

Review

Advanced Nanoformulations for Detection and Removal of Poly- and Perfluoroalkyl Substances (PFAS)

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Abstract: Perfluoroalkyl substances (PFAS), also known as “forever chemicals”, are a class of highly stable chemical compounds that slowly contaminate waterbodies and soil. The widespread presence of PFAS is associated with adverse human health effects and is a major environmental concern. The conventional, highly sensitive methods used for PFAS detection are LC-MS/MS and solid phase extraction, but they are very complex and expensive. Therefore, there is an urgent need for sensitive, low-cost, and fast methods for the detection and removal of PFAS compounds from water and soil resources. The advancement of nanotechnology has significantly impacted advanced disease diagnosis and treatment in the last few decades. Currently, these engineered nanomaterials (ENMs) have been exploited for the development of advanced nano-enabled techniques for the detection and removal of environmental pollutants. Nano-enabled techniques also offer improved performance over conventional methods. In this review, the details of the detection and removal of PFAS, as well as their optimization and limitations, and future perspectives are discussed. We focused on the implementation of nanomaterials such as nanoparticles, nanotubes, nanorods, and nano*filtration membranes for efficient PFAS detection and removal. We also included the recent literature and global guidelines for PFAS use and the effect of PFAS exposure on human health.

Keywords: PFAS detection and removal; nanoformulations; environmental pollution; nanotechnology; water treatment; disease from PFAS



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1. Introduction

Environmental contaminants have attracted global attention in the last two decades due to their high bioaccumulation, toxicity, and complex health problems. Among them, poly- and perfluoroalkyl substances (PFAS) are a group of persistent synthetic chemicals causing serious health problems [1,2]. PFAS are synthetic, organic, fluorinated chemicals that are both hydrophobic and oleophobic [3]. PFAS such as PFOA and PFOS prominently bioaccumulate in plant and animal tissues and move up the food chain. Due to their exceptional persistence in the water bodies, leading to their gradual accumulation in seafood and their consumption, they serve as source of primary exposure to humans [4]. They are also known as “forever chemicals” due to their extensive use across several sectors, serving as surface coatings in commercial items such as non-stick cookware, water-repellent clothing, paper, carpets, food packaging, photolithography, polymers, metal plating, semiconductors, aircraft, and construction [5]. They are a vital element of the aqueous film-forming foam (AFFF) used at airports and in military installations. However, their interaction with humans, as well as terrestrial and aquatic animals, is very hazardous,

resulting in acute to chronic diseases. The Agency for Toxic Substances and Disease Registry, Center for Disease Control and Prevention (CDC), reported that research studies suggest a positive correlation between PFAS and a broad spectrum of negative effects on human health [6].

The PFAS compounds are a subset of polyfluorinated compounds with a minimum of one perfluoroalkyl moiety ($-C_nF_{2n+1}-$) with one or more carbon (C) atoms that contain all hydrogen (H) being replaced by fluorine atoms (F) [7]. The physicochemical properties of PFAS are determined by the strong C–F bond, characterized by high thermal and chemical stability. Fluorine has high electronegativity, low polarizability, and a high ionization potential, and when bonded with carbon forms one of the strongest bonds observed in organic compounds, with $531.5 \text{ KJ mol}^{-1}$ of bond dissociation energy in PFAS [8]. PFAS exhibits extreme chemical and thermal stability, leading to persistent environmental degradation and the subsequent accumulation of PFAS in ecosystems [9]. PFAS compounds are included as polyfluorinated compounds in the Economic Co-operation and Development (OECD) 2018 report, with around 4700 unique substances recognized as PFAS, the most extensively studied chemical being perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid and (PFOS), and their toxicity has been well documented [7,9]. The other PFAS include their salts and related substances such as perfluorooctane sulfonyl fluoride (PFOSF) and PFHx, and the toxic effects of these derivatives have yet to be explored. PFAS were introduced by a US-based company “3M” in the early 1940s and 1950s; however, PFAS received global attention in the late 2000s, when they were first detected in arctic animals. In the United States, PFAS were detected in blood samples collected during a 1999–2000 national sampling. Surprisingly, PFAS compounds were identified in almost 98 percent of collected blood serum samples obtained from the general U.S. population, indicating extensive chemical exposure [10,11].

Over 4000 commercially manufactured PFAS chemicals have been documented, making their identification and selective detection a significant issue [12]. PFAS metabolites are progressively leaked into the environment during their manufacturing, use, and disposal, necessitating prompt action to mitigate their dissemination across trophic levels [13]. PFAS detection presents several challenges due to the absence of chromophores in their structure, complicating early colorimetric identification [9,11]. They exhibit significant chemical resistance to derivatization reactions, constraining the capacity to conduct most analytical evaluations typically performed on persistent organic pollutants (POPs). In the natural environment, PFAS can penetrate in various ecosystems, including water, air, and the soil and surface, and accumulate in trophic levels in the food chain [8,13]. In an aqueous environment, PFAS exist as surfactants, possessing remarkable electrostatic characteristics such as a critical micelle concentration, exceptionally low surface tension, a lack of Van der Waal forces, and excellent surface wettability, affecting their transport in aquatic environments [14]. In situ and real-time analyses of environmental samples are essential for accurately assessing the degree of contamination in significant PFAS sinks.

PFAS have been detected in food, soil, and drinking water, and evidence shows that the continued bioaccumulation of certain PFAS over time leads to harmful health effects even at low levels [15,16]. Widespread PFAS contamination is a potential threat to human health and several research studies link PFAS exposure to possible multiple human diseases, including cancer, neurodegeneration, and metabolic and immune-related disorders [9]. The pollution of drinking water presents a major obstacle because of the multitude of PFAS, the absence of integrated source data, constraints in treatment options, and the scarcity of research on some PFAS compounds. The development of new methods for their detection and removal might require research studies on renovating technologies and finding effective water treatment methods [9]. Current techniques typically focus on a

specific group of PFAS for which adequate studies are available. Strategies like risk-based geospatial studies and extensive research, and working on new management initiatives centered on total PFAS content, will accelerate the reduction in PFAS exposure when data are insufficient [6]. Figure 1 represents the global prevalence of the most common PFAS, such as PFOA and PFOS, in drinking water, groundwater, and surface water.

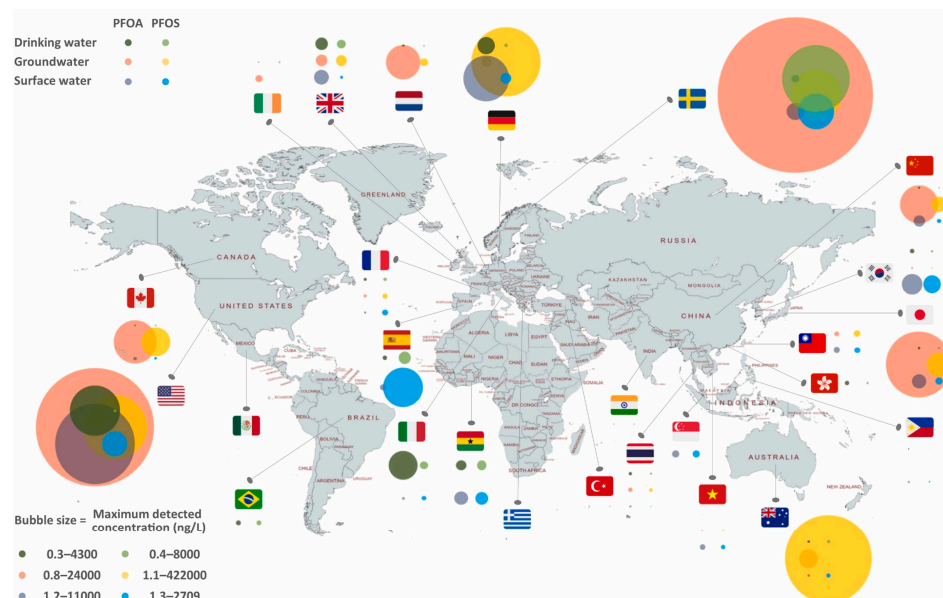


Figure 1. Global concentration of PFOA and PFOS, reprinted with permission from Reference [17].

The US Agency for Toxic Substances and Disease Registry (ATSDR) was established in 1980 for the implementation of laws related to health for the protection of the public from hazardous chemical and wastes. ATSDR flagged four PFAS for limited use, including PFOA and PFOS, corresponding to drinking water concentrations of 11 parts per trillion (ppt) and 7 ppt, respectively [18]. Moreover, the currently advised safe daily dosage limit for these substances, as established by the US Environmental Protection Agency (EPA), is 70 ppt, whether singly or in combination. Water treatment typically involves physical removal via nanofiltration (NF), reverse osmosis (RO), granular activated charcoal (GAC) treatment, ion-exchange resins, and high-pressure membrane systems [16]. The wastewater is treated using both physical and biological methods, but these treatment methods are not efficient in the removal of PFAS [18]. Recent advancements in wastewater treatment methods include nanofiltration, membrane separation, advanced oxidation/reduction processes (AOPs/ARPs), and membrane bioreactors. Among them, nanomembrane filtration is an effective method can eliminate PFAS with molecular weights of up to 300 g mol^{-1} . AOPs is a cutting-edge technology for PFAS removal; this method combines hydrogen peroxide with ozone/UV light and generates free hydroxyl radicals. Hydroxyl radicals further break down PFAS into less harmful compounds. Reverse osmosis may reduce PFAS content to 0.4 ng L^{-1} ; hence, RO is preferred over activated carbon filters. Both GAC and AOPs showed acceptable effectiveness for PFOA and PFOS removal at the microgram-per-liter concentration. However, due to the non-destructive nature of ion exchange, RO, nanofiltration, AOPs and GAC processes require a concentrated PFAS sample for effective treatment and removal, which differs from the presence of PFAS in aquatic ecosystems [9,18].

In the last few decades, PFOA was extensively used and, due to its significant durability, is identified globally in almost all life forms, with average concentrations reaching hundreds of ng L^{-1} . This is particularly elevated in coastal regions and riverbanks near

industrial urban centers [19]. A research study by Li et al. revealed that the average concentration of PFOA in coastal saltwater in Laizhou Bay reached 475 ng L^{-1} , endangering the health of coastal residents. The U.S. Environmental Protection Agency (EPA) advised a PFOA concentration of only 4 ng L^{-1} ($9.658 \times 10^{-12} \text{ M}$) in drinking water, while the Swedish National Food Administration established a higher action threshold of 90 ng L^{-1} ($2.17 \times 10^{-10} \text{ M}$) [20]. Additionally, the European Drinking Water Directive set the limit at 100 ng L^{-1} ($2.41 \times 10^{-10} \text{ M}$), reflecting a greater emphasis on safeguarding public health [20]. However, no states or tribes in the United States have approved PFOA water-quality guidelines for the protection of aquatic life. Several states have released draft or interim ecological screening-level values and benchmarks for the preservation of aquatic life, including both acute and chronic effects. Publicly accessible freshwater acute levels vary from $4.47 \times 10^6 \text{ ng L}^{-1}$ in Texas to $2 \times 10^6 \text{ ng L}^{-1}$ in Florida. The EPA's acute PFOA criteria for freshwater, $3 \times 10^{-6} \text{ ng L}^{-1}$, is marginally lower than the state-derived standards. No acute estuarine or marine criteria, standards, or protective values have been defined for PFOA other than the stated benchmark. Publicly accessible freshwater chronic readings vary from $2.2 \times 10^5 \text{ ng L}^{-1}$ in Australia/New Zealand to $2.27 \times 10^6 \text{ ng L}^{-1}$ in Texas, which are all above the EPA's freshwater chronic criteria of 10^5 ng L^{-1} . Chronic data for estuarine and marine environments are accessible for Australia/New Zealand and California [21]. Furthermore, studies indicate that the levels of poly- and perfluoroalkyl substances (PFAS) in seawater diminish as water salinity rises; hence, enhancing methodologies for PFOA risk assessment continues to be challenging when considering aquatic ecosystems [19]. A table summarizing average concentration and regulatory limits is presented in Table 1.

Table 1. Summary of PFOA concentrations and regulatory guidelines.

Country/State	Concentration	Regulatory Guidelines
Global Average Concentration	100 ng L^{-1}	Elevated levels, particularly in coastal regions and riverbanks near industrial urban centers [19].
Coastal Saltwater (Laizhou Bay, China)	475 ng L^{-1}	Reported concentrations pose potential health risks to coastal populations [19].
U.S. EPA Drinking Water Limit	4 ng L^{-1} ($9.658 \times 10^{-12} \text{ M}$)	Strict threshold recommended to ensure potable water safety [20].
Swedish National Food Administration Limit	90 ng L^{-1} ($2.17 \times 10^{-10} \text{ M}$)	Higher action level relative to U.S. EPA standards [20].
European Drinking Water Directive Limit	100 ng L^{-1} ($2.41 \times 10^{-10} \text{ M}$)	Emphasizes public health protection through stringent regulatory measures [20].
Freshwater Acute Benchmarks (U.S. States)	4.47 mg L^{-1} (Texas)– 20 mg L^{-1} (Florida)	State-derived acute benchmarks for short-term exposure's impacts on aquatic life [21].
U.S. EPA Acute Freshwater Criteria	3.1 mg L^{-1}	Slightly lower threshold than state-specific benchmarks for acute toxicity [21].
Freshwater Chronic Benchmarks (U.S. States)	0.22 mg L^{-1} (Australia/New Zealand)– 2.27 mg L^{-1} (Texas)	State-specific guidelines addressing the risks of long-term exposure to aquatic ecosystems [21].
U.S. EPA Chronic Freshwater Criteria	0.10 mg L^{-1}	More conservative approach to chronic toxicity compared to state standards [21].

The detection and removal of PFAS contamination from the environment is very difficult. The primary objective of this review is to assist researchers by rigorously evaluating the current methodologies for PFAS removal from wastewater, focusing on finding procedures that demonstrate potential in laboratory studies for the development of realistic,

industrial-scale, eco-friendly solutions. The negative health consequences of these chemicals will become evident in the coming decades. PFAS are omnipresent in the environment, including in soil, water supplies, food items, animals, people, sediments, textiles, sludge, and sediments [12]. Although PFAS are present everywhere, they are present in substantial quantities next to their production facilities and related sectors, and new PFAS are constantly being introduced to the market as substitutes for traditional PFAS. Emerging PFAS are often difficult to identify using conventional chemical methods, raising concerns about their detection and elimination [22]. At a higher concentration, chronic exposure to PFAS is associated with brain, liver, kidney, and testicular cancer, and presents a significant risk to both human health and the marine ecosystem [19]. Consequently, there is an urgent need for techniques to facilitate the real-time monitoring of PFOA to guarantee environmental safety. Therefore, identifying the physicochemical properties of PFAS is crucial for the development of, and research on, novel methods for their effective removal from the natural environment. Global PFAS research mostly studies the concentration and adverse health consequences of PFOA and PFOS, since these compounds are most commonly found in water, soil, and sediments [23].

Traditional methods such as LC-MS/MS, ion chromatography, and gas chromatography require additional sample concentration steps, including solid phase extraction adsorption using polystyrene–divinylbenzene and modified QuEChERS [24]. However, the emerging nanotechnology-empowered methods are not only single-step but also have excellent regeneration capabilities [25]. In terms of cost-effectiveness, traditional methods require trained laboratory professionals and cost around USD 300–600 per sample, whereas an MIP-based sensor can detect PFAS in river water for USD 50 [26,27].

The nanomaterial range of 1–100 nm in dimension increases the overall surface area, improves the surface-to-volume ratio, allows for more accurate control over material properties such as size, shape, surface area, and surface charge distribution, facilitates shorter response times, accelerates synthesis, minimizes reagent consumption, and enables the simultaneous detection of multiple target PFAS. Hence, the nano-enabled sensing of PFAS is currently being explored for water remediation [28]. Nanomaterials are highly tunable, selective, and targeted towards specific pollutants, making them an excellent choice for water treatment. Nanomaterials adsorb PFAS via their unique electrostatic and hydrophobic interaction with PFAS. Deeper insights into the chemical interaction between PFAS and engineered nanomaterials are currently being investigated using nanoelectrochemistry methods like single-particle-on-microelectrode collisions; these studies are supported by the spectroscopic results and theoretical studies. This new approach will help researchers to develop advanced, ultrasensitive methods for PFAS detection.

Additionally, nanomaterials can be reused via recycling during the treatment process and provide a sustainable solution for PFAS removal. Emerging methodologies using nanotechnology are gaining popularity due to their significant characteristics at this scale. This review provides a broad overview of the advanced methodologies for the detection, removal, and treatment of PFAS in polluted water.

2. PFAS Exposure and Human Health: Investigating the Link to Cancer, Immune Disorders, and Neurodegeneration

According to the EPA, PFAS have already been detected in the bloodstream of most people in the US, due to their extensive use and global presence. However, PFAS toxicity and related diseases depend on many parameters, such as the concentration, toxicity, exposure duration, and frequency. PFAS have been identified as exhibiting toxicity primarily via the proliferation of peroxisomes [22]. Research on animals, such as rats, has been conducted to gain insight into the toxicological consequences of PFAS. However, peroxisomal prolifer-

ation varies across different species and organisms, so it is not suitable to generalize the findings of these studies to humans [29]. Numerous case studies regarding the detrimental consequences of PFAS exposure in humans have shown a potential correlation between exposure to PFAS chemicals and the onset of diseases such as thyrotoxicosis, carcinogenesis, ulcerative colitis, testicular cancer, hypertension, increased cholesterol level, and decreased response to vaccines [16,21]. Researchers have shown that PFAS exposure affecting growth, learning, and behavior is more likely in infants and children than adults. This may be attributed to the elevated metabolism associated with the constantly evolving physiological systems of youngsters [22]. A recent analysis provided epidemiological evidence linking PFAS exposure in prenatal and/or early life stages with immune dysfunction, gastrointestinal and respiratory tract infections, cardiometabolic disorders, neurodevelopmental changes, thyroid dysfunction, renal issues, and the onset of puberty [22]. The primary sources of PFAS exposure are contaminated food and drinking water sources. Secondary sources are the consumption of vegetables and fruits contaminated with PFAS; other major sources are fish and crustaceans with high PFAS concentrations. For example, in China, the consumption of fish and crustaceans has been positively correlated with PFAS concentration in the plasma of a cohort of women [8,11].

2.1. PFAS Induced Carcinogenesis

The carcinogenic consequences of PFAS have mostly been investigated in humans. The studies primarily examined populations consuming polluted water and chemical industrial employees exposed to PFOA and PFOS, which are linked to frequently reported cases of kidney cancer and mortality [22]. The International Agency for Research on Cancer (IARC) declared PFOA a potential human carcinogen based on sufficient evidence of its carcinogenic effects on lab animals. Over the last decade, male workers exposed to PFOA in manufacturing facilities developed prostate cancer within a few years, and mortality increased by 3.3 times [30]. The correlation between carcinogenesis and PFOA exposure was clearly shown in two studies conducted in West Virginia and the Mid-Ohio Valley where the drinking water supply contained PFOA contamination due to its proximity to a chemical facility [31]. Another study examined medical history and PFOA blood levels in a group of adult participants from the C8 Health Project. The study revealed a significant correlation between serum PFOA concentrations and testicular and kidney cancer [31]. Similar results were reported in a case study including the general Danish population, which was conducted to assess cancer risk associated with PFOA and PFOS exposure [32]. Despite research on the correlation between PFAS exposure and cancer, robust evidence and extensive research involving a large population are needed to obtain a conclusion and improve the understanding of the toxicological outcomes and carcinogenic effects of PFOA and PFOS exposure [32].

Thyroid cancer incidence has been increasing over the past few decades, with over-diagnosis occurring partly due to improved diagnostic tools. However, there is growing evidence to support a true rise in thyroid cancer incidence. Endocrine-disrupting chemicals (EDCs), including PFAS, have been identified as a potential risk factor. A nested case-control study by Gerwan et al. (2023) found a 56% increased risk of thyroid cancer diagnosis per doubling of linear perfluorooctanesulfonic acid (n-PFOS) intensity [28]. The study confirmed the association between PFAS exposure and thyroid cancer using both a cross-sectional and longitudinal study design, enabling a unique investigation of exposure timing's influence on thyroid cancer. The study supports the hypothesis that PFAS exposure may be associated with an increased risk of thyroid cancer and underlines the need to remove/reduce PFAS from the potential exposure routes [28].

2.2. Metabolic and Immune Effects

The experimental studies show that PFOA and PFOS can alter immune responses, including inflammation and cytokine production [33]. In adult mice, exposure to PFOA and PFOS suppresses T-cell-dependent immunoglobulin M (IgM) production, a sensitive measure of immune function [33]. These findings support the immunotoxic effects of PFAS exposure. Studies on exposure to PFAS and their metabolic effects have been conducted across several age groups, such as children, adolescents, and adults, including pregnant women and diabetes patients [34]. Gilliland and Mandel conducted one of the first investigations examining the correlation between PFOF concentration in the blood serum of exposed workers and cholesterol, hepatic enzymes, and lipoprotein levels as a marker of liver function. The researchers measured exposure using total serum fluorine but found no substantial liver damage linked to the projected PFOA levels [35]. Another study by Liu et al. [36] assessed the impact of PFOA/PFOS in their linear and branched isomers on serum biochemical profiles. They used data from the National Health and Nutrition Examination Survey (NHANES) [36]. However, an absolute correlation was found between PFOA isomers and fasting glucose, high-density lipoprotein cholesterol (HDL-C), total cholesterol, and blood serum albumin. Research studies of the effects of PFOA and PFOS on immune dysfunction in animal models revealed that these chemicals induce variations in inflammatory responses and cytokine levels, therefore affecting both innate and acquired immunity [37]. Although data on PFAS immunotoxicity are limited, increasing evidence indicates that immunotoxicity in animals is correlated with PFAS exposure [37].

A very recent study published by Wang et al. (2023) suggested that PFOS altered liver metabolism and fatty acid metabolism in laboratory mice [38]. The study found that PFOS exposure led to hepatic steatosis, increased triglyceride levels, and reduced the expression of hepatokines like fibroblast growth factor-21 (Fgf-21), retinol-binding protein-4 (Rbp-4), and leukocyte cell-derived chemotaxin-2 (Lect-2) levels, which are involved in spermatogenesis and testosterone synthesis [38]. Testicular ATP and testosterone levels decreased, while peroxisome proliferator-activated receptor- α expression increased. PFOS was found in the testes and increased fatty acid metabolites, which are linked to inflammation and infertility [38].

2.3. Neurodegenerative Disorders

PFAS can cross and damage the blood–brain barrier, possibly facilitating the enhanced infiltration of other substances into the brain [39]. PFAS may traverse the placenta during gestation, resulting in accumulation inside the embryonic brain, and its exposure also disrupts several neurotransmitter systems, including the hypothalamus and hippocampus, which play a major role in the secretion of neurotransmitter dopamine and glutamate. PFAS interferes with the transmission of the neurotransmitters in these parts of the brain, a significant factor in neurotoxicity with negative neurological consequences [39]. The mechanisms that alter these neurotransmitters include mitochondrial dysfunction, altered metabolism, imbalanced calcium signaling, and altered protein expression and function [39,40]. The negative outcomes of these neurochemical alterations in people remain undetermined; however, a possible association between neurodegeneration and mental health issues was reported. A study involving adolescents and adults aged 50–65 in a contaminated area in the Netherlands found that exposure to PFOS and PFOA was linked to increased somnolence, inverse associations between boys' femininity and girls' masculinity, and a marginal decrease in sustained attention and cognitive performance [41]. However, in older adults, PFOS was associated with an increase in attention capacity and PFHxS, along with a significant increase in sustained attention. The study suggests that PFAS exposure can have neurological effects on individuals [41]. Consequently, the increasing data indi-

cate that chronic PFAS exposure leads to neurotoxicity are a significant concern regarding neuronal health, and the experimental data suggest that immediate actions should be taken to establish regulations against the commercial use of PFAS [39].

Nicolas Delcourt et al. (2024) reported the first evidence of the direct correlation between PFAS accumulation in the central nervous system and clinical and biological symptoms of Alzheimer's disease [42]. PFAS are manmade compounds that are extensively used in commercial and household applications, which are recognized for their detrimental effects on human health. Due to their bioaccumulation in the human brain and established neurotoxicity in animal models, they are presumed to contribute to neurodegenerative processes [42]. In this study, they evaluated the concentration of 18 PFAS in the cerebrospinal fluid (CSF) from eight individuals with suspected normal-pressure hydrocephalus [42]. Nicolas and his research group found a significant correlation between PFAS levels and both clinical and biological biomarkers of Alzheimer's disease pathology. They also demonstrated that PFOS and PFOA were present in all CSF samples obtained around a French area devoid of fluorochemical companies [42]. Furthermore, the study found significant differences in PFOS and PFOA concentrations in the cerebrospinal fluid of individuals with Alzheimer's disease signs and cognitive impairment compared to those with either condition [42]. Two prior investigations have shown that due to the compromised BBB integrity, PFAS concentrations in human CSF rise with age and could be a possible reason for the development of neurodegeneration in old age [42].

A very recent study by Karla M. Ríos-Bonilla et al. (2024), studying the neurotoxic effects of PFAS on the human blood, suggested that 12 PFAS compounds have adverse neurodevelopmental outcomes [43]. The experiments revealed no impairment of mitochondrial membrane potential or the activation of oxidative stress response at sub-cytotoxic doses. All combination components and formulated mixes reduced neurite outgrowth in differentiated neuronal cells at doses around or below cytotoxic levels [43].

2.4. Pre- and Postnatal PFAS Exposure

The risk from environmental pollutants can differ among species and is also determined by the duration of exposure. The negative consequences of these compounds not only affect those who are directly exposed but also, and particularly, affect future generations, who will experience the effects of toxic damage [44]. Mammalian gestation, including that of humans, is subjected daily to a multitude of environmental contaminants via ingestion, inhalation, or skin absorption. The distribution and elimination rates of these chemicals, which dictate their movement throughout the maternal body, are affected by the physicochemical characteristics of the substances, along with specific maternal and fetal conditions [44]. During pregnancy, some complex environmental chemicals may cross the placental barriers and penetrate fetal blood circulation, and early fetal exposure to foreign chemicals could be extremely challenging for their immature immune system [45].

A similar study published in the Journal of the National Cancer Institute [46] from Finland reported a strong correlation between PFAS exposure in pregnant women and acute lymphoblastic leukemia (ALL) in newborns [46]. The same observation was reported regarding the correlation with PFOS for pregnancies occurring from 1986 to 1995, the timeframe preceding their phase-out, during which concentrations were comparatively elevated relative to subsequent years. Although PFAS exposure is widespread, Finnish women had somewhat lower quantities than those seen in the general population in the U.S. and certain regions of Europe, potentially affecting childhood leukemia risk in these demographics [46].

In another research study by Jiwon oh et al. [47], childhood exposure to PFOA was linked to elevated risks of autism spectrum disorder (ASD) (odds ratio [OR] per ln ng/mL

increase: 1.99, 95% confidence interval [CI]: 1.20, 3.29) and developmental delay (DD) (OR: 2.16, 95% CI: 1.21, 3.84) compared to typical development (TD). Also, perfluoroheptanoic acid (PFHpA) was linked to elevated risks of autism spectrum disorder (ASD) (OR: 1.61, 95% CI: 1.21, 2.13). The child's sex and homeownership influenced the relationships between PFDA and developmental disorders (DD) and autism spectrum disorder (ASD), respectively [47].

A study of 1503 mother–child pairs found that 26% of children were hospitalized for infectious diseases. A doubling in maternal PFOS concentration was linked to a 23% increase in infection-related hospitalization [48]. Child sex was found to interact with PFOS and PFDA, but in opposite directions. Every doubling of PFOA or PFOS increased the risk of LRTI by 27% and 54%, respectively. Similar trends were observed for URTI and other infections. For GIs, the opposite pattern was observed, with HRs consistently below 1 [48].

3. Nano-Enabled Techniques for the Detection of PFAS

PFAS concentrations in the environment range from a few pico- to micrograms per liter, meaning that very sensitive detection methods are required for their accurate identification [14]. A great dynamic range is frequently as crucial as low limits of detection to ensure a sensing device is effective [14]. Since PFAS is observed in cationic, anionic, and zwitterionic forms, its charge and polarity fluctuate, complicating identification with a single analytical approach [12].

As of now, conventional methods such as liquid chromatography–mass spectrometry (LC-MS), liquid chromatography–tandem mass spectrometry (LC-MS-MS), and gas chromatography–mass spectrometry (GC-MS), as well as semi-quantitative methods such as total oxidizable precursors (TOP) and total organic content (TOC), are the leading approaches for the detection of PFOA [49]. However, these techniques often need complicated sample preparation procedures, expensive labor resources, and a substantial amount of equipment. Due to the critical significance of PFOA monitoring, there is an urgent need for a cost-effective, sensitive, reliable, field-deployable, and portable sensors for in situ PFOA detection. A significant barrier to the quantification of pollutant dispersion is the absence of a highly sensitive and selective monitoring platform, limiting the assessment of PFAS exposure and risk [49]. The implementation of newly developed methods at a large scale might solve this problem. Nanotechnology is an emerging field with promising environmental pollutant detection and removal approaches.

3.1. Nanotechnology-Based Sensors for PFAS Detection

According to the EPA, the use of nanotechnology to develop highly selective, cost-efficient, high-efficiency sensors might play a pivotal role in advancing PFAS detection [16]. Nanomaterials-based sensors are one of the major advancements in pollutant detection; sensors may generally be categorized according to their recognition probe, transducer type, and output signals. Nano-enabled PFAS sensors can be categorized as optical, electrochemical, fluorescent, spectrophotometric, and colorimetric [14]. A comparison of these nanosensors and an evaluation of their efficiency, selectivity, limit of detection, recovery, and biodegradability are the essential parameters used to assess their ability to detect PFAS at very low concentrations [14].

3.1.1. Photo-Electrochemical Sensors

Photo-electrochemical (PEC) sensors can provide an improved sensing performance and photoelectric efficiency due to their unique chemical and physical properties [50]. The working principle behind PEC sensors is that a photoelectric electrode is excited by a light source to produce a detectable current [51]. PEC sensors can be used in both non-

electroactive and electroactive species. For example, a photoactive electrode made up of silver iodide (AgI)-Bismuth oxyiodide (BiOI)-based nanocomposites with molecularly imprinted polymers (MIPs) were used for PFOA detection [52]. The development of these nanohybrids of nanomaterials and mixed metal oxides provides enhanced surface area and reactivity. A “signal-off” sensor was created when PFOA was added to the solution because it prevented electron transmission over the MIP layer, which, in turn, reduced the photocurrent. The term “signal-off” denotes the end of the electrical signals. The PEC sensing strip’s photocurrent transmission stopped as a consequence of this reaction, indicating the presence of the target analyte (PFOA) [52]. A recent study from ThanhThuy Tran. et al. reported the detection of a PFOS derivative, perfluorooctane sulfonyl fluoride (PFOSF/C8F18O2S), using specific binding with poly(acrylamide) [53]. The technique uses screen-printed electrodes enhanced with Bismuth oxyiodide (BiOI)-nanostructured arrays. It has a linear range and an LOD of 0.5–10 μM 86 ng mL^{-1} , making it very sensitive when used for PFOS detection in water samples [53].

3.1.2. Electrochemical Sensors

Electrochemical sensors are designed to convert the outcomes of electrochemical reactions into electrical signal detection for qualitative and quantitative signals [54]. Electrochemical sensing is mainly used for the selective identification of analytes within a mixture of compounds. It can detect very low concentrations of analyte up to ppb levels and is a very efficient category of PFAS detection methods [53]. Electrochemical sensors consist of three types of electrodes: a reference electrode (an electrode with a stable and known electrode potential), an anode/cathode, used as a working electrode, and a counter-electrode [55]. A change in electric current can be measured across the electrode upon the binding of the analyte molecule to the receptor, which is analyzed using the calibration curves to calculate the concentration of the detected analyte. However, there are numerous challenges with electrochemical PFAS detection. To start with, the majority of PFAS compounds have a decreasing potential window that is beyond the water potential window and varies between 2 and 3 V, as opposed to a standard hydrogen electrode (SHE) [18]. Consequently, it is not likely that PFAS can be directly identified using standard voltammetric or amperometric techniques. Second, it is usually necessary to identify PFAS indirectly by examining the products of digestion or how they affect electrochemical processes (like MIPs). However, changes in temperature, pH, electrolytes, or the presence of certain pollutants might lead to kinetic issues with the enzymatic process. Thirdly, additional materials—especially fatty acids—are frequently present in wastewater and remediation sites, interfering with EC sensors and lowering sensitivity by clogging electrode surfaces [15,39].

A recent study from Rebecca B. Clark et al. [56] uses the oxygen dissolved in river water for the quantitative measurement of PFOS. They used electrochemical impedance spectroscopy (EIS) and differential pulse voltammetry for the qualitative measurement of PFOS. The results indicate that the EIC exhibits an improved limit of detection (LOD) of 3.4 pM for PFOS compared to voltammetric techniques and other existing electrochemical methods, which can only detect analyte concentrations down to the sub-10 nM range [56].

A similar study from a research group developed an electrochemical sensor for the direct detection of PFOA using a selective perfluorinated anion-exchange ionomer (PFAEI) [57]. The sensor functions independently of redox probes with an LOD of about 6.51 ± 0.2 ppb (15 nM) in drinking water and buffered deionized water. When the sensor was tested using various ionomer electrode coatings, PFAEI interacted preferentially with PFOA anions compared to other, similar ions in the solution, due to a combination of Vander Waals interactions and electrostatic forces [57].

3.1.3. Electro-Chemifluorescence Sensors

Electro-chemiluminescence (ECL) is the process of light emission from the electrode during high-energy electron transfer [58]. ECL arises from the electroactive species near the electrode, which experience rapid electron transfer upon interaction with the target analyte, resulting in the formation of highly excited states. ECL systems consist of three major components: first, luminophores or emitters (to transduce electronic signals into optical signals), second, co-reactants (to facilitate the reaction), and third, electrodes (to drive the chemical reaction) [59]. For example, the use of poly(pyrrole) MIPs with ultra-thin graphene carbon nitride (utg-C₃N₄) as an electrochemiluminescence (ECL) emitter achieved an LOD of 0.01 µg/L (1000 times less sensitive than LC-MS/MS at 0.0001 µg/L) for the detection of perfluorooctanoic acid (PFOA). utg-C₃N₄ was identified as an exceptional electro-chemiluminescent material owing to the existence of surface-active regions facilitating particular interactions [60].

Zheng et al. (2019) developed guanidinocalix[5]arene, which can selectively and strongly bind to these substances [61]. This allows for the sensitive and quantitative detection of these compounds in contaminated water through a fluorescent indicator displacement assay. Additionally, co-assembling iron oxide nanoparticles with amphiphilic guanidinocalix[5]arene allows for the efficient removal of PFOS and PFOA from water. This approach, combining molecular recognition and the self-assembly of macrocyclic amphiphiles, is promising for the detection and remediation of water pollution. Figure 2a represents the mechanism of action of the developed detection method [61].

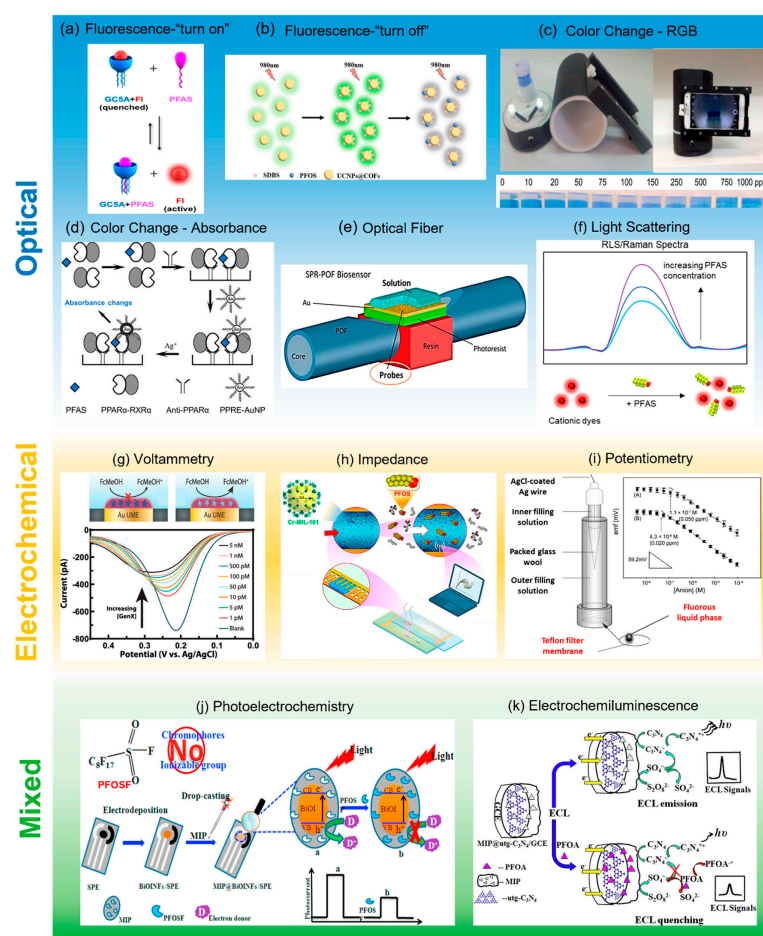


Figure 2. Types of PFAS optical sensors are presented in (a–f), electrochemical sensors are represented in (g–i), and mixed sensors are represented in (j,k). (a) A “turn-on” fluorescent sensor, fluorescein

-emissive species, and guanidinocalix[5]arenes-quencher. (b) A “turn-off” fluorescent sensor and sensing probe—UCNPs@COF nanoparticles. (c) A smartphone-operated calorimetric sensor for PFOA detection; (d) a modified gold nanoparticles probe-based calorimetric PFAS sensor. (e) An optical fiber biosensor with a monospecific antibody, used as a PFAS-capturing probe. (f) PFAS sensor based on light-scattering using cationic dyes. (g) Volumetric detection using an MIP-modified microelectrode (h) An MOF-based microfluidic impedance sensor for PFOS detection. (i) Potentiometric detection of PFAS using an anion exchange membrane. (j) PFAS detection using a photochemical sensing strip. (k) PFOA sensing using imprinted ultrathin graphitic carbon nitride nanosheets. Copyright © 2021, the Authors. Published by the American Chemical Society. This publication is licensed under CC-BY 4.0.

3.1.4. Living Cell Sensor for PFAS Detection

Researchers developed a fluorescent biosensor for the rapid detection of PFOA in drinking water and environmental water samples [62]. The biosensor, based on human liver fatty acid binding protein (hLFABP), was engineered to detect PFOA at the low levels set by regulatory officials. The sensor was conjugated with a circularly permuted green fluorescent protein (cp.GFP) and split hLFABP construct, allowing it to detect PFOA in PBS and environmental water samples with LODs of 236 and 330 ppb, respectively. The sensor also demonstrated the feasibility of whole-cell sensing in *E. coli* cells [62]. A genetically engineered bacterial biosensor was developed for the detection of perfluorinated compounds in water samples [63]. The biosensor integrates two genes, a regulatory (defluorinase gene) and a reporter gene (green fluorescence gene), using genetic engineering techniques. This was used to detect PFOA and PFOS in water samples [63]. The biosensor’s fluorescence emission was visualized using fluorescence microscopic images. The biosensor’s specificity was evaluated with various organic pollutants, including chlorinated compounds, polyaromatic hydrocarbons, and pesticides. It was used to detect perfluorinated compounds at nanogram levels in standard solutions and natural water samples, with a 24 h analysis time. The biosensor’s detection was validated using liquid chromatography coupled with a mass spectrometer [63]. A summary of the methods discussed above is provided in Table 2.

Table 2. Nanotechnology-based sensors for PFAS detection.

Method	Applications	Detection Limit	Selectivity and Sensitivity
Photo-electrochemical Sensors	Detection of PFOA and PFOS using nanohybrids and metal oxides.	LOD: 86 ng/mL for PFOS detection.	High sensitivity, enhanced surface area, selective detection [53].
Electrochemical Sensors	Quantitative and qualitative PFOS and PFOA detection.	LOD: 3.4 pM for PFOS, 6.51 ppb for PFOA.	Fast detection, low-cost, portable, suitable for field testing [55].
Electro-Chemifluorescence Sensors	Detection of PFOA and PFOS using luminescence and molecular probes.	LOD: 0.01 µg/L for PFOA.	Highly sensitive and selective detection at low concentrations [58].
Surface-Enhanced Raman Spectroscopy (SERS)	PFAS detection through signal enhancement using nanoparticles.	LOD: 10 ng/L for PFOA.	Highly specific, non-destructive, and fast analysis [59].

Table 2. Cont.

Method	Applications	Detection Limit	Selectivity and Sensitivity
Graphene-Based Sensors	Direct detection of PFAS using graphene nanostructures.	LOD: 1.2 ng/L for PFOS.	High surface-to-volume ratio, chemical stability, and fast response [60].
Molecularly Imprinted Polymers (MIPs)	Selective PFAS detection via molecular recognition templates.	LOD: 5.4 nM for PFOA.	Reusable, cost-effective, and high selectivity [62].
Fluorescent Nanoparticles	Detection of PFAS based on fluorescence-quenching or enhancement.	LOD: 15 ng/L for PFOS.	High sensitivity, multiplex detection, and portable nature [63].

4. Nano-Enabled Strategies for PFAS Removal

The high toxicity and persistence of PFAS in the environment have created an urgent need for the development of advanced strategies for PFAS detection and removal. Polyfluorinated compounds can degrade into perfluoroalkyl chemicals through both biological and abiotic processes [64]. PFAS molecules have a hydrophilic head of functional groups and a long-carbon-chain hydrophobic tail. PFAS are categorized based on their long-chain or short-chain hydrophobic tail [18]. The unique composition of PFAS contributes to their exceptional chemical and thermal stability, water-repellency, and resistance to acids and bases. PFAS are extensively utilized as basic ingredients in several consumer and commercial products [8].

Studies across many environmental contexts indicate that 80% of the PFAS released into the environment originate from the production and utilization of fluoropolymers [14]. PFOA and PFOS are the two most stable breakdown products, which are usually produced directly for both residential and industrial applications. Various processes and techniques have been established to eradicate PFAS from the water, soil, and environment, such as oxidation/reduction, nanofiltration, and UV radiation. Although some of these technologies are capable of eliminating or degrading PFAS, there are still obstacles and restrictions that need to be overcome [14]. Nanomaterials, including nanoparticles, nanotubes, nanocomposites, and nano-frameworks, are excellent substances with promising application in the removal of environmental pollutants. Some of the recent developments in the development of nanomaterials for PFAS removal are discussed below.

4.1. CNM/TiO₂ Composite for PFAS Removal

Studies have shown that a CNMs/TiO₂ composite enhanced the generation of hydroxyl radicals (OH) during a series of UV light-induced reactions, thereby catalyzing the PFOA breakdown [18]. Graphene is a semiconductor that has been used to prevent the formation of electron–hole pairs and increase OH production because of its high electron mobility and charge-separation capacity, hence enhancing the speed of the degradation of PFOA. Furthermore, graphene has a 0 eV band gap, facilitating interaction between the filled valence band and the vacant conduction band; in this way, it creates a new band for OH production [10]. Thus, it serves as an ideal electron-acceptor and is proficient in forming the transfer bridge.

The crucial function of CNMs in the above-described methodologies is enhanced by the supplementary electrochemical and photochemical degradation processes. Generally, CNMs may adsorb free PFAS in liquid media, resulting in their being concentrated on the surface of the CNMs [54]. Additionally, CNMs may facilitate electron migration from

other nanoparticles. The substantial quantity of nanoparticle attachment sites on carbon nanomaterials facilitates the uniform distribution of the nanomaterials in the solution.

4.2. Biomimetic Lignocellulosic Nano-Framework

Environmental pollution compromises exposure to harmful chemicals and their effect on human health and environmental sustainability, which involves serious health-related and ecological consequences. POPs such as PFAS are very complicated and costly to remove from the water, soil, and environment after their emission [65]. Therefore, innovative, cost-effective, and synergistic techniques are urgently needed for their immediate removal from the environment. A framework for this removal is referred to as the Renewable Artificial Plant for in situ Microbial Environmental Remediation (RAPIMER). PFAS compounds are rapidly absorbed by RAPIMER, with diversified adsorption capabilities for co-contaminants. In situ bioremediation is facilitated by RAPIMER with the fungus *Irpex lacteus*, promoting the removal of PFAS. RAPIMER provides a low-cost, promising solution for removing PFAS from biocompatible lignocellulosic sources, which would have a wide impact on sustainability and environmental bioremediation [65].

4.3. Photochemical Reactions

Photocatalysis is a powerful and efficient process for the elimination of organic contaminants from water. The photodecomposition of aqueous PFOA, a commonly studied PFAS, under UV light irradiation, was demonstrated to be successful using TiO_2 -based, In_2O_3 -based, and Ga_2O_3 -based photocatalysts made up of nanomaterials [8]. Altering the chemical composition, sizes, and shape of these photocatalysts can modify their photochemical characteristics and substantially enhance their effectiveness in PFAS removal. Some examples of the nanomaterials used for PFAS removal are stated below.

4.3.1. TiO_2 Based Nanomaterials

The introduction of metals to TiO_2 significantly improved its photoactivity. Transition metal-doped TiO_2 has been widely used in the photocatalytic degradation of PFOA [66]. A research study from Mingjie et al. offered new insights into the use of metallic nanoparticles for PFAS removal from polluted water [67]. They examined the photocatalytic degradation of PFOA in water using noble metallic NP-modified TiO_2 . The experimental results suggest that M- TiO_2 exhibited greater activity for PFOA breakdown compared to pure TiO_2 [67]. The primary products consisted of shorter-chain perfluorinated carboxylic acid. The significant photo-activities of M- TiO_2 were associated with noble metal NPs functioning as electron sinks, with Pt and Pd exhibiting higher work functions [67]. Pt- TiO_2 and Pd- TiO_2 exhibited enhanced photocatalytic activity. The pseudo-first-order rate constants for Pt-, Pd-, and Ag-modified TiO_2 were 0.7267, 0.4369, and 0.1257 h^{-1} , respectively, representing increases of 12.5, 7.5, and 2.2 times compared to those of TiO_2 . Pt- TiO_2 and Pd- TiO_2 exhibited enhanced photocatalytic activity [67].

4.3.2. Pb-Modified Nanoparticles

Lead (Pb) is an extremely toxic to human health and is one the most paradoxical elements but can be used for environmental remediation as Pb nanoparticles; for example, Chen et al. (2016) used Pb nanoparticles for PFAS removal. They showed that Pb doping functions as an electron trap under UV light, markedly reducing electron-hole recombination on the TiO_2 surface [68]. The researchers suggested that the reduced electron-hole recombination resulted in the generation of extra free hydroxyl radicals inside the UV/Pb-modified TiO_2 system, hence expediting the breakdown of PFOA. PFOA degradation in the UV/Pb-modified TiO_2 occurred at a much faster rate, 32.5, than in the pure TiO_2 /UV system, with a rate constant of 0.5136 h^{-1} , for the Pb-modified TiO_2 [67].

4.3.3. Ga₂O₃-Based Nanomaterials

Ga₂O₃ is a prevalent semiconductor with an ultra-wide band gap of 4.8 eV, which has many applications in photochemistry, gas sensing, phosphors, and power electronics. Ga₂O₃ exhibits five distinct polymorphs, of which monoclinic (β) is the most stable form and has been the subject of much research with various applications [69]. Ga₂O₃ exhibits a wider band gap and higher conduction band position in comparison to TiO₂. Recently, research studies have explored the photocatalytic efficacy of Ga₂O₃ in aqueous environments and discovered its PFOA degradation ability [70].

4.3.4. Titanate Nanotube

Titanate nanotubes are attracting attention as adsorbents and photocatalysts because of their photoelectronic reactivity [14]. Researchers examined an iron-modified composite for the concentration and degradation of PFOA. These nanocomposites are made up of activated carbon and titanate nanotubes (Fe/TNTs@AC). The material reduced over 90% of the surface-concentrated PFOA under UV exposure in just 4 h, with a 62% mineralization of PFOA into fluoride [71]. Photodegradation restored the material without the need for additional chemicals, and no substantial decline in performance was seen after six cycles [18].

4.4. Physical Adsorption

The PFAS contains both hydrophobic and oleophobic groups. It is complex due to its hydrophilic groups and C–F chains, as well as the existence of different forms. Its hydrophobic and hydrophilic characteristics contribute to its surfactant-like activity, but sorbent retention is influenced by several aspects that are typically ignored for other contaminants. Some of the physical methods used for the removal of PFAS are briefly mentioned in the following subsections [18].

4.4.1. Nanofiltration Membrane Separation

Nanofiltration is a pressure-driven separation method used for the separation of neutral components and divalent ions from aqueous solutions consisting of monovalent ions. A preliminary examination at a drinking water facility demonstrated that nanofiltration (NF) is effective (>98%) in eliminating PFAS [72]. Still, NF may not guarantee compliance with potable water standards, and the membranes remain susceptible to contamination. Nanocomposite, mixed-matrix membranes with new materials have the potential to transform reverse osmosis/nanofiltration processes. For instance, including metal oxide frameworks (MOF) in the polymeric membrane may enhance the selectivity and efficiency of PFAS capture, and analogous methods can be used for sorbents [73]. A zirconium-based MOF exhibited kinetic and regeneration properties that markedly overtook those of existing sorbents. Rapid sorption and almost complete elimination and recovery were achieved for the examined PFAS [74]. The fast separation and removal of this method make it an exceptional option for the development of integrated systems for capturing and eliminating PFAS [18].

Researchers have developed mixed-matrix-composite nanofiltration (MMCNF) membranes to improve PFAS removal. These membranes consist of thin polyelectrolyte multilayer films on thick polyethersulfone supports with β -cyclodextrin microparticles [75]. They achieved the near-complete removal of PFOA over longer filtration times. The membranes can be regenerated using ethanol, ensuring a high PFOA-removal performance. The eluent can be further concentrated via evaporation, enabling effective PFAS removal. Further concentration via evaporation is also possible [75].

4.4.2. Carbon Nanotubes

CNTs are carbon-based nanomaterials with extensive uses that are manufactured at a huge scale. CNTs have properties analogous to asbestos in terms of their size, shape, and persistence [76]. The development of nanotechnology-based methods for the decomposition of PFAS is a prominent research field [77]. Multiwalled carbon nanotubes (MWCNTs) are cylindrical-shaped, hollow allotropes of carbon, which are commonly used in polymer synthesis due to their strength and conductivity. Over the last decade, MWCNTs, due to their large surface area and strong affinity to organic molecules, have been used for the removal of environmental chemicals. Some of the recent developments that include PFAS removal using nanotubes are discussed below [18]. Recently, Fan Li et al. developed a cost-effective novel adsorptive photocatalyst, Fe/TNTs@AC, prepared using low-cost commercially activated carbon and TiO₂ [78]. The composite material demonstrated synergistic adsorption and photocatalytic activity, enabling a “concentrate-&-destroy” strategy for the fast and complete degradation of PFOA. This innovative strategy may degrade PFOA more cost-effectively [78].

Over the past decade, MWCNTs have been extensively studied due to their high surface area and high attraction for organic molecules. Xin-Li et al. [77] developed a layer-by-layer assembly technique to synthesize MWCNTs on magnetic carbon nanospheres. These nanospheres were used to extract six perfluoroalkyl substances (PFAS) from environmental water samples using ultra-high-performance liquid chromatography and tandem mass spectrometry. The sorbent showed wide linear ranges, low detection limits, and good repeatability. This method was applied to analyze six PFAS from environmental water samples, indicating its potential for the extraction and determination of PFAS [77].

4.4.3. Magnetic Iron-Oxide Nanoparticles

Magnetic nanoparticles (MNPs) have multiple unique magnetic properties, including superparamagnetism, elevated coercivity, a low Curie temperature, and high magnetic susceptibility [79]. MNPs are highly regarded by researchers across several fields, including data storage, magnetic fluids, biological applications, and chemical catalysis [79]. Modified magnetic iron oxide nanoparticles have a substantial sorption capacity and exhibit usefulness as reusable sorbents. MNPs were extensively studied, and a comprehensive overview of the recent research on their synthesis, modification, and environmental applications was released. Surface modification with specialized coatings might substantially enhance the capacity and selectivity of magnetic nanoparticles (MNPs), their composites, and other nanoscale (and larger) sorbents [18]. An ionic fluorogel efficiently eliminated 21 PFAS, including short-chain variants, from wastewater with more than 95% efficacy, demonstrating strong affinity and selectivity to amine-based sorbents. Specialized chemistries can be developed for magnetic nanoparticles (MNPs) and other nanoparticles (NPs), which may serve as the basis for diverse applications in the detection and removal of environmental pollutants [9,79].

4.4.4. Microbial PFAS Degradation

The primary method of the biodegradation of fluorinated compounds occurs through enzymatic secretion from environmental microbes [80]. Microbial enzymes use complex metabolic processes for the defluorination of organic compounds and use that energy for their growth and multiplication [80]. Although very few species are recognized as potentially degrading PFAS, when mixed with other substrates they can serve as a source of energy and nutrients. *Pseudomonas* have shown extreme tolerance to fluoride concentrations. Some strains of *Pseudomonas* even have the ability to catalyze defluorination [81].

A genetically engineered bacterial biosensor was developed by Sunantha et al. (2021) for the detection of perfluorinated compounds in water samples [63]. The biosensor integrates two genes, a regulatory gene (defluorinase gene) and a reporter gene (green fluorescence gene), and was used to detect PFOA and PFOS in water samples [63]. The biosensor's specificity was evaluated with various organic pollutants, and it was found to detect perfluorinated compounds at nanogram levels in both standard solutions and natural water samples. The detection was validated using liquid chromatography coupled with a mass spectrometer [63]. Similarly, *Dehalobacter* sp. (strain TeCB1) encodes the tetrachloroethene (PCE)-reductive dehalogenase, which is known to be essential for reductive de-chlorination [64]. These anaerobic bacteria use OHR (organohalide respiration) to survive, which provides the energy needed for metabolism. Enzyme P450 substitutes a transition metal for "F" in the F–C bond [64]. A summary of the methods discussed above is provided in Table 3.

Table 3. Nano-enabled strategies for PFAS removal.

Method	Mechanism of Action	Removal Efficiency	Advantages
CNM/TiO ₂ Composite	Photochemical degradation through hydroxyl radicals generated via UV light reactions.	>90% degradation of PFOA under UV exposure.	High efficiency, rapid degradation, reusable material [54].
Biomimetic Lignocellulosic Framework	Adsorption and in situ bioremediation using fungus <i>Irpex lacteus</i> .	High adsorption capacity and biodegradation.	Low-cost, eco-friendly, promotes sustainability [65].
Photochemical Reactions (TiO ₂ -based)	UV-induced photocatalysis using metal-doped TiO ₂ nanoparticles.	12.5–32.5 times higher efficiency compared to standard TiO ₂ .	Improved photoactivity, enhanced electron transfer, reusable [66].
Pb-Modified Nanoparticles	Electron trap mechanism reduces recombination, boosting degradation.	32.5 times faster degradation rate than pure TiO ₂ systems.	Enhanced electron–hole separation, higher hydroxyl radical generation [67,68].
Ga ₂ O ₃ -Based Nanomaterials	Photocatalysis using wide-band-gap (4.8 eV) semiconductor.	High degradation rate for PFOA in aqueous environments.	Stable under UV light, superior conduction band position [69,70].
Titanate Nanotubes	Adsorption and photocatalytic degradation under UV light.	>90% surface concentration reduction with 62% mineralization.	Reusable without chemicals, stable across multiple cycles [71].
Nanofiltration Membrane Separation	Pressure-driven separation for PFAS removal.	>98% removal efficiency in water treatment systems.	Scalable, high efficiency, effective for short-chain PFAS [73].
Carbon Nanotubes (CNTs)	Adsorption and photocatalysis using high-surface-area nanostructures.	>90% removal efficiency for PFAS.	High adsorption capacity, scalable, and reusable [78].
Magnetic Iron-Oxide Nanoparticles	Magnetic separation combined with adsorption for PFAS removal.	>95% removal of PFAS, including short-chain variants.	Reusable, selective adsorption, easy separation [79].
Microbial PFAS Degradation	Enzymatic defluorination and biodegradation by microbes.	Effective for specific PFAS compounds.	Biodegradable, environmentally friendly [81].

5. Conclusions and Future Perspectives

PFAS are synthetic chemicals that are continuously manufactured and added to the existing group. The newly developed PFAS provide alternatives to the existing group of compounds. The most common are PFOA and PFOS, which are extensively detected in soil, water, and sediments. They have been prohibited by several countries due to their ongoing detrimental effects on the environment and human health. PFAS are considered a group of new, emerging contaminants due to their toxic effects on aquatic animals and humans. The detection and removal of PFAS from the environment has become a major challenge and requires comprehensive and highly sensitive methods and tools. Recent research indicates that the conventional methods used for the detection and elimination of PFAS are very successful and promising, but each method possesses advantages and disadvantages, limiting their applications. Nanotechnology-based materials are cost-effective, ecologically sustainable, and more efficient. Carbon nanomaterials can be readily functionalized to include additional properties for the efficient detection and removal of PFAS and possess great hydrophobic, hydrophilic, or amphiphilic characteristics, enhancing their efficiency compared to traditional methods. The elimination of PFAS using nano-enabled techniques leads to notable adsorption efficiencies, with equilibrium times surpassing those of traditional adsorbents. These attributes might ensure the sustainable advancement of recyclable and reusable carbon nanomaterials with a specific selectivity for PFAS, subsequently facilitating their fair use and the removal of PFAS from aquatic bodies, reducing PFAS' further introduction into the food chain and the harmful complications they cause for plants and animals of higher trophic levels. Nanotechnology has consistently delivered effective results in the removal of PFAS from water samples. It is crucial to recognize that the toxicity of nanocomposites employed for the removal and mineralization of PFAS may present an additional barrier when addressing secondary contamination. Furthermore, they may pose significant risks to human health and the environment. Consequently, additional research is required to assess the stability and toxicity of these nanomaterials in an aqueous environment before their application to actual samples.

In this review, we addressed the nano-enabled methods used for the detection and removal of PFAS. The potential mechanisms of PFAS adsorption on different nanomaterials, including nanoparticles, nanotubes, nanomembranes, and modified nanostructures, were also evaluated. Alternative strategies for PFAS elimination were also discussed. Also, three-dimensional carbon nanomaterial structures, including carbon nanotube sponges, carbon nanotube electrodes, and carbon nanotube–graphene composite electrodes, are promising materials for the removal of PFAS. Additional research, including computational modeling, is required to elucidate the detection and removal of PFAS using nanomaterials. Additional attempts should be undertaken to develop advanced nanotechnology-based methods that can be utilized to readily and sustainably detect and remove PFAS.

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