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A comprehensive overview on the occurrence and removal of *per*- and polyfluoroalkyl substances through adsorption and biodegradation

Muhammad Zeeshan [a,b,c,*], Shamas Tabraiz [d], Safeerul Islam Hashmi [e], Arfa Iqbal [f], Daniel Dittmann [a], Zohaib Abbas [g], Cecilia L. MacLeod [h,i], Aki Sebastian Ruhl [a,b]

[a] German Environment Agency, Section II 3.3, Schichauweg 58, 12307, Berlin, Germany
[b] Technische Universität Berlin, Water Treatment, KF4, Str. des 17. Juni 135, 10623, Berlin, Germany
[c] Berlin School of Business and Innovation, Alte Post, Karl-Marx-Straße 97-99, 12043, Berlin, Germany
[d] Department of Civil and Environmental Engineering, Imperial College London, United Kingdom
[e] School of Aquatic Sciences, Fisheries and Aquaculture, Auburn University, Alabama, United States
[f] Institute of Environmental Engineering and Research, University of Engineering and Technology, Lahore, Pakistan
[g] Department of Environmental Science and Engineering, Government College University, Faisalabad, Pakistan
[h] School of Engineering, University of Greenwich, Chatham Maritime, Kent, United Kingdom
[i] Thrive Sciences Limited, United Kingdom

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS), known for their exceptional stability and hydrophobic properties, have become prominent environmental contaminants due to their persistence and toxicity. This review provides a comprehensive analysis of PFAS occurrence in groundwater and surface waters, their degradation by various microbial species and the effectiveness of different adsorbents in PFAS removal. Microbial degradation is a cost-effective and environmentally friendly method for PFAS removal, with aerobic biotransformation being more widely studied. Microbial strains, including *Acidimicrobium* sp. A6, *Pseudomonas*, and *Gordonia* sp. showed sustainable reduction (up to 99 %) in PFAS concentrations. Under aerobic and anaerobic conditions, microbial mechanisms differ significantly, requiring specific microbial strains or engineered systems to break the strong C-F bonds. Various adsorbents, such as carbonaceous materials, ion exchange resins, and other synthetic materials, have been used to remove PFAS from water. Positively charged adsorbents were more effective in removing PFAS than neutral or negatively charged ones. Ion exchange resins outperform other adsorbents in removing both long and short-chain PFAS. This review outlines significant research needs, including the need to understand the complex interactions between dissolved organic matter and PFAS removal, as well as the potential of advanced materials to improve adsorption processes. Future research should focus on scalable, cost-effective, and environmentally sustainable methods to reduce PFAS contamination and provide safer water resources for future generations.

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a group of synthetic organic chemicals widely used in various industrial and consumer products, such as non-stick cookware, water-repellent clothing, and fire-fighting foams (Calvert et al., 2022; Glüge et al., 2020; Sznajder-Katarzyńska et al., 2019). Some important substances of the PFAS group have a typical hydrophobic carbon chain of varying lengths (Table 1), with hydrogen atoms that are either partially (in the case of polyfluoroalkyl substances) or completely (perfluoroalkyl substances) replaced by fluorine atoms (Buck et al., 2011). Some well-known PFAS are classified based on their terminal functional groups, followed by chain length (long- and short-chain), as shown in Table 1. PFAS are highly resistant to degradation and remain in the environment for longer periods because they are hydrophilic and chemically and biologically stable (Cousins et al., 2020; Glüge et al., 2020; Henry et al., 2018). The occurrence of PFAS in the environment (air, water and soil) has been widely reported in recent years (Brusseau et al., 2020; Kurwadkar et al., 2022; Nakayama et al., 2019). Certain PFAS bioaccumulate in organisms, potentially leading to toxic levels in humans and other organisms

* Corresponding author at: German Environment Agency, Section II 3.3, Schichauweg 58, 12307, Berlin, Germany.
E-mail address: engr.m.zeeshan@live.com (M. Zeeshan).

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(Abunada et al., 2020; Brendel et al., 2018; Ma et al., 2022; Parolini et al., 2022). Several studies have reported adverse health effects of PFAS exposure, such as impaired kidney function, thyroid disease, and pregnancy complications (Blake and Fenton, 2020; DeWitt et al., 2019).

The high detection rate of PFAS due to advanced analytical methods means that they can persist in water bodies (Khan et al., 2023a; Liu et al., 2015; Nakayama et al., 2019). The development of the PFAS TOP (total oxidizable precursor) assay has helped to learn more about the precursors that may form terminal PFAS during degradation and to identify the source (Antell et al., 2023). Also, total organofluorine analysis might be a useful way to look for PFAS in general without finding specific PFAS (McDonough et al., 2019). Nevertheless, numerous countries have no drinking water monitoring scheme that could identify persistent and mobile organic substances (Montes et al., 2019; van der Hoek et al.,

Table 1
Classification of well-known PFAS, including abbreviations, chemical abstract service (CAS) number, molecular formula (MF) and the physio-chemical properties (molecular weight (MW in g/mol), solubility (in mol/L), n-octanol-water partitioning coefficient (log K_{ow}), organic carbon to water partition coefficient (log K_{oc}) and acid dissociation constant pKa) (Gagliano et al., 2020; Pauletto and Bandosz, 2022; Rahman et al., 2014; Saawarn et al., 2022).

Class	Compounds (abbreviations)	CAS No.	MF (MW)	Solubility	log K_{ow}	log K_{oc}	pK_a
Short-Chain (C_4-C_7) perfluoroalkyl carboxylic acids (PFCA)	Perfluorobutanoic acid (PFBA)	375–22-4	C_3F_7COOH (214.04)	2.1×10^{-3}	2.31	–	1.07
	Perfluoropentanoic acid (PFPeA)	2706-90-3	C_4F_9COOH (264.05)	4.5×10^{-4}	3.01	–	0.34
	Perfluorohexanoic acid (PFHxA)	307–24-4	$C_5F_{11}COOH$ (314.06)	9.3×10^{-5}	23.48	–	−0.16 −0.78
	Perfluoroheptanoic acid (PFHpA)	375–85-9	$C_6F_{13}COOH$ (364.07)	0.3	4.15	–	−2.29
Long- Chain (C_8-C_{14}) perfluoroalkyl carboxylic acids (PFCA)	Perfluorooctanoic acid (PFOA)	335–67-1	$C_7F_{15}COOH$ (414.07)	0.4×10^{-2}	4.81	1.47	−0.5 - 4.2
	Perfluorononanoic acid (PFNA)	375–95-1	$C_8F_{17}COOH$ (464.08)	2.8×10^{-3}	5.48	2.06	−6.51
	Perfluorodecanoic acid (PFDA)	335–76-2	$C_9F_{19}COOH$ (514.09)	5.3×10^{-3}	6.51	2.37	−5.2
	Perfluoroundecanoic acid (PFUnDA)	2058-94-8	$C_{10}F_{21}COOH$ (564.09)	1.6×10^{-4}	7.21	2.32	−5.2
	Perfluorododecanoic acid (PFDoDA)	307–55-1	$C_{11}F_{23}COOH$ (614.10)	–	7.92	–	−5.2
	Perfluorotridecanoic acid (PFTrDA)	72,629–94-8	$C_{12}F_{25}COOH$ (664.10)	–	8.62	–	−5.2
	Perfluorotetradecanoic acid (PFTeDA)	376–06-7	$C_{13}F_{27}COOH$ (714.12)	–	9.32	–	−5.2
Short-chain (C_4-C_5) perfluoroalkyl sulfonic acids (PFSA)	Perfluorobutane sulfonic acid (PFBS)	375–73-5	$C_4F_9SO_3H$ (300.09)	1.7×10^{-3}	1.82	–	−3.31
	Perfluoropentane sulfonic acid (PFPeS)	2706-91-4	$C_5F_{11}SO_3H$ (350.10)	3.1×10^{-2}	3.33	–	−3.32
	Perfluorohexane sulfonic acid (PFHxS)	355–46-4	$C_5F_{11}SO_3H$ (400.11)	6.1×10^{-4}	3.16	0.97	−3.32
	Perfluoroheptane sulfonic acid (PFHpS)	375–92-8	$C_7F_{15}SO_3H$ (450.12)	–	4.73	–	−3.32
Long- Chain (C_6-C_{14}) perfluoroalkyl sulfonic acids (PFSA)	Perfluorooctane sulfonic acid (PFOS)	1763-23-1	$C_8F_{17}SO3H$ (500.13)	1.1×10^{-3}	4.49	2.10	−3.32
	Perfluorononane sulfonic acid (PFNS)	474,511–07-4	$C_9F_{19}SO_3H$ (549.13)	–	6.13	–	−3.24
	Perfluorodecane sulfonic acid (PFDS)	335–77-3	$C_{10}F_{21}SO_3H$ (600.15)	–	6.83	–	−3.24
	Perfluoroundecane sulfonic acid (PFUnDS)	749,786–16-1	$C_{11}F_{23}SO_3H$ (650.15)	–	–	–	–
	Perfluorododecane sulfonic acid (PFDoDS)	79,780–39-5	$C_{12}F_{25}SO_3H$ (700.16)	–	8.23	–	−3.24
	Perfluorotridecane sulfonic acid (PFTrDS)	791,563–89-8	$C_{13}F_{27}SO_3H$ (750.17)	–	–	–	–
	Perfluorotetradecane sulfonic acid (PFTeDS)	1,379,460–39-5	$C_{14}F_{29}SO_3H$ (800.18)	–	–	–	–
Perfluoroalkane sulfonamides (FASA)	Perfluorooctane sulphonamide (FOSA)	754–91-6	$C_8H_2F_{17}NO_2S$ (499.1)	1.75	5.8	2.56	3.37
	Methyl perfluorooctane sulphonamide (MeFOSA)	31,506–32-8	$C_9H_4F_{17}NO_2S$ (513.2)	–	–	–	–
	Ethyl perfluorooctane sulphonamide (EtFOSA)	4151–50-2	$C_{10}H_6F_{17}NO_2S$ (527.2)	–	–	–	–
	Ethyl perfluorooctane sulfonamido ethanol (EtFOSE)	1691-99-2	$C_{12}H_{10}F_{17}NO_3S$ (571.3)	2.6×10^{-3}	6.96	–	–
	Methyl perfluorooctane sulfonamido ethanol (MeFOSE)	24,448–09-7	$C_{11}H_8F_{17}NO_3S$ (557.3)	1.5×10^{-6}	4.58	–	–
	Methyl perfluorooctane sulfonamido ethyl acrylate (MeFOSEA)	25,268–77-3	$C_{14}H_{10}F_{17}NO_4S$ (611.3)	–	–	–	–
Fluorotelomer alcohols (FTOH)	4:2 Fluorotelomer alcohol (4:2 FTOH)	2043-47-2	$C_6H_5F_9O$ (264.1)	2.7×10^{-3}	3.27	0.93	–
	6:2 Fluorotelomer alcohol (6:2 FTOH)	647–42-7	$C_8H_5F_{13}O$ (364.1)	4.8×10^{-5}	4.58	2.43	–
	10:2 Fluorotelomer alcohol (10:2 FTOH)	865–86-1	$C_{12}H_5F_{21}O$ (564.1)	8.5×10^{-8}	7.10	6.20	–

2014).

The United States Environmental Protection Agency (USEPA) set new maximum concentration levels for six PFAS in drinking water in April 2024. The concentrations of 4 ng/L of PFOS and PFOA, 10 ng/L of PFHxS and PFNA, 2000 ng/L of PFBS, and 10 ng/L of hexafluoropropylene oxide dimer acid or GenX are set as maximum levels. The regulation is extremely rigid to reduce exposure to persistent and potentially toxic substances and meet corresponding health hazards. It is challenging to decrease PFAS concentrations below permitted levels through conventional treatments like the activated sludge process due to the properties and stability of PFAS (Wanninayake, 2021). It is important to employ appropriate treatment methods to remove toxic substances like PFAS from drinking water, wastewater, or water in contaminated areas.

Several studies evaluated various physical and chemical processes for PFAS treatment, including advanced reduction processes, advanced oxidation processes (e.g., electrochemical oxidation, UV-induced oxidation, and supercritical water oxidation), membrane processes, adsorption using activated carbon or nanoparticles, pyrolysis/gasification, ion exchange, plasma-based processes, incineration, and sonochemical degradation (Ahmed et al., 2020; Ateia et al., 2019a; Barisci and Suri, 2021; Berg et al., 2022; Boyer et al., 2021; Saawarn et al., 2022). For example, electrochemical and UV-induced oxidation are highly effective in removing long- and short-chain PFAS but have extremely high energy requirements (Zhang et al., 2022). Dirani et al. (2024) reported that short-chain PFAS can better be removed using nanomaterial-based adsorption, e.g., iron oxide-coated biochar. However, most of these processes have very high energy requirements, making it unpracticable to employ them on a commercial scale.

Adsorption is widely recognized as a practical and effective treatment method due to its technological advancement and ease of implementation (Dirani et al., 2024; Lei et al., 2023). Several materials, such as activated carbon, clay, and sediments, have shown promising results in removing PFAS from water sources (Appleman et al., 2014; Lauwers et al., 2023; Lei et al., 2023; Medina et al., 2022; Militao et al., 2023). However, challenges exist in regenerating these adsorbents, which can incur additional costs and generate waste, and their efficiency significantly drops for short-chain PFAS due to lower adsorption affinity (Amen et al., 2023). Biodegradation has been reported to be affordable, sustainable, and capable of up to complete degradation of both long- and short-chain PFAS (Berhanu et al., 2023; Grgas et al., 2023; Wackett, 2021). Several studies investigating physicochemical and biological processes reported 62 to 100 % removal or degradation of PFAS from water (Appleman et al., 2013; Javed et al., 2020; Rahman et al., 2014; Rayne and Forest, 2009; Saawarn et al., 2022; Saleh et al., 2019). Recent advancements in bioaugmentation and bioreactor technologies have reported enhanced degradation, potentially achieving complete mineralization of PFAS (Zhang et al., 2022). Factors such as pH, temperature, and the presence of co-substrates significantly influence the biodegradation process, and incomplete degradation can result in the accumulation of intermediate by-products that may pose environmental risks (Ren et al., 2023).

This review addresses occurrences of PFAS in various water bodies worldwide. It also covers recent developments (over the last five years) in adsorption and biodegradation techniques, which remain practical and scalable options for PFAS removal. Scientific data from earlier studies conducted before the last five years were also included for comparison. Although adsorption and biodegradation methods are discussed in scientific reviews on PFAS removal technologies, these techniques represent only a small portion of the extensive research available. For example, one study explored PFAS removal using membrane filtration, advanced oxidation processes, adsorption, and microbial degradation with a brief discussion on adsorption and biodegradation processes (Saawarn et al., 2022). Another study examined the efficacy of adsorption, filtration, sonochemical destruction, and bioremediation for PFAS removal but only presented brief data on adsorption and biodegradation (Kucharzyk et al., 2017). In a recent study, Dirani et al. (2024) discussed the effectiveness of adsorption, membrane processes, and advanced oxidation methods in removing PFAS. However, the main focus of our review was to summarize possible microbial strains investigated for PFAS removal, including biological process influencing parameters. The effectiveness of ion exchange resin and carbon-based materials on PFAS removal, including process influencing parameters, was also comprehensively covered. The impact of dissolved organic matter (DOM) on adsorption and biodegradation was thoroughly examined. DOM may hinder or enhance PFAS adsorption and biodegradation (Qi et al., 2022; Xiao et al., 2020), and it exhibits seasonal fluctuations (Zeeshan et al., 2023a). Finally, this review identifies knowledge gaps and discusses research needs for effectively eliminating PFAS from water.

2. Bibliographic analysis

A thorough review of existing literature was carried out to investigate the presence of PFAS in groundwater and surface water, along with their removal using adsorption and biodegradation processes, as well as the factors that influence PFAS removal. PFAS, surface water, groundwater, biodegradation, adsorption, ion exchange resins, and regeneration were important keywords that were used to search for the most suitable papers within the Scopus database. Redundant or irrelevant literature was removed from the database. The database (with recent studies) was then used for a thorough analysis of the collected literature. Moreover, a bibliometric analysis was conducted using VOSviewer software (version 1.6.8; Fig. 1). The analysis identified a total of 603 keywords, with 50 meeting the predefined threshold of a minimum occurrence of five. The study utilized network visualization to systematically cluster the data, thereby elucidating relationships and connections among the various research topics. The network visualization map (Fig. 1a) displays connections between the nodes, representing the level of co-occurrence confidence among keywords. Additionally, the font size of the nodes is proportional to the frequency of keyword occurrence, while the linking lines between nodes indicate co-occurrences within the same articles. The network visualization (in Fig. 1a) reveals the presence of three distinct clustered groups, each denoted by a unique color code. The green cluster primarily emphasizes research related to PFAS removal through adsorption and is characterized by the terms activated carbon, biochar, and ion exchange resin. The red cluster is predominantly associated with terms related to the presence of PFAS in water systems, including water pollutants, drinking water, surface water, and groundwater. The blue cluster is centered on biodegradation, remediation, and bioaccumulation of PFAS.

Fig. 1b presents a network overlay visualization that elucidates the growing trends within PFAS research over the last five years. Studies published before 2020 primarily emphasized terms related to adsorption processes involving activated carbon and granular activated carbon. In contrast, the period from 2020 to 2023 has witnessed a discernible shift in research focus. Notably, there has been an increased emphasis on investigations concerning the prevalence of PFAS and adsorption mechanisms employing biochar and ion exchange resin. Furthermore, recent studies have displayed a growing interest in the biodegradation, remediation, and bioaccumulation of PFAS.

3. Occurrence of PFAS in the aquatic environment

Several PFAS are detected in surface waters and groundwater around the globe, with concentrations up to μg/L (Fig. S1). Seven PFAS, including PFPeA, PFOA, PFOS, PFBA, PFBS, PFHxA and PFHpA are often detected in groundwater and surface waters in many countries. The observed concentrations were higher in groundwater than in surface waters (Fig. S1). The variation is attributed to the dilution of PFAS in surface water bodies, which may also be affected by precipitation (Vo et al., 2020). For example, PFOA concentrations in China ranged from

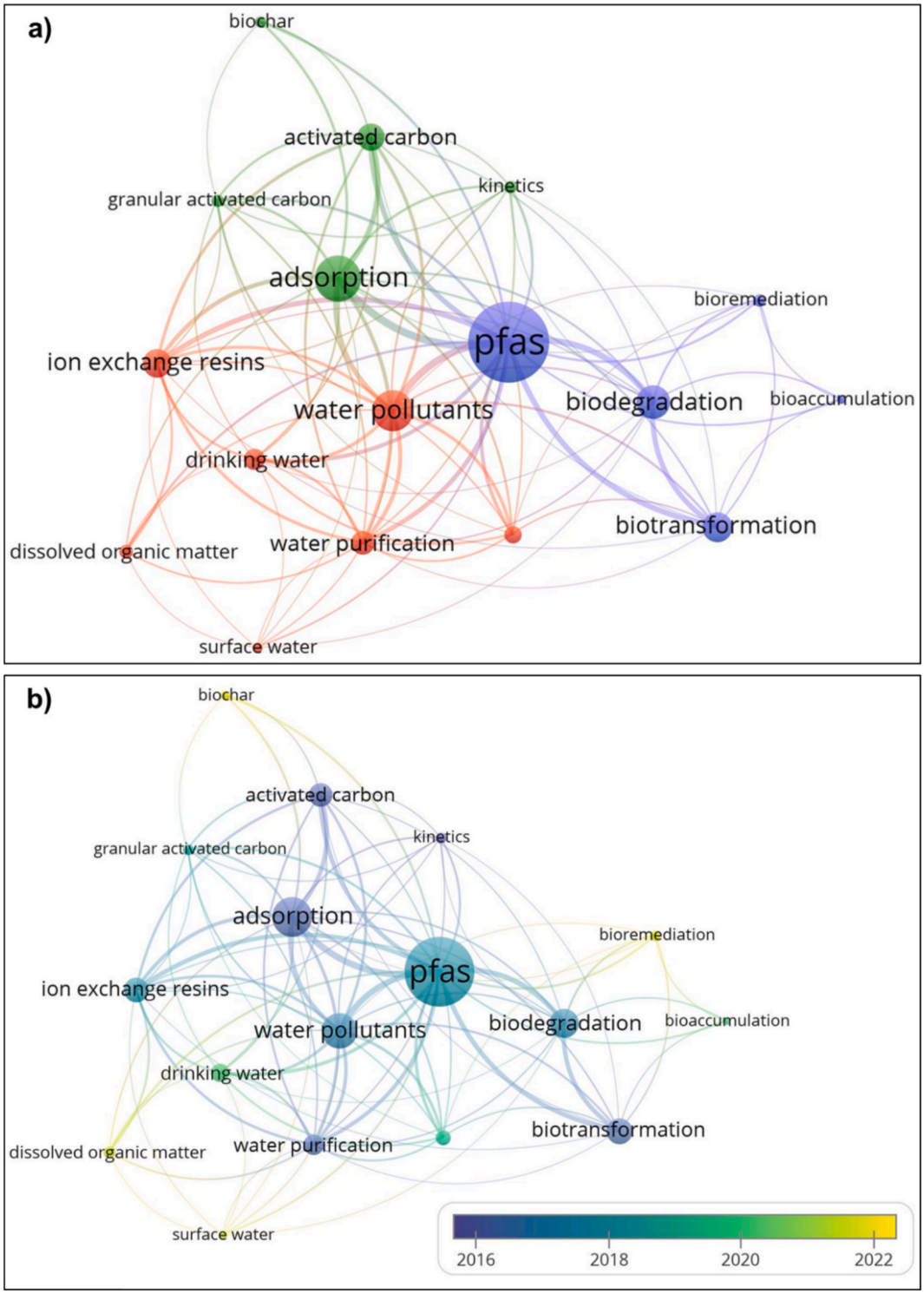

Fig. 1. Bibliometric analysis of PFAS occurrences, removal through adsorption and biodegradation, and factors affecting PFAS removal. **a)** Network visualization illustrating clusters of co-occurring terms and **b)** overlay visualization depicting literature trends over the last five years.

0.14 to 94 ng/L in surface waters (Wei et al., 2019) and from 105 to 2510 ng/L in groundwater (Bao et al., 2019).

Increased PFAS concentrations in groundwater are attributed to multiple direct sources (e.g., seepage, soil leaching, and irrigation) and indirect sources (such as air deposition, snow, and rainfall) (Wei et al., 2023; Xu et al., 2021; Yong et al., 2021). Prominent PFAS detected in groundwater are PFOA, PFHxA, PFBS, and PFBA (Fig. S1). Research in South Korea indicated that groundwater concentrations varied from non-detectable (N.D.) to 36.9 ng/L. Statistical disparities were also seen among groundwater samples collected from diverse land-use regions in South Korea. Groundwater downstream of an industrial area exhibited elevated mean concentrations of PFAS (26.4 ng/L) compared to upstream levels (4.3 ng/L) (Yong et al., 2021). Bräunig et al. (2017) conducted a study in which 10 groundwater samples from Oakey, Australia, were analyzed and found PFOS, PFHxS, PFHxA, PFBA, PFOA, PFBS, and PFPA, with PFOS exhibiting the highest mean concentration of 4300 ng/

L. Wang et al. (2022) reported the predominance of short-chain PFAS (such as PFBA, PFPeA and PFBS) in groundwater of southeastern China, with concentrations ranging from 1.27 to 381 ng/L.

The effluent discharged from many wastewater treatment facilities and runoff from non-point sources may contribute significantly to PFAS levels in surface waters (Vo et al., 2020; Wilkinson et al., 2017). Previous studies showed high concentrations of PFOA, PFOS, PFHxA, PFHpA, PFBS and PFBA in surface waters, compared to other PFAS (Fig. S1). PFOA is the most frequently detected PFAS in Finland, followed by PFHpA, PFHxA and PFOS (Junttila et al., 2019) and the total annual average PFAS concentrations in individual rivers varied between 1.8 ng/L and 42 ng/L. Bai and Son (2021) reported PFHxA, PFPeA, PFOA and PFBS as predominant PFAS in Nevada, USA. The concentration of PFAS in the Truckee River water was 441.7 ng/L, while Las Vegas Wash water contained 2234.3 ng/L of PFAS (Bai and Son, 2021). Furthermore, short-chain PFAS were predominant in surface water, whereas long-chain were prevalent in sediments.

Wei et al. (2023) found PFAS (PFOS, 6:2 FTS and 8:2 FTS) in gully pot sediment samples (in Sweden) and reported that stormwater runoff is a transport pathway for PFAS. Flanagan et al. (2021) found PFOS in the sediments of stormwater ponds, with concentrations ranging from <0.5 to 3.18 μg/kg. Across western China, notably high concentrations of PFBA, PFOS, and PFOA were reported in snow, with total PFAS concentrations varying substantially (0.88 ng/L to 3.97 ng/L) (Wang et al., 2019).

The large variations in PFAS concentrations across regions are most likely influenced by industrial activities, contamination sources, and targeted sampling in high-contamination areas, which may skew the representation of the actual distribution. For instance, Sims et al. (2022) reported that PFAS concentrations in groundwater and surface water strongly correlate with proximity to industrial areas, wastewater discharge sites, and firefighting activity zones. Localized contamination has been detected in Sweden near industrial zones, despite stringent regulatory procedures (Gobelius et al., 2018). A comparable trend is observable in Asia, where high PFAS levels are associated with untreated industrial wastewater discharges and inadequate regulatory enforcement (Tang et al., 2023). In Australia, although there are no PFAS manufacturing facilities, highly contaminated sites have been identified due to the extensive use of firefighting foam during emergency response activities (Ackerman Grunfeld et al., 2024).

Several studies reported that PFAS are either partially removed or not eliminated using conventional treatment methods (Kassar et al., 2022; Saleh et al., 2019; Schumann et al., 2023; Zeeshan et al., 2023d). For example, PFAS concentration in 77 tap water samples collected from different regions in China ranged from 4.59 ng/L to 365 ng/L (Chen et al., 2021). Li et al. (2022) reported PFBA, PFBS, PFPeA, PFHxA, PFHxS, PFOA, and PFOS in both surface waters and tap water in South Florida and the average PFAS concentration was higher in tap water (86.3 ng/L) than in surface waters (46.3 ng/L).

Reducing PFAS concentrations below permissible limits remains challenging when using conventional treatment methods due to the persistent nature of these compounds (Wanninayake, 2021). Therefore, appropriate treatment technologies are needed to effectively remove these potentially toxic substances from water supplies.

4. Biodegradation of PFAS

PFAS are recognized for their persistence in the natural environment due to their non-degradable characteristics. Several treatment methods have been employed to remove different organic pollutants from water (Zeeshan et al., 2023b, 2023c). Numerous researchers have attempted to enhance the biological degradation of PFAS (specifically PFOS and PFOA) using various microbial species but faced challenges in biotransforming PFAS (Ross et al., 2018; Shahsavari et al., 2021; von der Trenck et al., 2018; Zhang et al., 2022). Liou et al. (2010) investigated biodegradation of PFOA using five different types of microbial species collected from various sources, including sewage treatment plants, industrial site sediment, agricultural fields, and two tire training facilities and observed that PFOA was not transformed under all tested conditions. In contrast, several studies have reported biodegradation through microbial action as an effective method to degrade PFAS under both aerobic and anaerobic conditions, with aerobic biotransformation being more frequently reported (Table 2). Most available literature describes PFAS biotransformation using mixed cultures obtained from sources such as activated sludge, digestion sludge, soil, or sediments (Chiavola et al., 2020; Huang and Jaffé, 2019; Kwon et al., 2014; Li et al., 2018). This significant achievement has raised hopes for degrading these persistent compounds through biological processes, even when transformation relies heavily on the biodegradation triangle, which consists of the structural complexity of PFAS, the composition of microorganisms present, and other environmental conditions.

Huang and Jaffé (2019) found that a pure culture of *Acidimicrobium* sp. strain A6 showed significant potential to transform PFOA and PFOS at mg/L concentrations under controlled lab conditions. Ruiz-Urigüen et al. (2022) observed 77 % degradation of PFOA in a microbial electrolysis cell inoculated with *Acidimicrobiaceae* sp. In addition, fluoride as well as PFAA were produced. Huang et al. (2022) studied the anaerobic degradation of PFOA using *Acidimicrobium* sp. strain A6 in biosolids. A decrease in PFOA concentrations was observed, and this process required augmentation with A6 and ferric iron. Additionally, higher PFOA levels influenced the composition of the microbial community in the biosolids.

Several studies reported that activated sludge from sewage treatment plants can reduce the concentration of short-chain PFAS (Berhanu et al., 2023; Fincker and Spormann, 2017; Grgas et al., 2023). Schultz et al. (2006) reported that the levels of perfluoroalkylsulfonates, specifically PFOS and PFDS, increased during the activated sludge process due to the biodegradation of their precursor substances. On the other hand, Ochoa-Herrera et al. (2016) found no strong evidence of microbial degradation of perfluoroalkylsulfonates under either anaerobic or aerobic conditions when using various sources of inoculation, including sludge from different wastewater treatment plants.

The selection of microbial stains is critical for the biodegradation process. For instance, a co-culture of bacteria from aerobic sludge only degraded 6 % of 6:2 FTSA (Wang et al., 2011). In contrast, a co-culture of bacteria from aerobic sediment successfully decomposed 80 % of 6:2 FTSA (Zhang et al., 2016). Previous studies have demonstrated that different microbial strains, including *Mycobacterium vaccae* JOB5, *Pseudomonas butanovora, Gloeophyllum trabeum,* and *Trametes versicolor*, show distinct differences in their aerobic biotransformation of 6:2 FTOH and its metabolites (Kim et al., 2014; Merino et al., 2018; Tseng et al., 2014).

The microbial degradation of PFAS depends on characteristics of individual compounds. According to Wackett (2021), a microbe must (1) internalize the fluorinated compound, (2) utilize a recently evolved enzyme capable of catalyzing the cleavage of the robust C-F bonds, (3) manage the fluoride ions produced during this process, and (4) protect itself from fluoride toxicity, potentially through a fluoride-proton antiporter. These steps are crucial for PFAS biodegraders, but significant physiological challenges arise due to the lack of direct cellular benefits from PFAS biodegradation.

In contrast to dechlorination, where bacteria can use polychlorinated compounds as electron acceptors in energy metabolism, polyfluorinated compounds cannot serve this function because of their negative redox potential (Fincker and Spormann, 2017; Sun et al., 2021). Microbial degradation typically begins with the enzymatic attack on the C-F bonds by reductase enzymes, facilitating reductive defluorination, where fluorine atoms are replaced by hydrogen, leading to intermediate products like partially fluorinated hydrocarbons (Alalm and Boffito, 2022; Berhanu et al., 2023; Kumar et al., 2023). However, the fluorine shield (dense electronegative barrier of fluorine atoms) around the carbon-carbon bonds significantly impedes this process, making PFAS

Table 2
Microbial transformation of PFAS.

Compound(s), initial concentration, microbial strain	Operational conditions	Results/key findings	Reference
Compounds: PFOA, PFOS **Initial concentration:** 0.1 and 100 mg/L **Microbial strain:** *Acidmicrobium* sp. A6	**Duration:** 100 days **pH:** 4.5–5 **Temp.:** 30 °C **Redox condition:** Anaerobic	60 % degradation of PFOA and PFOS Transformation products for PFOA: perfluorobutanoic acid (HFBA), PFPeA, PFHxA, PFHpA are the transformation products of PFOA. HFBA and PFBS are the transformation products of PFOS.	(Huang and Jaffé, 2019)
Compound: PFOA **Initial concentration:** 0.2 and 10 mg/L **Microbial strain:** *Acidmicrobium* sp. A6	**Duration:** 150 days **pH:** 4.5–5 **Temp.:** 25 °C **Redox condition:** Anaerobic	50 % degradation and transformation products are PFBA, PFPeA, PFHxA, PFHpA, F^-	(Huang et al., 2022)
Compound: PFOA **Initial concentration:** 47 mg/L **Microbial strain:** *Acidmicrobium* sp. A6	**Duration:** 18 days **pH:** 5–5.5 **Redox condition:** Anaerobic	77 % reduction and transformation products are PFBA, PFPeA, PFHxA, PFHpA and F^-	(Ruiz-Urigüen et al., 2022)
Compound: PFOA **Initial concentration:** 500 mg/L **Microbial strain:** *Pseudomonas parafulva* YAB1	**Duration:** 96 days **pH:** 7 **Temp.:** 30 °C **Redox condition:** Aerobic	Up to 48.1 % reduction in PFOA with glucose supplementation	(Yi et al., 2016)
Compound: PFOS **Initial concentration:** 1.4–1.8 mg/L **Microbial strain:** *Pseudomonas aeruginosa* strain HJ4	**Duration:** 48 days **pH:** 7 **Temp.:** 35 °C **Redox condition:** Aerobic	Up to 67 % PFOS reduction, transformation products are PFBS and PFHxS (4–26 ng/L), no fluoride production	(Kwon et al., 2014)
Compound: PFOS **Initial concentration:** 1000 mg/L **Microbial strain:** *Pseudomonas plecoglossicida* 2.4-D	**Duration:** 6 days **pH:** 7 **Temp.:** 28 °C **Redox condition:** Aerobic	Complete transformation, transformation products are PFHpA and F^- release, 75 % elimination in soil within 6 months	(Chetverikov et al., 2017)
Compound: 8:2 FTOH **Initial concentration:** 0.337 mg/L **Microbial strain:** Activated sludge	**Duration:** 28 days **pH:** 7 **Redox condition:** Aerobic	Transformation products are 8:2 FTCA (27 %), 8:2 FTUCA (6 %), PFOA (2.1 %), 2H, 2H, 3H-perfluorodecanoic acid (7:3 acid, 2.3 %)	(Wang et al., 2005)
Compound: 8:2 FTOH **Initial concentration:** 100 mg/kg **Microbial strain:** Soil microcosm and pure cultures: *Pseudomonas* OCY4, *Pseudomonas* OCW	**Duration:** 7 and 60 days **Temp.:** 30 °C **Redox condition:** Aerobic	Transformation products are 7:2 sFTOH, 8:2 FTCA, PFOA and minor products (PFPeA, 7:3 FTCA, 7:3 FTUCA)	(Liu et al., 2007)
Compound: 8:2 FTOH **Initial concentration:** 40 mg/L **Microbial strain:** Bacterial culture (*Pseudomonas butanovora*, *Pseudomonas oleovorans*)	**Duration:** 28 days **pH:** 7 **Temp.:** 30 °C **Redox condition:** Aerobic	Transformation products are 8:2 FTUCA, 7:2 ketone, 7:2 sFTOH, PFOA, PFHxA	(Kim et al., 2012)
Compound: 8:2 FTOH **Initial concentration:** 0.5 mg/L **Microbial strain:** Activated WWTP sludge	**Duration:** 1 day **pH:** 7 **Redox condition:** Aerobic	Transformation products are PFBA, PFPeA, PFHxA, PFHpA, transient metabolites (7:2 sFTOH, 8:2 FTUCA)	(Yu et al., 2016)
Compound: 8:2 FTOH **Initial concentration:** 7.9 mg/L, spiked to 232 mg/L **Microbial strain:** Landfill leachate and sediment	**Duration:** 90 days **pH:** 7.9 **Temp.:** 20 °C **Redox condition:** Aerobic	Transformation products are PFOA (2.8 %), PFHxA (0.5 %), PFHpA (0.5 %), 7:3 FTCA (1.5 %), FTUCA (1.8 %)	(Hamid et al., 2020)
Compound: 8:2 FTOH **Initial concentration:** 0.147 mg/L **Microbial strain:** Activated WWTP sludge	**Duration:** 150 days **pH:** 7.5 **Redox condition:** Anaerobic	Transformation products are 8:2 FTUA, 8:2 FTUCA, 7:2 sFTOH, 7:3 acid, terminal: PFOA (17 %), PFPeA (1.2 %), PFBA (1.9 %), PFHxA (5.4 %), PFHpA (8.9 %), F^- release	(Li et al., 2018)
Compound: 6:2 FTSA **Initial concentration:** 64.23 mg/L **Microbial strain:** *Dietzia aurantiaca* J3 enrichment	**Duration:** 7 days **pH:** 7 **Temp.:** 30 °C **Redox condition:** Aerobic	6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, PFHxA, PFPeA	(Méndez et al., 2022)
Compound: 6:2 FTSA **Initial concentration:** 25.69 mg/L **Microbial strain:** *Gordonia* sp. strain NB4-1Y	**Duration:** 7 days **Temp.:** 30 °C **Redox condition:** Aerobic, sulfur limiting	99.9 % degradation, transformation products are 6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, 4:3 FTCA, PFHxA, PFPeA, PFBA, 5:2 FT-ketone, 6:2 FTOH, 5:2 sFTOH	(Shaw et al., 2019)
Compound: PFHxS **Initial concentration:** 20 mg/L **Microbial strain:** *Pseudomonas* strains PS27 and PDMF10	**Duration:** 5 days **Temp.:** 27 °C **Redox condition:** Aerobic	Bioaccumulation of PFHxS: 28–32 % by resting cells and up to 40 % by mixed culture of two isolates	(Presentato et al., 2020)

compounds resistant to environmental degradation and microbial utilization as carbon and energy sources (Douna and Yousefi, 2023).

4.1. Factors affecting PFAS biodegradation

Several factors influence the biodegradation of specific PFAS, including the concentration (along with their toxicity and bioavailability), organic matter, redox conditions, and the characteristics of the microbial community (such as the composition of microbial consortia, metabolic potential, population density and ability to produce biosurfactants) (Berhanu et al., 2023; Cai et al., 2020; Cao et al., 2022a, 2022b; Zhang et al., 2022). Furthermore, temperature and pH affect enzymes and biochemical reactions, including the biodegradation of PFAS. For example, the sorption of PFAS to matrices is affected by pH, which influences their bioavailability for microbes (Berhanu et al., 2023; Wackett, 2021, 2022).

The temperature variation between 25 °C ($t_{1/2}$ = 44 days) and 4 °C

($t_{1/2}$ = 160 days) significantly influences the biodegradation of N-ethylperfluorooctane sulfonamidoethanol (EtFOSE) in marine sediments (Benskin et al., 2013). Zhang et al. (2017) indicated that the biodegradation of EtFOSE was four to five times higher in alkaline soil (pH = 7.8) than in acidic soil (pH = 5.5). Shaw et al. (2019) found that 6:2 FTSA and 6:2 FTAB can undergo biodegradation within one week under sulfur-limiting conditions. The degrading process required much more time (sometimes several months) with supplementary sulfur sources (Berhanu et al., 2023; D'Agostino and Mabury, 2017; Zhang et al., 2016). This indicates that bacteria generally prioritize the breakdown of 6:2 FTSA and 6:2 FTAB only after exhausting other accessible sulfur sources (Shaw et al., 2019).

5. PFAS removal through adsorption

Adsorption is a highly practical and efficient approach for the removal of PFAS from water (Appleman et al., 2014; Lauwers et al., 2023; Medina et al., 2022; Militao et al., 2023). Several researchers have employed various adsorbents, including carbonaceous materials, ion exchange resin, and other synthetic materials (Appleman et al., 2014; Boyer et al., 2021; Cantoni et al., 2021; Sørmo et al., 2021). According to Collivignarelli et al. (2023), one of the most common mechanisms for PFAS adsorption is electrostatic interactions (Fig. 2a), during which PFAS can interact with positively charged groups existing on the adsorbent surface. Another adsorption mechanism is hydrogen bonding that occurs between the oxygen atoms of PFAS structure and hydroxyl/carboxyl/amino groups present on the adsorbent surface (Fig. 2b). Fig. 2c indicates hydrophobic interactions that take place either between hydrophobic adsorbent surface and the hydrophobic tail of PFAS or through self-aggregation, which permits PFAS to form different structures (bilayer, micelles, or hemimicelles). During the ligand exchange mechanism, the adsorption phenomenon occurs on the surface of minerals like aluminum oxyhydroxide, silica and iron oxides having abundant hydroxyl groups. Through a ligand exchange mechanism, hydroxyl groups can be substituted by the negatively charged PFAS (Collivignarelli et al., 2023).

5.1. Carbonaceous materials

Comprehensive studies have been conducted on the elimination of PFAS utilizing carbonaceous substances, with the focus mainly on the removal of PFOA and PFOS (Dey et al., 2024; Saha et al., 2021; Teymourian et al., 2021). The preference is due to their predominant production and regulatory attention. Takagi et al. (2011) revealed that employing granular activated carbon (GAC) for only one year may eliminate over 69 % of PFOA and PFOS. The removal of PFOA and PFOS with powdered activated carbon (PAC) was higher than that from granular activated carbon (GAC) (Chen et al., 2017; Yu et al., 2009). Deng et al. (2015) and Park et al. (2020) reported the enlargement of micropores in GAC could facilitate the adsorption of hydrophilic PFAS (PFOS and PFOA). Furthermore, positively charged adsorbents often demonstrated higher PFAS removal than neutral or negatively charged adsorbents (Cantoni et al., 2021; Park et al., 2020). The source material and surface area of GAC are decisive in PFAS removal from water, with coconut charcoal demonstrating superior sorption capabilities compared to other alternative activated carbon sources. In another study, sub-micron powdered activated carbon showed 200 to 2000 times better sorption capacity than GAC for PFAS removal (Quinnan et al., 2022).

PFAS removal by activated carbon (AC) is typically attributed to hydrophobic interactions. PFAS molecules characterized by long fluorinated chains and weakly polar terminal groups have increased resistance to removal (Ateia et al., 2019b; Liu and Sun, 2021; Vu and Wu, 2022; Wu et al., 2020). Several studies found that long-chain PFAS are better adsorbed by GAC than short-chain PFAS due to a reduction in polarity with increasing chain length (Hansen et al., 2010; Medina et al., 2022; Ochoa-Herrera and Sierra-Alvarez, 2008; Riegel et al., 2023). In addition, the adsorbability of PFSA was higher than PFCA (McCleaf et al., 2017). PAC (wood-based) showed >70 % removal for eight PFAS, below 40 % removal for PFPeA and insignificant removal of short-chain PFAS (such as PFBS, PFBA, and others) (Dudley, 2012). The study was conducted with an initial adsorbate concentration of 500 ng/L and an adsorbent dosage of 15 mg/L for a contact time of 15 min. Tan et al. (2023) reported that GAC removed more PFAS under an acidic medium, whereas PAC removed hydrophobic PFAS better under a neutral medium and the adsorption of PFOA and PFOS was enhanced by higher ionic strength and the presence of Ca^{2+} ions. In contrast, the presence of Suwannee River fulvic acid and butylcarbitol reduced the adsorption of PFOS and PFOA (Hakimabadi et al., 2023).

Recent studies showed that adsorption onto biochar (BC) is a potentially cost-effective method for PFAS removal (Zhang et al., 2019). However, the multiple negatively charged sites on the BC surface

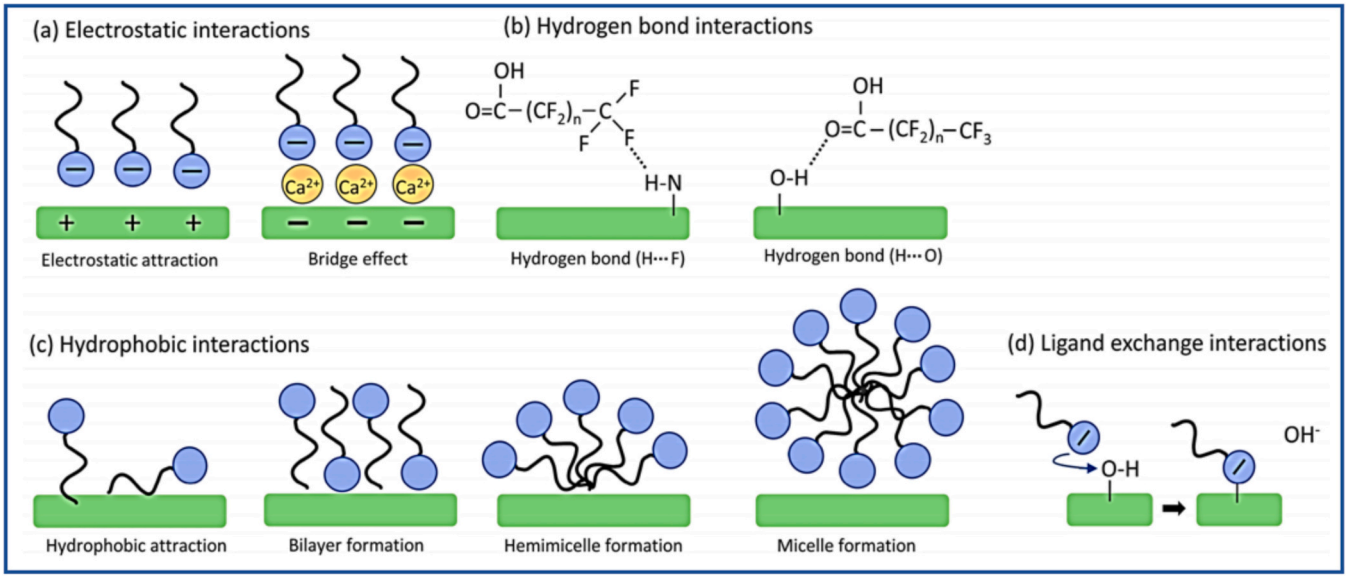

Fig. 2. Removal of PFAS from aqueous solutions through various adsorption mechanisms: **a)** electrostatic interactions, **b)** hydrogen bond interactions, **c)** hydrophobic interactions, and **d)** ligand exchange interactions (Reproduced from Collivignarelli et al. (2023)).

constrain the sorption of anions (such as PFAA) in water (Xiao et al., 2017). Furthermore, due to the pronounced water solubility of short-chain PFAA like PFBA and PFBS, the efficacy of BC in their removal is limited (Xiao et al., 2017; Zhang et al., 2019). Yu et al. (2023) reported that polypyrrole functionalized BC composites showed a greater ability to remove PFAS from water than pristine biochar. Additionally, the removals of four anionic PFAA (PFBA (≥79 %), PFBS (≥94 %), PFOA (≥99 %) and PFOS (≥99 %)) were almost unaffected under a wide pH range from 3 to 9. Biochar-alginate beads (rice straw-derived) was reported to remove PFAS from water (Militao et al., 2023). These beads achieved remarkable PFOS removal (up to 99 %) within 16 h and nearly 40 % removal of PFBS in <48 h.

Pore structure and specific surface area significantly affect the adsorption performance of carbonaceous materials for PFAS removal. Materials with high surface areas, like nitrogen-doped porous carbon beads, have shown improved PFAS adsorption by optimizing their pore architecture and functional surface groups to interact better with PFAS molecules (Czech et al., 2023). Biomass-sourced activated carbon, for example, coffee grounds, can have a surface area as high as 1050 m^2/g, showing that pore size distribution is an important variable in the efficiency of adsorption (Sklepova et al., 2023). Nitrogen-doped carbons derived from polyaniline further illustrate how tailored porosity combined with surface modifications optimizes adsorption mechanisms; hence, these materials are reported to be promising for the removal of PFAS from contaminated water sources (Khan et al., 2023b).

Different carbonaceous materials show different adsorption capabilities: AC exhibits high specific surface area and microporosity, suitable for the removal of organic contaminants; GAC is suitable for water treatment; BC, obtained from biomass, could be a low-cost alternative with a medium surface area and various functional groups.

5.2. Ion exchange resins

Ion exchange (IX) is a widely used purification method where exchangeable *co*-ions replace target ions in a solution on the surface of polymeric resins. Due to its small footprints, high efficiency and regenerative capabilities, IX resins are utilized in water and wastewater treatment to eliminate various pollutants (Bolisetty et al., 2019; Huang et al., 2020; Li et al., 2021). IX resins can be categorized into cationic exchange resins (CER) and anionic exchange resins (AER). CER effectively remove positively charged ions (such as iron and lead), while AER are adept at eliminating negatively charged ions (such as fluoride and sulfate) (Bezzina et al., 2019, 2020; Grzegorzek et al., 2020). Since many PFAS are anionic at the pH range of natural water, previous studies have shown that AER resins outperformed activated carbon in removing PFAS, especially short-chain PFAS (Gagliano et al., 2020; Woodard et al., 2017). The AER resins possess extreme affinity towards the negatively charged PFAS ions and the reversible nature of the process allows resin regeneration after the saturation phase (Boyer et al., 2021; Dixit et al., 2021b; Park et al., 2020). As a result, studies using AER to remove PFAS from contaminated water increased in these years (Table 3) (Dixit et al., 2021b; Karbassiyazdi et al., 2023; Liu and Sun, 2021; Saawarn et al., 2022). The performance of numerous commercially available AER (including macroporous polystyrene resin, gel-type resin, and polyacrylic resin) for PFAS removal under various experimental conditions is shown in Table 3 (Boyer et al., 2021; Dixit et al., 2021b; Saawarn et al., 2022).

Table 3 summarizes breakthrough results of bench and pilot-scale studies for PFAS removal by AER. The initial PFAS concentration increased, while the bed volumes (BV) to breakthrough dropped, indicating that AER removes PFAS via stoichiometric exchange with finite ion-exchange capacity. The properties of the resin and the test water composition were two factors that affected this tendency. Specifically, more BV were required to reach the breakthrough when testing with deionized water (DI) and simple water matrices compared to real water samples. Additionally, polystyrene AER exhibited more BV to

Table 3
PFAS removal by ion exchange resin in column studies with bed volumes (BV) to breakthrough (Boyer et al., 2021).

Compound, pH, resin, test water	Initial concentration (ng/L)	BV to breakthrough	Reference
Compound: PFBS **Resin:** LanXESS Lewatit® TP108DW, Purolite D9279, Evoqua APR-2, Evoqua PSR2+, and ResinTech SIR-110-HP **Test water:** non-potable groundwater	14	Completely removed (50 % breakthrough up to 220,000 BV)	(Pannu et al., 2024)
Compound: PFBS **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	15 17.5 18 42 220	Up to 300,000 Up to 200,000 > 1,100,000 Up to 350,000 > 75,000	(Zeng et al., 2020)
Compound: PFBS **pH:** 6.9 **Resin:** AER1 (PS/G/type I) **Test water:** spiked groundwater	53	> 30,000	(Schaefer et al., 2019)
Compound: PFBS **pH:** 6.9 **Resin:** AER1 **Test water:** spiked groundwater	53 1270	> 40,000 (breakthrough defined at 50 %) > 15,000	(Schaefer et al., 2019)
Compound: PFBS **pH:** 6.9 **Resin:** AER2 (PS/G/type I) **Test water:** spiked groundwater	53 1270	> 40,000 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFBS **Resin:** A600 **Test water:** PFAS occurring groundwater	11	> 18,600	(Franke et al., 2019)
Compound: PFBS **Resin:** A600 **Test water:** Spiked groundwater	81	13,368	(McCleaf et al., 2017)
Compound: PFBS **pH:** 7.0 **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 41,000 (at 2.5 mg/L initial DOC concentration) Up to 26,000 (at 5 mg/L initial DOC concentration) Up to 16,000 (at 10 mg/L initial DOC concentration) Up to 13,000 (at 20 mg/L initial DOC concentration)	(Dixit et al., 2020)
Compound: PFBS **pH:** 7.2 **Resin:** A860 **Test water:** PFAS occurring surface water	10,000	Up to 22,500	(Dixit et al., 2020)
Compound: PFHxS **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	30 75 85.5 110 1800	> 1,100,000 Up to 200,500 > 1,100,000 > 315,000 > 75,000	(Zeng et al., 2020)
Compound: PFHxS **Resin:** A600 **Test water:** PFAS occurring groundwater	110	> 18,600 (breakthrough defined at 50 %)	(Franke et al., 2019)

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Table 3 (*continued*)

Compound, pH, resin, test water	Initial concentration (ng/L)	BV to breakthrough	Reference
Compound: PFHxS **pH:** 7.9 **Resin:** A600 **Test water:** Membrane reject water	470	28,717 (breakthrough defined at 50 %)	(Franke et al., 2019)
Compound: PFHxS **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	315	> 30,000 > 40,000 (rapid small-scale column)	(Schaefer et al., 2019)
Compound: PFHxS **pH:** 6.9 **Resin:** AER2 **Test water:** Spiked groundwater	315	> 40,000 (rapid small-scale column)	(Schaefer et al., 2019)
Compound: PFOS **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	9.7 29 32 85 2900	Up to 400,000 > 315,000 Up to 200,000 > 1,100,000 > 75,000	(Zeng et al., 2020)
Compound: PFOS **Resin:** A600 **Test water:** PFAS occurring groundwater	39	> 18,600 (breakthrough defined at 50 %)	(Franke et al., 2019)
Compound: PFOS **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	481 8620	> 30,000 > 40,000 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFOS **pH:** 6.9 **Resin:** AER2 **Test water:** Spiked groundwater	481 8620	> 40,000 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFOS **pH:** 7.5–8.5 **Resin:** DOWEX PSR-2 **Test water:** PFAS occurring groundwater	2900	> 75,000	(Zeng et al., 2020)
Compound: PFOS **pH:** 7.0 **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 33,000 (at 2.5 mg/L initial DOC concentration) Up to 27,000 (at 5 mg/L initial DOC concentration) Up to 15,500 (at 10 mg/L initial DOC concentration) Up to 10,500 (at 20 mg/L initial DOC concentration)	(Dixit et al., 2020)
Compound: PFOS **pH:** 7.2 **Resin:** A860 **Test water:** PFAS occurring surface water	10,000	21,250	(Dixit et al., 2020)
Compound: PFBA **Resin:** LanXESS Lewatit® TP108DW, Purolite D9279, Evoqua APR-2, Evoqua PSR2+, and ResinTech SIR-110-HP **Test water:** non-potable groundwater	29	< 25,000	(Pannu et al., 2024)
Compound: PFBA **pH:** 6.9 **Resin:** AER1	323	Up to 6000 > 13,000 (rapid small-scale column)	(Schaefer et al., 2019)

Table 3 (*continued*)

Compound, pH, resin, test water	Initial concentration (ng/L)	BV to breakthrough	Reference
Test water: Spiked groundwater	5210	> 15,000	
Compound: PFBA **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	323 5210	< 6500 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFBA **pH:** 7.0 **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 36,500 (at 2.5 mg/L initial DOC concentration) Up to 25,500 (at 5 mg/L initial DOC concentration) Up to 15,000 (at 10 mg/L initial DOC concentration) Up to 11,300 (at 20 mg/L initial DOC concentration)	(Dixit et al., 2020)
Compound: PFBA **pH:** 7.2 **Resin:** A860 **Test water:** PFAS occurring surface water	10,000	21,875	(Dixit et al., 2020)
Compound: PFBA **Resin:** LanXESS Lewatit® TP108DW, Purolite D9279, Evoqua APR-2, Evoqua PSR2+, and ResinTech SIR-110-HP **Test water:** non-potable groundwater	7	< 25,000	(Pannu et al., 2024)
Compound: PFPeA **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	742 16,000	< 15,000 Up to 32,500 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFPeA **pH:** 6.9 **Resin:** AER2 **Test water:** Spiked groundwater	742 16,000	< 6500 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFHxA **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	14 16 490 1000	> 18,600 Up to 35,000 Up to 50,000 Up to 45,000	(Zeng et al., 2020)
Compound: PFHxA **pH:** 7.9 **Resin:** A600 **Test water:** Membrane reject water	65	7487 (breakthrough defined at 50 %)	(Franke et al., 2019)
Compound: PFHxA **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	652 12,000	Up to 15,000 (rapid small-scale column) > 40,000 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFHxA **pH:** 6.9 **Resin:** AER2 **Test water:** Spiked groundwater	652 12,000	> 13,000 (rapid small-scale column) > 15,000	(Schaefer et al., 2019)
Compound: PFHpA **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	3.7 4.1 13 21.5 210 300	< 210,000 < 190,000 < 700,000 < 1,100,000 Up to 1,100,000 > 70,000	(Zeng et al., 2020)

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Table 3 (*continued*)

Compound, pH, resin, test water	Initial concentration (ng/L)	BV to breakthrough	Reference
Compound: PFHpA **Resin:** A600 **Test water:** PFAS occurring groundwater	8	> 11,000 (breakthrough defined at 50 %)	(Franke et al., 2019)
Compound: PFOA **pH:** 7.5–8.5 **Resin:** PFA694E **Test water:** PFAS occurring groundwater	5 6.2 29 43.5 120 820	Up to 290,000 Up to 190,000 > 1,100,000 > 1,100,000 Up to 300,000 > 75,000	(Zeng et al., 2020)
Compound: PFOA **Resin:** A600E **Test water:** PFAS occurring groundwater	430	Up to 42,500	(Zaggia et al., 2016)
Compound: PFOA **pH:** 6.9 **Resin:** AER1 **Test water:** Spiked groundwater	243	Up to 30,000 > 40,000 (rapid small-scale column)	(Schaefer et al., 2019)
Compound: PFHxA **pH:** 6.9 **Resin:** AER2 **Test water:** Spiked groundwater	243	> 40,000 (rapid small-scale column)	(Schaefer et al., 2019)
Compound: PFOS **pH:** 7.0 **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 28,000 (at 2.5 mg/L initial DOC concentration) Up to 22,000 (at 5 mg/L initial DOC concentration) Up to 14,500 (at 10 mg/L initial DOC concentration) Up to 9500 (at 20 mg/L initial DOC concentration)	(Dixit et al., 2020)
Compound: PFNA **Resin:** A600 **Test water:** Spiked drinking water	89	12,938	(McCleaf et al., 2017)
Compound: PFDA **Resin:** A600 **Test water:** Spiked drinking water	73	12,974	(McCleaf et al., 2017)
Compound: GenX **pH:** 7.0 **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 32,000 (at 2.5 mg/L initial DOC concentration) Up to 24,000 (at 5 mg/L initial DOC concentration) Up to 15,000 (at 10 mg/L initial DOC concentration) Up to 10,500 (at 20 mg/L initial DOC concentration)	(Dixit et al., 2020)
Compound: 6:2 FTSA **Resin:** Sorbix A3F **Test water:** PFAS occurring groundwater	18,000	Up to 10,000	(Woodard et al., 2017)
Compound: 8:2 FTSA **Resin:** A860 **Test water:** Spiked DI water	10,000	Up to 15,000	(Dixit et al., 2021b)

breakthrough than polyacrylic AER and WB-AER. For short-chain PFAS (such as PFBS, PFBA, PFPeA, and PFHxA) and polystyrene AER, the results suggested that gel-structured resins required more BV to reach breakthrough compared to macroporous resins.

Electrostatic interactions are important in PFAS adsorption because they form ionic interactions between anionic PFAS and positively charged adsorbents. According to the studies conducted by Johnson et al. (2007) and Xiao et al. (2011) PFAS develops a negative shell around the positively charged core molecule instead of a negative charge around a functional group. It occurs because of the high electronegativity of fluorine atoms. Each fluorine atom has a strong electronegativity (negative dipole) together with the three lone pairs of electrons (as Lewis base sites), which are hypothesized to allow weak interactions with charged adsorbent surfaces (Xiao et al., 2011).

Numerous studies have observed that PFAS can overcome electrostatic repulsion between molecules and are adsorbed onto negatively charged surfaces of adsorbents (Ahrens et al., 2011; You et al., 2010; Zhang et al., 2013). Notably, the concentration of adsorbed hydrophobic PFAS is significantly higher than that of less hydrophobic PFAS, especially when employing hydrophobic adsorbents (Deng et al., 2012). Additionally, increasing the hydrophobicity of materials, such as zeolites and carbon nanotubes, enhances their adsorption capacity (Ochoa-Herrera and Sierra-Alvarez, 2008). These findings highlight the importance of hydrophobic interactions in the adsorption of PFAS.

The hydrophobic chains in PFAS can repel water molecules (He et al., 2013). As a result, hydrogen bonds can form between PFAS and hydrogen atoms bonded to functional groups (such as nitrogen or oxygen, for example, –NH, –OH, and –COOH groups) in the adsorbents. Oxygen atoms in the functional group of PFAS can act as acceptors in hydrogen bonding with these groups (Gao and Chorover, 2012; Xiao et al., 2012). Several studies showed that this type of hydrogen bonding results in the adsorption of PFAS (Takayose et al., 2012; Xu et al., 2013). In addition, functional groups with oxygen-carrying moieties can capture water molecules through hydrogen bonds from the bulk solution, potentially leading to the adsorption of water with PFAS and the formation of clusters on adsorbent surfaces (such as carbon nanotube), which may impede PFAS access (Deng et al., 2012; Zhang et al., 2009). Consequently, the role of hydrogen bonding may become less significant in adsorbing PFAS.

The sustainability and cost-effectiveness of PFAS removal might be improved with adsorbent regeneration (Baskar et al., 2022; Lei et al., 2023; Vakili et al., 2024). Recent studies showed various regeneration methods that recover the adsorption capacity of different adsorbents, allowing for their repeated use. For example, Kim et al. (2024) found that hydrotalcite could be thermally regenerated at 400 °C without experiencing a significant loss in its ability to adsorb PFOS and PFBA and showing constant performance throughout several cycles. Liu and Sun (2021) reported that up to 90 % of the adsorption capacity of a resin may be recovered using a regeneration solution containing 2 % NaCl and 10 % methanol. Depending on the type of PFAS, Lauwers et al. (2023) showed how to regenerate all-silica zeolite β using either thermal methods or particular solvents. Calculations revealed that thermal regeneration at 500 °C for two hours was sufficient to recover over 95 % of the adsorption capacity for PFOA. These findings highlight the possibility of regenerating different adsorbents, thereby potentially reducing operational costs and increasing the viability of large-scale PFAS treatment systems.

5.3. *Factors affecting PFAS adsorption*

Adsorption is influenced by several parameters such as surface area of the adsorbent, pore size distribution, and physicochemical properties of the solution such as pH, presence of co-existing solutes, and dissolved organic compounds (Iqbal et al., 2022, 2023, 2024; Song et al., 2018; Yadav et al., 2022). For example, the adsorption capacity of IX resins for PFAS is dependent on the type and concentration of organic and inorganic substances, as well as the specific affinity of the used ion-exchange resin for PFAS (Dixit et al., 2021b; Lampert et al., 2007). Sulfate is the most competitive inorganic ions, followed by phosphates and nitrate (according to charge density) (Dixit et al., 2021a, 2021b). Organic

matter, specifically the humic fraction with the highest molecular weight, had the most pronounced negative impact on PFAS adsorption (Dixit et al., 2019; Gagliano et al., 2020). According to results from several studies, this fraction can clog the pores of IX resins, limiting the exchange capacity of microporous resins (Dixit et al., 2019, 2021b; Pereira et al., 2018). Fulvic acids and other organic components with lower molecular weights can compete for active IX uptake sites, potentially reducing PFAS removal (Dixit et al., 2019). Some resins exhibit a lower sensitivity to inorganic ions, displaying a relatively higher affinity for PFAS (Barisci and Suri, 2021). However, most PFAS molecules possess anionic functional groups, with only a few exceptions having cationic functional groups. Some PFAS are classified as zwitterionic, showing both anionic and cationic functional groups. This charge of individual PFAS plays a key role in their interactions with various adsorption materials (Barisci and Suri, 2021).

Zhang et al. (2019) reported that the adsorption of PFAS is primarily defined by factors such as pH, sorbent characteristics, and the hydrophobic properties of PFAS. Lower pH levels have been associated with increased PFAS adsorption; however, this effect is contingent upon the point of zero charge of the sorbent (Zhang et al., 2019). Conversely, several studies have indicated that raising the pH levels can result in a decrease in the adsorption capacity of various adsorbents (Deng et al., 2010; Gao et al., 2017). PFAS adsorption on IX resins decreased with increasing pH (Deng et al., 2015; Gagliano et al., 2020; Zhang et al., 2019). These findings align with existing literature on PFAS adsorption using other anionic adsorbents such as alumina, boehmite, and various mineral sediments (Gagliano et al., 2020; Wang et al., 2016).

Zeta potential and pH remain two critical factors in PFAS adsorption that can directly affect the electrostatic interactions between PFAS and adsorbent (Wu et al., 2020). While increasing the pH, the zeta potential decreased (Lee et al., 2020). PFAS adsorption to surfaces of adsorbents decreases the zeta potential of adsorbents, which may be enhanced in a chain-length-dependent manner (Xiao et al., 2011). Wu et al. (2020) reported that zeta potential was hardly related to the weighted average of PFAS removal at 4 h. Therefore, hydrophobic interactions are dominating in the adsorption of PFAS on AC. In another study, Zenobio et al. (2022) found that the fluctuations in temperature (between 4 °C and 20 °C) and hydrophobicity of the container surfaces affect the adsorption of PFAS.

Recent bench-scale experiments confirmed that the polymer matrix is another critical factor influencing the adsorption capacity of resins. Specifically, polyacrylic resins exhibited relatively higher and more rapid removal of PFOS compared to polystyrene resins (Deng et al., 2010; Dixit et al., 2021b). The adsorption of PFOS on polyacrylic resins was in the range of 4–5 mmol/g. In the case of PFHxS removal, IRA910 resin showed high efficiency (Maimaiti et al., 2018). Using IRA67 (a polyacrylic-DVB, gel-type resin) resulted in a higher adsorption capacity than styrene-type resins (Gagliano et al., 2020). Gao et al. (2017) reported the effectiveness of polyacrylic resin for PFAS (F-53B) removal, with the adsorption capacity of IRA67 reaching approximately 2400 mg/g in batch experiments.

According to Zaggia et al. (2016), the porosity of the adsorbent material significantly affects the removal of PFAS. Deng et al. (2010) reported that microporous AER allows faster PFOS diffusion than gel-type resins due to increased pore size. Various studies investigating the adsorption capacity of two distinct resins, IRA910 (polystyrene-DVB, macroporous-type) and IRA400 (styrene-DVB, gel-type), in the context of PFOS removal, have consistently reported similar findings (Maimaiti et al., 2018; Vu and Wu, 2022; Wang et al., 2016). These observations highlight the critical significance of material porosity in regulating PFAS removal.

Advanced IX resins, such as MIEX® GOLD demonstrated high efficiency (over 99 %) in removing various PFAS. These materials show robustness against interference from both organic and inorganic matter, making them promising candidates for large-scale applications (Tamanna et al., 2023). Novel adsorbent materials (such as surfactant-modified electrodes) have been developed to degrade PFAS by electroreduction, combining adsorption with chemical transformation to achieve effective remediation (Wang et al., 2023).

6. Conclusion and future work

This review provides a comprehensive overview of PFAS occurrence in aquatic environments and critically examines recent advancements in PFAS elimination through biodegradation and adsorption. PFAS contamination, including compounds such as PFPeA, PFOA, PFOS, PFBA, PFBS, PFHxA, and PFHpA, is pervasive in many surface waters and groundwaters, with higher concentrations reported for groundwater due to reduced dilutions compared to surface water systems. Notable, PFBA, PFOS, and PFOA concentrations have been found at substantial levels in surface snow, emphasizing the transport and persistence of PFAS.

Recent advancements in bioremediation have expanded the scope of biodegradable PFAS species, with mixed microbial cultures and specific pure microbial strains showing degradation capabilities. Achieving complete degradations required multi-pathway involvement and optimization of critical factors such as pH, temperature, and redox conditions, which significantly influence microbial activity and reaction efficiency. While biodegradation might offer sustainable solutions, overcoming slow kinetics and incomplete breakdown of PFAS remain challenges.

Current research on adsorption demonstrates a gap between laboratory conditions and real-world applications. Many studies apply higher adsorbent dosages and high PFAS concentrations, limiting their practical relevance to treatment plants and natural environments.

Ion exchange resins are reported to outperform other adsorbents in removing both long and short-chain PFAS. Short-chain PFAS adsorption was reported to be limited by reduced adsorption capacities and competitive interactions with long-chain PFCA and PFSA. Additionally, factors such as polymer matrix composition, solution pH, DOM and PFAS hydrophobicity play pivotal roles for adsorption.

Overall, this review emphasizes the critical need for refining technologies to enhance scalability and applicability under realistic environmental conditions.

Further in-depth research is needed on the following issues.

1. Isolating and identifying indigenous microbial species capable of PFAS degradation might enhance future implementation. Indigenous-lab-optimized microbes can be promising bio-agents for In-situ PFAS degradation in water and wastewater treatment.
2. Current PFAS biodegradation may not fully degrade due to limitations in microbial enzyme responses, making bioremediation challenging. Future studies could combine biodegradation and physiochemical methods to improve PFAS removal. For example, chemical destruction methods such as electrochemical processes may produce biodegradable intermediates, potentially resulting in total PFAS decomposition.
3. Research should focus on the impact of biodegradable DOM, microbial diversity, and water treatment systems on PFAS removal.
4. Further studies are needed to investigate the removal of PFAS through adsorption, especially for short-chain PFAS, under realistic conditions, considering factors such as adsorbent dose, empty bed contact time, pH, and PFAS concentrations.
5. Life cycle assessment and cost estimation might help to identify most sustainable and effective treatment options.

CRediT authorship contribution statement

Muhammad Zeeshan: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Shamas Tabraiz:** Writing – review & editing. **Safeerul Islam Hashmi:** Writing – review & editing, Data

curation. **Arfa Iqbal:** Writing – review & editing. **Daniel Dittmann:** Writing – review & editing. **Zohaib Abbas:** Writing – review & editing. **Cecilia L. MacLeod:** Writing – review & editing. **Aki Sebastian Ruhl:** Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biteb.2025.102077.

Data availability

Data will be made available on request.

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