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Review

From Sources to Environmental Risks: Research Progress on Per- and Polyfluoroalkyl Substances (PFASs) in River and Lake Environments

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Abstract

Per- and polyfluoroalkyl substances (PFASs) have attracted global attention due to their persistence and biological toxicity, becoming critical emerging contaminants in river and lake environments worldwide. Building upon existing studies, this work aims to comprehensively understand the pollution patterns, environmental behaviors, and potential risks of PFASs in freshwater systems, thereby providing scientific evidence and technical support for precise pollution control, risk prevention, and the protection of aquatic ecosystems and human health. Based on publications from 2002 to 2025 indexed in the Web of Science (WoS), bibliometric analysis was used to explore the temporal evolution and research hotspots of PFASs, and to systematically review their input pathways, pollution characteristics, environmental behaviors, influencing factors, and ecological and health risks in river and lake environments. Results show that PFAS inputs originate from both direct and indirect pathways. Direct emissions mainly stem from industrial production, consumer product use, and waste disposal, while indirect emissions arise from precursor transformation, secondary releases from wastewater treatment plants (WWTPs), and long-range atmospheric transport (LRAT). Affected by source distribution, physicochemical properties, and environmental conditions, PFASs display pronounced spatial variability among environmental media. Their partitioning, degradation, and migration are jointly controlled by molecular properties, aquatic physicochemical conditions, and interactions with dissolved organic matter (DOM). Current risk assessments indicate that PFASs generally pose low risks in non-industrial areas, yet elevated ecological and health risks persist in industrial clusters and regions with intensive aqueous film-forming foam (AFFF) use. Quantitative evaluation of mixture toxicity and chronic low-dose exposure risks remains insufficient and warrants further investigation. This study reveals the complex, dynamic environmental behaviors of PFASs in river and lake systems. Considering the interactions between PFASs and coexisting components, future research should emphasize mechanisms, key influencing factors, and synergistic control strategies under multi-media co-pollution. Developing quantitative risk assessment frameworks capable of characterizing integrated mixture toxicity will provide a scientific basis for the precise identification and effective management of PFAS pollution in aquatic environments.

Keywords: per- and polyfluoroalkyl substances (PFASs); bibliometrics; pollution characteristics; environmental behaviors; environmental risks

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1. Introduction

Per- and polyfluoroalkyl substances (PFASs) are a group of synthetic organofluorine compounds that have been widely used as surfactants, coatings, fire-fighting foams, and in electronic manufacturing [1]. Their unique carbon–fluorine (C–F) bonds confer hydrophobicity, thermal stability, and oxidation resistance, resulting in their exceptional persistence and stability in natural environments [2,3]. PFASs are highly resistant to natural degradation processes and can persist and accumulate in the environment for extended periods; hence, they are often referred to as "forever chemicals" [4–6]. In addition, PFASs can undergo long-range atmospheric transport (LRAT), enabling their migration across regional and even global scales [7,8]. During production, use, and disposal, they may be released into the environment through various pathways, including industrial effluents, domestic wastewater, and landfill leachates [7,9–11]. Currently, PFASs have been widely detected in various environmental media, including water, sediments, soils, and the atmosphere (in both gaseous and particulate phases), and have become contaminants of global concern [12,13]. Once introduced into the environment, PFASs exhibit strong bioaccumulative potential and can be transferred and magnified along the food chain, ultimately accumulating in higher trophic organisms [14,15]. Previous studies have demonstrated that PFAS exposure can induce multiple toxic effects in biota, including endocrine disruption, reproductive and developmental inhibition, as well as hepatic and immunological toxicity [16–18].

PFASs are typically categorized into two main groups: non-polymer and polymer compounds [19,20]. Some polymeric compounds, such as polytetrafluoroethylene (PTFE), fluorinated ethylene-propylene (FEP), and ethylene-tetrafluoroethylene copolymer (ETFE), are classified as "low-concern polymers" due to their high molecular weight and exceptional chemical stability [20,21]. In contrast, non-polymeric PFASs encompass a much broader range of species, including perfluoroalkyl acids (PFAAs) (including perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkane sulfonic acids (PFSAs)), perfluoroalkyl ether acids (including perfluoroalkyl ether carboxylic acids, PFECAs, and perfluoroalkyl ether sulfonic acids, PFESAs) and perfluoroalkane sulfonamides [20]. These compounds generally exhibit higher toxicity and pose greater potential environmental risks. To address the structural diversity of PFASs, analytical and detection technologies have advanced rapidly in recent years. Conventional targeted analyses of PFASs commonly rely on triple quadrupole tandem mass spectrometry (QqQ-MS/MS), which provides high selectivity and sensitivity. However, its applicability is limited by the availability of reference standards, resulting in poor qualitative performance for certain short-chain PFASs (SC-PFASs) and ultra-short-chain PFASs (C2–C3 PFASs) [22–24]. In comparison, non-target screening (NTR) based on high-resolution mass spectrometry (HRMS)—such as time-of-flight mass spectrometry (TOF-MS), Orbitrap mass spectrometry (OT-MS), and Fourier-transform ion cyclotron resonance mass spectrometry (FTICR-MS)—enables the simultaneous analysis of target compounds, suspect screening, and the identification of unknowns [23–27]. Despite challenges such as complex data processing and a higher risk of false positives [28], these methods have significantly expanded the breadth and depth of PFAS monitoring in environmental studies.

As global awareness of PFAS pollution risks has increased, regulatory policies for PFASs have gradually evolved worldwide. As early as 2009, perfluorooctane sulfonic acid (PFOS) was formally listed in Annex B ("Restriction of Use") of the Stockholm Convention on Persistent Organic Pollutants (POPs), requiring Parties to gradually eliminate its production and use except for specific applications. In 2019, perfluorooctanoic acid (PFOA) was included in Annex A ("Global Elimination") [29–31]. More recently, at the twelfth meeting of the Conference of the Parties (COP-12) in 2025, long-chain perfluoroalkyl carboxylic acids

(LC-PFCAs) were further added to the regulatory scope [32]. Alongside the expanding inclusion of PFASs within the global POPs control framework, many countries have introduced new regulations, extending the regulatory coverage from representative substances such as PFOA and PFOS to compounds including perfluorohexanoic acid (PFHxA) and its precursors, the GenX series, and total PFAS content. For example, Canada has established drinking water standards for a total of 25 PFASs [33]. Regulatory focus has also shifted from individual PFAS compounds to substance classes. The European Union has proposed a group-based management strategy according to PFAS use [34], and the U.S. Environmental Protection Agency (U.S. EPA) has incorporated multiple PFAS categories into the Toxics Release Inventory (TRI) [35]. Moreover, regulatory approaches are transitioning from passive control to proactive elimination and substitution. The EU, for instance, has implemented the "essential use" principle and plans to phase out non-essential applications over the next decade [36]. Similarly, countries in Europe and North America have developed comprehensive governance frameworks based on legislation such as CERCLA and TSCA, combined with total fluorine analysis and substance grouping technologies [37,38]. These regulatory developments reflect a clear trend toward a shift from fragmented PFAS management to systematic governance.

Rivers and lakes, as typical freshwater ecosystems, serve as critical compartments for the accumulation, transport, and fate of PFASs. Rivers, with their high hydrological connectivity and intense interactions with upstream and surrounding land use, often act as key conduits transferring PFASs to downstream waters [39,40]. Lakes, due to their long water residence times and limited hydrological exchange, become natural accumulation sites and "reservoirs" for PFASs [8,41]. In recent years, with continuous advances in analytical and detection technologies and growing awareness of health risks, the occurrence, transport mechanisms, and ecological effects of PFASs in river and lake environments have received increasing attention, leading to a rapid growth in related studies. However, a systematic review of the existing body of research is still lacking, and our understanding of current research trends and hotspots concerning PFASs in freshwater systems remains incomplete and insufficiently comprehensive.

PFASs originating from anthropogenic activities are highly persistent and bioaccumulative, and may pose long-term risks to aquatic ecosystems and human health. Therefore, conducting a systematic review of the occurrence and environmental risks of PFASs in river and lake systems holds significant scientific and practical value. This review takes 2002 as a starting point (although PFASs were widely used as early as the mid-20th century, systematic research on their occurrence, environmental behavior, and ecological risks in river and lake environments began around 2002) and comprehensively synthesizes research trends and key findings from 2002 to 2025. This study aims to elucidate the primary sources, occurrence patterns, and environmental behaviors of PFASs in river and lake systems, as well as their combined effects under the influence of coexisting components and the current status of environmental risks. The findings are intended to provide a theoretical basis for a deeper understanding of PFAS environmental behavior and potential risks, and to support the development of effective pollution control and environmental protection strategies.

2. Bibliometric Overview of PFASs Research in River and Lake Environments

2.1. Data Sources and Analytical Methods

Literature on PFAS pollution in river and lake environments published between 2002 and 2025 was retrieved from the Web of Science (WoS) Core Collection, with the data sources and search strategy presented in Figure 1. The search was limited to research articles and review papers. Each record was manually screened by title, author keywords,

and abstract to exclude irrelevant studies, those with mismatched research subjects, or duplicate and redundant publications, resulting in a final dataset of 1761 articles.

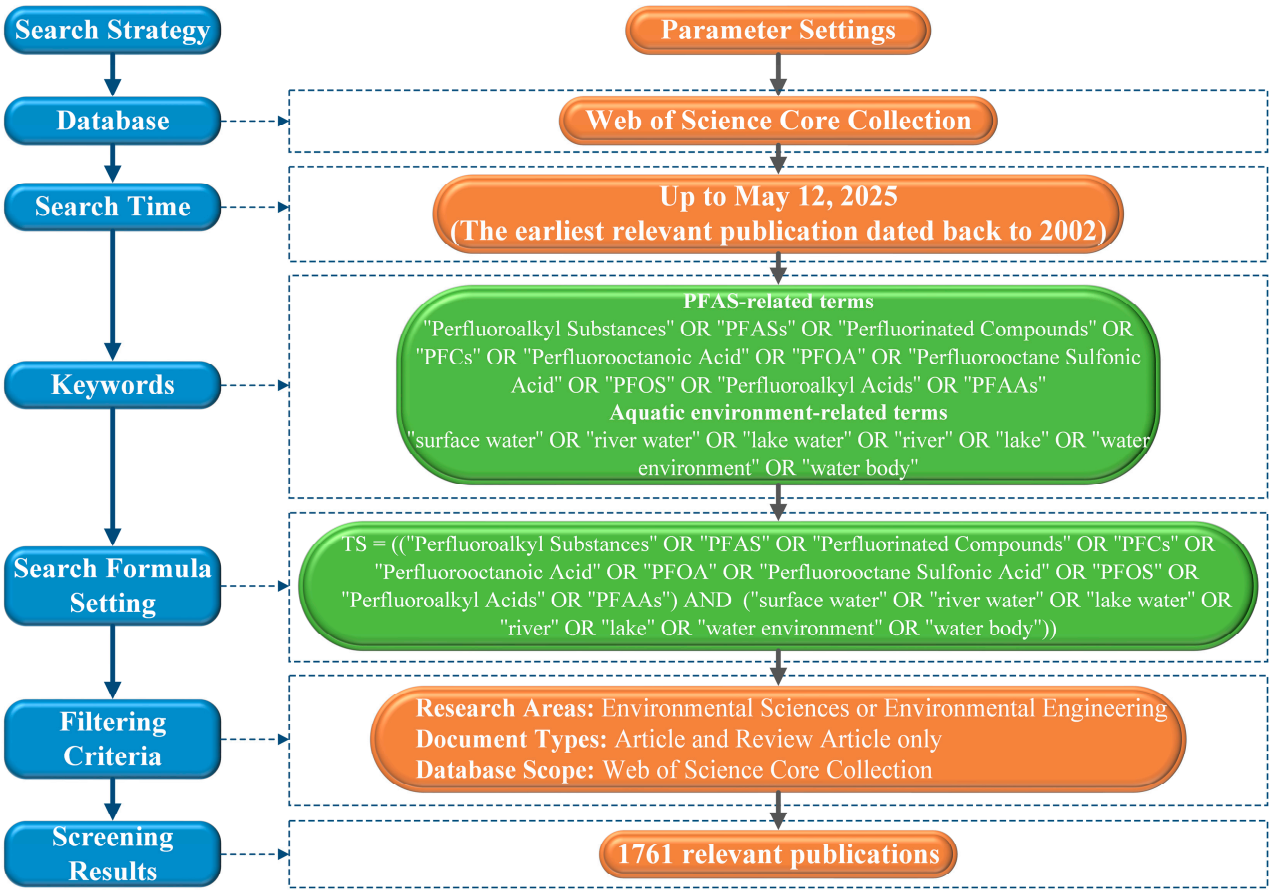


Figure 1. Data sources and search strategy.

For data analysis, Origin 2024 was used to visualize publication trends, citation counts, and research topics through Sankey diagrams, providing an intuitive overview of research development and thematic evolution. Co-occurrence analysis of high-frequency keywords was performed using VOSviewer (version 1.6.20) to construct keyword co-occurrence networks, allowing identification of major research themes and the evolution of research hotspots. The co-occurrence matrix generated by VOSviewer was further processed in Pajek to visualize research hotspots and their temporal evolution.

2.2. Publication Trends, Evolutionary Patterns, and Research Hotspots

As shown in Figure 2a, the number of publications related to PFASs in river and lake water environments within the WoS Core Collections has exhibited significant annual growth. The total publication count steadily increased from 4 in 2002 to 1761 in 2025. Annual total citations began fluctuating upward around 2010, peaking between 2019 and 2021. Citations have slightly declined in recent years, primarily because the newly published literature has had a short accumulation period and still needs time to consolidate. Nevertheless, the overall trend remains upward.

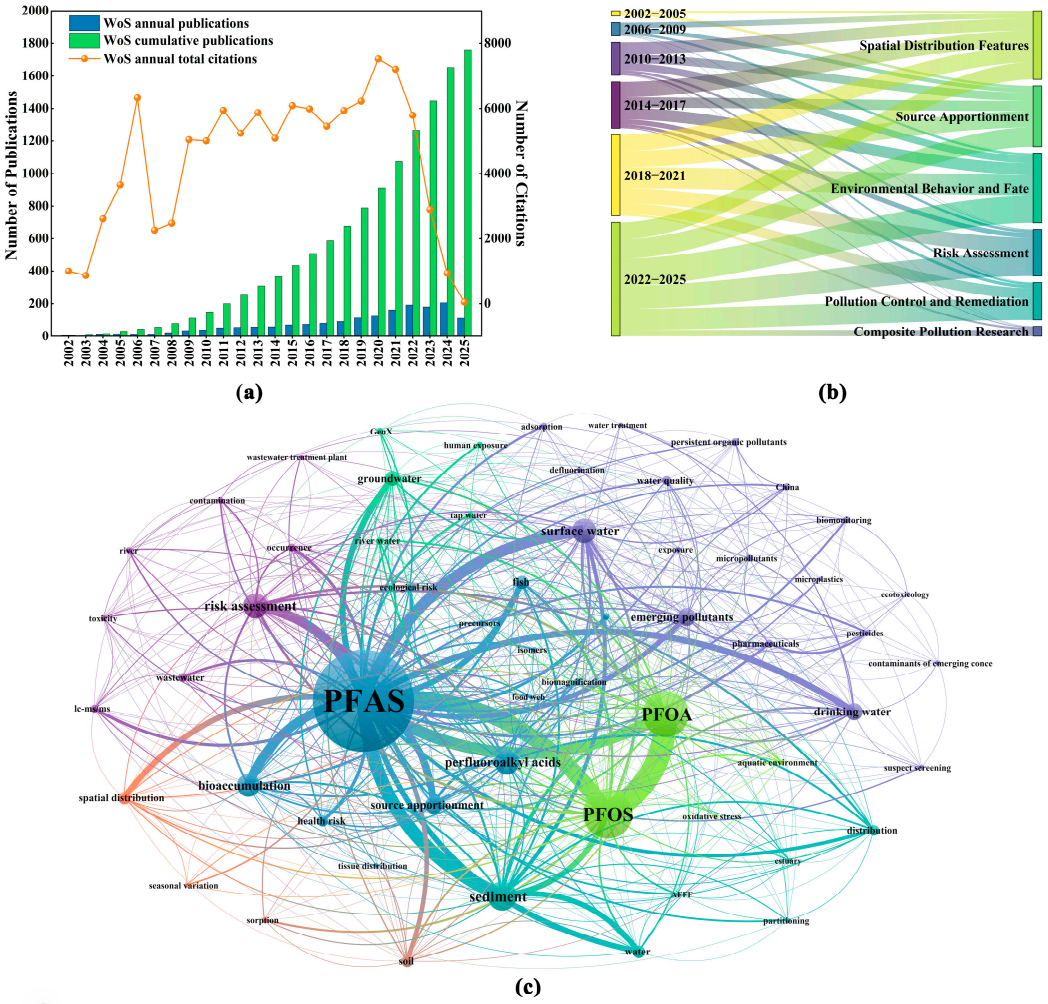


Figure 2. (**a**) Annual and cumulative publication volume and annual citation trend in Web of Science; (**b**) WoS hotspot evolution map; (**c**) high-frequency keyword co-occurrence network.

From the perspective of temporal evolution (Figure 2b), early studies primarily focused on the occurrence and mechanistic exploration of PFAS pollution. Studies conducted between 2002 and 2005 mainly focused on the spatial distribution and source apportionment of typical PFASs such as PFOS and PFOA. Between 2006 and 2009, although some exploratory studies emerged on environmental behaviors and fate, risk assessment, pollution control, remediation, as well as compound pollution, their volume remained limited. From 2010 to 2013, research on environmental behaviors and fate expanded, primarily focusing on the migration, degradation, and accumulation of PFASs in water–sediment systems. In recent years, an increasing number of studies have focused on the risk levels of PFASs and strategies for their pollution control. Between 2014 and 2017, studies on risk assessment increased significantly, with some research beginning to develop exposure models for aquatic ecosystems and human health related to the pollution of PFASs. Between 2018 and 2021, research on PFAS pollution control and remediation, represented by techniques such as adsorption, membrane separation, advanced oxidation, and bioremediation, gradually became a new focus, demonstrating certain removal potential in both laboratory and engineering applications [42–44]. However, the application of these technologies in river and lake environments remains limited. For example, adsorption exhibits low removal efficiency and poor regeneration for SC-PFASs [42]; advanced oxidation can achieve complete degradation but is often energy-intensive and requires harsh operational conditions [44]; and bioremediation is constrained by its insufficient capacity to

degrade PFASs [42]. Since 2022, the combined pollution of PFASs with other contaminants has emerged as a hot topic, particularly regarding the synergistic effects of PFASs with microplastics (MPs) and organic pollutants, as well as their combined impacts on aquatic ecosystems and drinking water safety. PFASs and MPs widely co-occur in the environment, and both exhibit long-term persistence and bioaccumulative potential [45–47]. Studies have shown that PFASs can adsorb onto MPs through electrostatic interactions, hydrophobic interactions, and functional group binding, with adsorption behavior influenced by PFAS carbon chain length, MP material, and degree of aging [45,48–50]. Co-exposure to MPs and PFASs may produce synergistic, antagonistic, or enhanced effects, thereby affecting PFAS accumulation and toxicity in organisms [51–53]. Although some mechanisms have been elucidated, the long-term environmental behavior of PFASs in combination with MPs and their ecological and human health risks remain to be further investigated [45].

The keyword co-occurrence network (Figure 2c) further illustrates high clustering and close connections among terms such as PFASs, PFOA, PFOS, PFAAs, surface water, sediment, drinking water, bioaccumulation, risk assessment, source apportionment, and emerging pollutants. This pattern indicates that, with the maturation of PFAS detection technologies, deeper investigation into environmental behavior mechanisms, and strengthened regulatory requirements, research in this field has gradually expanded from single-substance identification and regional distribution to more comprehensive areas, including non-target screening, exposure and risk assessment, combined effects of co-occurring pollutants, and pollution control technologies. Overall, driven by the transition of global PFAS regulatory policies from fragmented control to systematic governance, studies on PFASs in river and lake environments are entering a new stage of diversified and coordinated development, moving from qualitative identification to quantitative evaluation and integrated management.

3. Sources and Pollution Characteristics of PFASs in River and Lake Environments

3.1. Sources of PFASs

PFASs are primarily derived from anthropogenic activities and are not naturally produced in the environment [54,55]. Their pollution sources primarily include the production, use, and disposal of industrial, commercial, and consumer products, with their inputs into rivers and lakes categorized into direct emissions and indirect emissions (Figure 3).

3.1.1. Direct Input Pathways

The primary direct sources of PFASs entering river and lake environments include industrial production and discharge, emissions from specific sectors such as paper manufacturing, electronics, and semiconductor production, as well as firefighting applications and the use and disposal of consumer products in human activities. These direct inputs often result in PFASs exhibiting pronounced industry-specific patterns and spatial clustering in aquatic environments.

Industrial production and discharge in concentrated industrial zones represent the primary pathway for PFASs entering river and lake environments. In China, for example, typical coastal industrial areas such as the Bohai Bay region host fluoropolymer manufacturers, corrosion inhibitor factories, and Aqueous Film-Forming Foam (AFFF) production facilities, which are considered the main sources of PFAS contamination in the area. This is closely associated with the dense distribution of heavy chemical industries in the region [5,56]. Meanwhile, research on PFASs contamination in Asia's largest fluorochemical industrial park in Changshu, Jiangsu Province, China, further confirms the significant contribution of industrial clusters to PFASs pollution in aquatic environments. Test results

reveal total PFASs concentrations in surface water as high as 8250 ng/L, with PFHxA, PFOA, and 6:2 fluorotelomer carboxylic acid (6:2 FTCA) accounting for over 80% of the total concentration [57]. The relatively high proportion of PFHxA may be linked to the long-term use of this compound as a processing aid in polyvinylidene fluoride (PVDF) production within the park [57]. Similar industry-specific and regional clustering effects have also been reported internationally. For example, a study on the Mersey River in the United Kingdom identified upstream industrial zones as the primary sources of PFASs through load analysis, with PFOS concentrations reaching up to 270 ng/L [58]. In the Nairobi River basin in Kenya, emissions from specific industries led to localized accumulation of PFASs in both water and sediments, resulting in total PFAS concentrations in downstream sediments of up to 51.9 ng/L and posing potential environmental risks [59].

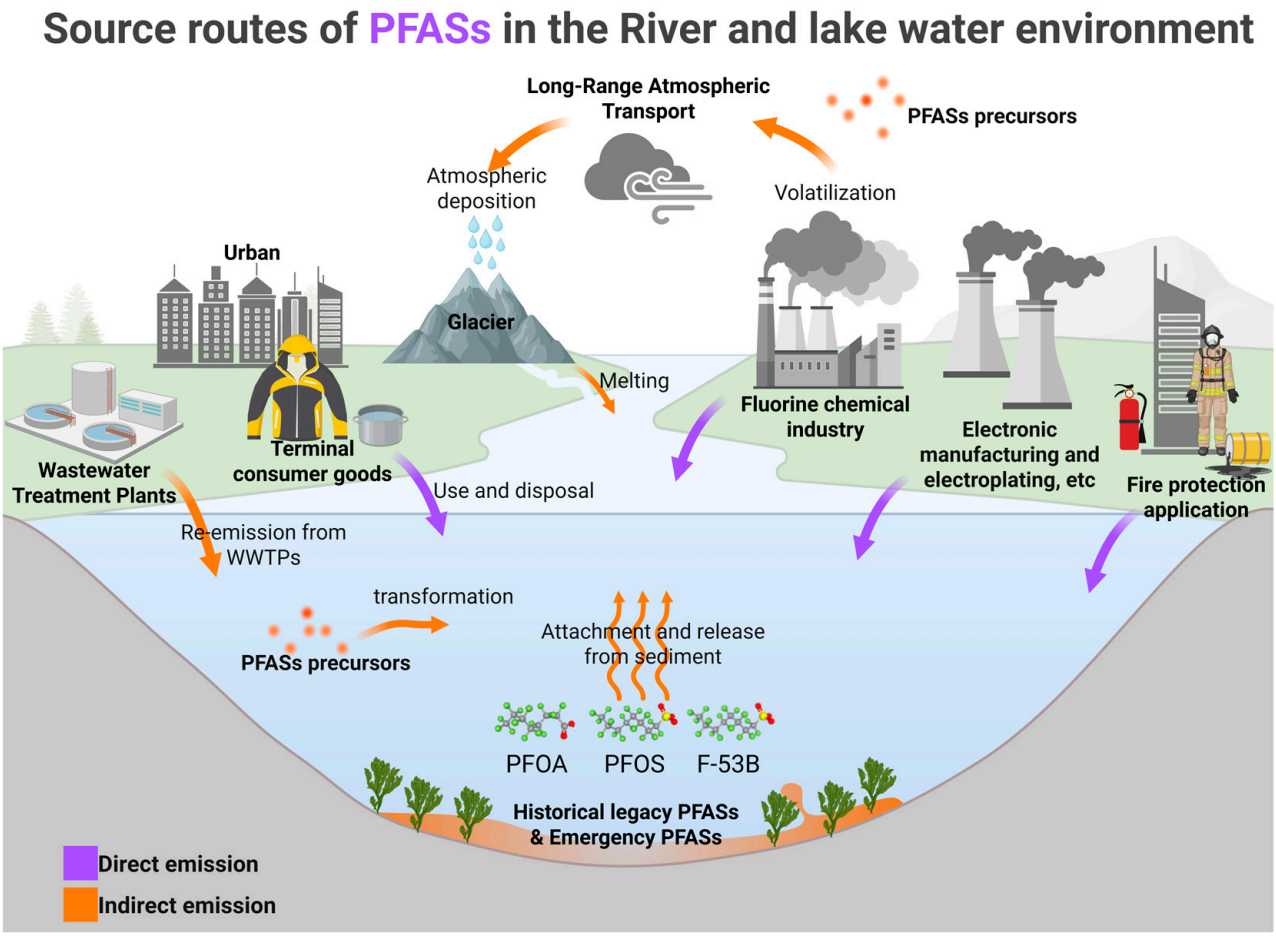


Figure 3. Sources of PFASs in river and lake aquatic environments.

Beyond emissions from fluorochemical industries, the use and discharge of PFASs in specific sectors, as well as the use and disposal of consumer products, represent important direct pathways for PFASs entering aquatic environments. In Europe, sediments from Lake Tyrifjorden in Norway are widely contaminated by paper manufacturing plants, with major PFAS classes including preFOS, SAmPAP, long-chain FTS, and PFCAs, reaching concentrations of several hundred to several thousand μg/kg. Historical discharges from these plants account for more than 90% of the total PFAS contamination in the lake [60]. Griffin et al. [61] reported that the semiconductor industry is also a significant PFAS source, with wastewater dominated by SC/C2–C3 PFASs, such as trifluoroacetic acid (TFA), perfluoropropanoic acid (PFPrA), and perfluorobutanoic acid (PFBA), reaching maximum concentrations of 96,413, 11,796, and 504 ng/L, respectively. These compounds are difficult to remove effectively using conventional water treatment technologies. In China, surveys

of the five major lake regions similarly highlighted the significant contribution of specific industries to PFAS contamination in water bodies. Key sources included food packaging, textiles, electroplating, firefighting, and semiconductor industries, with C4–C14 PFCAs and PFOS contributing up to 77.7% and 22.3% of the total PFAS concentrations, respectively. These findings indicate that the use of consumer products and emissions from specific application scenarios are increasingly important input pathways for PFASs [62].

Overall, the industrial and consumption structures of different regions determine the primary sources and compositional profiles of PFASs, with industrial clusters posing particularly high environmental risks that warrant close attention to their potential impacts on surrounding river and lake ecosystems.

3.1.2. Indirect Input Pathways

Indirect emissions of PFASs mainly involve the transformation and release of PFAS precursors, the long-term contributions of historical emissions and replacement products, secondary discharges from wastewater treatment plants (WWTPs), and inputs through LRAT. These processes are often highly concealed, long-lasting, and difficult to identify or quantify using conventional means, making them a key challenge in current pollution research of PFASs.

PFAS precursors can persist in the environment for a time and convert into structurally stable PFASs through processes like oxidation, photolysis, or biodegradation. This makes them ongoing sources of contamination. Research has shown that short-chain precursors, such as 6:2 fluorotelomer sulfonate (6:2 FTSA), can be oxidized or broken down into terminal products like PFHxA in the environment. The first detection of PFHxA in lakes on the Tibetan Plateau suggests that precursor transformation happens widely, even in remote areas with little human activity [8]. Other studies have found that perfluorocarbon C6 alkyl structure products can be important precursors of 6:2 FTCA in water near fluorochemical industrial parks [57]. Additionally, volatile precursors like fluorotelomer alcohols (FTOHs) can undergo atmospheric degradation to form PFCAs, which then enter river and lake environments through wet deposition, further increasing indirect PFASs inputs [11]. These findings suggest that behind seemingly "clean" emissions lies a complex pool of precursor compounds whose long-term environmental impacts have not yet been fully understood or included in risk assessment frameworks.

Historical emissions of PFASs and the ongoing use of their substitutes also act as significant indirect sources of pollution in aquatic environments. Legacy PFASs and their precursors can stay in the environment for decades, continuously releasing into water through chemical or biological changes. Although many new substitutes of PFASs are designed to be less bio-accumulative, they degrade very slowly and can still persist, continually contributing to pollution. Additionally, degradation products can migrate between water, soil, and the atmosphere, further extending their environmental residence times [63]. For example, source apportionment of PFASs in the Poyang Lake Basin showed contributions of 8.2% from legacy emissions and 26% from emerging PFAS substitutes, highlighting the combined impacts of historical use and current replacements on PFAS pollution [64].

WWTPs serve as major points for indirect emissions of PFASs in rivers and lakes because of limitations in treatment processes. Studies have shown that conventional wastewater treatments cannot effectively remove PFASs and their precursors and may even cause transformation of precursors during treatment [11,63]. In parts of the U.S. where AFFF has been widely used, WWTPs have become key pathways for the secondary release of PFASs [65]. Some studies even found that the concentrations of PFASs in treated effluents are much higher than in influents, showing how WWTPs can amplify indirect emissions [11,66].

LRAT can carry PFASs from emission zones to remote ecosystems through the atmospheric movement and deposition of highly volatile PFASs such as perfluorooctanesulfonyl fluoride (POSF) and fluorotelomer sulfonamides (FTSAs) [7,8,67,68]. Studies have detected PFOA and PFHxS in various lakes (e.g., Yamdrok Lake, Namco, and Ranwu Lake) on the Tibetan Plateau, mainly attributing their presence to LRAT driven by the Indian monsoon [8]. In industrialized regions, the combined effects of LRAT and local emissions create a multi-source input pattern of PFASs [11]. Additionally, the concentrations of PFASs in glacial rivers and lakes in the Arctic highlands have risen significantly with glacier melt, indicating that climate change and LRAT could further increase PFAS pollution risks in previously less-affected polar environments [69].

Based on this, numerous studies have employed models such as principal component analysis (PCA), multiple linear regression (MLR), and positive matrix factorization (PMF) to conduct source apportionment and fate studies of PFASs [68,70–72]. While these methods are well-suited for identifying known pollution sources, they still face limitations in addressing the complex degradation pathways of precursors, multi-compartment transport behaviors, and emerging sources from substitutes of PFASs. Recently, advanced techniques such as non-target screening, high-resolution mass spectrometry (e.g., LC-Orbitrap-MS), and isotope tracing (e.g., ^{13}C-FTOH) have been increasingly used to improve the accuracy and depth of source apportionment [27,41,73,74]. Meanwhile, new data-driven models, such as machine learning, are emerging as powerful tools for resolving complex pollution sources, offering new opportunities for advancing multi-source apportionment of PFASs [75,76].

3.2. Pollution Characteristics of PFASs

The types and spatial distribution of PFAS contamination in aquatic environments are primarily determined by the intensity of human activities, the nature of pollution sources, and environmental conditions. PFAS concentrations in rivers and lakes vary markedly across countries, generally exhibiting higher levels near industrial and urban emission sources compared to background waters. This pattern reflects the dominant influence of pollution source types and human activity intensity on their distribution (Table 1). The most frequently detected PFASs in rivers and lakes are PFCAs and PFSAs, with PFOS, PFOA, PFHxS, and perfluorononanoic acid (PFNA) showing relatively high detection frequencies [77–79]. Globally, the total concentration of PFASs in background aquatic environments generally ranges from nd to 10^3 ng/L [4,78,80]. In highly urbanized areas—especially near WWTPs, industrial discharge sites, or locations where AFFF is used, such as fire-training sites—PFASs levels are often much higher, reaching several thousand to tens of thousands of ng/L in some rivers and lakes [4,79,81]. For instance, in several lakes and rivers in the Great Basin of Nevada and California, United States, PFAS concentrations climbed up to 4754 ng/L due to urban development and WWTP discharges [65]. In rivers close to a decommissioned fluorochemical plant in Hubei Province, China, levels of perfluorobutanesulfonic acid (PFBS) remained as high as 11,462.9 ng/L, indicating that legacy point-source pollution can have long-lasting effects on nearby surface water [81].

In China, the concentration of PFASs in river and lake environments is generally within the middle-to-high range compared to global reports and exhibits distinct regional patterns [79,82]. Surface waters in the eastern plain lake regions and downstream urban clusters tend to have higher concentrations than those in the western and plateau lake regions [4,11,82]. In certain fluorochemical industry clusters and downstream receiving areas, the level of PFASs approaches the highest reported worldwide, while non-hotspot areas typically show concentrations within ND to 10^2 ng/L [41,48]. For example, the Xiao Qing River Basin is among China's most heavily contaminated areas, with 2020 monitoring

revealing the total PFASs level up to 25,429 ng/L in surface water, and emerging PFASs such as hexafluoropropylene oxide trimer acid (HFPO-TA) and hexafluoropropylene oxide dimer acid (HFPO-DA) reaching 1039 ng/L and 164 ng/L, respectively [82], indicating significant industrial discharge impacts. PFASs are also frequently detected in river and lake sediments, exhibiting significant spatial variations in their distribution. A study covering lakes in China's five major regions indicated that the total PFASs concentration in surface sediments ranged from 0.086 to 5.79 ng/g dry weight (dw), with an average of about 1.15 ng/g dw, dominated by long-chain perfluoroalkyl substances (LC-PFASs) [62]. The average total PFASs concentration in sediments from the eastern plain lake region was 1.72 ng/g dw—about 3.0 to 6.3 times higher than in the other four lake regions—and the profiles were dominated by PFOS, PFOA, and other LC-PFASs, indicating stronger human and industrial influences [62,83]. Overall, the pollution of PFASs in China is heavily influenced by the distribution of industrial activity and population density, especially in the east, where high population density, intense industrial and agricultural emissions, and inadequate wastewater treatment are the main driving factors [4].

The Yangtze River Basin spans diverse natural and socioeconomic zones across eastern and western China, encompassing plateau hills, urban plains, agricultural areas and industrial zones, making it a representative for understanding spatial differences in PFAS pollution. For example [84], in the upper reaches near Chongqing, PFNA, PFHxA, PFOA, PFOS, and perfluorodecanoic acid (PFDA) are frequently detected, with PFOA contributing 90.3–93.6% of total PFASs, and concentrations increasing downstream, indicating accumulation along the flow path. In the middle reaches near Wuhan, concentrations of eight PFASs in the Han River ranged from 8.60 to 568 ng/L, with widespread contamination largely attributed to discharges from nearby industrial parks; concentrations decreased markedly at the confluence of the Han and Yangtze rivers, likely due to dilution by the Yangtze mainstream [85]. In the lower reaches near Shanghai, surface waters are dominated by PFBS and PFOA, with generally higher concentrations in the northwest than in the southeast [10]. The primary PFASs input to the Yangtze Estuary comes from the Huangpu River, where PFOA is the dominant pollutant, likely linked to industrial production along the river [10].

Table 1. Concentrations of PFASs in surface water from representative regions (ng/L).

Country	Site	Total Concentration Range
China	Chongqing section of the Yangtze River [84]	1.41~53.8
	Wuhan section of the Yangtze River [85]	8.60~568
	Shanghai section of the Yangtze River [10]	113.38~362.37
	Surface water in Shanghai [86]	284~3018
	Yellow River [87]	15.57~36.42
	Liao River [88]	0.38~127.88
	Xiaoqing River [82]	25,429
	Hulun Lake [70]	3.67~8.84
	Chaohu Lake [89]	13.6~90.0
	Yangzonghai Lake [68]	14.95~26.42
America	Truckee River [90]	441.7
	Las Vegas Wash [90]	2234.3
	Great Lakes [91]	1~96
	New Jersey [92]	22.9~279.5
Korea	Asan Lake [13]	17.7~467
	Changwon region [93]	0.00~43.25
Poland	Oder River [94]	7.62~68.01
Brazil	Pampulha Lake [95]	191~12,400

4. Environmental Behaviors and Key Influencing Factors of PFASs in River and Lake Environments

PFASs entering river and lake systems typically exhibit distinct distribution characteristics across environmental media such as surface water and sediments. These patterns are primarily governed by a series of environmental behaviors of PFASs in aquatic systems, including their partitioning, degradation, and transport (Figure 4).

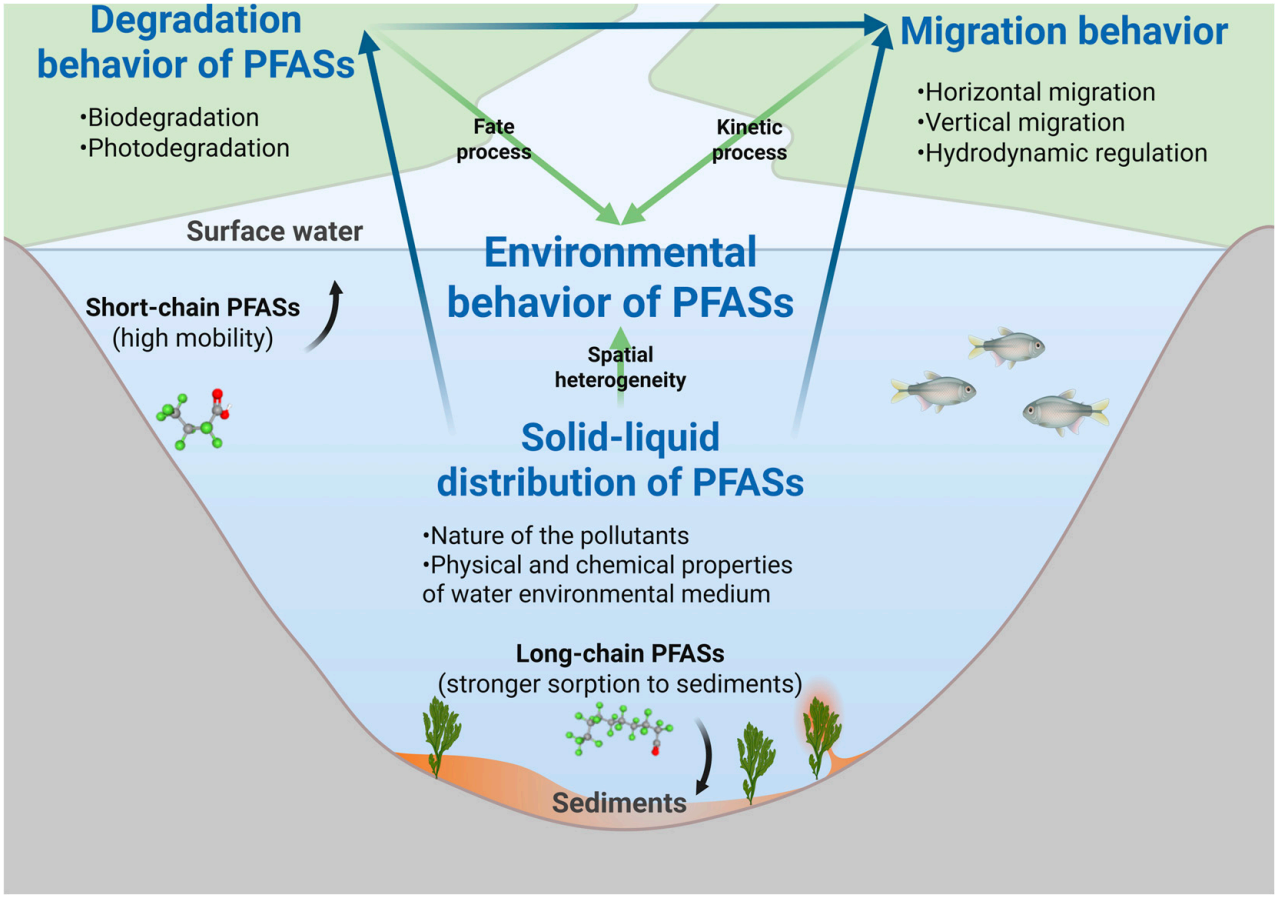


Figure 4. Environmental behaviors of PFASs in river and lake environments.

4.1. Water–Sediment Partitioning of PFASs

The partitioning behavior of PFASs in aquatic environments is influenced by both their intrinsic structural properties (e.g., chemical stability, carbon chain length, and functional group types) and environmental conditions (e.g., pH, salinity, hydrodynamics, and dissolved organic matter (DOM)) (Table 2). Firstly, the high chemical stability of PFASs makes them resistant to natural degradation processes, allowing them to persist and accumulate in environmental media such as water and sediments [96,97]. This persistence is mainly due to the replacement of hydrogen atoms on carbon chains with highly electronegative fluorine atoms (substituting C–H with C–F bonds), which confer oxidative resistance and reduce the electron density and reactivity of functional groups [96]. Secondly, PFASs with different carbon chain lengths exhibit distinct hydrophobic and hydrophilic properties, which in turn influence their partitioning behavior among various environmental media. Additionally, the different functional groups of PFASs can influence their distribution between water and sediments by affecting their solubility and adsorption behavior. Generally, PFASs with carboxylic groups (–COOH) tend to remain in the aqueous phase more than those with sulfonic groups (–SO_3H), as the higher hydrophobicity and larger molecular volume of –SO_3H enhance complexation with OC, promoting sediment association [31].

Table 2. Major structural properties and environmental factors influencing the partitioning of PFASs between water and sediments.

Classification	Factor	Influence Mechanism	Typical Behavioral Characteristics	Reference
PFASs structural properties	Chemical stability	Enhances resistance to oxidation and reduces reactivity	PFASs are resistant to natural degradation and tend to accumulate in environmental media	[96,97]
	Carbon chain length	Affects hydrophobicity and OC binding capacity	LC-PFASs tend to adsorb to sediments; SC-PFASs (e.g., PFHxA) are more mobile	[90,92]
	Functional group type	Influences solubility and adsorption behavior	$–SO_3H$ increases hydrophobicity and sediment deposition; –COOH favors the aqueous phase	[31]
Environmental conditions	pH	Affects particle surface charge and adsorption strength	High pH promotes desorption and mobility	[96,98,99]
	Salinity and ionic strength	Affects particle adsorption and bridging interactions	High salinity or high Ca^{2+} enhances PFAS accumulation in sediments	[100–102]
	Sediment properties	Influences adsorption capacity	High OC, fine particles, and iron/aluminum oxides enhance adsorption	[102–105]
	DOM	Competes for adsorption sites, affects desorption	DOM increases PFAS mobility; short-chain PFASs desorb more readily	[106–111]

Environmental factors such as water pH, salinity, ionic strength, redox conditions, sediment organic carbon (OC), particle size, and mineral composition can also influence the dynamic partitioning of PFASs by influencing their adsorption and desorption processes (Table 2). Under high-pH conditions, deprotonation of surface functional groups on particles and sediments weakens the electrostatic sorption of molecules of PFASs, increasing their desorption rates and promoting their downstream transport [96,98,99]. In high-salinity environments (e.g., estuaries), PFASs accumulation on suspended particles increases significantly [100,101]. Furthermore, when the ionic strength of surface water rises, the bridging effect boosts hydrophobic interactions between PFASs and sediments or suspended particles, thus increasing the solid-phase partitioning of PFASs [102]. In sediments, finer particles with higher OC contents usually have stronger sorption capacities for PFASs. The hydrophobic parts of sediment OC help bind PFAS molecular tails, creating stable sorption sites for LC-PFASs (e.g., PFOS and PFOA) [102,103]. Particle fineness and mineral makeup also influence sorption behavior. In sediments containing iron and aluminum oxides, low pH increases the positive charge on these mineral particles, enhancing their sorption capacity for negatively charged PFASs [104,105]. Moreover, aged sediments show lower PFASs desorption rates compared to fresh sediments, indicating that long-term contact can trap PFASs in micropores of sedimentary OC or form irreversible associations with organic functional groups, thereby reducing their desorption potential [105].

DOM, as a widespread form of natural organic matter in aquatic environments, can compete with PFASs for sorption sites on environmental media, thereby influencing their solid–liquid partitioning [106–108]. For instance, in a simulated brackish water–clay system, Jeon et al. found that DOM reduced the retention of PFOS and PFOA in the solid phase, likely due to competition for clay sorption sites. The concentrations of PFOS and PFOA in the aqueous phase were 1.26 to 1.39 times and 1.15 to 1.17 times higher, respectively, than in the DOM-free system, indicating that DOM increased PFASs mobility by competing for sorption sites [106,109]. DOM may also promote the desorption of relatively hydrophilic short-chain PFASs, increasing their release into the environment [110,111].

4.2. Biodegradation and Photodegradation of PFASs

PFASs possess highly stable molecular structures and generally exhibit low degradation efficiencies, although degradation may occur under specific conditions. In river and lake environments, PFAS degradation mainly occurs through two pathways: biodegradation and photodegradation. Biodegradation primarily refers to microbial-mediated breakdown of PFAS chemical structures. Certain bacterial strains, such as Pseudomonas, Ensifer, and Acidimicrobium, can degrade PFAAs under specific conditions, generating short-chain intermediates and fluoride ions (F^-) through reactions such as decarboxylation and desulfonation [112]. Although some studies have reported partial PFAS degradation by specific strains or microbial communities under laboratory conditions, microbial degradation is generally limited by the chemical inertness of the substrate and the metabolic capabilities of microorganisms, and typically relies on particular microbial populations and optimized aerobic or anaerobic conditions [113,114]. Furthermore, enzyme-catalyzed processes, as an emerging research direction, have shown potential for PFAS degradation but remain at an early stage, with challenges in reproducibility and scale-up application [115]. Notably, DOM can indirectly influence PFAS biodegradation by modulating microbial community composition and metabolic activity. As an important carbon source for microbes, the organic and inorganic nutrients released during DOM degradation can either promote or suppress the growth of particular microbial populations, thus impacting the bioavailability and microbial degradation efficiency of PFASs [116–119]. Compared with microbial degradation, plants cannot break down PFASs; however, their roots can uptake and accumulate PFASs, thereby altering their environmental distribution. The presence of DOM can modulate PFASs uptake by affecting their desorption and mobility in the rhizosphere [120,121]. On the one hand, DOM can increase the desorption of PFASs (especially PFOS) from soil, boosting their bioavailability in soil porewater and promoting their uptake by plant roots [122]. On the other hand, DOM can interact with PFASs through hydrophobic interactions, electrostatic forces, or functional group binding, which alters their distribution in the rhizosphere [123–125]. However, excessively high organic matter content can strengthen the solid-phase sorption of PFASs, reducing their bioavailability and making them less accessible for plants to take up [126,127].

Studies on PFAS photodegradation generally indicate that under natural sunlight most PFASs exhibit minimal light absorption, resulting in low rates of photochemical degradation [128]. However, the introduction of photocatalytic or specific photosensitized systems—such as intense UV irradiation, semiconductor photocatalysts, or photoelectrochemical setups—can substantially enhance degradation efficiency [129]. For instance, PFOS can be photochemically degraded under 254 nm UV irradiation using oxygen (O_2) as the terminal oxidant, producing short-chain PFCAs, F^-, $SO_4{}^{2-}$, and other intermediates [130]. Under visible light or simulated sunlight conditions, ZIS photocatalysts achieve higher degradation efficiency of sodium p-perfluorous nonenoxybenzene sulfonate (OBS, a novel PFAS) compared to conventional P25 TiO_2 [131]. The presence of DOM further complicates PFAS photodegradation processes. On the one hand, chromophoric groups in DOM can absorb UV light and generate reactive oxygen species, thereby inducing photochemical reactions and promoting the degradation of organic pollutants [129,132]. On the other hand, DOM may also inhibit PFASs photodegradation by competing for light absorption or quenching reactive species, and this inhibitory effect becomes more pronounced with prolonged irradiation [129,133,134]. For example, studies have shown that the photodegradation rate of PFOA is much lower in the presence of humic acid (HA) than under HA-free conditions, although the difference between the two rates gradually diminishes as the reaction proceeds [106,135].

Overall, natural sunlight and conventional microbial processes generally result in low degradation rates of PFASs, whereas engineered photocatalytic or photochemical systems can markedly enhance their degradation efficiency within a relatively short time.

4.3. Spatial Transport and Dynamic Release of PFASs

The transport behavior of PFASs in river and lake environments is governed not only by the intrinsic properties of the compounds and conventional hydrodynamic processes but also strongly depends on dynamic exchanges at the sediment–water interface. In particular, under extreme hydrological events (e.g., floods and droughts) or changes in water chemistry, the spatiotemporal distribution of PFASs is significantly influenced by processes such as resuspension, desorption, and release from sediments.

At the horizontal scale, flow velocity influenced by hydrodynamic conditions is a key factor determining the transport extent and rate of PFASs. Studies have shown that in lakes, river reaches, and reservoirs with low flow velocities, PFASs tend to attach to suspended particles and settle, reducing their horizontal transport distance [4,136,137]. Conversely, in high-flow river channels or sections affected by water diversion projects, increased hydrodynamic disturbances can resuspend PFASs from sediments into the overlying water, enhancing their downstream movement [93,101]. Some studies have further quantified the influence of flow velocity on PFASs sedimentation–desorption processes using partial least squares structural equation modeling (PLS-SEM), demonstrating a significant negative correlation between flow velocity and the particle–water partition coefficient (Kd) [40]. Recent experiments using confluence flumes have further elucidated the regulatory role of the sediment–water interface on the remobilization of PFASs under complex hydrodynamic conditions. Studies have shown [138] that in regions where tributaries and main streams converge, hydrodynamic partitioning leads to significant spatial variability in PFAS release from sediments. As the confluence ratio increases, the total amount of PFASs released from sediments into the water can increase by approximately 0.75-fold, with the concentration of SC-PFASs (e.g., PFBA) in the aqueous phase rising from 286 ng/L to 492 ng/L, whereas the release of LC-PFASs (e.g., PFOS) increases from 69 ng/L to 100 ng/L.

At the vertical scale, PFASs exhibit stratified transport characteristics across the sediment–water interface, the vadose zone, and the saturated zone, with their behavior jointly influenced by molecular structure and environmental media properties. The chain length and functional group composition of PFAS molecules affect their retention during vertical migration; LC-PFASs owing to higher solid-phase adsorption coefficients, are more readily retained by sediment particles and exhibit limited mobility. In contrast, SC PFASs (e.g., PFHxS), with stronger penetration capability and lower affinity for sediments, are more likely to rapidly percolate into deeper aquifers [139–141]. Additionally, during water level fluctuations or artificial recharge events, enhanced surface water infiltration can create preferential flow pathways in heterogeneous strata, thereby accelerating the vertical migration of PFASs [40,139]. Furthermore, changes in water chemistry, such as saline intrusion, can significantly influence the interfacial release and transport of PFASs. Column experiments have shown that high concentrations of ions (e.g., Na^+, Ca^{2+}, Mg^{2+}) in saline water can alter the adsorption of PFASs onto solid matrices [142]. Specifically, PFASs with terminal negative charges exhibit delayed release in soils with high ionic strength, whereas PFASs with terminal positive charges show accelerated release in matrices with high organic matter content. In addition, branched PFASs, due to their compact structure and weaker adsorption, tend to be preferentially released from solids, increasing their risk of reaching deeper aquifers [142]. These findings indicate that in coastal areas experiencing saltwater intrusion or rising groundwater salinity, the mechanisms governing vertical migration and release of PFASs may be substantially altered.

In addition, the increasing frequency of extreme weather events driven by climate change may further influence the transport and release of PFASs at the sediment–water interface [142,143]. For example, flooding can enhance runoff and sediment disturbance, promoting the resuspension and desorption of PFASs from sediments, whereas drought followed by water level recovery may alter the redox conditions and water chemistry in riparian porewater, affecting the adsorption–desorption equilibrium of PFASs between solid and aqueous phases. Recent studies have shown [143] that in rivers and lakes subject to long-term alternation between inundation and drying, SC-PFASs are more readily mobilized from sediments into the water. Under such conditions, protein-like substances in sedimentary DOM decrease while oxygen-containing functional groups increase, weakening co-adsorption and enhancing competitive adsorption between DOM and PFASs, thereby overall accelerating the dynamic release of PFASs at the sediment–water interface [143].

5. Environmental Risks of PFASs in River and Lake Environments

The environmental risks of PFASs in river and lake systems typically encompass both ecological risk and human health risk. Commonly used assessment methods include the risk quotient (RQ) approach, species sensitivity distribution (SSD) method, ecological dose–response model (EDRM), the AQUATOX mechanistic model, as well as the hazard quotient (HQ) and mixed-exposure assessment method (HQmix). Different methods analyze distinct risk dimensions for ecological systems and human health, and can be applied across various environmental settings, including urban rivers and lakes, industrial areas, and natural background regions; these approaches have been widely adopted in both domestic and international studies (Table 3). Overall, PFAS concentrations in most background areas remain low, with correspondingly minimal ecological and health risks. However, in point-source pollution sites, historical contamination zones, or media with elevated concentrations, the environmental risks posed by PFASs may increase substantially.

Table 3. Comparison of PFASs Risk Assessment Methods and Their Applicable Scenarios in River and Lake Environments.

Assessment Method		Principle and Features	Advantages	Limitations	Main Applicable Scenarios	Case Studies
Ecological Risk Assessment	RQ	Compares environmental PFAS concentrations with ecological toxicity reference values to identify potential risk	Simple and intuitive; suitable for preliminary cumulative risk assessment	Provides only qualitative or semi-quantitative results; limited in addressing mixture pollution and long-term accumulation effects	Preliminary ecological risk screening	Ecological risk assessment of the Bohai Sea and its inflowing rivers [5]; Water environment ecological risk assessment in Southwest China karst region [144]
	SSD	Constructs concentration–effect probability curves based on multiple toxicity data to derive predicted no-effect concentration (PNEC)	Accounts for interspecies sensitivity differences; strong capability for extrapolating risk thresholds	Limited toxicity data; applicability to emerging PFASs is insufficient	Overall risk assessment (suitable for PFOS/PFOA)	Ecological risk assessment of rivers near electroplating factories [145]
	EDRM	Fits dose–effect relationships to predict long-term low-dose cumulative effects; can consider hormesis effects	Captures long-term exposure and low-dose stimulation effects, enhancing risk identification	Data-intensive; interspecies sensitivity differences increase uncertainty	Long-term ecological risk assessment	–

Table 3. *Cont.*

Assessment Method		Principle and Features	Advantages	Limitations	Main Applicable Scenarios	Case Studies
Ecological Risk Assessment	AQUATOX	Mechanistic model simulating PFAS fate, exposure, and toxic effects in aquatic ecosystems	Integrates multiple factors; suitable for long-term predictions in complex aquatic environments	Requires extensive model parameters; high workload for construction and validation	Multi-media accumulation and long-term exposure risk analysis	Ecological risk assessment of shallow lakes near fluorochemical industrial areas [146]
Human Health Risk Assessment	HQ	Compares environmental concentrations with reference doses (RfD) to evaluate human exposure risk	Simple and intuitive; suitable for preliminary health risk assessment	Does not consider mixture exposure or population sensitivity differences	Preliminary human or food chain health risk assessment	Health risk assessment of regional large-scale regulated lakes [147]
	HQmix	Aggregates cumulative exposure effects from multiple PFASs or other pollutants	Can reflect risks from multi-component mixed exposure	Data-demanding; limited treatment of synergistic or antagonistic effects	Health risk analysis under combined pollution scenarios	Mixed exposure risk assessment of PFASs in urban rivers [148]

5.1. Ecological Risk Assessment of PFASs

As shown in Table 3, different assessment methods vary in their principles and characteristics, leading to differences in advantages, limitations, and primary applicable scenarios. The RQ method allows for a rapid estimation of potential ecological risks by comparing the measured environmental concentrations of PFASs with ecological toxicity reference values. This approach is simple, intuitive, and suitable for preliminary cumulative risk screening. For example, Meng et al. [5] evaluated PFASs in the Bohai Sea and its inflowing rivers and found that the RQ values for individual compounds were all below 1, indicating low risk. Similarly, Qiao et al. [144] reported that the concentrations of PFOA and PFOS in the water environment of the karst region in southwestern China were not sufficient to pose significant threats to the environment or human health.

The SSD approach integrates toxicity data of PFASs across multiple organisms to construct concentration–response probability curves, from which the PNEC can be derived. This method provides a more comprehensive reflection of interspecies sensitivity differences. However, its applicability is limited by the availability of toxicity data, and current studies mainly focus on PFOS and PFOA, offering insufficient coverage for emerging PFASs with unclear toxicological profiles [149]. The EDRM focuses on fitting dose–response relationships and can predict cumulative effects of long-term, low-dose pollutant exposure. However, it requires extensive toxicity data, and interspecies differences in sensitivity can increase the uncertainty of the assessment. Recently, researchers have proposed incorporating hormesis (low-dose stimulation and high-dose inhibition) type dose–response relationships into the EDRM framework to better describe potential stimulatory effects at low doses, thereby enhancing the sensitivity of risk detection [150]. Distinct from these methods, the AQUATOX model is a mechanistic tool used to simulate the fate and effects of pollutants in aquatic ecosystems. For example, Ma [46] simulated the effects of PFASs concentrations on the biomass of aquatic organisms in Taihu Lake. The results indicated an overall low ecological risk, although certain species, such as carp, may still face potential biomass changes.

Numerous studies and risk assessment results [28,149,151–155] indicate that ecological risk values in river and lake background areas—where PFASs concentrations approach environmental baseline levels—are generally below 1, with some studies reporting values even below 0.1, suggesting an overall low risk. However, in areas near fluorochemical plants, firefighting training sites, airports with AFFF usage, or historically contaminated waters, PFASs concentrations may exceed ecological toxicity thresholds, posing relatively

high potential risks [88,155–157]. Therefore, the ecological risk pattern of PFASs in river and lake environments can be summarized as the coexistence of "generally low risk" and "localized high risk". Future risk assessments and management efforts should focus on hotspot areas and sensitive species, while also considering long-term accumulation and scenarios of mixed pollution.

5.2. Human Health Risk Assessment of PFASs

Human health risks of PFASs are commonly evaluated using the health risk model recommended by the U.S. EPA [158]. This model calculates individual excess risk and the average annual excess risk based on factors like drinking water intake, body weight, and reference dose. The HQ is the primary indicator in this model, used to assess the health risk of a single chemical. Typically, HQ $\geq$ 1 signals a significant health risk, 0.2 $\leq$ HQ < 1 suggests a potential risk, and HQ < 0.2 is considered negligible. This method is simple and straightforward, but the results may be influenced by individual variability. To address the issue of combined PFASs exposure in water environments, some researchers have proposed a cumulative health risk formula—HQmix—based on the "funnel hypothesis" to preliminarily assess the joint health threats of multiple PFASs [148]. This approach offers a novel perspective for studying human health risks under mixed contamination scenarios; its applicable contexts and characteristics are summarized in Table 3.

Current studies generally show that HQ values of PFASs are mostly below 1 in most drinking water systems and among the general population, and the overall risk is considered "low" or "negligible" [159–161]. However, in certain areas with legacy fluorochemical contamination or near water sources affected by AFFF and industrial emissions, PFAS concentrations can be extremely high (hundreds to tens of thousands ng/L), significantly increasing human health risks under such exposure conditions [59,162,163]. Furthermore, in its recently released EPA Document No. 815R24006, the U.S. EPA has further lowered the health reference values for PFOA and PFOS, indicating that future human health risk assessments of PFASs may become more strict and conservative. At the same time, this also highlights the need for future management and policy measures to implement stricter controls and monitoring for high-risk areas, contaminated water sources, and specific vulnerable populations.

6. Conclusions and Perspectives

Over the past twenty years, research on PFASs in river and lake environments has made significant progress in understanding their pollution characteristics, sources, environmental behaviors, fate, and ecological and health risks. Studies have shown that PFASs are widespread in river and lake environments, with diverse species and highly variable concentrations. Their main sources include fluorochemical manufacturing, the use and disposal of consumer products, and both point- and nonpoint-source discharges. The spatial distribution of PFASs in the environment is influenced by the combined effects of human activity levels, source types, and environmental media properties. Their high mobility and environmental persistence enable them to persist in water–sediment systems over the long term and exhibit complex environmental behaviors. Risk assessments using methods such as RQ, SSD, and HQ have offered initial insights into the ecological and human health risks of PFASs in river and lake environments, although a quantitative evaluation of the combined toxic effects and long-term low-dose exposure risks of multiple PFASs is still limited.

Future research on PFASs in riverine and lacustrine environments should prioritize four interrelated directions. First, understanding of PFAS precursors and emerging substitutes remains limited. Advanced analytical approaches, such as high-resolution mass

spectrometry and non-targeted screening, should be combined with molecular structure prediction and transformation product analysis to elucidate environmental transformation pathways and assess associated ecological risks. Second, the transport and fate of PFASs have predominantly been studied within single environmental media or localized processes, leaving multi-media spatiotemporal dynamics insufficiently characterized. Future studies should implement integrated multi-media and multi-process fate modeling, incorporating isotope tracing and machine learning techniques, to resolve the mechanisms governing PFASs distribution and transformation. Third, PFASs often coexist with other emerging contaminants, such as microplastics and antibiotics, which may induce synergistic adsorption, co-migration, and combined toxicological effects. Comprehensive assessment of these interactions—including environmental behavior, joint toxicity, and cumulative human health risk—is critical to fully understand the implications of complex pollution scenarios. Finally, given the limitations of current remediation technologies, research should focus on developing efficient, scalable, and economically feasible treatment and restoration strategies, integrated with watershed-scale management, source control, and early warning systems, thereby establishing a full-chain pollution mitigation framework. Collectively, a systematic research paradigm encompassing "occurrence characterization—transport and transformation mechanisms—risk assessment—pollution mitigation" is essential for constructing a multi-dimensional PFAS risk management system and providing a robust scientific foundation for safeguarding freshwater ecosystem integrity.

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