-modified biochar eliminated 75–85% of PFBS, GenX, and perfluorooctane sulfonamide (PFOS).¹⁷⁹ When compared to pristine biochar, modified biochar showed enhanced adsorption potentials for both short- and long-chain PFAS. The main processes of biochar-mediated PFAS elimination include hydrophobic and electrostatic attraction. Since short-chain PFAS are not often removed, more research is required to find sustainable removal strategies that target short-chain PFAS in particular during wastewater treatment.

6.3.3. Miscellaneous adsorbents. Current research focuses on exploring innovative adsorbents with lower cost and higher adsorption capacity. Some novel materials, such as cross-linked chitosan beads, effectively removed PFOS from aqueous solution at pH 3.¹⁸⁰ At pH 3, amino groups (99.4%) on the chitosan beads were protonated, and consequently, PFOS, carrying a negative charge, is adsorbed easily, which leads to enhanced adsorption capacity. At an equilibrium concentration of 0.33 mmol L⁻¹, the maximum adsorption capacity for chitosan biosorbent reached 5.5 mmol g⁻¹ for PFOS, and it was greater than for traditional adsorbents. Chen *et al.*¹⁸¹ reported other cost-effective adsorbents including maize straw-origin ash, straw- and willow-derived chars, and carbon nanotubes. Single-walled carbon nanotubes and maize ash showed greater adsorption capacity for PFOS at ~700 mg g⁻¹. This may be due to the formation of hemimicelles and hydrophobic interactions preferring the binding of PFOS. Carbon nanotubes can serve as effective adsorbents for the adsorption of PFAS because of the greater adsorption capacity as well as equilibrium reached in a short time; nevertheless, ash attained equilibrium within 48 h along with an adsorption capacity similar to carbon nanotubes.



Furthermore, the cost of the ash is relatively low, and it is obtained easily *via* incineration of biomass; therefore, its utilization is a favorable option for other biomaterials, along with high recovery and reuse of agro-waste.^{182,183} The regeneration of a cross-linked adsorbent was found to be quick and was achieved in merely five minutes without any significant decrease in the adsorption capacity for PFOS in the presence of other organic pollutants.

Adsorption is a well-developed method for the remediation of PFAS from an aqueous solution, but it suffers from some limitations. Generally, adsorption studies are focused on the remediation of specific PFAS, whereas actual aqueous solutions contain a mixture of PFAS with varying chain lengths and functional groups.^{184,185} Also, further research is necessary to understand the interference of inorganic ions and organic substances. This inexorably restricts the adsorption capacity, which is important for the regeneration of the adsorbents after adsorption saturation; regenerated adsorbents demonstrate lower reutilization capacity when the concentration of PFAS in the aqueous solution is relatively low.¹⁸⁶

6.4 Electrochemical approach

Certain electrochemical approaches, such as electrosorption, electrochemical oxidation, and electrocoagulation, are used in various applications.^{187–191} The electrocoagulation approach is comparatively attractive due to its low capital expenditure and simple operation. The results obtained from Niu *et al.*¹⁸⁷ revealed the use of zinc as an anode and stainless steel as a cathode for the remediation of a PFOA aqueous solution at an initial concentration of 200 mg L⁻¹. The maximum removal efficiency reached 99.7%. The surface of the electrode becomes passive if a single electrode is used for a prolonged time. For the remediation of PFOA, Liu *et al.*¹⁹² applied periodically reversing electrocoagulation (PREC) with an Al–Zn electrode. The mechanism behind the sorption was the hydrophobic interaction between PFOA and Al–Zn hydroxide flocs. The direct oxidation of the anode and the indirect oxidation of active radicals are the main mechanisms behind the electrochemical oxidation approach for the treatment of PFAS. Oxygen evolution potential (OEP), the electron transfer ability of the anode material, and the potential to produce $\cdot\text{OH}$ are the factors affecting the removal efficiency.^{193,194} Schaefer *et al.*¹⁸⁸ observed that PFOA and PFOS are efficiently oxidized *via* boron-doped diamond anodes (electrochemical method), and the dominant mechanism is direct anodic oxidation. Although various studies suggested that the generation of $\cdot\text{OH}$ through the anode could not break the PFOA, the rate-limiting step for PFOA breakdown was the electron transfer at the anode surface.^{195,196}

6.5 Treatment-train techniques

The goal of SERDP Project ER-2423 was to provide an economical *in situ* technique for treating groundwater that contains per- and polyfluoroalkyl compounds (PFAS). Horizontal reactive media wells were utilised in the project, also known as “*in situ* chemical oxidation of sorbed contaminants (ISCO-SC)”, to remediate contaminated groundwater on-site.

This technique is particularly helpful for locations affected by aqueous film-forming foam (AFFF). In treatment-train techniques, there is a combination of *in situ* and *ex situ* remediation techniques, which results in efficient remediation techniques as compared to any single method. Crimi,¹⁹⁷ in SERDP project 2423, integrates two potential remediation technologies: sorption *via* GAC and PFAS destruction using activated persulfate.¹⁹² It was believed that the technology gives information on the regeneration of carbon, the effectiveness of persulfate treatment, and degradation products. Further SERDP ER-2426 technology was used for the investigation of reductive or oxidative methods applying [0] valent metals and bimetals (Pd/Fe, Mg, Pd/Mg) and synthesis using clay and vitamin B12-assisted co-solvents.^{198,199}

6.6 Bioremediation

For many years, it was suggested that biotechnology is not an efficient technique for the elimination of PFAS. It was difficult for the microorganisms to utilize PFAS as their sole carbon source for cell growth. The cleavage of the C–F bond requires more energy, which is greater than the capacity of microorganisms. Currently, some bacterial strains under anammox and anaerobic conditions have appeared to be feasible to transform as well as degrade PFAS.^{196,200,201} Anammox is a type of reaction in which ammonium gets oxidized in the absence of oxygen into dinitrogen gas using nitrite as the electron acceptor.^{202,203} Early research indicated that the bioremediation of PFAS is not feasible because of the strong C–F bonds; however, a recent study²⁰² has shown that the biodegradation of PFAS is possible. Fluorotelomeric structures having CH₂ groups on the C–F backbone have shown sequential microbial defluorination instead of perfluorinated substances. Fluorotelomeric alcohols consisting of carboxylic and sulfonic acids were moderately biotransformed into perfluoroalkyl acids under aerobic as well as anaerobic environments.

The breakdown of fluorinated alkyl substances needs a minimum of one H atom in the alkyl chain for the place of the prime attack.² The main challenge in the oxidative replacement of fluorine atoms is the potential formation of a highly hydrophobic layer near the carbon–carbon bonds, which hinders the oxidative degradation. Due to this feature, the fluorine-saturated carbon chain element restricted the oxidation *via* the employment of microorganisms as sources of carbon as well as energy.²⁰⁴ Only a few studies have been carried out on the microbial degradation of PFAS.^{205,206} There are several contradicting reports, and every report needs further research work to understand the biotic transformations of these substances.²⁰⁷ Various strains of bacteria were applied for the breakdown of the PFAS in an aerobic environment. Sulfonates containing a terminal hydrogen, such as H-PFOS and 2,2,2-trifluoroethane, can be partially degraded by the *Pseudomonas* strain D2 through defluorination under aerobic, sulfur-limited conditions.²⁰⁸ Six products formed during the defluorination reaction carried only oxygen and fluorine, without sulfur. *Pseudomonas butanova* was applied for the breakdown of a 6–2 fluorotelomer alcohol precursor and this strain is not effective for the



degradation of 8–2 FTOH.²⁰⁹ Additionally, studies on the breakdown of PFAS using fungal species are ongoing,²¹⁰ because of a broad spectrum of substrate reduction catalyzed by extracellular ligninolytic enzymes. Under aerobic conditions, white-rot fungus *Phanerochaete chrysosporium* can reduce the 45% concentration of 6:2 FTOH within 35 days, along with the generation of various shorter-chain metabolites, for example, perfluorobutyric acid (PFBA) and perfluoropentanoic acid (PFPeA).^{197,211} There are various approaches that have been applied in the bioremediation of PFAS from aqueous solutions, as shown in Fig. 4.

6.6.1. Phytoremediation. Phytoremediation is the most promising approach for the extraction of toxic pollutants from the environment by employing chosen plants. In the current scenario, the utilization of plants for the removal of pollutants from water becomes the optimized alternative. Phytoremediation comprises different types, as depicted in Fig. 4.²¹²

The absorption of the pollutant through the root is known as phytoextraction. Phytostabilization is defined as the incorporation of the organic or inorganic substance into the lignin. In phytovolatilization, the hazardous substance is converted into the least toxic form by absorption of the pollutants through the roots. In phytodegradation, the breakdown of toxic substances is feasible by means of specific enzyme activity. In the rhizofiltration process, terrestrial plants are used for the absorption of different types of pollutants from water or soil through their roots.²¹³

There are typically four steps involved in the elimination of PFAS from the environment, which are phyto uptake, translocation, bioaccumulation and biodegradation. Biodegradation in plants was previously recognized through the “plant detoxification mechanisms” of transformation, conjugation, and compartmentalization, which were also recognized as the

“green liver model”.²¹⁴ It has also been observed that the elimination of PFAS from wetlands was possible due to three major mechanisms: phytouptake, bioaccumulation, and translocation.^{215,216} The phytouptake of PFAS in plants occurs through two pathways; one is passive and the second is active,²¹⁷ as shown in Fig. 5.

Anthracene-9-carboxylic acid, 4,4'-diisothiocyanostibene-2,2'-disulfonate, and 5-nitro 2-(3-phenylpropylamine) benzoic acid are examples of anion channel blockers, and metabolic inhibitors (NaN₃ and Na₃VO₄), and aquaporin competitors (AgNO₃ and glycerol) are examples of inhibitors that can be used to measure the active and passive transport of PFAS.^{219,220} In maize (*Zea mays* L. cv. TY2), metabolic pathways contributed 43–83% and zero, respectively, to the transfer of PFOA and PFOS.²¹⁹ As a result, although PFOS uptake may be somewhat passive, PFOA uptake may be partially active. Nonetheless, wheat (*Triticum aestivum* L.) reduced its uptake of PFOA and PFOS by 58–87% and 64–94%, respectively, suggesting that both PFOA and PFOS are mostly transported in wheat.²²⁰ If all factors are considered, then the transport mechanism of PFOS in plants may vary by species. Furthermore, PFAS transport in plants may be facilitated by aquaporin and anion channels.²¹⁹

The mechanism behind the biodegradation of PFAS by plants as well as microorganisms is still not clear. There are various parameters, such as physicochemical properties, ionic strength, host plant species, exposure time, pH, chain length, and initial concentration, which affect the extraction process of PFAS from the environment.^{215,221}

Phytofiltration is the main procedure by which aquatic plants eliminate PFAS.²²² The phytofiltration of PFAS into the aqueous phase mostly occurs in plant roots and submerged organs by the adsorption and precipitation mechanisms.²²³ Based on diverse mechanisms of uptake, aquatic plants (free-

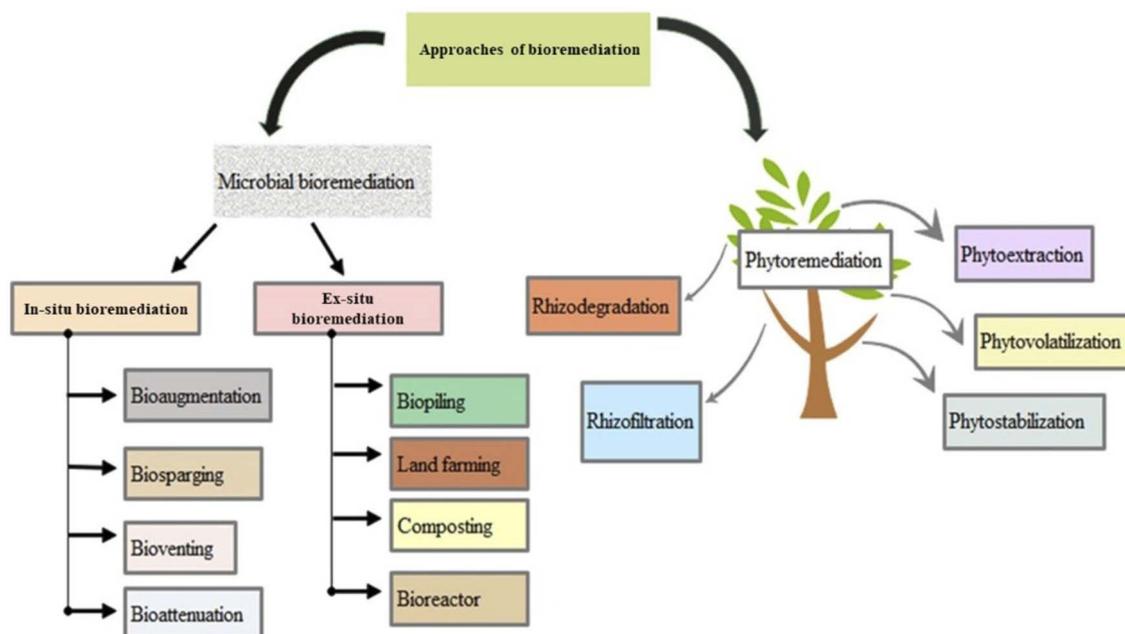


Fig. 4 Different approaches applied for the bioremediation of PFAS.



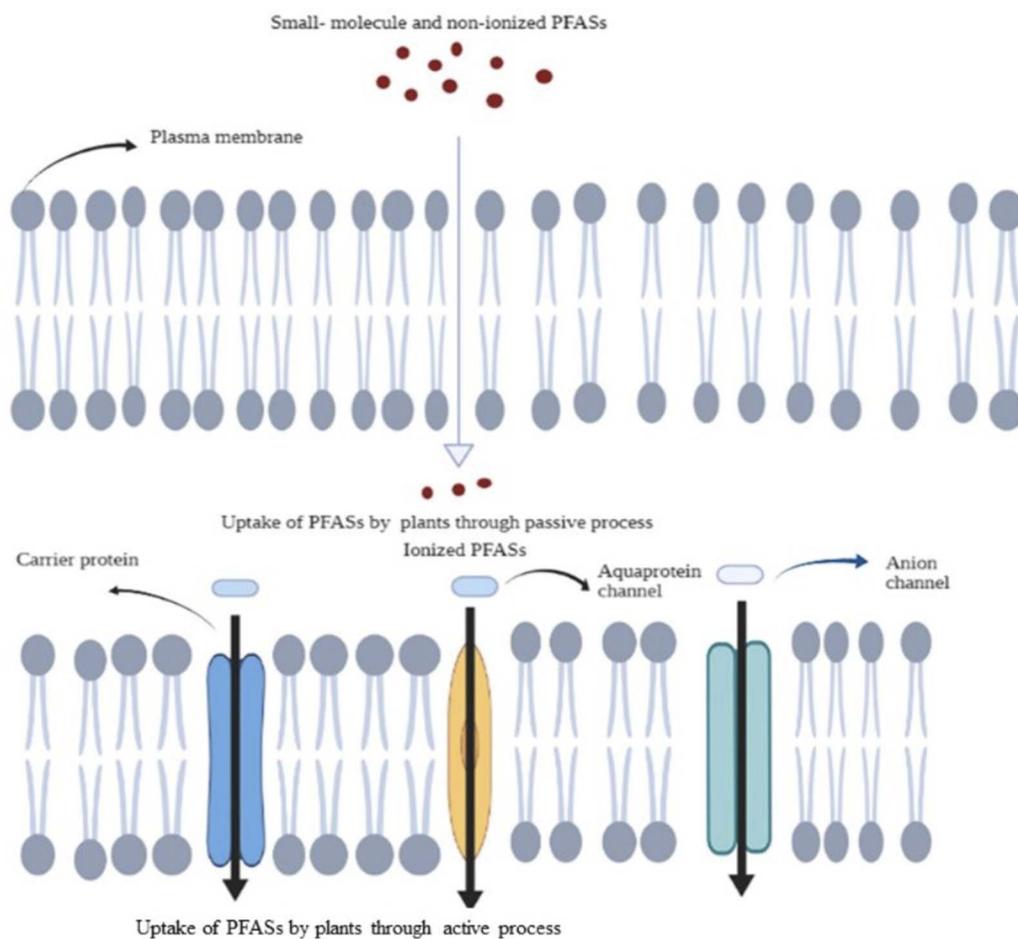


Fig. 5 Uptake of PFAS via plants through (a) a passive process and (b) an active process [modified from Wang *et al.*²¹⁸ with permission from Elsevier (ref. 218) (license number: 6157540752757 dated 28th November 2025)].

floating plants as well as submersed species) would be able to attract pollutants through the means of precipitation as well as adsorption. On the other hand, various terrestrial plants can eliminate pollutants *via* root uptake and bioaccumulation. Previous research suggests that submersed wetland plants with higher growth of root systems could attain higher uptake of PFOs.²²⁴ The results of a study on *Echinodorus horemanii* (submersed plants) and *Eichhornia crassipes* (free-floating plants) found that the *E. horemanii* exhibits faster uptake of PFAS compared to *E. crassipes*, which may be due to its greater biosorption as well as leaf-water exchange capacities.²²⁵ Consequently, the findings suggest that submersed plants exhibit greater uptake of PFAS compared to free-floating plants owing to their well-developed root systems as well as leaf-water exchange capacity in an aqueous medium.^{225,226}

The prime mechanisms behind the elimination of organic contaminants are phytodegradation, phytofiltration, and phytovolatilization.^{227–229} Moreover, it is important to note that PFAS do not undergo biodegradation, chemical disintegration or photolysis.²¹⁷ As the biodegradation of PFAS is restricted, the best alternative is plant uptake at the remediation site of the PFAS.¹⁹⁸ Past researchers indicate that the uptake mechanism of PFAS from the soil or water typically occurs *via* the plant's roots

by means of passive and active processes, as shown in Fig. 5.^{218,230} The main passive transportation depends on the diffusion of small-molecular and non-ionized PFAS as well as the driving force of the transpiration stream.²³¹ Once the PFAS goes into the root apoplast through diffusion, it is translocated to the part of the plant that is above the ground, for example, fruits, stems, leaves, and shoots, by the transpiration stream.²³² Meanwhile, active transportation largely involves selective adsorption through particular transporters such as ion channels and aquaporins for the transportation of ionized PFAS into plant cells.²¹⁹ However, airborne PFAS and analogues in the vapor phase and particulates could be adsorbed in plants through aerial parts, for example, foliage and bark.²³³

Yin *et al.*²³⁴ investigated the removal efficiency for constructed wetlands of PFAS. In this study, the authors used eleven PFAS, out of which seven PFCAs were perfluoroalkyl carboxylic acids and perfluoroalkane sulfonates along with 7 PFAA precursors. The level of the PFAS changes from 1269 to 7661 mg L⁻¹. Throughout the year (sampling time), the composition of the PFAS remains the same in comparison with the PFCAs. It was believed that the CW treatment could eliminate approximately 61% of the total PFAS, along with 50–96% for individual PFAS. The reed bed shows a high efficiency from 42 to 49%.



Zhang and Liang²²¹ conducted an experiment to determine the efficiency of duckweed for the removal of PFAS from an aqueous solution. It was observed that removal of perfluorobutanoic acid (PFBA), perfluoropentanoic acid (PFPeA), perfluorobutanesulfonic acid (PFBS), and perfluorohexanoic acid (PFHxA) was marginal and much lower than those of perfluoroheptanoic acid (PFHpA), perfluorohexanesulfonic acid (PFHxS), perfluorooctanoic acid (PFOA), and perfluorooctanesulfonic acid (PFOS). The duckweed showed about 80% removal efficiency for PFOA and PFOS at pH 2.3 at an initial concentration of 0.200 mg L⁻¹.

7. Challenges for the remediation of PFAS from aqueous solutions and conclusion

PFAS remediation from water is also very challenging, and the advantages and disadvantages of the methods are discussed in the following. The techniques used for the remediation of PFAS from aqueous solutions are not efficient due to the special molecular structures and physicochemical properties of PFAS.^{197,220} Furthermore, the bioremediation of PFAS requires further research to be successfully applied for effective PFAS degradation. Additionally, the choice of an appropriate adsorbent for the adsorption of PFAS is also a big challenge. The removal of short-chain PFAS is more challenging compared to that of long-chain PFAS. Innovative remediation approaches need specific conditions, more chemicals, materials, and high energy, and are expensive.^{141,235,236}

Poly- and perfluoroalkyl substances are large groups of synthetic chemicals that are persistent and environmentally stable. These chemicals have a wide range of industrial applications, and with the rapid escalation in PFAS usage, their concentration in aqueous solutions has concurrently increased. These chemical substances are toxic and can seriously threaten human and animal health. This review discusses the sources of PFAS in aqueous solutions and their negative impacts on humans. Furthermore, various remediation techniques such as electrochemical treatment, adsorption, bioremediation, and membrane technology have been discussed critically, along with a special focus on the phytoremediation of PFAS. However, the phytoremediation of PFAS still needs to be extensively explored, because the survival rate of the plants is relatively low. Future research should focus on developing advanced, cost-effective, and eco-friendly remediation technologies for PFAS removal from aqueous media, with improved efficiency.

Conflicts of interest

The authors declare that they do not have any personal or financial conflicts of interest.

Data availability

No primary research results, software or code have been included and no new data were generated or analysed as part of this review.

Acknowledgements

The authors did not receive support from any organization for the submitted work.

References

- 1 S. Taniyasu, K. Kannan, Y. Horii, N. Hanari and N. Yamashita, A survey of perfluorooctane sulfonate and related perfluorinated organic compounds in water, fish, birds, and humans from Japan, *Environ. Sci. Technol.*, 2003, **37**(12), 2634–2639, DOI: [10.1021/es0303440](https://doi.org/10.1021/es0303440).
- 2 K. Prevedouros, I. T. Cousins, R. C. Buck and S. H. Korzeniewski, Sources, fate and transport of perfluorocarboxylates, *Environ. Sci. Technol.*, 2006, **40**(1), 32–44, DOI: [10.1021/es0512475](https://doi.org/10.1021/es0512475).
- 3 A. Shetty and A. Goyal, Application of nanofiltration systems in wastewater reclamation & its long-term performance, *AIP Conf. Proc.*, 2022, **2357**(1), 030015.
- 4 R. Takkar, B. Kaur, S. Rana, P. Kumar, P. Khanra and M. Dhiman, Usage of magnetic spinel nano-ferrites in waste water treatment, *ECS Trans.*, 2022, **107**(1), 10237.
- 5 R. C. Buck, J. Franklin, U. Berger, J. M. Conder, I. T. Cousins, P. de Voogt, *et al.*, Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins, *Integr. Environ. Assess. Manage.*, 2011, **7**(4), 513–541, DOI: [10.1002/ieam.258](https://doi.org/10.1002/ieam.258).
- 6 C. A. Moody and J. A. Field, Perfluorinated surfactants and the environmental implications of their use in fire-fighting foams, *Environ. Sci. Technol.*, 2000, **34**(18), 3864–3870, DOI: [10.1021/es991359u](https://doi.org/10.1021/es991359u).
- 7 A. Kärrman, K. Elgh-Dalgren, C. Lafossas and T. Møskeland, Environmental levels and distribution of structural isomers of perfluoroalkyl acids after aqueous fire-fighting foam contamination, *Environ. Chem.*, 2011, **8**(4), 372–380, DOI: [10.1071/EN10145](https://doi.org/10.1071/EN10145).
- 8 J. A. Simon, S. Abrams, T. Bradburne, D. Bryant, M. Burns, D. Cassidy, *et al.*, PFAS experts symposium: statements on regulatory policy..., *Remediat J.*, 2019, **29**(4), 31–48, DOI: [10.1002/rem.21624](https://doi.org/10.1002/rem.21624).
- 9 M. M. Schultz, D. F. Barofsky and J. A. Field, Quantitative determination of fluorotelomer sulfonates in groundwater by LC-MS/MS, *Environ. Sci. Technol.*, 2004, **38**(6), 1828–1835, DOI: [10.1021/es035031j](https://doi.org/10.1021/es035031j).
- 10 J. W. Washington, J. Ellington, T. M. Jenkins, J. J. Evans, H. Yoo and S. C. Hafner, Degradability of an acrylate-linked fluorotelomer polymer in soil, *Environ. Sci. Technol.*, 2009, **43**(17), 6617–6623, DOI: [10.1021/es9002668](https://doi.org/10.1021/es9002668).
- 11 C. Yang, H. K. Lee, Y. Zhang, L. L. Jiang, Z. F. Chen, A. C. K. Chung, *et al.*, In situ detection and imaging of PFOS in mouse kidney by MALDI imaging MS, *Anal. Chem.*, 2019, **91**(14), 8783–8788, DOI: [10.1021/acs.analchem.9b00711](https://doi.org/10.1021/acs.analchem.9b00711).
- 12 J. Seow, *Fire Fighting Foams with Fluorochemicals: Environmental Review*, Department of Environment and Conservation Australia, 2013.



- 13 Z. Wang, I. T. Cousins, M. Scheringer, R. C. Buck and K. Hungerbühler, Global emission inventories... Part II, *Environ. Int.*, 2014, **69**, 166–176, DOI: [10.1016/j.envint.2014.04.006](https://doi.org/10.1016/j.envint.2014.04.006).
- 14 Z. Wang, I. T. Cousins, M. Scheringer, R. C. Buck and K. Hungerbühler, Global emission inventories... Part I, *Environ. Int.*, 2014, **70**, 62–75.
- 15 E. F. Houtz, R. Sutton, J. S. Park and M. Sedlak, Poly- and perfluoroalkyl substances in wastewater, *Water Res.*, 2016, **95**, 142–149.
- 16 B. Weiner, L. W. Yeung, E. B. Marchington, L. A. D'Agostino and S. A. Mabury, Organic fluorine content in AFFFs and biodegradation..., *Environ. Chem.*, 2013, **10**(6), 486–493.
- 17 R. Hetzer, F. Kümmerlen, K. Wirz and D. Blunk, Fire testing a new fluorine-free AFFF, *Fire Saf. Sci.*, 2014, **11**, 1261–1270.
- 18 S. A. Milley, I. Koch, P. Fortin, J. Archer, D. Reynolds and K. P. Weber, Estimating airports potentially contaminated..., *J. Environ. Manage.*, 2018, **222**, 122–131.
- 19 A. Wittenberg, After controversy, U.S. releases..., *Science*, 2018, DOI: [10.1126/science.aau5402](https://doi.org/10.1126/science.aau5402).
- 20 X. C. Hu, D. Q. Andrews, A. B. Lindstrom, T. A. Bruton, L. A. Schaidler, P. Grandjean, *et al.*, PFAS in U.S. drinking water..., *Environ. Sci. Technol. Lett.*, 2016, **3**, 344–350.
- 21 T. Stahl, D. Mattern and H. Brunn, Toxicology of perfluorinated compounds, *Environ. Sci. Eur.*, 2011, **23**, 38.
- 22 USEPA, PFOA & PFOS Drinking Water Health Advisories, EPA 800-F-16-003, 2016.
- 23 R. Naidu, P. Nadebaum, C. Fang, I. Cousins, K. Pennell, J. Conder, *et al.*, PFAS: current status and research needs, *Environ. Technol. Innovation*, 2020, **19**, 100915, DOI: [10.1016/j.eti.2020.100915](https://doi.org/10.1016/j.eti.2020.100915).
- 24 T. L. Coggan, T. Anumol, P. James, J. Shimeta and B. O. Clarke, A single analytical method for 53 PFAS, *Anal. Bioanal. Chem.*, 2019, **411**(16), 3507–3520, DOI: [10.1007/s00216-019-01829-8](https://doi.org/10.1007/s00216-019-01829-8).
- 25 T. L. Coggan, D. Moodie, K. Adam, D. Szabo, J. Shimeta, N. D. Crosbie, *et al.*, PFAS in 19 Australian WWTPs, *Heliyon*, 2019, **5**(8), e02316.
- 26 G. M. Sinclair, S. M. Long and O. A. H. Jones, PFAS exposure at environmental levels, *Chemosphere*, 2020, **258**, 127340.
- 27 Z. Wang, I. T. Cousins, M. Scheringer and K. Hungerbühler, Fluorinated alternatives to long-chain PFCs, *Environ. Int.*, 2013, **60**, 242–248.
- 28 L. Ahrens and M. Bundschuh, Fate and effects of PFAS in aquatic environment, *Environ. Toxicol. Chem.*, 2014, **33**, 1921–1929.
- 29 F. M. Hekster, R. W. Laane and P. de Voogt, Environmental and toxicity effects of PFAS, *Rev. Environ. Contam. Toxicol.*, 2003, **179**, 99–121.
- 30 D. Herzke, E. Olsson and S. Posner, PFAS in consumer products in Norway, *Chemosphere*, 2012, **88**, 980–987.
- 31 M. Chropeňová, P. Karásková, R. Kallenborn, E. K. Gregušková and P. Čupr, Pine needles for the screening of PFASs along ski tracks, *Environ. Sci. Technol.*, 2016, **50**(17), 9487–9496.
- 32 S. Fujii, C. Polprasert, S. Tanaka, H. Lien, N. Pham and Y. Qiu, New POPs: PFAS in water environment, *J. Water Supply: Res. Technol.-AQUA*, 2007, **56**, 313–326.
- 33 D. Benford, J. De Boer, A. Carere, *et al.*, Opinion on PFOS and PFOA, *EFSA J.*, 2008, **653**, 1–131.
- 34 K. Kannan, S. Corsolini, J. Falandysz, *et al.*, PFOS and related chemicals in human blood, *Environ. Sci. Technol.*, 2004, **38**, 4489–4495.
- 35 M. F. Rahman, S. Peldszus and W. B. Anderson, Fate of PFAS in drinking water treatment, *Water Res.*, 2014, **50**, 318–340.
- 36 H. A. Kaboré, S. Vo Duy, G. Munoz, *et al.*, Worldwide drinking water PFAS occurrence, *Sci. Total Environ.*, 2018, **616–17**, 1089–1100.
- 37 ITRC, *Naming Conventions and Physical Properties of PFAS*, 2017.
- 38 J. Hölzer, T. Göen, K. Rauchfuss, *et al.*, One-year follow-up of PFAS in plasma, *Int. J. Hyg. Environ. Health*, 2009, **212**, 499–504.
- 39 O. S. Arvaniti and A. S. Stasinakis, PFAS in wastewater treatment, *Sci. Total Environ.*, 2015, **524–25**, 81–92.
- 40 G. Munoz, P. Labadie, F. Botta, *et al.*, PFAS in tropical groundwater, *Sci. Total Environ.*, 2017, **607–8**, 243–252.
- 41 M. Ateia, A. Maroli, N. Tharayil and T. Karanfil, Short-chain PFAS review, *Chemosphere*, 2019, **220**, 866–882.
- 42 T. D. Appleman, E. R. V. Dickenson, C. Bellona and C. P. Higgins, Nanofiltration and GAC removal of PFAS, *J. Hazard. Mater.*, 2013, **260**, 740–746.
- 43 D. Szabo, T. L. Coggan, T. C. Robson, M. Currell and B. O. Clarke, PFAS from recycled water, *Sci. Total Environ.*, 2018, **644**, 1409–1417.
- 44 OECD/UNEP Global PFC Group, *Synthesis Paper on PFCs*, 2013, pp. 1–58.
- 45 S. Brendel, É. Fetter, C. Staude, L. Vierke and A. Biegel-Engler, Short-chain PFAS regulatory strategy, *Environ. Sci. Eur.*, 2018, **30**, 13.
- 46 M. Ikhlayel, Integrated e-waste management systems, *J. Cleaner Prod.*, 2018, **170**, 119–130.
- 47 C. Aimin, D. K. Xia and H. Shuk-mei, Developmental neurotoxicants in e-waste, *Environ. Health Perspect.*, 2011, **119**, 431–438.
- 48 J. Seeberger, R. Grandhi, S. S. Kim, *et al.*, E-waste management & public health, *J. Environ. Health*, 2016, **79**, 8–17.
- 49 A. Bhattacharya and S. K. Khare, Toxicants from e-waste, *Curr. Sci.*, 2016, **111**(12), 1946–1954.
- 50 M. Singh, M. Sankhla, M. Kumari, *et al.*, Effect of e-waste, *IOSR J. Environ. Sci., Toxicol. Food Technol.*, 2016, **10**, 98–104.
- 51 C. Gallen, D. Drage, S. Kaserzon, *et al.*, PFAS in landfill leachate, *J. Hazard. Mater.*, 2016, **312**, 55–64.
- 52 C. Hongkachok, S. K. Boontanon, *et al.*, PFAS in Thai groundwater, *Water Sci. Technol.*, 2017, **75**, 457–466.
- 53 S. Garg, P. Kumar, V. Mishra, *et al.*, PFAS from electronics, *J. Water Process Eng.*, 2020, **38**, 101683.
- 54 B. Tansel, From products to e-waste, *Environ. Int.*, 2017, **98**, 35–45.



- 55 Y. Fan, X. Li, Q. Xu, *et al.*, Albumin mediates PFAS lipid effects, *Environ. Pollut.*, 2020, **266**, 115138.
- 56 S. Palmer, A. Alaeus, J. Albert and S. Cox, HIV drug susceptibility, *AIDS Res. Hum. Retroviruses*, 1998, **14**(2), 157–162.
- 57 E. R. Barry, T. Morikawa, B. L. Butler, *et al.*, YAP restriction of intestinal stem cell expansion, *Nature*, 2013, **493**(7430), 106–110.
- 58 V. M. Vieira, K. Hoffman, H. M. Shin, *et al.*, PFOA exposure and cancer, *Environ. Health Perspect.*, 2013, **121**(3), 318–323.
- 59 K. M. Rappazzo, E. Coffman and E. P. Hines, PFAS exposure & child health, *Int. J. Environ. Res. Public Health*, 2017, **14**(7), 691.
- 60 P. Mishra, C. M. Pandey, U. Singh, *et al.*, Descriptive statistics, *Ann. Card Anaesth.*, 2019, **22**(1), 67–72.
- 61 W. Zhou, S. Zhao, C. Tong, *et al.*, Dietary intake and PFAS in Chinese women, *Environ. Int.*, 2019, **127**, 487–494.
- 62 F. D. Gilliland and J. S. Mandel, Mortality in PFOA workers, *J. Occup. Med.*, 1993, **35**, 950–954.
- 63 K. K. Raleigh, B. H. Alexander, G. W. Olsen, *et al.*, Mortality & cancer incidence in PFOA workers, *Occup. Environ. Med.*, 2014, **71**, 500–506.
- 64 K. T. Eriksen, M. Sørensen, J. K. McLaughlin, *et al.*, PFOS/PFOA plasma levels & cancer risk, *J. Natl. Cancer Inst.*, 2009, **101**(8), 605–613.
- 65 J. C. DeWitt, M. M. Peden-Adams, J. M. Keller and D. R. Germolec, Immunotoxicity of PFAS, *Toxicol. Pathol.*, 2012, **40**(2), 300–311.
- 66 E. Corsini, R. W. Luebke, D. R. Germolec and J. C. DeWitt, PFAS immunotoxicity, *Toxicol. Lett.*, 2014, **230**, 263–270.
- 67 E. M. Sunderland, X. C. Hu, C. Dassuncao, *et al.*, PFAS exposure pathways & health, *J. Exposure Sci. Environ. Epidemiol.*, 2019, **29**(2), 131–147.
- 68 P. Grandjean, C. Heilmann, P. Weihe, *et al.*, PFAS exposure & antibody response, *J. Immunotoxicol.*, 2017, **14**(1), 188–195.
- 69 C. Y. Lin, P. C. Chen, Y. C. Lin and L. Y. Lin, PFAS and metabolic syndrome, *Diabetes Care*, 2009, **32**(4), 702–707.
- 70 S. J. Frisbee, A. Shankar, S. S. Knox, *et al.*, PFAS and lipids in children, *Arch. Pediatr. Adolesc. Med.*, 2010, **164**(9), 860–869.
- 71 J. W. Nelson, E. E. Hatch and T. F. Webster, PFAS & cholesterol, *Environ. Health Perspect.*, 2010, **118**(2), 197–202.
- 72 N. Zhang, W. S. Wang, W. J. Li, *et al.*, PFOS effects on placenta hormone secretion, *Placenta*, 2015, **36**, 575–580.
- 73 F. Pérez, M. Nadal, A. Navarro-Ortega, *et al.*, PFAS accumulation in human tissues, *Environ. Int.*, 2013, **59**, 354–362.
- 74 J. P. Vanden Heuvel, J. T. Thompson, S. R. Frame and P. J. Gillies, Nuclear receptor activation by PFAS, *Toxicol. Sci.*, 2006, **92**(2), 476–489.
- 75 ATSDR, *PFAS in the U.S. Population*, 2017, [cited 2025 May 28], Available from: <https://www.atsdr.cdc.gov/PFAS>.
- 76 B. E. Blake and S. E. Fenton, Early-life PFAS exposure: placenta involvement, *Toxicology*, 2020, **443**, 152565.
- 77 A. P. Starling, J. L. Adgate, R. F. Hamman, *et al.*, PFAS during pregnancy & birth adiposity, *Environ. Health Perspect.*, 2017, **125**(6), 067016.
- 78 S. K. Sagiv, S. L. Rifas-Shiman, A. F. Fleisch, T. F. Webster, A. M. Calafat, X. Ye, *et al.*, Early-pregnancy plasma concentrations of perfluoroalkyl substances and birth outcomes in Project Viva: confounded by pregnancy hemodynamics?, *Am. J. Epidemiol.*, 2018, **187**, 793–802.
- 79 K. J. Marks, A. J. Cutler, Z. Jeddy, K. Northstone, K. Kato and T. J. Hartman, Maternal serum concentrations of perfluoroalkyl substances and birth size in British boys, *Int. J. Hyg. Environ. Health*, 2019, **222**, 889–895.
- 80 C. Xu, S. Yin, Y. Liu, F. Chen, Z. Zhong and F. Li, Prenatal exposure to chlorinated polyfluoroalkyl ether sulfonic acids and perfluoroalkyl acids: potential role of maternal determinants and associations with birth outcomes, *J. Hazard. Mater.*, 2019, **380**, 120867.
- 81 R. Huang, Q. Chen, L. Zhang, K. Luo, L. Chen and S. Zhao, Prenatal exposure to perfluoroalkyl and polyfluoroalkyl substances and risk of hypertensive disorders of pregnancy, *Environ. Health*, 2019, **18**, 5.
- 82 C. R. Stein, D. A. Savitz and M. Dougan, Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome, *Am. J. Epidemiol.*, 2009, **170**, 837–846.
- 83 C. C. Bach, A. Vested, K. T. Jørgensen, J. P. E. Bonde, T. B. Henriksen and G. Toft, Perfluoroalkyl and polyfluoroalkyl substances and measures of human fertility: a systematic review, *Crit. Rev. Toxicol.*, 2016, **46**, 735–755.
- 84 K. J. Lum, R. Sundaram, D. B. Barr, T. A. Louis and G. M. Buck Louis, Perfluoroalkyl chemicals, menstrual cycle length, and fecundity: findings from a prospective pregnancy study, *Epidemiology*, 2017, **28**, 90–98.
- 85 C. Zhang, R. Sundaram, J. Maisog, A. M. Calafat, D. B. Barr and G. M. Louis, A prospective study of prepregnancy serum concentrations of perfluorochemicals and risk of gestational diabetes, *Fertil. Steril.*, 2015, **103**, 184–189.
- 86 M. Boas, K. M. Main and U. Feldt-Rasmussen, Environmental chemicals and thyroid function: an update, *Curr. Opin. Endocrinol., Diabetes Obes.*, 2009, **16**, 385–391.
- 87 J. M. Weiss, P. L. Andersson, M. H. Lamoree, P. E. Leonards, S. P. van Leeuwen and T. Hamers, Competitive binding of poly- and perfluorinated compounds to thyroid hormone transport protein transthyretin, *Toxicol. Sci.*, 2009, **109**, 206–217.
- 88 G. M. Webster, S. A. Venners, A. Mattman and J. W. Martin, Associations between perfluoroalkyl acids and maternal thyroid hormones in early pregnancy: a population-based cohort study, *Environ. Res.*, 2014, **133**, 338–347.
- 89 F. Coperchini, P. Pignatti, S. Lacerenza, S. Negri, R. Sideri, C. Testoni, *et al.*, Exposure to perfluorinated compounds: in vitro study on thyroid cells, *Environ. Sci. Pollut. Res.*, 2015, **22**, 2287–2294.
- 90 D. Hershkovitz, Z. Burbea, K. Skorecki and B. M. Brenner, Fetal programming of adult kidney disease: cellular and molecular mechanisms, *Clin. J. Am. Soc. Nephrol.*, 2007, **2**, 334–342.
- 91 J. Dötsch, C. Plank and K. Amann, Fetal programming of renal function, *Pediatr. Nephrol.*, 2012, **27**, 513–520.



- 92 L. Liu, Z. Wang, Q. Zhang, Y. Mei, L. Li, H. Liu, *et al.*, Ion mobility mass spectrometry for the separation and characterization of small molecules, *Anal. Chem.*, 2023, **95**, 134–151.
- 93 M. Alikord, A. Mohammadi, M. Kamankesh and N. Shariatifar, Food safety and quality assessment: comprehensive review and recent trends in applications of ion mobility spectrometry, *Crit. Rev. Food Sci. Nutr.*, 2022, **62**, 4833–4866.
- 94 E. Te Brinke, A. Arrizabalaga-Larrañaga and M. H. Blokland, Insights of ion mobility spectrometry and its application on food safety and authenticity: a review, *Anal. Chim. Acta*, 2022, **1222**, 340039.
- 95 J. W. McCabe, M. J. Hebert, M. Shirzadeh, C. S. Mallis, J. K. Denton, T. E. Walker, *et al.*, The IMS paradox: a perspective on structural ion mobility–mass spectrometry, *Mass Spectrom. Rev.*, 2021, **40**, 280–305.
- 96 K. Masike, M. A. Stander and A. de Villiers, Recent applications of ion mobility spectrometry in natural product research, *J. Pharm. Biomed. Anal.*, 2021, **195**, 113846.
- 97 T. Causon, D. Petrusheva, E. Bogeve, V. Ivanova, and S. Hann, *Incorporating Standardized Drift-Tube Ion Mobility to Enhance Non-targeted Assessment of Wine Metabolome (LC×IM-MS)*, 2017.
- 98 X. Li, H. Wang, M. Jiang, M. Ding, X. Xu, B. Xu, *et al.*, Collision cross section prediction based on machine learning, *Molecules*, 2023, **28**, 4050.
- 99 E. Olesti, J. Boccard, G. Visconti, V. Gonzalez-Ruiz and S. Rudaz, From a single steroid to the steroidome: trends and analytical challenges, *J. Steroid Biochem. Mol. Biol.*, 2021, **206**, 105797.
- 100 K. E. Burnum-Johnson, X. Zheng, J. N. Dodds, J. Ash, D. Fourches, C. D. Nicora, *et al.*, Ion mobility spectrometry and the omics: distinguishing isomers, molecular classes and contaminant ions in complex samples, *Trends Anal. Chem.*, 2019, **116**, 292–299.
- 101 L. M. San Emeterio, I. M. Pérez-Ramos, M. T. Domínguez, J. M. R. Arranz, F. J. González-Vila, and J. González-Pérez, *Pyrolysis-Compound Specific Isotope Analysis for Characterization of Lignin in Soils*, 2021.
- 102 V. Palma, N. T. Jiménez Morillo, F. Sauro, B. Cubero, A. T. Caldeira, J. A. González-Pérez, *et al.*, *Assessing the Origin of Lipid Biomarkers in Volcanic Cave Environments*, 2021.
- 103 K. M. Hines, D. H. Ross, K. L. Davidson, M. F. Bush and L. Xu, Large-scale structural characterization of drug and drug-like compounds by high-throughput ion mobility-mass spectrometry, *Anal. Chem.*, 2017, **89**, 9023–9030.
- 104 A. Kaufmann, The use of UHPLC, IMS, and HRMS in multiresidue analytical methods: a critical review, *J. Chromatogr. B*, 2020, **1158**, 122369.
- 105 M. Hernández-Mesa, D. Ropartz, A. M. García-Campaña, H. Rogniaux, G. Dervilly-Pinel and B. Le Bizec, Ion mobility spectrometry in food analysis: principles, current applications and future trends, *Molecules*, 2019, **24**, 2706.
- 106 C. Capitain and P. Weller, Non-targeted screening for profiling volatile organic compounds by GC-IMS and machine learning, *Molecules*, 2021, **26**, 5457.
- 107 V. d'Atri, T. Causon, O. Hernandez-Alba, A. Mutabazi, J. L. Veuthey, S. Cianferani, *et al.*, Adding a new separation dimension to MS and LC-MS: utility of ion mobility spectrometry, *J. Sep. Sci.*, 2018, **41**, 20–67.
- 108 X. Garcia, M. D. M. Sabate, J. Aubets, J. M. Jansat and S. Sentellas, Ion mobility–mass spectrometry for bioanalysis, *Separations*, 2021, **8**, 33.
- 109 O. E. Olajide, K. Y. Kartowikromo, and A. M. Hamid, *Ion Mobility Mass Spectrometry: Instrumentation and Applications*, 2023.
- 110 S. Yukioka, S. Tanaka, Y. Suzuki, S. Fujii and S. Echigo, A new method to search for PFAS by linking fragmentation flags with molecular ions using IMS, *Chemosphere*, 2020, **239**, 124644.
- 111 Y. S. Luo, N. A. Aly, J. McCord, M. J. Strynar, W. A. Chiu, J. N. Dodds, *et al.*, Rapid characterization of emerging PFAS in AFFF using IMS-MS, *Environ. Sci. Technol.*, 2020, **54**, 15024–15034.
- 112 S. Yukioka, S. Tanaka, Y. Suzuki, S. Echigo and S. Fujii, Data-independent acquisition with IMS-MS for suspect screening of PFAS in environmental water, *J. Chromatogr. A*, 2021, **1638**, 461899.
- 113 H. Mu, Z. Yang, L. Chen, C. Gu, H. Ren and B. Wu, Suspect and non-target screening of PFAS based on IMS-MS and machine learning, *J. Hazard. Mater.*, 2024, **461**, 132669.
- 114 L. Liu, Y. Liu, B. Gao, R. Ji, C. Li and S. Wang, Removal of PFOA and PFOS from water by carbonaceous nanomaterials: a review, *Crit. Rev. Environ. Sci. Technol.*, 2020, **50**, 2379–2414.
- 115 C. Prakash, M. Mishra, P. Kumar, V. Kumar and D. Sharma, Dehydroepiandrosterone alleviates oxidative stress and apoptosis in iron-induced epilepsy via Nrf2/ARE pathway, *Brain Res. Bull.*, 2019, **153**, 181–190.
- 116 T. Ohya, N. Kudo, E. Suzuki and Y. Kawashima, Determination of perfluorinated carboxylic acids in biological samples by HPLC, *J. Chromatogr. B*, 1998, **710**, 1–7.
- 117 K. J. Hansen, L. A. Clemen, M. E. Ellefson and H. O. Johnson, Compound-specific quantitation of organic fluorochemicals in biological matrices, *Environ. Sci. Technol.*, 2001, **35**, 766–770.
- 118 J. Bao, W. Liu, L. Liu, Y. Jin, X. Ran and Z. Zhang, Perfluorinated compounds in river sediments from Guangzhou and Shanghai, *Chemosphere*, 2010, **80**, 123–130.
- 119 J. E. Naile, J. S. Khim, T. Wang, C. Chen, W. Luo, B. O. Kwon, *et al.*, Perfluorinated compounds in water, sediment, soil and biota from Korea, *Environ. Pollut.*, 2010, **158**, 1237–1244.
- 120 Y. Tian, Y. Yao, S. Chang, Z. Zhao, Y. Zhao, X. Yuan, *et al.*, Neutral and ionizable PFAS in atmosphere and plant leaves near landfills: Tianjin, China, *Environ. Sci. Technol.*, 2018, **52**, 1301–1310.
- 121 A. Nakayama, M. Yoshida, N. Kagawa and T. Nagao, Acetamidiprid and imidacloprid impair neurogenesis and



- alter microglia in mouse neonates, *J. Appl. Toxicol.*, 2019, **39**, 877–887.
- 122 T. P. Riedel, J. R. Lang, M. J. Strynar, A. B. Lindstrom and J. H. Offenber, Gas-phase detection of fluorotelomer alcohols and oxygenated PFAS by CI-MS, *Environ. Sci. Technol. Lett.*, 2019, **6**, 289–293.
- 123 J. W. Martin, S. A. Mabury, K. R. Solomon and D. C. Muir, Dietary accumulation of perfluorinated acids in juvenile trout, *Environ. Toxicol. Chem.*, 2003, **22**, 189–195.
- 124 M. Shoeib, T. Harner and P. Vlahos, Perfluorinated chemicals in Arctic atmosphere, *Environ. Sci. Technol.*, 2006, **40**, 7577–7583.
- 125 R. Alzaga, C. Salgado-Petinal, E. Jover and J. M. Bayona, Determination of perfluorocarboxylic acids in sediments by PFE-HS-SPME-GC-MS, *J. Chromatogr. A*, 2005, **1083**, 1–6.
- 126 J. F. Ayala-Cabrera, A. Contreras-Llin, E. Moyano and F. J. Santos, Determination of neutral PFAS in water by GC-APPI-HRMS, *Anal. Chim. Acta*, 2020, **1100**, 97–106.
- 127 R. Ranaweera, C. Ghafari and L. J. Luo, Bubble nucleation-based electrochemical detection of surfactants, *Anal. Chem.*, 2019, **91**, 7744–7748.
- 128 L. D. Chen, C.-Z. Lai, L. P. Granda, M. A. Fierke, D. Mandal, A. Stein, *et al.*, Fluorous membrane ion-selective electrodes for PFAS: trace-level detection and monitoring, *Anal. Chem.*, 2013, **85**, 7471–7479.
- 129 N. Karimian, A. M. Stortini, L. M. Moretto, C. Costantino, S. Bogianni and P. Ugo, Electrochemosensor for trace PFOS in water based on MIP poly(o-phenylenediamine), *ACS Sens.*, 2018, **3**, 1291–1298.
- 130 T. Zhang, H. Zhao, A. Lei and X. Quan, Electrochemical biosensor for detection of perfluorooctane sulfonate based on inhibition biocatalysis of enzymatic fuel cell, *Electrochemistry*, 2014, **82**, 94–99.
- 131 N. Cennamo, F. Arcadio, C. Perri, L. Zeni, F. Sequeira, L. Bilro, *et al.*, Water monitoring in smart cities exploiting plastic optical fibers and molecularly imprinted polymers: The case of PFBS detection, in *2019 IEEE International Symposium on Measurements & Networking (M&N)*, Catania, Italy, 2019, pp. 1–6.
- 132 J. Blotevogel, R. J. Giraud and T. Borch, Reductive defluorination of perfluorooctanoic acid by zero-valent iron and zinc: A DFT-based kinetic model, *Chem. Eng. J.*, 2018, **335**, 248–254.
- 133 P. Yin, Z. Hu, X. Song, J. Liu and N. Lin, Activated persulfate oxidation of perfluorooctanoic acid (PFOA) in groundwater under acidic conditions, *Int. J. Environ. Res. Public Health*, 2016, **13**, 602.
- 134 R. J. Watts and A. L. Teel, Chemistry of modified Fenton's reagent (catalyzed H₂O₂ propagations-CHP) for in situ soil and groundwater remediation, *J. Environ. Eng.*, 2005, **131**(4), 612–622.
- 135 S. M. Mitchell, M. Ahmad, A. L. Teel and R. J. Watts, Degradation of perfluorooctanoic acid by reactive species generated through catalyzed H₂O₂ propagation reactions, *Environ. Sci. Technol. Lett.*, 2014, **1**, 117–121.
- 136 S. Park, L. S. Lee, V. F. Medina, A. Zull and S. Waisner, Heat-activated persulfate oxidation of PFOA, 6:2 fluorotelomer sulfonate, and PFOS under conditions suitable for in-situ groundwater remediation, *Chemosphere*, 2016, **145**, 376–383.
- 137 T. A. Bruton and D. L. Sedlak, Treatment of perfluoroalkyl acids by heat-activated persulfate under conditions representative of in situ chemical oxidation, *Chemosphere*, 2018, **206**, 457–464.
- 138 T. P. C. Ezeorba, E. S. Okeke, C. E. Nwankwo, S. C. Emencheta, A. E. Enochoghene, V. C. Okeke, *et al.*, Emerging eco-friendly technologies for remediation of per- and poly-fluoroalkyl substances (PFAS) in water and wastewater: A pathway to environmental sustainability, *Chemosphere*, 2024, **364**, 143168.
- 139 T. D. Appleman, C. P. Higgins, O. Quiñones, B. J. Vanderford, C. Kolstad, J. C. Zeigler-Holady, *et al.*, Treatment of poly- and perfluoroalkyl substances in U.S. full-scale water treatment systems, *Water Res.*, 2020, **51**, 246–255.
- 140 Á. Soriano, D. Gorri and A. Urriaga, Efficient treatment of perfluorohexanoic acid by nanofiltration followed by electrochemical degradation of the NF concentrate, *Water Res.*, 2017, **112**, 147–156.
- 141 V. Franke, P. McCleaf, K. Lindgren and L. Ahrens, Efficient removal of per- and polyfluoroalkyl substances in drinking water treatment: nanofiltration combined with active carbon or anion exchange, *Environ. Sci.:Water Res. Technol.*, 2019, **5**, 1836–1843.
- 142 B. Xu, S. Liu, J. L. Zhou, C. Zheng, J. Weifeng, B. Chen, *et al.*, PFAS and their substitutes in groundwater: Occurrence, transformation and remediation, *J. Hazard. Mater.*, 2021, 125159.
- 143 M. N. Taher, S. A. Al-Mutwalli, S. Barisci, D. Y. Koseoglu-Imer, L. F. Dumée and M. M. Shirazi, Progress on remediation of PFAS from water and wastewater using membrane technologies: A review, *J. Water Process Eng.*, 2024, **59**, 104858.
- 144 A. O. Örnheim Gal, M. E. Åström and M. E. Filipsson, A review of selected alternative remediation methods for PFAS contamination, *Remediat. J.*, 2025, **35**(2), e70007.
- 145 W. Yu, J. Teng, X. Cai, L. Shen, G. Yu, B. Li, *et al.*, Advanced membrane technology for PFAS removal: Innovations in membrane materials and hybrid systems, *J. Environ. Manage.*, 2025, **394**, 127384.
- 146 A. Boonya-atichart, S. K. Boontanon and N. Boontanon, Removal of perfluorooctanoic acid in groundwater by nanofiltration membrane, *Water Sci. Technol.*, 2016, **74**, 2627–2633.
- 147 T. Jin, M. Peydayesh and R. Mezzenga, Membrane-based technologies for PFAS removal from water: Removal mechanisms, applications, challenges and perspectives, *Environ. Int.*, 2021, **157**, 106876.
- 148 C. Y. Tang, Q. S. Fu, A. P. Robertson, C. S. Criddle and J. O. Leckie, Use of reverse osmosis membranes to remove PFOS from semiconductor wastewater, *Environ. Sci. Technol.*, 2006, **40**, 7343–7349.
- 149 E. Steinle-Darling and M. Reinhard, Nanofiltration for trace organic contaminant removal: membrane fouling effects



- on rejection of perfluorochemicals, *Environ. Sci. Technol.*, 2008, **42**, 5292–5297.
- 150 S. Yadav, H. Saleem, I. Ibrar, O. Naji, A. A. Hawari, A. A. Alanezi, *et al.*, Recent developments in forward osmosis membranes using carbon-based nanomaterials, *Desalination*, 2020, **482**, 114375.
- 151 S. Thompson, A. M. Gutierrez, J. Bukowski and D. Bhattacharyya, Microfiltration membrane pore functionalization with primary and quaternary amines for PFAS remediation: Capture, regeneration and reuse, *Molecules*, 2024, **29**(17), 4229.
- 152 C. Y. Tang, Q. S. Fu, C. S. Criddle and J. O. Leckie, Effect of flux and membrane properties on fouling and rejection treating PFOS-containing wastewater, *Environ. Sci. Technol.*, 2007, **41**(6), 2008–2014.
- 153 P. Podkoscielny, A. Dabrowski and O. Marijuk, Heterogeneity of active carbons in adsorption of phenol aqueous solutions, *Appl. Surf. Sci.*, 2003, **205**, 297–303.
- 154 P. A. Quinlivan, L. Li and D. R. U. Knappe, Effects of activated carbon characteristics on adsorption of micropollutants and natural organic matter, *Water Res.*, 2005, **39**, 1663–1673.
- 155 H. Vo, H. Ngo, W. Guo, T. M. Nguyen, J. Li, H. Liang, *et al.*, Poly and perfluoroalkyl substances in water and wastewater: A comprehensive review, *J. Water Process Eng.*, 2020, **36**, 101393.
- 156 S. Bharti, PFAS in water systems: A critical review, *Water Conserv. Sci. Eng.*, 2025, **10**(2), 1–24.
- 157 L. Lee, *Quantification of in situ Chemical Reductive Defluorination of Perfluoroalkyl Acids in Groundwater Impacted by AFFFs*, ER-2426, 2014.
- 158 J. Yu, J. Y. Hu, S. Tanaka and S. Fujii, PFOS and PFOA in sewage treatment plants, *Water Res.*, 2009, **43**(9), 2399–2408.
- 159 Q. Yu, R. Q. Zhang, S. B. Deng, J. Huang and G. Yu, Sorption of PFOS and PFOA on activated carbons and resin, *Water Res.*, 2009, **43**(4), 1150–1158.
- 160 S. Senevirathna, S. Tanaka, S. Fujii, C. Kunacheva, H. Harada, B. Ariyadasa, *et al.*, Adsorption of PFOS onto non-ion exchange polymers and GAC: Batch & column test, *Desalination*, 2010, **260**, 29–33.
- 161 Z. W. Du, S. B. Deng, Y. G. Chen, B. Wang, J. Huang, Y. J. Wang, *et al.*, Removal of perfluorinated carboxylates using activated carbons and resins, *J. Hazard. Mater.*, 2015, **286**, 136–143.
- 162 Y. Zhi and J. X. Liu, Adsorption of perfluoroalkyl acids by carbonaceous adsorbents: effect of surface chemistry, *Environ. Pollut.*, 2015, **202**, 168–176.
- 163 Y. Hu, M. Guo, S. Zhang, W. Jiang, T. Xiu, S. Yang, *et al.*, Microwave synthesis of MOF absorbents for PFOS & PFOA removal, *Microporous Mesoporous Mater.*, 2022, **333**, 111740.
- 164 S. Patel, M. H. Marzbali, I. G. Hakeem, G. Veluswamy, N. Rathnayake, K. Nahar, *et al.*, Production of H₂ and CNM via biochar and PFAS adsorption, *Waste Manag.*, 2023, **159**, 146–153.
- 165 B. A. Mohamed, N. R. Nicomel, H. Hamid and L. Y. Li, Sludge-based activated carbon optimization for PFAS removal, *Sci. Total Environ.*, 2023, **874**, 162392.
- 166 I. M. Militao, F. Roddick, L. Fan, L. C. Zepeda, R. Parthasarathy and R. Bergamasco, PFAS removal by alginate-encapsulated albumin & biochar, *J. Water Process Eng.*, 2023, **53**, 103616.
- 167 J. Pala, T. Le, M. Kasula and M. R. Esfahani, PFOS adsorption by MOFs and composites, *Sep. Purif. Technol.*, 2023, **309**, 123025.
- 168 R. Amen, H. Bashir, I. Bibi, S. M. Shaheen, N. K. Niazi, M. Shahid, *et al.*, A review on arsenic removal using biochar-based sorbents, *Chem. Eng. J.*, 2020, **396**, 125195.
- 169 B. Teng, Z. Zhao, L. Xia, J. Wu and H. Wang, Iron-modified biochar for PFAS removal, *Chemosphere*, 2024, **369**, 143844.
- 170 X. Li, Q. Shen, D. Zhang, X. Mei, W. Ran, Y. Xu, *et al.*, Functional groups determine biochar properties, *PLoS One*, 2013, **8**, e65949.
- 171 D. Liang, C. Li, H. Chen, E. Sørmo, G. Cornelissen, Y. Gao, *et al.*, Biochar for remediation of PFAS-contaminated soil and water, *Sci. Total Environ.*, 2024, **951**, 174962.
- 172 M. C. Ndoun, H. A. Elliott, H. E. Preisendanz, C. F. Williams, A. Knopf and J. E. Watson, Adsorption of pharmaceuticals using biochar, *Biochar*, 2021, **3**, 89–104.
- 173 M. Inyang, B. Gao, Y. Yao, Y. Xue, A. R. Zimmerman, P. Pullammanappallil, *et al.*, Heavy metal removal by biochars, *Bioresour. Technol.*, 2012, **110**, 50–56.
- 174 Z. Wang, A. Alinezhad, R. Sun, F. Xiao and J. J. Pignatello, Thermal treatment strategies for biochar reactivation to remove PFAS, *ACS ES&T Eng.*, 2023, **3**, 193–200.
- 175 K. M. Krahn, G. Cornelissen, G. Castro, H. P. Arp, A. G. Asimakopoulos, R. Wolf, *et al.*, Sewage sludge biochars as PFAS-sorbents, *J. Hazard. Mater.*, 2023, **445**, 130449.
- 176 J. M. Steigerwald, S. Peng and J. R. Ray, PFOS-imprinted polymer on coffee-grounds biochar, *ACS ES&T Eng.*, 2023, **3**, 520–532.
- 177 J. Fabregat-Palau, M. Vidal and A. Rigol, Sorption of PFAS in biochars and carbon-rich materials, *Chemosphere*, 2022, **302**, 134733.
- 178 Y. Wu, L. Qi and G. Chen, Mechanical investigation of PFOA adsorption by engineered biochar, *J. Cleaner Prod.*, 2022, **340**, 130742.
- 179 P. M. Rodrigo, C. Navarathna, M. T. Pham, S. J. McClain, S. Stokes, X. Zhang, *et al.*, Sorption of PFAS on Douglas fir biochar and hybrids, *Chemosphere*, 2022, **308**, 136155.
- 180 X. Cheng, J. Tang, Y. Chen, X. Bai, Y. Liao, X. Ouyang, *et al.*, Dual-functional monomer imprinted polymer for electrochemical PFAS detection, *J. Hazard. Mater.*, 2025, 138422.
- 181 X. Chen, X. H. Xia, X. L. Wang, J. P. Qiao and H. T. Chen, Sorption of PFOS by chars, ash and CNTs, *Chemosphere*, 2011, **83**(10), 1313–1319.
- 182 F. M. Cao, L. Wang, X. H. Ren and H. W. Sun, PFOA molecularly imprinted polymer for selective removal, *J. Appl. Polym. Sci.*, 2016, **133**(15), DOI: [10.1002/app.43192](https://doi.org/10.1002/app.43192).



Review

- 183 F. Xiao, Emerging PFAS in the aquatic environment: Review, *Water Res.*, 2017, **124**, 482–495.
- 184 A. Zaggia, L. Conte, L. Falletti, M. Fant and A. Chiorboli, Removal of PFAS using strong anion exchange resins, *Water Res.*, 2016, **91**, 137–146.
- 185 Y. Gao, Y. Liang, K. Gao, Y. Wang, C. Wang, J. Fu, *et al.*, PFAS in soil, groundwater and tap water around a manufactory in China, *Chemosphere*, 2019, **227**, 305–314.
- 186 B. Ji, P. Kang, T. Wei and Y. Zhao, Challenges of aqueous PFASs and foreseeable removal strategies, *Chemosphere*, 2020, 126316.
- 187 J. Niu, Y. Li, E. Shang, Z. Xu and J. Liu, Electrochemical oxidation of perfluorinated compounds in water, *Chemosphere*, 2016, **146**, 526–538.
- 188 C. E. Schaefer, C. Andaya, A. Burant, C. W. Condee, A. Urriaga, T. J. Strathmann, *et al.*, Electrochemical treatment of PFOA and PFOS: Mechanisms and application, *Chem. Eng. J.*, 2017, **317**, 424–432.
- 189 B. Gomez-Ruiz, P. Ribao, N. Diban, M. J. Rivero, I. Ortiz and A. Urriaga, Photocatalytic degradation of PFOA using TiO₂-rGO, *J. Hazard. Mater.*, 2018, **344**, 950–957.
- 190 D. A. Grunfeld, A. M. Jones, J. Sun, S. T. Le, R. Pickford, Q. Huang, *et al.*, Electrochemical degradation of a C6-PFAS using activated carbon cathode, *Environ. Sci.:Water Res. Technol.*, 2024, **10**(1), 272–287.
- 191 M. Ahmadi, O. Adeleye, G. Vonalt and M. Nazemi, Identifying sweet spots for redox reactions in electrochemical advanced oxidation and reduction processes for pollutant degradation and resource recovery, *Environ. Sci. Technol.*, 2025, **59**(34), 17952–17977.
- 192 Y. Liu, X. M. Hu, Y. Zhao, J. Wang, M. X. Lu, F. H. Peng, *et al.*, Removal of PFOA with different electrodes by electrocoagulation, *Chemosphere*, 2018, **201**, 303–309.
- 193 Q. Zhuo, S. Deng, B. Yang, J. Huang and G. Yu, Electrochemical oxidation of PFOA using Ti/SnO₂-Sb-Bi anode, *Environ. Sci. Technol.*, 2011, **45**, 2973–2979.
- 194 Q. F. Zhuo, S. B. Deng, B. Yang, J. Huang, B. Wang, T. T. Zhang, *et al.*, Degradation of perfluorinated compounds on BDD electrode, *Electrochim. Acta*, 2012, **77**, 17–22.
- 195 B. D. Turner, S. W. Sloan and G. R. Currell, Remediation of PFAS using hemp protein powder, *Chemosphere*, 2019, **229**, 22–31.
- 196 S. Huang and P. R. Jaffé, Defluorination of PFOA and PFOS by *Acidimicrobium* sp. Strain A6, *Environ. Sci. Technol.*, 2019, **53**, 11410–11419.
- 197 M. Crimi, *In situ Treatment Train for PFAS-Contaminated Groundwater: ISCO-SC. ER-2423*, 2014.
- 198 H. Lee, A. G. Tevlin, S. A. Mabury and S. A. Mabury, Fate of polyfluoroalkyl phosphate diesters in biosolids-applied soil, *Environ. Sci. Technol.*, 2014, **48**(1), 340–349.
- 199 D. Lu, S. Sha, J. Luo, Z. Huang and X. J. Zhang, Treatment train approaches for PFAS remediation: A critical review, *J. Hazard. Mater.*, 2020, **386**, 121963.
- 200 S. Rayne and K. Forest, Perfluoroalkyl sulfonic and carboxylic acids: a critical review of physicochemical properties, levels and patterns in waters and wastewaters, and treatment methods, *J. Environ. Sci. Health*, 2009, **44**, 1145–1199.
- 201 Y. Ke, J. Chen, X. Hu, T. Tong, J. Huang and S. Xie, Emerging perfluoroalkyl substance impacts soil microbial community and ammonia oxidation, *Environ. Pollut.*, 2020, **257**, 113615.
- 202 B. L. Hu, L. D. Shen, X. Y. Xu and P. Zheng, Anaerobic ammonium oxidation (anammox) in different natural ecosystems, *Biochem. Soc. Trans.*, 2011, **39**(6), 1811–1816.
- 203 L. M. Colosi, R. A. Pinto, Q. Huang and W. J. Weber Jr, Peroxidase mediated degradation of perfluorooctanoic acid, *Environ. Toxicol. Chem.*, 2009, **28**, 264–271.
- 204 J. Liu and S. M. Avendano, Microbial degradation of polyfluoroalkyl chemicals in the environment: a review, *Environ. Int.*, 2013, **61**, 98–114.
- 205 L. P. Wackett, Nothing lasts forever: understanding microbial biodegradation of polyfluorinated compounds and perfluorinated alkyl substances, *Microb. Biotechnol.*, 2022, **15**(3), 773–792.
- 206 C. M. Smorada, M. W. Sima and P. R. Jaffé, Bacterial degradation of perfluoroalkyl acids, *Curr. Opin. Biotechnol.*, 2024, **88**, 103170.
- 207 B. D. Key, R. D. Howell and C. S. Criddle, Defluorination of organofluorine sulphur compounds by *Pseudomonas* sp. Strain D2, *Environ. Sci. Technol.*, 1998, **32**, 2283–2287.
- 208 M. H. Kim, *Factors Affecting Biodefluorination of Fluorotelomer Alcohols (FTOHs): Degradative Microorganisms, Transformation Metabolites and Pathways, and Effects of Co-substrates*, Texas A&M University, College Station (TX), 2012.
- 209 N. Tseng, *Feasibility of Biodegradation of Polyfluoroalkyl and Perfluoroalkyl Substances*, UCLA, Los Angeles (CA), 2012.
- 210 M. F. Khan, G. J. Paul, R. K. Thampi, J. A. Sullivan and C. D. Murphy, Enhanced removal of perfluorooctanoic acid with sequential photocatalysis and fungal treatment, *Environ. Sci. Pollut. Res.*, 2023, **30**(39), 91478–91486.
- 211 K. H. Kucharzyk, R. Darlington, M. Benotti, R. Deeb and E. Hawley, Novel treatment technologies for PFAS compounds: A critical review, *J. Environ. Manage.*, 2017, **204**(Pt 2), 757–764.
- 212 S. Kuppusamy, T. Palanisami, M. Megharaj, K. Venkateswarlu and R. Naidu, Ex-situ remediation technologies for environmental pollutants: a critical perspective, *Rev. Environ. Contam. Toxicol.*, 2016, **236**, 117–192.
- 213 S. R. Vishnoi, and P. N. Srivastava, Phytoremediation – green for environmental clean, in *Proceedings of Taal 2007: 12th World Lake Conference*, 2007, pp. 1016–1021.
- 214 M. Arslan, A. Imran, Q. M. Khan and M. Afzal, Plant-bacteria partnerships for the remediation of persistent organic pollutants, *Environ. Sci. Pollut. Res.*, 2017, **24**(5), 4322–4336.
- 215 S. Dalahmeh, S. Tirgani, A. J. Komakech, C. B. Niwagaba and L. Ahrens, Per- and polyfluoroalkyl substances (PFASs) in water, soil and plants in wetlands and agricultural areas in Kampala, Uganda, *Sci. Total Environ.*, 2018, **631**, 660–667.



- 216 B. Wen, Y. Wu, H. Zhang, Y. Liu, X. Hu, H. Huang, *et al.*, The roles of protein and lipid in the accumulation and distribution of PFOS and PFOA in plants grown in biosolids-amended soils, *Environ. Pollut.*, 2016, **216**, 682–688.
- 217 M. W. Sima and P. R. Jaffé, A critical review of modeling poly- and perfluoroalkyl substances (PFAS) in the soil-water environment, *Sci. Total Environ.*, 2021, **757**, 143793.
- 218 T. T. Wang, G. G. Ying, L. Y. He, Y. S. Liu and J. L. Zhao, Uptake mechanism, subcellular distribution, and uptake process of PFOA and PFOS by wetland plant *Alisma orientale*, *Sci. Total Environ.*, 2020, **733**, 139383.
- 219 B. Wen, L. Li, Y. Liu, H. Zhang, X. Hu, X. Q. Shan, *et al.*, Mechanistic studies of PFOS, PFOA uptake by maize (*Zea mays* L. cv. TY2), *Plant Soil*, 2013, **370**(1), 345–354.
- 220 W. Zhang, D. Zhang and Y. Liang, Nanotechnology in remediation of water contaminated by poly- and perfluoroalkyl substances: A review, *Environ. Pollut.*, 2019, **247**, 266–276.
- 221 W. Zhang and Y. Liang, Removal of eight perfluoroalkyl acids from aqueous solutions by aeration and duckweed, *Sci. Total Environ.*, 2020, **724**, 138357.
- 222 J. Pratas, C. Paulo, P. J. Favas and P. Venkatachalam, Potential of aquatic plants for phytofiltration of uranium-contaminated waters in laboratory conditions, *Ecol. Eng.*, 2014, **69**, 170–176.
- 223 B. E. Pivetz, *Phytoremediation of Contaminated Soil and Ground Water at Hazardous Waste Sites*, US EPA, Washington (DC), 2001.
- 224 W. Qiao, R. Li, T. Tang and A. A. Zuh, Removal, distribution and plant uptake of PFOS in a simulated constructed wetland system, *Front. Environ. Sci. Eng.*, 2021, **15**(2), 20.
- 225 N. Pi, J. Z. Ng and B. C. Kelly, Uptake and elimination kinetics of PFAS in aquatic macrophytes: mesocosm experiments with *Echinodorus horemanii* and *Eichhornia crassipes*, *Water Res.*, 2017, **117**, 167–174.
- 226 J. B. N. Mudumbi, S. K. O. Ntwampe, M. Muganza and J. O. Okonkwo, Susceptibility of riparian wetland plants to PFOA accumulation, *Int. J. Phytorem.*, 2014, **16**(9), 926–936.
- 227 L. Gobelius, J. Lewis and L. Ahrens, Plant uptake of PFAS at a contaminated fire training facility to evaluate phytoremediation potential, *Environ. Sci. Technol.*, 2017, **51**(21), 12602–12610.
- 228 D. Q. Zhang, M. Wang, Q. He, X. Niu and Y. Liang, Distribution of PFASs in aquatic plant-based systems: from soil adsorption and plant uptake to effects on microbial community, *Environ. Pollut.*, 2020, **257**, 113575.
- 229 A. Lee, J. K. Choe, K. D. Zoh, C. Lee and Y. Choi, Development of ionic-liquid-impregnated activated carbon for sorptive removal of PFAS in drinking water treatment, *Chemosphere*, 2024, **355**, 141872.
- 230 T. Stahl, R. A. Riebe, S. Falk, K. Failing and H. Brunn, Long-term lysimeter experiment to investigate the leaching of PFASs and the carry-over from soil to plants: pilot study results, *J. Agric. Food Chem.*, 2013, **61**(8), 1784–1793.
- 231 A. C. Blaine, C. D. Rich, E. M. Sedlacko, K. C. Hyland, C. Stushnoff, E. R. Dickenson, *et al.*, Perfluoroalkyl acid uptake in lettuce and strawberry irrigated with reclaimed water, *Environ. Sci. Technol.*, 2014, **48**(24), 14361–14368.
- 232 Y. H. Su, T. Liu and Y. C. Liang, Transport via xylem of trichloroethylene in wheat, corn, and tomato seedlings, *J. Hazard. Mater.*, 2010, **182**(1–3), 472–476.
- 233 T. Tian, H. X. Zhang, C. P. He, S. Fan, Y. L. Zhu, C. Qi, *et al.*, Surface functionalized exosomes as targeted drug delivery vehicles for cerebral ischemia therapy, *Biomaterials*, 2018, **150**, 137–149.
- 234 T. Yin, H. Chen, M. Reinhard, X. Yi, Y. He and K. Y. H. Gin, PFAS removal in a full-scale tropical constructed wetland system treating landfill leachate, *Water Res.*, 2017, **125**, 418–426.
- 235 W. Zhang, D. Zhang and Y. Liang, Nanotechnology in remediation of water contaminated by poly- and perfluoroalkyl substances: A review, *Environ. Pollut.*, 2019, **247**, 266–276.
- 236 M. Andrunik and M. Smol, Life after adsorption: regeneration, management, and sustainability of PFAS adsorbents in water treatment, *Water*, 2025, **17**(19), 2813.

