

Per- and polyfluoroalkyl (PFAS) eternal pollutants: Sources, environmental impacts and treatment processes

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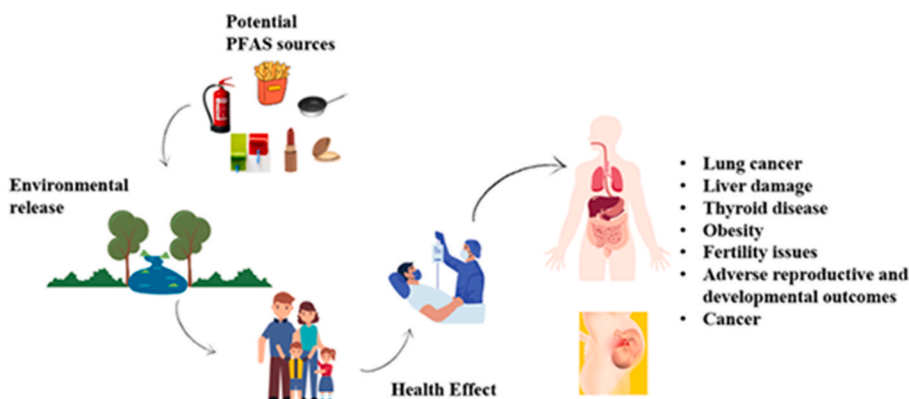
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HIGHLIGHTS

- PFAS pollution is a growing environmental concern.
- Among PFAS, PFOA and PFOS remain by far the most studied compounds.
- Different processes are studied for the removal of PFAS from waters.
- The removal processes are classified into separation and degradation processes.

GRAPHICAL ABSTRACT



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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) have become a growing environmental concern due to their tangible impacts on human health. However, due to the large number of PFAS compounds and the analytical difficulty to identify all of them, there are still some knowledge gaps not only on their impact on human health, but also on how to manage them and achieve their effective degradation. PFAS compounds originate from man-made chemicals that are resistant to degradation because of the presence of the strong carbon-fluorine bonds in their chemical structure. This review consists of two parts. In the first part, the environmental effects of fluorinated compound contamination in water are covered with the objective to highlight how their presence in the environment adversely impacts the human health. In the second part, the focus is put on the different techniques available for the degradation and/or separation of PFAS compounds in different types of waters. Examples of removal/treatment of PFAS present in either surface or ground water are presented.

1. Introduction

Per- and polyfluoroalkyl (PFAS) compounds constitute a class of

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Abbreviations and nomenclature

PFAS	Per- and polyfluoroalkyl substances	FOSA	Perfluorooctane sulfonamide
PFOS	Perfluorooctanesulfonic acid	6:2 FTS	6:2 fluorotelomersulfonate
PFOA	Perfluorooctanoic Acid	FTSA	FluoroTelomer Sulfonic Acid
PFCA	Perfluorocarboxylic acids	PFOSAmS	Perfluorooctane Sulfonamide Quaternary Ammonium Salt
PFAAs	Perfluoroalkanoic acids	PFOAAmS	Perfluorooctaneamido Quaternary Ammonium Salt
PFBA	Pentafluorobenzoic acid	SDWA	Safe Drinking Water Act
PFBS	Perfluorobutane sulfonate	EPA	Environmental Protection Agency
PFHpA	Perrfluoroheptanoic Acid (C7)	NOAEL	No Observed Adverse Effect Level
PFHxA	Perfluorohexanoic Acid (C6)	GAC	Granular Activated Carbon
PFHxS	Perfluorohexanesulphonic acid	AIX	Anion-Exchange Resin
PFNA	Perfluorononanoic Acid (C9)	PEI-CNC	Polyethylene Imine-Modified Cellulose Nanocrystals
PFPeA	Perfluoropentanoic Acid(C5)	EC	Electrocoagulation
PFSA	Perfluorosulfonic acid	OECD	The Organization for Economic Co-operation and Development
POPs	Persistent Organic Pollutant	ECHA	The European Chemical Agency
PTFE	Teflon	U.S EPA	The U.S. Environmental Protection Agency
REACH	Registration, Evaluation, Authorization, and Restriction of chemicals	TURA	Toxics Use Reduction Act
AFFFs	Aqueous film-forming foam	NDAA	The National Defense Authorization Act
		SCP	The California Safer Consumer Products

substances that belong to a large family of man-made fluorinated organic compounds with different varieties in functional groups and chain length (KEMI, 2015; B. Wang et al., 2020a,b). Carbon-Fluorine (C-F) bond in the structure of PFAS compounds allow perfluoroalkyl moiety (C_nF_{2n+1}) to offer chemical and thermal stability (e.g., stronger acidity, higher surface activity at very low concentrations, stability, and/or water- and oil repellency) as well as hydrophobic and hydrophilic properties (Buck et al., 2011a). The hydrophobic characteristics of PFASs are mainly carbon chains linked to fluorine atoms. PFAS have hydrophobic properties, which means they have a strong aversion to water, thanks to this carbon-fluorine combination (Mohona et al., 2023). The PFAS often have a hydrophilic functional group, such as a hydrogen atom or a charged chemical group, despite the fact that most of their carbon chain is hydrophobic. The hydrophobic and hydrophilic properties of PFAS make them adaptable surface-active substances that repel grease and dirt. For instance, PFAS coatings on a given fabric makes it stain-resistant water-repelling. The perfluoroalkanoic acids (PFAAs) are among the most widely known PFAS compounds (Kwiatkowski et al., 2020). These PFAAs are fully fluorinated, have a polar end group and are subdivided into perfluorocarboxylic acids (PFCA) such as perfluorooctanoic Acid (PFOA), and perfluorosulfonic acids (PFSA) such as perfluorooctanesulfonic acid (PFOS) (Fenton et al., 2021; Zhang et al., 2022). Moreover, the functional head groups in PFAS make them particularly interesting for surfactant applications (Brusseau and Van Glubt, 2019; Psillakis et al., 2009; Schaefer et al., 2019b; Silva et al., 2019). Because of these unique properties, PFAS have been used in a broad range of applications such as paints, carpets, textiles, non-stick cookware, aqueous film forming foam, and oil production to name only a few. Since their fortuitous discovery in 1930 (Tokranov et al., 2019; Wang et al., 2017). The PFAS production is exploding in many places in the world. For instance, the US Environmental Protection Agency (EPA) estimated that at least 118 PFAS are active in the US market and 85,000 tons of PFAS are produced domestically each year, of which 50% ended up in direct and/or indirect releases to the environment (EPA, 2021; Paul et al 2009).

PFAS are among the components that have gained much attention in the few from both their environmental/human health concerns and their management and risks mitigation. Although primary concerns regarding PFAS and their associated environmental impacts have begun in the early 1990s, the development of the analytical techniques and Instrumentation capable of detecting PFAS compounds is rather recent and still ongoing. These sophisticated analytical methods enabled the

researchers to detect PFAS in different sources of water and link their presence to possible adverse effects on health and environmental. The spectacular rise of interest in PFAS-related research is well. The spectacular rise of interest in PFAS-related research is well illustrated by the number of publications (on Elsevier's database by using the keyword "PFAS" in the title, abstract, and keywords) is clearly seen to sky-rocketing after 2017 (passing from only ~24 in 2020 to more than 1030 papers in 2022). It is worth mentioning that the significant increase in PFAS-related publications is correlated with the growing concerns about production and release of PFAS compounds in the environment. The term "PFAS" lacks a commonly agreed-upon definition. There isn't a single, accepted definition of PFAS because different regulatory organizations and academic institutions have established different definitions. There isn't a single, widely agreed-upon definition of PFAS, despite the USEPA developing guideline limits on PFAS in drinking water. The absence of a common definition has made it difficult to regulate and conduct research on these substances because different definitions can lead to different chemicals being classified as PFAS. The definition of PFAS is changing, reflecting ongoing research on these substances as well as the various organizations' and researchers' operating standards. Furthermore, the US EPA has broadened its definition of PFAS, reflecting the evolving understanding of these substances and their environmental impact (USEPA, 2019). This expansion of the definition signifies the recognition of new compounds and the need for comprehensive regulation and management. Additionally, the OECD has introduced a new definition for PFAS, aligning with the global efforts to address these substances effectively (OECD, 2021). This reflects the international dimension of PFAS management and the importance of harmonizing definitions and regulatory approaches across countries. The evolving definitions of PFAS underscore the complexity and significance of these substances in environmental and regulatory contexts, emphasizing the need for a unified and comprehensive understanding to address their potential risks and impacts effectively.

According to the previous Table 1, the definition of PFAS varies among different organizations and researchers. The U.S EPA, specifically the Office of Pollution Prevention and Toxics (OPPT), previously had a "working definition" that was narrower than the definition used by other organizations, such as the OECD (Maier, 2021; OECD, 2021). However, in 2023, the EPA announced that it is no longer using a "working definition" to characterize the universe of PFAS subject to the office's various testing, reporting, and regulatory efforts. Instead, the EPA intends to explain the rationale for identifying specific PFAS substances it

Table 1
Different PFAS definitions.

Organization	Year	PFAS definition	References
Buck et al.	2011	A subset of fluorinated substances refers to highly fluorinated aliphatic compounds that have one or more carbon atoms, where all the hydrogen substituents have been replaced by fluorine atoms, creating the perfluoroalkyl moiety C_nF_{2n+1} .	Buck et al. (2011b)
The European Chemical Agency (ECHA)	2021	PFAS are substances containing at least one aliphatic CF_2 or CF_3 element.	Hassold et al. (2021)
The Organization for Economic Co-operation and Development (OECD)	2021	The PFAS are substances containing at least one fully fluorinated methyl or methylene carbon atom.	OECD (2021)
The U.S. Environmental Protection Agency (US.EPA)	2021	The per- and polyfluorinated compounds (PFAS) that have the unit $R-(CF_2)_n-C(F)(R_1)R_2$ structurally. There can be no hydrogen atom in any of the R groups (R , R_1 , or R_2) since the CF_2 and CF moieties are saturated carbons.	(Barber, 2021)
Toxics Use Reduction Act (TURA)	2023	PFAS contain a perfluoroalkyl moiety with three or more carbons (e.g., $-C_nF_{2n-}$, $n \geq 3$; or $CF_3-C_nF_{2n-}$, $n \geq 2$) or a perfluoroalkyl ether moiety with two or more carbons (e.g., $-C_nF_{2n}OC_mF_{2m-}$ or $-C_nF_{2n}OC_mF_{m-}$, where n and $m \geq 1$).	Guidance+for+Reporting+of+PFAS+under+TURA+June+2023
The National Defense Authorization Act (NDAA)	2020	Perfluoroalkyl and polyfluoroalkyl compounds, which are man-made compounds containing at least one totally fluorinated carbon atom, are referred to as PFAS. According to the NDAA, a fully fluorinated carbon atom is one on which fluorine has completely replaced all the hydrogen substituents.	Andrews and Naidenko (2020)
Glüge et al.	2020	PFAS are compounds with C_nF_{2n+1} , where $n \geq 1$. Other compounds in the category of highly fluorinated aliphatic substances are (i) aromatic compounds with perfluoroalkyl moieties on the side chains, (ii) compounds where a perfluorocarbon chain relates to functional groups on both ends, and (iii) fluorinated cycloaliphatic substances. Additionally, non-polymeric substances that solely include a $-CF_3$ or $-CF_2-$ moiety are excluded, apart from perfluoroalkyl ethers and substances based on per- and polyfluoroalkyl ethers. Polymeric PFAS with the $-CF_2-$ moiety and non-polymeric PFAS with the $-CF_2-CF_2-$ moiety is included. This group also includes substances with a $-CF_2OCF_2-$ or $-CF_2OCFHCF-$ moiety.	Glüge et al. (2020)

believes are appropriate to include within the scope of each individual action. Any compound having at least one totally fluorinated carbon atom mostly that carbon-fluorine bond is classified as PFAS by the OECD. This definition drew on a three-year, multi-stakeholder, international review of the literature. ([ENVIRONMENT DIRECTORATE CHEMICALS AND BIOTECHNOLOGY COMMITTEE, 2021](#)). The California Safer Consumer Products (SCP) program does not provide a specific definition of PFAS. However, the program focuses on reducing toxic chemicals in consumer products and identifies priority products for regulation based on potential hazard and consumer exposure ([Hammel et al., 2022](#)). The Massachusetts Toxics Use Reduction Act (TURA) Program, in 2023, acknowledged the definitions provided by [Buck et al. \(2011a\)](#) and [OECD \(2021\)](#) while also considering the definition of PFAS to include various fluorinated substances ([Buck et al., 2011b](#); [OECD, 2021](#); “[Table 1 TURA/TRI PFAS Substances to Continue Reporting Individually](#)”). Overall, the absence of a universally accepted definition of PFAS has led to varied interpretations and approaches by different organizations and researchers, which can impact regulatory efforts and the understanding of these substances.

2. Classification of PFAS compounds

PFAS can be classified into two main categories, namely polymers and non-polymers. The polymer type of PFAS is commonly used material such as Teflon (PTFE) and similar plastics though non-polymer is mostly the subject of research because of their wide detection in the surface water. Perfluoroalkyl and polyfluoroalkyl substances are the two sub-categories of non-polymer. The main differences between these two categories are attributed to the degree of the C-F bond in molecular structure. For instance, perfluoroalkyl substances have fully fluorinated carbon chains at all possible bonding sites with a terminal functional group while polyfluoroalkyl substances are only partially fluorinated.

As mentioned previously, the unique property of PFAS compounds is primarily associated with the fluorine atoms that replace hydrogen along carbon chain. On the other hand, other properties such as chain length and functional group play vital roles in the physical and chemical properties of PFAS. Understanding these properties are important for finding an appropriate treatment technique and their release into the environment ([Team, 2020](#)). The high electronegativity of fluorine (4.0 for fluorine vs. 2.5 for carbon) offers the carbon-fluorine bond a significant polarity/dipole moment. This together with small size of fluorine make the C-F bond one of the strongest covalent bonds in chemistry and of the most unreactive compounds. Therefore, it is expected that PFAS show a high dissociation resistance towards different strong oxidants (i.e., $\bullet OH$), high temperature, chemical and biological degradation ([Kissa, 2001](#); [Team, 2020](#)). Carbon chain length is another parameter which also has a significant effect on PFAS compound properties. In particular, the longer chain increases amphiphilic property of PFAS ([Gagliano et al., 2020](#); [Team, 2020](#)). In addition, the presence of different functional groups has been reported to affect the properties of PFAS ([Teymourian et al., 2021](#)). The most common functional groups introduced to carbon bonds are carboxylates, sulfonates, and hydroxyl groups where affect degradation pathways of PFAS compounds. For instance, PFCA showed higher degradation in comparison with PFSA (functional groups are carboxylates and sulfonate, respectively) within the same amount of carbon in their chain ([Trautmann et al., 2015](#)).

3. Occurrence of PFAS in environment and adverse impacts

The toxicological effects of PFAS are being increasingly studied due to their ubiquitous presence and bioaccumulation in the environment. Except for PFOS and PFOA, there are currently few toxicological data available for many of the PFAS compounds. Per and polyfluoroalkyl substances (PFAS) have been found to have significant ecotoxicological

effects on various organisms and the environment. PFAS are widely used in industrial and commercial applications, leading to their global presence and persistence in the environment. Studies have shown that PFAS can have toxic effects on animals, humans, plants, and microbial communities. For example, PFAS have been associated with negative effects on human health, including cancer, reducing fertility and other health issues, due to environmental pollution (Ankley et al., 2021). Furthermore, research has shown that PFAS can have toxic effects on animals, plants, and microbial communities. For example, PFAS have been found to be toxic to earthworms, affecting their growth, reproduction, and survival.

3.1. The effects on aquatic ecosystems

PFAS pose a substantial threat to aquatic ecosystems, with potential adverse effects on aquatic plankton, invertebrates, and microorganisms (Ma et al., 2023a). These emerging pollutants can accumulate in benthic invertebrates, and their bioaccumulation is influenced by the organic carbon content, with higher organic carbon leading to decreased bioaccumulation (Yun et al., 2023). PFAS, particularly PFOS, exhibit significant toxicity to aquatic organisms, with PFOS being more toxic than PFOA. PFOS exhibits significant toxicity to aquatic organisms, with PFOS being more toxic than PFOA. Research has shown that PFOS is more toxic to aquatic invertebrates such as *Daphnia magna*, with a 48-h immobility NOEC value of 0.8 mg/L and a 21-day lethality NOEC of 5.3 mg/L, while the toxicity of PFOA is classified as low for *D. magna* (Ma et al., 2023a). Additionally, PFOS has been found to have toxic effects on aquatic vertebrates, such as fish, with a 96-h LC50 value of 22.56 mg/L for red sea bream (Ma et al., 2023b). The high stability and bioaccumulation of PFASs, including PFOS, contribute to their adverse impact on aquatic organisms. (Ma et al., 2023b). The toxicity of PFASs, including PFOS, to aquatic organisms is a growing concern due to their widespread contamination of aquatic environments and their potential health risks to humans through the consumption of PFAS-contaminated fish (Teunen et al., 2021). The U.S EPA is attempting to determine and forecast the toxicity of PFAS and PFAS combinations of highest ecological concern by investigating the links between PFAS toxicity and chemical structure for a number of aquatic species (Viera Valencia and Garcia Giraldo, 2019). Additionally, the toxicity of PFAS to benthic organisms and the relationship between toxicity and sediment concentrations are still being studied, with the need for systematic toxic effect investigations on aquatic organisms. It is essential to gain a more complete understanding of the adverse effects of PFAS on various levels of the aquatic ecosystem to effectively assess and mitigate their impact.

3.2. The effects on wildlife

Recent studies have revealed the widespread contamination of various animal species with PFAS, indicating the potential for adverse effects on wildlife. An analysis by the Environmental Working Group (EWG) found PFAS in a range of species, including scorpions, pandas, Siberian tigers, turtles, horses, dogs and wild boar. The study highlighted the sobering breadth of PFAS contamination in wildlife and emphasized the need for greater restrictions to prevent further environmental contamination (Andrews et al., 2023). Furthermore, a study on legacy and novel PFAS in juvenile seabirds from the US Atlantic Coast demonstrated the presence of PFAS in these birds, indicating the pervasive nature of PFAS exposure in avian species. The research suggested associations between PFAS concentrations and reproductive parameters, morphometric characteristics, and other adverse effects in seabirds, underscoring the potential risks posed by PFAS to avian wildlife (Robuck et al., 2020). Furthermore, Recent studies have shed light on the toxicology of birds exposed to PFAS. Monitoring studies have indicated that a wide variety of avian species, including gulls, wading birds, piscivorous and insectivorous birds, and terrestrial raptors, are exposed to PFAS, with some PFAS known to bioaccumulate and

biomagnify in avian food webs. However, the distribution and movement of PFAS in avian food webs, as well as the effects of PFAS on birds, are still poorly understood (Hopkins et al., 2023). Furthermore, investigations have indicated that PFAS may have an impact on bird reproduction. As a result, comprehensive toxicokinetic behaviour studies of PFAS in various bird species and developmental stages are required. Furthermore, avian chronic toxicity values (CTVs) for PFAS that are specific to species and tissues have been determined, offering important information on the possible negative effects of PFAS on birds (Groffen et al., 2023; Robuck et al., 2020). These findings collectively emphasize the extensive reach of PFAS contamination in diverse animal species, raising concerns about the potential impact on wildlife health and ecosystems. The studies underscore the need for continued research and regulatory measures to mitigate the adverse effects of PFAS on a global scale.

3.3. The effects on terrestrial plants

The ecotoxicological effects of PFAS on terrestrial plants are a topic of growing concern due to their potential impact on terrestrial ecosystems. Limited data is available on the specific effects of PFAS on plant health, with most existing studies focusing on acute ecotoxicological effects, such as impacts on plant growth, reproduction, and survival. Recent research has also highlighted the potential negative effects of PFAS at elevated concentrations on the root structures of plants. The translocation of PFAS from the environment to plant tissues primarily occurs through root uptake, which is considered a main source of PFASs to plants and is primarily a diffusive process (Costello and Lee, 2020; Xu et al., 2022). Both active and passive transport mechanisms are involved in the transfer of PFAS from roots to shoots, and the effects and uptake of PFAS differ among and within plant species (Costello and Lee, 2020). It was discovered that there was no significant correlation between the root concentration factor (RCF) of PFAS and hydrophobicity in plants cultivated in soil. This suggests that variables other than the chemical features of the substances influence the translocation of PFAS within the plant (Mei et al., 2021; Melo et al., 2022). These findings highlight the significance of root uptake as the primary pathway for PFAS translocation to plants and the complex nature of PFAS translocation and bioaccumulation within plant tissues. However, the current understanding of the ecotoxicological effects of PFAS on terrestrial plants remains limited, emphasizing the need for further comprehensive research to fully assess the potential risks associated with PFAS contamination in terrestrial ecosystems. The goal of this study is to present a summary of the body of research on the ecotoxicological effects of PFAS on terrestrial plants, providing important new information on this important area of PFAS toxicity and its consequences for ecosystem sustainability and environmental health.

PFAS are used in industrial processes, which may lead to their releases from industrial facilities into the environment. Releases can also occur from landfills and wastewater treatment plants, and due to the reuse of biosolids from wastewater treatment plants. Many studies have demonstrated that PFAS are transported long distances through the atmosphere, in water bodies and within groundwater (Environment and Climate Change Canada/Health Canada, 2023; Kurwadkar et al., 2022). Existing studies on PFOA and PFOS indicate moderate ecotoxicity to aquatic species (Hekster et al., 2003). For instance, the lowest known No Observed Adverse Effect Level (NOAEL) for PFOS is 2.3 µg/L for the insect *Chironomus* (Moermond and Verbruggen, 2010). The NOAEL term is used in toxicology under defined experimental conditions to mention the highest concentration observed in a specific test organism. In ecotoxicology studies aiming at determining possible impacts of pollutants on aquatic ecosystems, these insects are frequently utilized as a model organism. Furthermore, the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) classifies PFOS and PFOA as toxic compounds. According to REACH criteria, PFOA and PFOS are classified as reproductive and chronic toxicity, respectively (ECHA,

2014). This chronic toxicity was demonstrated in zooplankton colonies whose total numbers decreased after exposure to PFOS (For et al., 2004). On the other hand, fish exposed to PFOS showed no significant decrease in the egg hatches time and reproductive rate, and their mortality and growth rates were unaffected (Codling et al., 2020). Other studies carried out on the transfer of PFAS compounds from soils and aqueous solutions to plants have shown that PFAS can be absorbed and taken up by plants (Abunada et al., 2020; Lechner and Knapp, 2011). The uptake and accumulation of PFAS in plants depend on several factors, in particular, length of the chains, hydrophobic character, type of plant and properties of the environment (soil or water) (Jeon et al., 2011). In Canada, PFAS are mostly incorporated into goods or become components of produced goods. There are two main causes of PFAS pollution in the aquatic environment: specific and diffuse sources. Non-point sources of PFAS may include surface runoff from urban areas and wet/dry atmospheric deposition. The most common point sources of PFAS contamination are associated with the use of aqueous film-forming foam (AFFFs) to extinguish fuel fires or during firefighting training sessions (Beesoon et al., 2012; De Silva et al., 2012; Eriksson and Kärman, 2015; Guerra et al., 2014; Hamid et al., 2018; Shoeib et al., 2011; Tittlemier et al., 2007).

There are limited data regarding PFAS in Canadian freshwater sources and drinking water. The existing summation methodologies for PFAS legislation in Canada do not take into account specific PFAS variations in toxicity. The precautionary approach, which emphasizes the need to address less commonly recognized PFAS in regulations, views less popular PFAS as equally worrying unless proven otherwise, encouraging the implementation of comprehensive PFAS summing procedures across jurisdictions (Sauvé et al., 2023). A study focused on examining the presence of PFAS in freshwater found perfluorobutanoic acid (PFBA), perfluoropentanoic acid (PFPeA), perfluorohexanoic acid (PFHxA), perfluoroheptanoic acid (PFHpA), PFOA and PFOS to have higher detection frequencies than the other PFAS (Lalonde and Garron, 2022). The amounts of PFBA and PFPeA showed a noticeable increase between 2013 and 2023, but PFHpA, PFOA, and PFOS concentrations decreased. The researchers also determined that perfluorobutane sulfonate (PFBS) and PFHxA had the greatest concentrations as 138 ng/L. In addition, neither PFOA nor PFOS were found in treated drinking water in six out of seven Saskatchewan municipalities (Management, 2011). The Ontario Ministry of the Environment, Conservation, and Parks conducted a thorough investigation between 2012 and 2016 to determine the presence and levels of 14 PFAS in 25 drinking water systems, including water intakes and treated drinking water (Findings, 2023). This study revealed that all the systems under investigation contained PFOS, PFOA, and Perfluorohexanesulphonic acid (PFHxS). The most often observed PFAS were determined to be PFOS (6 ng/L) and PFOA (3 ng/L), both of which had the highest quantities. A comparison was made between the concentrations observed between 2007 and 2008 and the maximum surface water concentration collected from the same site. According to the findings, there were noticeably higher amounts of PFOA (66 ng/L) and PFOS (8.8 ng/L) throughout that time. Additionally, at least 30% of the samples taken from the Saint Lawrence River and other rivers contained at least one of the five PFAS compounds (PFHxA, PFHpA, PFOA, Perfluorononanoic Acid (PFNA), and PFOS) (Environment and Climate Change CanadaHealth Canada, 2023). The most frequently found chemical among them was PFOA and PFHxA, with rates of 72% and 59%, respectively. The median value for both was 2 ng/L, while the maximum concentration was 6 ng/L. In lake Memphremagog's raw water samples, a median concentration of 1 ng/L and a maximum concentration of 2 ng/L of PFOA and a median concentration of 1.5 ng/L and a maximum concentration of 3 ng/L of PFHxA were found additionally (Environment and Climate Change CanadaHealth Canada, 2023). These chemicals were found in treated drinking water at median and maximum concentrations of 1 ng/L each. In groundwater sources, different samples have been taken (Santé, 2023). It has been found that PFPeA represent 14%–17% of the samples, with a median

value of 4 ng/L and a maximum concentration of 48 ng/L. A median concentration of 3 ng/L and a maximum concentration of 30 ng/L of PFHxA were both found in 14%–17% of the samples. On the other hand, the samples containing PFOA and PFOS, were found to have between 6% and 4% of each, with median concentrations of 2 ng/L and 2 ng/L and maximum concentrations of 4 ng/L and 3 ng/L, respectively. (Ministère de l'Environnement et de la Lutte contre les changements climatiques (MELCC), 2022).

Also, the ecotoxicological effects of PFAS have been a subject of concern in other countries than Canada, including the USA, China, and Europe. Studies have highlighted the diverse impacts of PFAS on aquatic organisms, with evidence of adverse effects on amphibians, larval fish, and mollusks (Flynn et al., 2022; Gebreab et al., 2022; Lech et al., 2022). In the USA, PFAS have been found to contaminate water supplies, with significant contributions to human exposure through drinking water, especially downstream of contaminated areas (Smalling et al., 2023). This has raised concerns about the widespread drinking water contamination and its impact on human health (Brunn et al., 2023). In China, a global inventory for the emission of long-chain PFCA (C₄–C₁₄) has been published, quantifying different entry routes of PFAS to the environment (Brunn et al., 2023). The various applications of PFAS resulted in a varied environmental input in China, which has added to its pervasiveness and possible ecological damage. The study conducted by Pan et al. (2014) revealed that in the muscle tissues of nine fish species obtained from rivers in south China, only long-chain perfluorinated alkyl carboxylic and sulfonic acids from C₈ upwards were identified.

The ecotoxicology of PFAS in Europe is a topic of growing concern, as evidenced by recent research. A study published in the SETAC Journals by Ankley et al. provides a comprehensive assessment of the ecological risks associated with PFAS, highlighting the current state of the science and proposing a path forward (Ankley et al., 2021). Furthermore, the proposed ban on PFAS in the European Union has sparked discussions about its potential unintended consequences and the need for a careful consideration of the policy's environmental impact. The ban, if approved, would affect more than 12,000 PFAS, with implications for various industries, including automotive, aerospace, chemical, and electronics sectors (Ankley et al., 2021). These findings underscore the significance of considering not only the ecological risks but also the broader implications of regulatory measures related to PFAS in Europe.

4. Regulation of PFAS across the world

Considering the extended health-related harmful effects and persistence of PFAS compounds, different governmental agencies have imposed regulations and guidelines to decrease the adverse threat of PFAS to their own environment and people. For instance, a comprehensive action plan on PFAS in the environment was released by the EPA of the United States in February of 2019 (Country information - OECD). This plan mainly dealt with Safe Drinking Water Act (SDWA) and addressed PFAS limits in drinking water together with measurement guidelines for per fluorinated chemicals in water (Dalvi and Rossky, 2010). According to Method numbers 533 and 537.1, the current protocols to measure PFAS in drinking water includes those with carbon chain length between 4 and 12 within PFAS categories (Development, O. of R.&, n.d.). Following this regulation, the National Primary Drinking Water Regulation (NPDWR) establishes a Maximum Contaminant Level (MCL) for primary drinking water. The two most prevalent PFAS was reduced by the EPA in March 2023 to set a new limit of 4 ng/L for PFOA and PFOS, respectively, in drinking water (EPA, 2010). Nevertheless, it is worth noting that the unsafe level of all PFAS compound in different sectors (human health, aquatic life and industrial wastewater) is still not defined by EPA (Safe Drinking Water Act). Canada has also developed guidelines for PFOS and PFOA under the collaboration between the Canadian Environmental Protection Act, 1999 Health and the Federal-Provincial Territorial Committee on Drinking water. To replace the rules put in place in 2012, the Canadian government introduced the

Prohibition of Certain Toxic Substances Regulations, in May 2022. In particular, the use, import, manufacturing, and sale of PFOS, PFOA would be subject to significant restrictions under the new regulations. Besides, Canada reduced the concentration of PFOS and PFOA to 600 and 200 ng/L, respectively in drinking water (Health Canada, 2018).

In the European Union (EU) countries, guidelines to enhance protection of both environmental and human health from different chemicals are addressed by REACH ("Understanding REACH - ECHA," n.d.). The REACH program was first established by EU Regulation No. 1907/2006. REACH is a regulatory framework that came into action on June 1, 2007. This program included procedures, safety sheets, and appeal procedures correlated to variant chemicals listed under different classifications. According to this program, PFOA (added in June 2013), PFHxS, (added in July 2017), and PFBS, (added in January 2020) were listed as substances of very high concern ("Candidate List of substances of very high concern for Authorisation - ECHA," n.d.). However, PFOS is not listed in the mentioned category, but it is considered as a restricted compound under this program. More recently in June 2019, EU Regulation No. 2019.1021 was adopted and imposed more restriction and exceptions for PFOS in terms of manufacturing and exploitation in addition to the pre-existing EU Directive 2006/112/EC from 2006 (Regulation). Accordingly, food with a PFAS content (PFOA, PFNA, PFHxS, and PFOS) higher than 4.4 ng/kg body weight has been restricted. A similar restriction was imposed on semi-finished products and textiles or other coated materials where the PFOS concentration exceeded by 0.1 mass.% and 1 µg/m² per coated materials, respectively (Directive). As for drinking water, there is still not a universal value across the EU for the amount of PFOS and PFOA, and hence each country has set its own concentration limits (Olsavsky et al., 2020a). In Sweden, for example, the Swedish National Food Agency has imposed a total limit of 90 ng/L for a sum of 11 PFAS including PFBS, PFHxS, PFOS, 6:2 fluorotelomersulfonate (6:2 FTS), PFBA, and PFOA ("Risk management PFAS in drinking water and self-caught fish," n.d.). In the United Kingdom (UK), the Drinking Water Inspectorate (DWI) had adopted the limit concentration of PFOS in drinking water should not exceed 100 ng/L.

At the international regulatory level, persistent organic pollutants (POPs) have different classifications in terms of risk level within The

Stockholm Convention ("Listing of POPs in the Stockholm Convention," n.d.). The Annex A of this convention includes the most harmful substances subjected to elimination and not to be used by members. Currently, PFOA and all related PFOA compounds are categorized under Annex A because of being persistent, toxic and bioaccumulative. PFOS and all its related compounds, however, are listed in Annex B of the Stockholm Convention as a less harmful class of compounds ("Listing of POPs in the Stockholm Convention," n.d.). The presence of PFAS in drinking water is a major public health concern. To address this issue, governments and environmental agencies have established guidelines and standards for acceptable levels of PFAS in drinking water. A summary on regulation of different PFAS compounds in different countries is provided in Table 1. This Table compares the guidelines from Australia, Netherland, Sweden, United Kingdom, Canada, Denmark, Germany and Italy with those recommended by the US EPA and Europe Union (EU). Guidelines proposed by Denmark and US EPA are more restrictive for PFOA and PFOS. Denmark environmental agencies respectively proposed 2 ng/L and 2 ng/L, compared to 4 ng/L and 4 ng/L proposed by US EPA. The highest PFOS concentration of 600 ng/L is proposed by Canadian Health Guidelines, whereas the highest concentration of 560 ng/L for PFOA is proposed by Australia Department of Health. By comparison, a concentration of 500 ng/L is recommended by Europe Union irrespective of the type of fourteen PFAS mentioned in Table 2. It is worth noting that the guidelines imposed in each country take into account local realities.

Table 2 and Fig. 1, shows some examples of the most two detected PFAS compounds that have been identified in drinking water and their standard limits in a number of European countries, the US and Canada (Santé, 2023; Toft and Malaiyandi, 1984).

5. Management of PFAS compounds

All techniques dealing with degradation of PFAS can be categorized either in separation or degradation techniques (Fig. 2). For separation techniques, the aim is to separate PFAS compounds either physically or chemically, and then subsequently subject them to a degradation process. For degradation techniques, the goal is to decompose PFAS during the treatment. This section discusses the most used techniques/

Table 2
Summary on regulation of different PFAS compounds in different regions.

PFAS (ng/L)	US EPA ^a	Europe Union ^b	Countries							
			Australia ^c	Netherland ^d	Sweden ^e	UK ^f	Canada ^g	Denmark ^h	Germany ⁱ	Italy ^j
PFOA	4	500	560	90	90	100	200	2	90	500
PFOS	4	500	70	–	90	100	600	2	90	30
PFNA	1	500	–	–	–	–	–	–	–	–
PFBA	–	500	–	–	–	–	–	–	–	–
PFBS	1	500	–	–	–	–	–	–	–	–
PFHxS	1	500	70	–	–	–	–	–	–	–
PFHxA	–	500	–	–	–	–	–	–	–	–
PFPeA	–	500	–	–	–	–	–	–	–	–
PFHpA	–	500	–	–	–	–	–	–	–	–
PFOSA	–	500	–	–	–	–	–	–	–	–
PFDA	–	500	–	–	–	–	–	–	–	–
6:2 FTS	–	500	–	–	–	–	–	–	–	–
8:2FTS	–	500	–	–	–	–	–	–	–	–
Gen-X	1	500	–	–	–	–	–	–	–	–

^a (EPA, 2010): National Primary Drinking Water Regulation (NPDWR).

^b ("Understanding REACH - ECHA," n.d.): The European Union (EU).

^c (Effects, 2017): Dept. Of Health -Australia (DOH).

^d (Noorlander et al., 2010): Environmental Protection Agency- Netherland.

^e ("Understanding REACH - ECHA," n.d.):The European Union (EU).

^f (DWI, 2021): Drinking Water Inspectorate - United Kingdom (DWI).

^g (Environment and Climate Change Canada and Health Canada, 2023): Health Canada (HC).

^h (Danish Environmental Protection Agency, 2015):Environmental Protection Agency (EPA).

ⁱ (UBA, 2022): Germany Ministry of Health (GMH).

^j (Mastrantonio et al., 2018): National Health Institute-Italy.

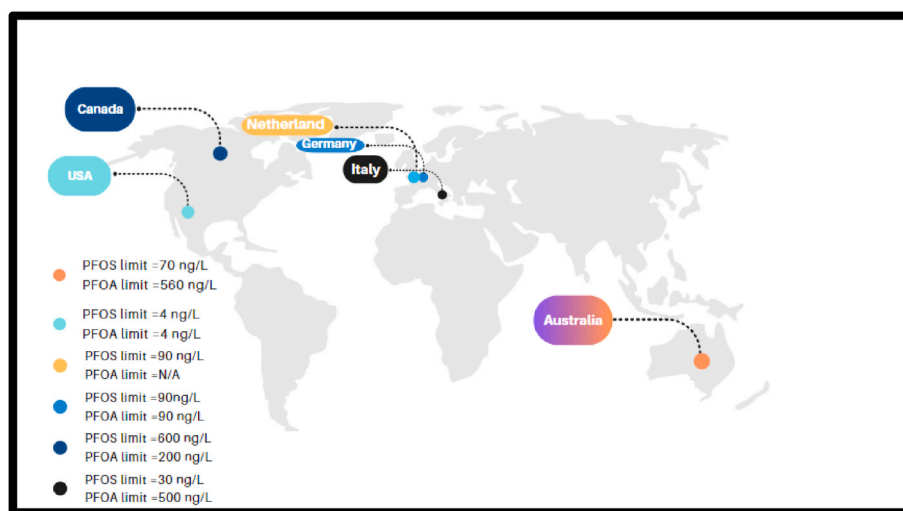


Fig. 1. Regulation of PFOA and PFOS for drinking water in selected countries.

approaches for complete degradation of PFAS compounds. Although some of these degradation approaches are still being used at the lab scale, some others have already been used in the field. Fig. 2 summaries possible techniques to treat PFAS.

5.1. Separation techniques

5.1.1. Activated carbon

Activated carbon is recognized as one of the most widely used adsorbents that can reduce PFAS concentrations in water to very low levels (ng/L) because of its high surface area and adsorption properties (Zhi and Liu, 2016). The PFAS molecules are drawn and bound to its surface, which effectively removes them from the water. However, the removal efficiency of activated carbon considerably varies depending on the PFAS compound type (Appleman et al., 2014). In general, short-chain PFAS are adsorbed less than long chain PFAS, and perfluorosulfonic acids are adsorbed better than perfluorocarboxylic acids with the same chain length (Eschauzier et al., 2012). Therefore, depending on the particular molecule, the point at which the adsorbent gets fully saturated and the adsorbate exits the column in the effluent may vary. (Kah et al., 2017). For instance, one study compared the same amount of

activated carbon powder for removal of three PFAS compounds (Hopkins et al., 2018). The results showed under the same experimental conditions 30%, 80%, and 95% removal was achieved for GenX, PFOA, and PFOS, respectively. Most of the studies on PFAS adsorption by activated carbon have been performed at the laboratory scale, and the information on its long-term application under real conditions is rather scarce. Among the documented cases, an installation has been monitored for PFAS removal under real conditions over a period of five years. The number of bed volumes treated before the breakthrough was reported as 60,000 for PFOS, 30,000 for PFHxA + PFOA, and 5000 for PFBA (Appleman et al., 2014). To conclude, it has been shown that operational parameters of AC technology, such as the type of coal, bed size, and hydraulic loading rate, also affect its filter run time.

It's important to note that activated carbon is not a long-term PFAS treatment option and may need to be periodically replaced or reactivated to keep its absorbing capacity working. Additionally, it's important to manage the used activated carbon appropriately to stop PFAS from being released back into the environment. In general, activated carbon is a widely used and efficient treatment method for removing PFAS; however, its effectiveness can change depending on the circumstances. To completely remove PFAS from water sources, it is

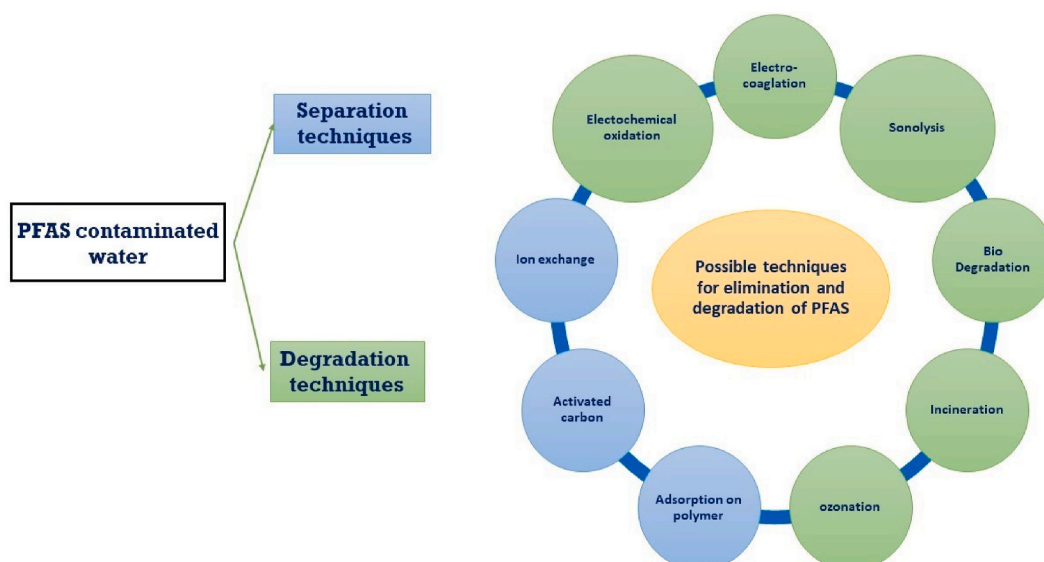


Fig. 2. Possible techniques for PFAS degradation and elimination.

frequently employed in combination with other treatment technologies (Du et al., 2014).

5.1.2. Ion exchange resins

Ion exchangers are solid carriers usually in the form of small porous resin beads often called resins. Ion exchangers consist of an immobile cross-linked polymer matrix with charge carrying functional groups and mobile counterions. The PFAS ions are attracted to the functional groups of the resin when PFAS-containing water flows through the resin bed, where they are exchanged with other ions like chloride or sulphate ions. The PFAS can be efficiently eliminated from the water with this technique. In water treatment, synthetic ion exchangers are based on polystyrene or polyacrylate in form of porous plastic beads with a diameter ranging from 0.3 to 1.3 mm (Edel et al., 2018). A resin's affinity for an ion is often expressed as a separation factor or selectivity factor. The exposure separation factor is equal to the distribution of ions between phases, which depends on the distribution of the second ions between the two phases. By considering the molecular structure of PFAS compounds, ion-exchange resins could effectively remove the majority of PFAS compounds from water. This is due to fluorinated carbon chains with a hydrophobic nonionic tail and a negatively charged anionic head allow.

In neutral pH, surface/ground waters contaminated with PFAS are negatively charged due to dissociation of carboxylic acid or sulfonic acid functional groups. The hydrophobic tail of PFAS binds to hydrophobic backbone of the resin while positively charged ion-exchange sites attract the anionic head. For instance, it has been shown that positively charged anion exchange resin resulted in a greater removal of PFAS with the sulfonic acid head (with removal rates of up to 90–99%) (Lampert et al., 2007). Individual resins' capacity to remove PFAS depends on a variety of conditions. For instance, PFAS capacity of a particular resin is highly affected by the water's inorganic content including the ion's type and concentration.

Resin type and properties have a direct effect on degradation efficiency of PFAS compounds. For example, while the A-714 ion exchanger (an anion exchange resin) could remove over 99% of PFOA and PFOS after 25 h of contact, the A-244 ion exchanger achieved only 33% removal at the same period (Lampert et al., 2007). Amberlite® IRA-400 resin has been also reported to exhibit a high-binding capacity for PFOS and PFOA (Dixit et al., 2021c). Batch tests have shown that polyacrylic resins have higher reaction rates and binding capacities for PFOS and PFBS than polystyrene resins (Rahman et al., 2022). Another study by Carter et al. (Carter and Farrell, 2008) showed that PFOS removal was more effective than PFBS, probably due to stronger hydrophobic interactions between the resin polymer backbone and longer carbon chains. A systematic study of three different resins (AE, Purolite® A600) showed that they had greater affinity for long-chain PFAS and stronger affinity for perfluoro sulfonic acid for the same chain (McCleaf et al., 2017). Apart from type of resin, PFAS chain length influences degradation efficiency. For instance, ion exchangers may release short-chain PFAS due to higher binding of long-chain PFAS resulting in aggregation or micelle formation. In the case of treating municipal wastewater, anion-exchange resin (AIX) properties, such as the porosity and polymer matrix, can affect PFAS treatment efficacy, and strong base AIX resins are effective in removing PFAS species that exist as anions in drinking water. Although AIX resins have greater adsorption capacity than GAC, they are typically limited to a single use for drinking water applications (Chularueangakorn et al., 2014; Pétré et al., 2022; To, P.A., Discharge, P.; Zaggia et al., 2016). The capacity of ion exchange resins to remove PFAS is constrained, and once the resin reaches saturation, it must be regenerated or replaced. To prevent any leakage into the environment, the spent resin that contains trapped PFAS needs to be disposed of properly.

Ion exchange is frequently used in conjunction with other treatment techniques, much like activated carbon, to produce the best PFAS elimination. It can be a good alternative, especially for wastewater with

high PFAS concentrations or as part of a multi-stage treatment system. To ascertain the effectiveness and suitability of ion exchange for a specific application, the application's unique circumstances and PFAS characteristics should be considered.

5.1.3. Adsorption on polymers

Beside activated carbon and ion exchangers, the use of polymer-based adsorbents is one of the methods for removing PFAS-contaminated water. Because of their high surface area and porosity, polymer adsorbents can effectively remove PFAS from water (La et al., 2019). The specific functional groups on the polymer surface area chosen to have a high affinity for PFAS, facilitating thus their attraction and binding. The type of polymer utilized, the structure and concentration of PFAS compound, and the operating circumstances of the treatment system are among several parameters that affect the adsorption efficiency of PFAS by those polymers. Two studies have been conducted to investigate the efficacy of polymer-based adsorbents in the treatment of PFAS. Taking the example of Liu et al., who investigated the efficiency of a polydopamine-coated magnetic nanocomposite in removing PFAS from water. With a maximum adsorption capacity of 70.3 mg/g, the nanocomposite effectively removed PFAS from water (Tang et al., 2020). By using another polymeric adsorbent, namely polyethyleneimine-modified cellulose nanocrystals (PEI-CNC), Wang et al. reported that a maximum adsorption capacity as high as 525.4 mg/g for PFAS can be achieved (Ateia et al., 2018). However, the use of polymer-based sorbents also has some drawbacks, such as the slowing down of the adsorption process limiting thereby the overall throughput of the system. Also, the sorbent can become saturated with the adsorbent PFAS, reducing its effectiveness, and requiring its replacement. Another limitation of polymeric sorbents is their relatively high manufacturing cost, limiting thereby their use for large-scale treatment of PFAS-contaminated water. In sum, polymer-based sorbents have the potential to remove effectively PFAS from water, but they also have some drawbacks. R&D effort is still needed to come up with more efficient and cheaper methods to remove PFAS-contaminated water.

In order to study the treatment achievability of PFAS, using different methods namely GAC, AIX, and Polymer adsorption it has been shown that these methods can effectively reduce the concentrations of shorter chain PFCA and PFSA to below detection limits. However, achieving such low concentrations (of <1–2 ng/L for individual PFAS) requires proper configuration and operation of the treatment systems. It should be noted that achieving such low concentrations can lead to challenging operating conditions, such as long-empty bed contact times or frequent media regeneration or replacement, which may not be feasible for some water treatment facilities from a practical or economic standpoints (Santé, 2023).

5.1.4. Electrocoagulation

Electrocoagulation (EC) is an electrochemical process in which coagulating agents are *in situ* generated by using an electric current (Wang et al., 2016). This technique has been successfully used for treating PFAS-contaminated water and wastewater and found to be quite promising. For instance, Hou et al. have shown that aluminum electrodes are more effective than iron ones in removing PFAS by means of EC. Removal rates of up to 90% were achieved for some compounds (Merino et al., 2016). PFAS can be removed by electrocoagulation by adding zinc hydroxide as flocculants (Lin et al., 2015). Indeed, it has been reported that zinc hydroxide-based EC can remove more than 95% (within 20 min) of PFOA and PFOS aqueous solutions polluted in the range of mg/L to g/L (Liu et al., 2018). Electrocoagulation generated flocs of zinc hydroxide had a significantly higher capacity and adsorption rate. Electrocoagulation experiments using a Fe electrode have been also reported to result in 99% of PFOS removal after 50 min of treatment (Yang et al., 2016). Li et al. have used EC with iron electrodes to remove up to 98% of PFAS from landfill leachate (Yang et al., 2016). They have found that PFAS removal efficiency increases with increasing current

density and initial PFAS concentration. Aluminum electrodes were also used with EC to removal of PFAS from aqueous solution, and were shown to be capable of achieving high removal rates for several PFAS compounds (with removal rates of up to 99.9% for some compounds) (Shi et al., 2021). This being said, there are still insufficient data to fully evaluate fully and rationally the efficiency of the EC process. In particular, the extent to which EC can effectively remove short-chain PFAS chemicals is still a subject of discussions among the research community. In addition, EC treated waters with high salt content prevents the groundwater reservoir from being simply drained into the ground. As a result, the total cost effectiveness of this method seems to be limited.

Among PFAS tested, PFOA and PFOS remain by far the compounds which have been most studied to show the performance of separation techniques for the removal of such refractory organic compounds. Table 3 compares some physicochemical separation techniques used to remove PFAS from water. From this table, it can be seen that physicochemical treatments using granular activated carbon and ion-exchange resins are effective processes to remove PFAS. Removal rates of up to 100% can be recorded using either granular activated carbon or ion exchange resins. Electrocoagulation processes using either iron or aluminum electrodes can also be effectively used to remove PFOS and PFOA from waters. It is worth noting that the two technological approaches more or less established for PFAS removal from water are the adsorption using granular activated carbon and filtration on ion exchange resin. However, this type of treatment leads to a pollution transfer (from the liquid phase into the solid phase), rather than a real depletion. This results in a supplementary step for granular activated carbon management or ion exchange resin regeneration. These drawbacks reinforce the necessity to develop advanced oxidation

technologies for PFAS degradation. Moreover, electrocoagulation (EC) reactors, which range from basic to sophisticated designs, have demonstrated high efficiency, compactness, relatively low cost, and complete automation in the treatment of water contaminated with various pollutants, including PFAS such as PFOA and PFOS (Giri et al., 2011a). According to recent research, PFAS can rapidly sorb on the surface of zinc hydroxide particles produced by EC. (Wang et al., 2010). The efficiency of EC for PFAS removal can be influenced by factors such as the specific composition of the water, electrode material, current density, and operational parameters. Recent study has shown that PFAS concentration during EC treatment at relatively high current densities ($>1 \text{ mA cm}^{-2}$) can lead to the formation of foam, which facilitates the separation of PFAS from wastewater with high PFAS concentrations. According to this study, high PFAS concentrations (individual concentrations larger than $0.1 \text{ }\mu\text{M}$) during electrocoagulation resulted in the generation of EC-derived foam. Current densities greater than 1 mA cm^{-2} were also observed to be associated with this phenomenon. The distribution of PFAS in the various phases was affected by the fractionation of PFAS into foam. In addition to the surfactant qualities of PFAS, the formation of foam appeared to be caused by gases that developed on the electrodes (O_2 on the anode and H_2 on the cathode), and it was also correlated with the PFAS concentration. Foam formation can be facilitated by the study's demonstration that a sizable portion of the removed PFAS segregated into foam under EC treatment conditions. This can be useful for separating PFAS from wastewater that contains high concentrations of PFAS, such as reverse osmosis retentate and wastewater treated by electroplating. On the flocs produced *in situ* during EC, PFAS is rapidly adsorbed for low PFAS concentration fluids, such as groundwater (Guo et al., 2019). By decreasing mass loading into

Table 3
PFAS removal using separation techniques.

Separation processes	Matrix	PFAS	Operating conditions	Results and comments	Ref.
Granular activated carbon	Drinking water	PFOS	$T = 20^\circ\text{C}$, Treatment time = 24 h, $\text{pH} = 3$, $C_{\text{PFOA}} = 1.0 \text{ mg/L}$, $C_{\text{PFOS}} = 1.0 \text{ mg/L}$	98.6% removal rate of PFOS and PFOA	Zhang et al. (2021)
	Synthetic water	PFOA	$T = 25^\circ\text{C}$, Treatment time = 24 h, $\text{pH} = 3$, $C_{\text{PFOA}} = 250 \text{ mg/L}$	97% removal rate of PFOA	Deng et al. (2012)
	Synthetic water	PFOA	$T = 25^\circ\text{C}$, Treatment time = 24 h, $\text{pH} = 4$, $C_{\text{PFOA}} = 100 \text{ mg/L}$	Adsorption capacity = 368.2 mg/g 100% removal rate of PFOA	Lei et al. (2022)
Ion exchange resin	Synthetic water	PEDoDA, PFOA, PFOS, PFBA, PFBS, GenX, 8:2FTS	Two types of resins were used: i) Poly-acrylic organic scavenger resin (A860) having an exchange capacity of 0.8 meq/mL ; ii) Polystyrenic PFAS-specific resin (A592) having an exchange capacity of 0.9 meq/mL . Initial conc. of individual PFAS = $10 \text{ }\mu\text{g/L}$. Initial conc. of DOC = 5 mg/L	A860 resin removed 60–70% DOC in addition to PFAS A592 resin was selective for PFAS removal; Complete removal could be achieved below 10 ng/L .	Dixit et al. (2021b)
	Groundwater	PFBA, PFBS, PFHxA, PFHxS, PFOA, PFOS	Four types of resins were used: i) AER ^a = RA67, ii) AER ^a = IRA910, iii) AER ^a = IRA96, iv) AER ^a = IRA400 Initial conc. of individual PFAS = $0.1 \cdot 10^{-3} \text{ mol/L}$ Charge Density = 1.0 eq/L $\text{pH} = 6$	IRA910 resin was selective as the effective resin to remove 90% of PFOS, 50% of PFHxS, 30% of PFOA and below 25% of each PFBA, PFBS and PFHxA. IRA67 is the most effective adsorbent for removing long-chain PFAS. IRA400 and IRA96 resins exhibits more effective adsorption performance for short-chain PFAS.	Maimaiti et al. (2018)
	Groundwater	PFOA, PFDA, PFBA, PFBS, PFOS	One anionic macroporous acrylate-based resin Purolite® A860, Initial conc. of individual PFAS = $10 \text{ }\mu\text{g/L}$ Charge density = 0.8 meq/mL ($221 \text{ mg dry weight} = 1 \text{ mL}$)	Total PFAS removal with $>70\%$ DOC	Dixit et al. (2021a)
Electrocoagulation	Synthetic groundwater	PFOS	$C_{\text{PFOS}} = 1 \text{ mg/L}$, $\text{pH} = 7$, $C_{\text{NaCl}} = 1 \text{ g/L}$; Electrodes: aluminum, iron, and zinc anodes ($6 \text{ cm} \times 4 \text{ cm} \times 0.2 \text{ cm}$), Current intensity = 3 A	PFOS removal rate = 100%	Bao et al. (2020)
	Synthetic water	PFOA	$C_{\text{PFOA}} = 0.25 \text{ mM}$, $\text{pH} = 7$, $C_{\text{NaCl}} = 10 \text{ mM}$, Electrodes: zinc anodes (3.8 cm diameter and thickness = 0.2 mm)	PFOA removal = 100%	Mu et al. (2021)
	synthetic groundwater	PFBS	$C_{\text{PFBS}} = 1 \text{ mg/L}$, $\text{pH} = 5$, current intensity = 3 A , electrodes: Fe–Fe and Al–Zn ($6 \times 4 \times 0.2 \text{ cm}$), $C_{\text{NaCl}} = 2 \text{ ng/L}$,	PFBS removal = 81%	Liu et al. (2023)

a = anion exchange resin.

b = bed volumes to breakthrough.

downstream processes like medium or membrane filtration, the EC process may be used as a pretreatment stage. Industrial wastewater treatment already makes substantial use of small-scale EC units, and EC offers some promise for removing PFAS from water (Guo et al., 2019).

5.2. Degradation techniques

5.2.1. Photocatalytic process

Light assisted techniques (such as photolysis and photocatalysis) where degradation occurs via an interaction of emitted photons with target contaminants or by triggering a chain of reactions after light absorption by the catalyst. As well explained elsewhere, when a semiconductor (SC) catalyst absorbs a photon with an energy ($h\nu$) equal or greater than its band gap, an electron is bumped from the valence band to the conduction band of the SC creating a pair of electron-hole. If the excited electron hole could be separated, then the electron moves toward the bulk and consequently the hole migrate to the catalyst's surface (in the case of n-type semiconductor and vice versa for p-type semiconductors). Separated photocharges may subsequently participate in reaction with an electron acceptor/donor species already accumulated on the interface of semiconductors/electrolyte leading to reduce or oxidize them. By considering TiO_2 (one of the most studied photocatalysts), photoexcitation occurs when the energy of incoming photons is greater than 3.2 eV (or 3.02 eV), for the anatase (or rutile) phase, respectively. Moreover, by appropriately doping the TiO_2 (by N atoms for example), its band gap can be narrowed down to 2.2 eV, making it more suitable to use visible sunlight for photocatalytic reactions (Delegan et al., 2014; Pandiyan et al., 2016; Yan et al., 2013). Accordingly, holes on the surface of TiO_2 nanomaterials can trigger a chain of oxidation reactions in which hydroxyl radicals ($\cdot\text{OH}$) - as one of the most oxidative radicals - is produced as a product (Guo et al., 2019).

In the case of photolysis, comparing C-F dissociation energy (544 kJ/mol) and photon energy of most exploited UV sources (254 nm) as 471 kJ/mol reveals low efficiency of PFAS degradation by direct photolysis at long reaction time (Giri et al., 2011b). However, exploiting shorter wavelength UV light (i.e., 185 nm, 646.6 kJ/mol) have been shown to degrade efficiently PFAS compounds (Merino et al., 2016). Although, the efficiency of photolysis also depends on the environment parameters; for instance, it has been reported that the presence of chemical reagents such as Fe^{3+} , TiO_2 , $\text{S}_2\text{O}_8^{2-}$, CO_3^{2-} , and IO_4^- in the electrolyte media lead to an increase in the efficiency of PFAS degradation thanks to formation of reactive and potent oxidative species, i.e., $\cdot\text{OH}$, $\text{CO}_3^{\cdot-}$, $\text{H}\cdot$, etc. (Wang et al., 2010). As for photocatalysis, different metal oxides such as titanium dioxide (TiO_2), gallium oxide (Ga_2O_3), and indium trioxide (In_2O_3) have been reported to be effective for PFAS removal (Chen et al., 2011; Li et al., 2012; Yu et al., 2009; Zhao et al., 2015a). Nevertheless, one important issue in photocatalysis is the organic contaminants adsorption on the catalyst surface, which leads to its poisoning and significantly reduces its catalytic activity. Therefore, surface modification could be considered as a strategy to increase the performance of photocatalytic system (Sharma and Bhattacharya, 2016).

Previous works have studied the performance of pure and modified TiO_2 because of its attractive features, namely its high chemical and physical stability, nontoxicity, low cost and availability (Robert, 2007; Westerhoff et al., 2011). However, it is generally admitted that pure TiO_2 is not effective for PFAS degradation because of its relatively large band gap and relatively high recombination rates (Panchangam et al., 2009). Among the most promising approaches to overcome this disadvantage and make TiO_2 even sensitive to visible light, we can mention its co-doping (by N and W, for example) (Abdullah et al., 2018; Delegan et al., 2018) or its nano decoration by CoO nanoparticle catalysts (Favet et al., 2020, 2023). These approaches have been shown to reduce the electron-hole recombination and to increase the number of absorbed photons and hence the density of generated photo charges, all leading to an effective increase of the oxidizing species concentration. Other

groups have shown that PFOA degradation by molecularly-imprinted-polymer-modified- TiO_2 composites was much more effective than that of pure TiO_2 .

Gallium oxide ($\beta\text{-Ga}_2\text{O}_3$) is another photocatalyst that has been used for PFAS decontamination. Gallium oxide has a band gap of 4.8 eV which is even wider than that of TiO_2 (3.02–3.2 eV), and so has a higher oxidation-reduction potential. Interestingly, complete degradation of PFOA has been reported when using Ga_2O_3 photocatalysts (Shao et al., 2013; Zhao and Zhang, 2009; Zhao et al., 2016). In particular, Shao et al. (2013) compared two types of $\beta\text{-Ga}_2\text{O}_3$, namely sheath-like and commercial ones, where the difference was mainly attributed to their adsorption and reaction sites with the enlarged surface area. Based on their result, photocatalytic performance of PFOA was 100% (in 45 min) and 15% (in 180 min) for $\beta\text{-Ga}_2\text{O}_3$ and commercial Ga_2O_3 , respectively. However, the contribution of UV source on degradation of PFAS compounds should be further studied.

5.2.2. Incineration process

Incineration is an effective, yet primitive, approach for the destruction of many forms of PFAS and other organic compounds. Materials containing PFAS may experience thermal degradation when heated to a high temperature (600 °C–1400 °C) in an incinerator, which will result in the destruction of the compounds (Olsavsky et al., 2020b). Such temperature is capable of destroying C–C or C–F bonds and potentially lead to the degradation of PFAS compounds (Tsang et al., 1998). However, the effectiveness of this method of PFAS compounds degradation is currently limited to the industrial scale (Ahmed et al., 2020; Petridis and Dey, 2018). It is also worth mentioning that industrial incineration cannot break down organo-fluorinated compounds. Thus, incomplete destruction of PFAS compounds can result in the production of toxic and volatile chemical by-products such as 1-H-perfluoroheptane (Krusic and Roe, 2004). For instance, 11 PFAS compounds out of 31 polytetrafluoroethylene (PTFE) materials were detected after subjecting to incineration under some specific incineration conditions (Aleksandrov et al., 2019). Therefore, the whole environmental impact of incineration, including the energy requirements and potential toxic emissions from the incineration process, must also be taken into account. Any pollutants released during incineration should be captured and treated using appropriate air pollution control systems, such as scrubbers or filters. In sum, incineration can be used for the degradation of PFAS containing matrices, but it needs to be done with careful attention to the generated by-products along with appropriate operating conditions while mitigating the adverse environmental effects. To maintain the process effectiveness and environmental safety, it is crucial to adhere to legal requirements and standards for incineration and to handle appropriately the leftovers that result from such a process.

5.2.3. Biodegradation

Biodegradation of PFAS is another promising approach involving the use of identified microorganisms that are capable of breaking down PFAS compounds. Biodegradation is a natural process that biologically recycles essential elements within the earth's biogeochemical cycles. In other words, biodegradation processes are typically catalyzed by enzymes arranged in pathways that convert chemicals into end products via a series of intermediates. Most of the preliminary results reported so far were disappointing due to the high stability and the complexity of the PFAS molecules, as mentioned before. Nevertheless, several attempts aiming to degrade PFAS with various microbial species have been made (Chen et al., 2019; Gun et al., 2014; Senevirathna et al., 2022). Recently, some research has been done on different microbial species or mixed cultures that may degrade some PFAS species such as PFOA and PFOS precursors (Badgujar et al., 2020). However, the degradation mechanisms proposed by these authors involved pathways and some intermediate metabolites which were not experimentally detected. In any case, the biodegradation of PFAS cannot deliver the percentages required for the removal of the non-fluorinated fraction reported

(Shahsavari et al., 2021). Although, Luo et al. (2018) also found that the biodegradation of PFOA is difficult and limited. However, PFOA is degraded using the oxidation reaction of enzyme-catalyzed humus through the medium of soybean meal, which is a natural organic matter. This study has shown that the PFOA molecules in soybeans are broken down by the free radicals in soybeans (Luo et al., 2018). The authors found that in soil samples, the most significant degradation was for the carbon-carbon bonds in PFOA, and this can explain the 40% of PFOA which was degraded in 140 days while in soil and water it reached 24%.

Most of the previously reported biodegradation processes of PFAS species are limited to removal of non-fluorinated fractions rather than defluorination via C–F bond scission. Due to the different structures of PFAS, the proposed biotransformation mechanisms differ according to its PFAS categories (PFOA, PFOS, neutral PFAS, etc.). For a better understanding, the expression of the *Rhodococcus jostii* RHA gene was kept under an investigation of bacteria capable of degrading in the presence of 6:2 FTS oxygenase, dehalogenase and desulphurize. The head group of sulphonic acid prevents defluorination. Fluorotelomer alcohols are easily excreted under treat. In the absence of a sulphur-containing medium, alkanesulfonate monooxygenase was able to desulfurize 6:2 FTS. This desulphurization promoting subsequent defluorination by alkane monooxygenase (Yang et al., 2022). Furthermore, it can be designed a future degradation mechanism to involve these enzymes in degradation, to use synthetic materials to mimic enzymatic degradation. Also, a synergistic system can be designed to improve decomposition of PFAS. The unique structure of PFAS species contains C–F bonds, which are thought to be the strongest in organic chemistry (Leung et al., 2022). With a bond dissociation energy of up to 130 kcal/mol, fluorine forms the single strongest bond with carbon. C–F bond cleavage has been successfully reported using energy-intensive approaches (e.g., advanced oxidation or reduction processes), but it has rarely been demonstrated in detail during biological processes (Cui et al., 2020). Nevertheless, fluorine ions were detected in some PFAS biodegradation trials, indicating that microbes can break C–F bonds under aerobic or anaerobic conditions (Sáez et al., 2008).

A more complete defluorination seems to be impossible due to the high strength of C–F bonds (Tiedt et al., 2016). Although the precise pathways for PFOA and PFOS degradation are currently unknown, the production of shorter-chain PFAS species, including perfluorobutanoic acid (HFBA), PFPeA, PFHxA, and PFHpA, during PFOA degradation and PFBS, HFBA, and sulphate ions during PFOS degradation, indicated that the C–F bond, C–C bond were cleaved during the Feammox process (i.e., anaerobic oxidation of ammonium under iron-reducing conditions) for PFOA and PFOS biodegradation (Zhang et al., 2022).

So far, most of the reported biodegradation processes for PFAS species are limited to the removal of non-fluorinated moieties, rather than defluorination by breaking the C–F bonds. Based on different structures of PFAS, the proposed biotransformation mechanisms vary among PFAS categories (e.g., PFAA, FluoroTelomer Sulfonic Acid (FTSA), neutral PFAS) (Douna and Yousefi, 2023). Several authors studied different types of PFAS by-products such as Perfluorooctaneamido quaternary ammonium salt (PFOAms) and perfluorooctane sulfonamide quaternary ammonium salt (PFOSAmS) (Zhang et al., 2022). These different by-products were used in AFFFs. Mejia et al. were the first to study the primary biotransformation product of PFOAms was PFOA (30%), while the products of PFOSAmS were FOSA (<0.005%), PFOS (0.3%), and perfluorooctanesulfonamidoamino amines (PFOSAm) (10%) (Zhang et al., 2022). This study demonstrated that PFOAms and PFOSAmS are biodegradable. But since there was no breaking of the C–F bonds, it was clear that the biotransformation processes were limited to the non-fluorinated regions.

5.2.4. Sonolysis process

Over the past two decades, sonolysis treatment has attracted much attention due to its high efficiency to degrade a wide range of contaminants (such as dyes, antibiotics, surfactants, etc) in aquatic media

(Douna and Yousefi, 2023; Mason and Lorimer, 2002). When ultrasonic waves propagate in a liquid medium, voids or cavities are formed. It is reported in literature that local temperature and pressure in the vicinity of the cavities can reach up to 5000 K and 1000 atm, respectively (Suslick and Flannigan, 2008). Therefore, it is expected that such high local temperature and pressure leads to pyrolysis of H₂O and O₂ molecules. Accordingly, sonolysis is associated with the production of hydroxyl radicals (•OH), hydrogen (•H), and superoxide radicals (O₂•[−]) that can oxidize the contaminants (Collings et al., 2010). During sonolysis treatment, operational parameters such as medium temperature, ultrasonic power density, and frequency may affect the degradation efficiency (Collings et al., 2010).

In the case of PFAS degradation by sonolysis, the effect of such operating parameters on degradation efficiency has been reported by several groups. For instance, Campbell et al. (2009), investigated the effect of acoustic frequency on removal efficiency of different PFAS compounds, where the frequency was varied from 202 to 1060 KHz. According to their results, the maximum degradation of individual PFAS compounds may not follow the same order of frequency. They have found that while the maximum degradation of PFOS, PFHpA and PFHpS was achieved at 358 KHz, it was achieved at 610 KHz for other compounds like as PFBA and PFBS (Sidnell et al., 2022). A similar tendency was also reported by James et al. (2020) where they found that the maximum degradation of PFOS (96.9 %) occurred at a frequency of 400 KHz. Increasing the ultrasonic frequency to 1000 KHz was found to decrease slightly from the degradation to 91%. Therefore, it seems that there is a threshold frequency for each individual PFAS compound. Going beyond that frequency does not necessarily increase their degradation. Since the degradation of PFAS mainly depends on the adsorption at the bubble-water interface, this may be explained by a certain correlation between frequency and size of produced bubbles. In general, it has been reported that working at relatively low frequencies resulted in larger bubbles with higher energy, and vice versa (Cheng et al., 2010).

On the other hand, removal efficiency depends also on the characteristics of individual PFAS compounds. In fact, sonolysis can potentially destroy a variety of PFAS compounds including both long- and short chains (Fernandez et al., 2016). Taking two PFAS compounds with the same chain length as a case study (PFOS and 6:2 fluorotelomer sulfonate), Fernandez et al. (2016) showed that PFOS degrades 2.3 times faster than the 6:2 fluorotelomer sulfonate under the same experimental and operational conditions. In the same study, it was shown that degradation rate of perfluoroalkyl sulfonic acids decreases with the increase of chain length. Therefore, the PFBS degrades 1.9 times less than PFOS. The same behaviour was observed in the sonolysis treatment of perfluoroalkyl carboxylic acids, and it shows that the PFBS degrades 2.3 times slower than PFOA (Cheng et al., 2010).

Although the main advantage of sonolysis is that there is no production of unwanted by-products (e.g., hydrogen fluoride, chlorine gas, bromate and perchlorate), there are nevertheless some drawbacks of this technique. The efficacy of sonolysis has been demonstrated in the PFAS laboratory, but has not been yet implemented in real conditions (Fernandez et al., 2016). Moreover, sonolysis treatment usually takes several hours and this may disqualify this process as a suitable candidate for continuous treatment. Lastly, most of the studies are performed at very high initial concentrations of PFAS. So, it is not yet known whether sonolysis is capable of degrading PFAS at low concentrations to meet the regulations.

5.2.5. Electrochemical oxidation

Electrochemical oxidation (EO) is one of the most widely used approaches in wastewater treatments to degrade pollutants. Electro-oxidation can be performed either in three electrodes or two electrodes' configurations. However, the two electrode configurations (simply consists of an anode and a cathode) is the preferred one for industrial and large-scale applications. In this technique, a constant

potential (potentiostatic electrolysis) or current (galvanostatic) is driven between anode and cathode in the presence of a supporting electrolyte. It consequently results in the formation of highly reactive oxidizing species that react with the target compounds. In principle, two types of oxidation are responsible for degradation reactions, direct and indirect oxidation. In direct oxidation, hydroxyl radicals as one of the strongest oxidation species with a redox potential of 2.8 V are formed at the anode surface and react with the contaminants. Indirect oxidation occurs when appropriate oxidizing species are available in the electrolyte either from supporting electrode or target matrix. For instance, it is reported that $\text{SO}_4^{\cdot-}$, ClO^- , H_2O_2 , HOCl , Cl_2 , etc. oxidant species are formed during electrolysis. These oxidizing species are released in the solution and they can simultaneously react with pollutants.

Electrochemical oxidation comes with some advantages such as low energy cost, working in ambient condition, simplicity, capability in mobile and static configurations, and it does not require the use of any strong oxidative reagents (Garcia-Segura et al., 2018). On the contrary, there are some disadvantages associated with EO such as unselective interaction, high potential of toxic or more complex by-products, and loss of efficiency due to the mineral deposition on the anode surface (Nzeribe et al., 2019; Schaefer et al., 2019a). Nevertheless, it has been demonstrated that EO can successfully degrade PFAS at both lab and pilot scales (Nzeribe et al., 2019). The PFAS degradation mechanism is induced by both direct and indirect oxidation. In direct oxidation, the applied electrical energy forces the electron transfer from oxygen atoms of the functional group (i.e., carboxyl group or sulfonate group) to the anode. In indirect oxidation, on the other hand, PFAS degradation mechanism involves a chain of reactions due to electrochemical formed oxidant species (i.e., hydroxyl radical) in which intermediate products decompose from the parent compounds and defluorination occurs as a consequence (Nzeribe et al., 2019; Schaefer et al., 2019a; Tang et al., 2021).

The performance of the EO treatment performance depends on several parameters such as electrodes' nature and properties, electrochemical surface area, applied current density, initial concentration of target compounds, etc. The main feature of an appropriate electrode for EO application is its high physical and chemical stability (i.e., the atoms of electrode materials do not participate in electrochemical oxidation and hence their oxidation state remain intact during EO process). Accordingly, the most studied effective inactive electrodes are doped SnO_2 , PbO_2 , sub-stoichiometric TiO_2 (Magneli), and Boron Doped Diamond (BDD) electrodes. Among these electrodes, BDD has attracted much attention in EO techniques and also for PFAS degradation. This is due to the fact that BDD electrodes are highly stable under wide potential anodic polarization thanks to carbon atoms being in sp^3 hybridization. Interestingly, it has been shown that BDD have the highest reactivity with PFOA and PFOS degradation (Le et al., 2019; Schaefer et al., 2017; Zhuo et al., 2012). Recently the researchers explored the use of a $\text{Ti}/\text{SnO}_2\text{-Sb}$ anode for the electrochemical degradation of PFOA in seawater. The researchers optimized the operating conditions and achieved a high efficiency of over 95% degradation. They proposed that the degradation mechanism involved the formation of reactive oxygen species (Lin et al., 2012). Magneli-phase- Ti_4O_7 has also been identified as a crucial anode material for PFAS degradation with EO-based processes. Because of their notable persistence, PFAS can be difficult to degrade using conventional procedures. Using Ti_4O_7 as an anode, electrochemical processing presents a viable strategy for converting PFAS into less hazardous chemicals (G. Wang et al., 2020a,b). A recent study by Lin et al. showed that the porous Ti_4O_7 ceramic anode has higher PFAS oxidation rates than other electrodes, including Ce-doped PbO_2 and Ti/BDD electrodes (Lin et al., 2018a). The porous Ti_4O_7 ceramic anode showed higher PFAS oxidation rates than other electrodes, including Ce-doped PbO_2 and Ti/BDD electrodes (Lin et al., 2018a).

The EO process time depends on many parameters such as current density, initial concentration, etc. nevertheless appropriate degradation is usually obtained on the time scale of hours. However, combination of

EO with other techniques allows optimizing the process efficiency in terms of cost and time. For instance, the use of a reactive EO membrane system has been shown to be effective in reducing the treatment time to seconds. Indeed, when operated at particular electrical potential, the Magneli phase Ti_4O_7 reactive electrochemical membrane (REM) allows for rapid oxidation, achieving about 5-log removal of this compound in a single pass through the REM with extremely brief residence times of about 11 s for PFOA and 3.6 s for PFOS (Le et al., 2019). Another challenge with the EO technique is its non-selectivity while treating real wastewater where different cations and anions could be found. The presence of chlorides in water leads to the formation of perchlorate (ClO_4^-) which is electrochemically favored to the detriment of PFAS degradation. Recently, Yang et al. (2019) have used a BDD anode to study its efficiency to treat PFOA and PFOS in a chloride containing electrolyte support. They found that introducing hydrogen peroxide can significantly inhibit the perchlorate formation. Based on their result, lack of injecting hydrogen peroxide resulted in 40% perchlorate formation converted form available chloride in the medium. At the same experimental conditions, adding hydrogen peroxide decreased perchlorate formation to only 5% during the process thanks to quenching of chlorine and hydroxyl radicals by hydrogen peroxide. However, the suggested mechanism confirmed that direct electron transfer oxidation is the dominant factor for oxidation of PFOA and PFOS.

The following Table 4 presents a summary of the efficiency of various treatment techniques for the degradation of PFOA and PFOS. The techniques considered in this table include electrochemical oxidation and photocatalytic mechanism.

From Tables 4 and it can be seen that degradation techniques using electrochemical oxidation and photo-catalytic oxidation using appropriate catalysts (ex. BN/TiO_2 , rGO/TiO_2 , $\text{In}_2\text{O}_3\text{-GRs}$) are the most effective processes to oxidize PFOA and PFOS. For instance, removal rates up to 99% (for PFOA) and 98% (for PFOS) can be recorded using electro-oxidation process (Lin et al., 2018b; Shi et al., 2019). By comparison, 93%–100% of PFOA was removed using photo-catalytic process (Duan et al., 2022; Gomez-Ruiz et al., 2018; Xu et al., 2019).

6. Analytical Considerations

The two standard methods approved by the U.S EPA for the quantification of PFAS in drinking water are the Methods #533 and #537.1 (Li, 2015). The EPA Method-533 is an isotope dilution/anion exchange/chromatography/tandem mass spectrometry (LC-MS/MS) technique. The EPA-Method-533 measures 25 specific PFAS compounds, including per fluorinated acids, sulfonates, fluorotelomer sulfonates, and poly/per fluorinated ether carboxylic acids, while the EPA-Method-537.1 measures 18 specific PFAS compounds, including perfluoro carboxylic acids up to 14 carbons in length. Due to their dependence on distinct techniques for sample concentration, the two methods necessitate distinct sample preparation procedures and cannot be merged into a solitary analysis. Considering the following example, in Canada, the EPA-Method-537.1 is generally approved for use in laboratories. Nevertheless, it is acknowledged that the EPA-method-533 would offer superior coverage of the PFAS identified in Canadian data mentioned in the exposure considerations section. To prevent duplication, drinking water authorities may require the use of a specific EPA method due to the overlap between the two methods, utilities should consider the potential sources of PFAS that may be present in the drinking water. Also, site characterization guidance is available from other sources. Total PFAS should include all PFAS listed in a method and detected in a sample, even if the value is below the detection limit, which is assigned a value of zero. Utilities should strive to analyze as many PFAS compounds as possible to better identify the PFAS present in the drinking water and select the appropriate treatment to reduce exposure to the greatest extent possible. If more than one method is used, duplicate analyses are not necessary, but if conducted the highest

Table 4
Degradation efficiency of different treatment techniques for PFAS in aqueous solutions.

Degradation processes	Matrix	PFAS	Operating conditions	Results and comments	References
Electrochemical Oxidation	Synthetic water	PFOS/PFOA mixture	C_i PFOA = 0.5 mM, C_i PFOS = 0.1 mM, $C_{Na_2SO_4}$ = 0.25 M T = 25 ± 1 °C, Anodes: Porous Ti_4O_7 Current density = 10 mA/cm ² , Treatment time = 3 h	PFOA degradation = 99.9% PFOS degradation = 9.1%	Lin et al. (2018b)
	Synthetic water	PFOS	C_i PFOS = 2.0 µM; $C_{Na_2SO_4}$ = 100 mM; T = 25 ± 1 °C Anodes: Magneli phase Ti_nO_{2n-1} ; pH = 6 Current density = 10 mA/cm ² , Treatment time = 2 h	PFOS degradation = 98.30 ± 0.51%	Shi et al. (2019)
	Groundwater	PFOS/PFOA mixture	C_i PFOA+ PFOS = 5 mg/L; $C_{Na_2SO_4}$ = 500 mg/L, pH = 7.4 Anodes: Mixed metal oxide (MMO) Current density = 10 mA/cm ² , Treatment time = 8 h	PFOS degradation = 90 % PFOA degradation = 90%	Schaefer et al. (2015)
	Aqueous solution	PFOA, PFOS, PFHxS	C_i PFAS = 1 mg/L, $C_{Na_2SO_4}$ = 10 mM, C_{NaCl} = 2.10^{-3} mol/L Current density = 1.8 mA/cm ² , Electrode material: Si/BDD, pH = 7, Treatment time = 5 h	PFOA degradation = 98% PFOS degradation = 99% PFHxS degradation = 95%	Sukeesan et al. (2021)
	Synthetic water	PFOA GenX PFBA 6:2 FTCA	C_i PFAS = 20 mg/L, $C_{Na_2SO_4}$ = 0.02 mg/L Anodes: Boron-doped diamond, Current density = 10 mA/cm ² , Treatment time = 120 min, pH = 7	PFOA degradation = 97.9% ± 0.1 GenX degradation = 65.6 % ± 5.0 PFBA degradation = 84.9% ± 3.3 6:2 FTCA degradation = 99.4% ± 0.0	Asadi Zeidabadi et al. (2024)
Photocatalytic process	Surface water	PFOS	C_i PFOS = 10764 µg/L, $C_{Na_2SO_4}$ = 20 mM, Anodes: pristine Ti_4O_7 , ($Ti_{1-x}Ce_x$) ₄ O ₇ (x = 1%), ($Ti_{1-x}Ce_x$) ₄ O ₇ (x = 3%), $C_{Na_2SO_4}$ = 10 M, Current density = 20 mA/cm ² , Treatment time = 180 min, pH = 7.2	PFOS degradation = 99%	Lin et al. (2021)
	Synthetic water	PFOA	Photocatalyst = BN/ TiO_2 , C_i PFOA = 100 µM; $C_{catalyst}$ = 0.5 g/L; pH = 3.2, treatment time = 4 h	PFOA degradation = 97.6%	Duan et al. (2022)
	Synthetic water	PFOA	Photocatalysts = rGO/ TiO_2 , C_i PFOA = 100 mg/L, $C_{catalyst}$ = 1.0 g/L, pH = 7, treatment time = 12 h	PFOA degradation = 93 ± 7%	Gomez-Ruiz et al. (2018)
	Synthetic water	PFOA	Photocatalysts = In_2O_3 -GRs, C_i PFOA = 30 mg/L $C_{catalyst}$ = 0.4 g/L, pH = neutral/alkalin, treatment time = 3 h	PFOA degradation = 100%	Xu et al. (2019)
	Synthetic water	PFOA	Photocatalysts = TiO_2 , C_i PFOA = 59 mg/L, $C_{catalyst}$ = 0.25 g/L, pH = 3, treatment time = 8 h	PFOA degradation = 98.8%	Xu et al. (2020)
	Synthetic water	PFOA	Photocatalysts = β -Ga ₂ O ₃ , C_i PFOA = 10 mg/L, $C_{catalyst}$ = 0.5 g/L, pH = 7, treatment time = 1.5 h	PFOA degradation = 100%	Zhao et al. (2015b)
	Synthetic water	PFOA	Photocatalysts = CeO ₂ / In_2O_3 , C_i PFOA = 100 mg/L, $C_{catalyst}$ = 0.4 g L ⁻¹ , pH = 2.84, treatment time = 1 h	PFOA degradation = 100%	Jiang et al. (2016)

individual PFAS result should be used for summation which should not exceed the proposed objective value of 30 ng/L (Pfas, 2020). The two above-mentioned methods are widely used and accepted by regulatory/governmental agencies to monitor and regulate PFAS levels in drinking water. The methods produce accurate and reliable results, which are critical for protecting public health and ensuring regulatory compliance.

7. Economic aspect

The treatment of PFAS is a complex and costly process, and the remediation of PFAS-contaminated sites has significant economic implications. The cost of PFAS treatment varies according to the extent and severity of the contamination, as well as the specific treatment technologies and strategies employed. Furthermore, the economic impacts of PFAS contamination are not limited to the cost of treatment, but also include broader costs to public health, environmental damage, and economic activity. A study published in 2021 estimated the economic costs of PFAS contamination in Europe, and found that the total costs could exceed €52–84 billions (The cost of inaction, n.d.). The study identified a range of economic impacts, including lost productivity, healthcare costs, and damage to the tourism and fishing industries.

In the United States, the economic impacts of PFAS contamination have also been significant. A study by the Environmental Working Group estimated that the costs of PFAS contamination in the United States could reach \$2 trillion, taking into account the costs of the cleanup, lost

economic activity, and health effects (Water et al., 2020). These economic impacts underscore the need for proactive measures to prevent PFAS contamination, as well as more efficient and cost-effective approaches to remediation. Some industries and governments are taking steps to address PFAS contamination, such as banning or phasing out the use of certain PFAS chemicals, implementing more stringent regulations, and investing in research and development of new treatment technologies. For example, in the United States, the EPA has set a health advisory level for PFAS in drinking water and has established regulatory limits for some PFAS chemicals in water systems (FACT SHEET PFOA, 2016). The state of Michigan has also taken proactive measures to address PFAS contamination, including establishing a PFAS Action Response Team and implementing a comprehensive strategy to manage and prevent PFAS contamination (Hendershott, 2022).

Overall, the economic implications of PFAS treatment are significant, and there is an urgent need for continued investment in research and development of new treatment technologies and strategies. While there are promising innovations being developed, there is also a need for greater collaboration between researchers, industry, and government to identify and implement cost-effective solutions for PFAS remediation.

8. Conclusion and future trends

In a comprehensive review, this work discussed various aspects of PFAS compounds ranging from their uses, sources, classification, to their environmental and health impacts alongside regulatory measures. Due

to Carbon–Fluorine (C–F) bond in the structure of PFAS compounds, they have a high chemical and thermal stability, and hence they leave significant environmental footprint, primarily through water and soil contamination, causing adverse ecotoxicological and toxicological effects to both wildlife and humans. Management strategies for PFAS, as detailed in the review, span from separation techniques like activated carbon, ion exchange resins, electrocoagulation and adsorption on polymers, to advanced degradation methods including photocatalytic mechanisms, incineration, biodegradation, sonolysis, and electrochemical oxidation. These methodologies underscore the multi-faceted approach required to tackle the pervasive challenge of PFAS pollution. Future R&D effort should be devoted to the development of efficient PFAS degradation techniques at the source. The recommended method for future research incorporates two key steps to remediate water contaminated with PFAS. In the first stage, the PFAS pollutants should be concentrated in a smaller volume by using either ion exchange resin (IER) or granular activated carbon (GAC). In the second step, the smaller volume should be subjected to an oxidation process (using for instance advanced electro-catalytic process) for PFAS degradation into H₂O and CO₂.

CRedit authorship contribution statement

Hela Rekik: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Data curation. **Hamed Arab:** Writing – review & editing, Writing – original draft, Validation, Software, Investigation, Formal analysis. **Loick Pichon:** Writing – review & editing, Resources, Methodology, Investigation. **My Ali El Khakani:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition. **Patrick Drogui:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Funding acquisition.

Declaration of Competing interest

The authors declare the following financial interests/personal relationships, which may be considered as potential competing interests: Pôle Recherche Innovation Matériaux Avancés (PRIMA) Research and innovation project in advanced materials.

Data availability

We have shared the link to our data

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