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Review

Poly- and perfluoroalkyl substances (PFAS) pollution in South Korean water systems: A critical review of occurrence patterns and regulatory gaps

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) pollution in South Korean water systems is emerging environmental and public health concern, yet nationwide assessments remain limited. This review systematic evaluates PFAS occurrence across South Korea's primary water bodies and wastewater treatment plants (WWTPs), integrating historical and recent datasets from 49 facilities. Surface water monitoring (2004-2006) revealed widespread contamination, with 1.3-45.2 ng/L for PFOS and 2.9-19.7 ng/L for PFOA, particularly in the Hangang and Nakdonggang rivers, where downstream gradients reached ~40 ng/L (PFOA) and 38-59 ng/L (PFOS). Temporal analysis demonstrated a compositional shift from legacy long-chain PFAS (2006) to short-chain alternatives (2023-2024), reflecting regulatory-driven industrial transitions. WWTP analysis (2010) revealed industrial facilities consistently exceeded domestic counterparts, with PFOS reaching 829.44 ± 1837.41 ng/L in electronics manufacturing, of which data, collected in 2010, may not fully reflect the current situation. Conventional treatment proved ineffective, as effluent concentrations being 84-191% of influent levels. Current South Korean guidelines list only some PFAS as monitoring substances, with outdated values (70 ng/L for PFOA and PFOS) compared to stricter global standards. These findings highlight widespread PFAS detection and a critical regulatory gap, underscoring the urgent need for enforceable standards and advanced treatment technologies to support effective PFAS management.

Keywords: PFAS, Regulation, Removal efficiency, South Korea, Surface water, Wastewater

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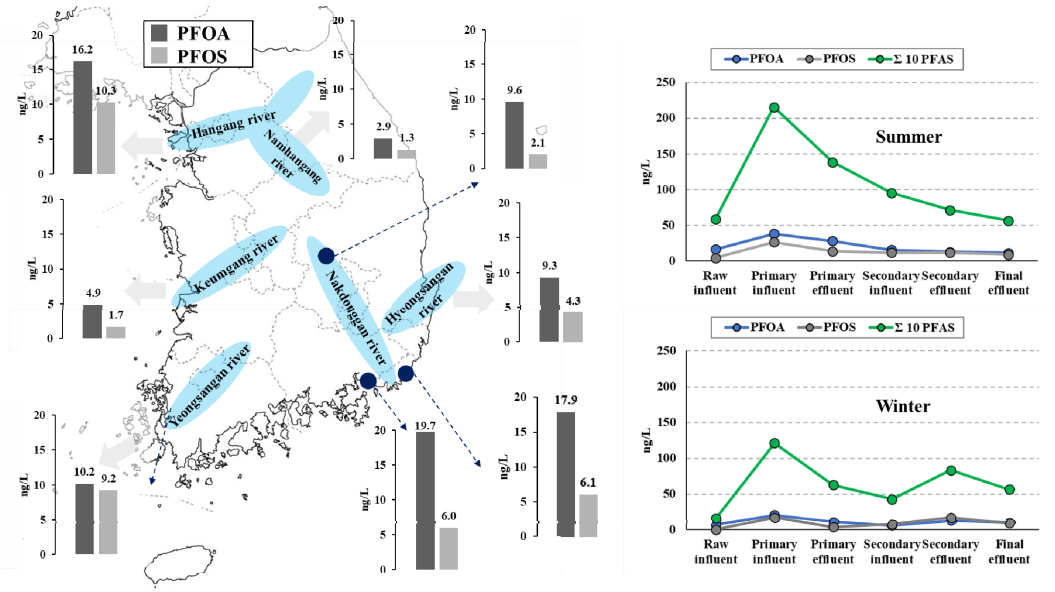
Graphical Abstract



[PFAS profile in surface water system] [PFAS profile in wastewater system]

1. Introduction

Per- and polyfluoroalkyl substances (PFAS) are a large group of synthetic chemicals defined by the presence of at least one fully fluorinated carbon atom (–CF_3 or –CF_2–) without direct attachment to hydrogen, chlorine, bromine, or iodine atoms that have been widely used in various applications since the 1950s [1]. In 2024, the United States Environmental Protection Agency (EPA) announced the Final PFAS National Primary Drinking Water Regulation (NPDWR), establishing maximum contaminant levels (MCLs) for six PFAS compounds— perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS), perfluorobutane sulfonic acid (PFBS) and hexafluoropropylene oxide dimer acid (HFPO-DA or GenX)—making the culmination of a regulatory process spanning over two decades [2]. In conjunction with this landmark regulation, many jurisdictions around the world including the European Union (EU), the United Kingdom (UK), Canada, Australia, and Japan have accelerated the establishment and strengthening of their own regulatory measures [3, 4]. These worldwide regulatory trends underscore the growing importance of scientific research to support strengthened regulations through systematic data collection, including regular monitoring, occurrence pattern analysis, and source identification studies. Such comprehensive scientific evidence has become critical for developing effective, evidence-based policies in countries currently lacking adequate PFAS oversight.

PFAS compounds possess exceptional properties including thermal stability, chemical inertness from robust C-F bonds, and dual repellent characteristics against water and oil due to their hydrophobic C-F chains and hydrophilic functional groups [5, 6]. These unique properties have driven the extensive use of approximately 14,000 PFAS compounds across diverse industrial applications and consumer products [7-9]. Industrial applications span paper production, textile manufacturing, leather processing, medical device fabrication, petroleum refining, mineral extraction, and electroplating operations [10-12]. Consumer products widely incorporate PFAS in aqueous film-forming foams (AFFFs), food contact materials, cosmetics, personal care items, paints, non-stick cookware, and waterproofing treatments [13-17]. This widespread use has inevitably led to extensive aquatic contamination through multiple discharge pathways, including municipal sewage, industrial effluents, landfill leachate, urban runoff, and agricultural drainage [18-21].

Aquatic contamination represents a critical exposure route for PFAS, posing direct threats to both human health and aquatic ecosystems [22]. Epidemiological studies have established strong associations between PFAS exposure and numerous adverse health effects, including thyroid dysfunction, asthma, anxiety disorders, obesity, pediatric allergies, immune system impairment, kidney disease, carcinogenicity, and hepatic damage [23-25]. Given these significant health risks, global research efforts have extensively investigated PFAS contamination in various water systems, including wastewater treatment plants (WWTPs), surface water, and groundwater over recent decades [1]. However, research for monitoring PFAS in WWTPs has been geographically concentrated in developed nations, with China leading at 31% of studies, followed by Europe at 30% and North America at 16%—where advanced analytical infrastructure and established PFAS regulations have prioritized comprehensive contamination assessments [1, 26, 27]. Conversely, developing countries demonstrate limited PFAS research capacity, particularly in regions where regulations are

sparse or addressed only indirectly through general environmental protection frameworks [28]. This disparity, coupled with insufficient resources for appropriate PFAS sampling and analytical methodologies, has created substantial knowledge gaps regarding PFAS contamination patterns in these regions. These deficiencies not only hinder comprehensive understanding of PFAS behavior but also impede the development and implementation of effective treatment technologies.

In South Korea, PFAS are managed under monitoring rules in drinking water; however, legally enforceable PFAS-specific standards have not yet been established, making the country a representative example of this research gap [29]. Previous studies have examined PFAS occurrence across South Korea, and some results are publicly available. However, these studies were limited to specific regions or focused only on certain compounds such as PFOA and PFOS; thus, no systematic and comprehensive comparison—including temporal and regional trends— has been conducted to date [18, 30, 31] (Table S1). Consequently, the available data remained fragmented and difficult to compare comprehensively. In light of the recent global reinforcement of PFAS regulations, it is crucial to summarize and compare the published PFAS data in South Korea. Such an effort will help bridge knowledge gaps between South Korea and countries that have already established systematic investigations and regulatory frameworks, thereby underscoring the urgent need for future regulatory actions.

Hence, this review addresses these critical knowledge gaps through three novel approaches: (1) conducting the comprehensive nationwide assessment of PFAS occurrence patterns, incorporating recent and extensive data across South Korea's primary water bodies and WWTPs, with specific focus on the influence of wastewater types, industrial sectors, and main treatment processes, (2) analyzing spatial and temporal variations, and (3) establishing a comparative framework with nations having PFAS-specific regulations to identify regulatory gaps and inform policy development. Given the absence of specific regulatory limits for PFAS in WWTP effluents in South Korea, comparing measured concentrations against drinking water standards provides a practical risk assessment approach. This comparison is further justified because WWTP effluents discharged into rivers that serve as drinking water sources, where insufficient dilution can maintain elevated PFAS levels in source water. Since conventional drinking water treatment processes in South Korea are generally ineffective at removing PFAS [30], PFAS concentrations in source water directly impact final drinking water quality, making this comparative assessment essential for public health protection. The findings will contribute to the development of evidence-based policies, standardized monitoring protocols, and source control strategies in regions currently lacking comprehensive PFAS regulations by providing essential baseline data.

2. Methodology

2.1. Data Collection

The review study primarily focuses on the occurrence and fate of PFAS in surface water bodies and WWTPs in South Korea. Literature was surveyed using Google Scholar as the main search platform, and results were cross-checked with Web of Science and Scopus to improve coverage and completeness. The search period was set from Jan 2000 to June 2025. A combination of the following keywords was used to identify relevant literature: PFAS, PFOA, PFOS, surface water, wastewater, occurrence, fate, removal efficiency, regulation, Korea, standards. Studies that reported quantitative data on PFAS concentrations, distribution patterns, and removal efficiency in surface waters and wastewater treatment systems were given priority during the selection process for the review. Irrelevant studies were excluded through title and abstract screening, followed by full-text assessment to confirm data availability, resulting in 21 publications from 2008 to 2024 that met the inclusion criteria. While South Korea was the primary geographical focus of this review, studies from other countries—including the US, EU, Japan, Australia, and China—were also incorporated to facilitate international comparison. This review primarily focused on legacy PFAS compounds, such as perfluorocarboxylic acids (PFCAs) and perfluorosulfonic acids (PFSAs), which are central to regulatory frameworks due to their well-documented health risks. In contrast, emerging or novel PFAS, although increasingly used in industrial applications, were not considered primary targets in this study due to lack of sufficient data. The literature reviewed in this study primarily consisted of peer-reviewed journal articles, and was complemented by official reports and technical documents published by governmental and environmental agencies. All abbreviations of PFAS compounds investigated in this study are explained in supplementary materials Table S2.

2.2. Data Extraction and Analysis

The selected literature was categorized based on several criteria, including the type of PFAS analyzed, study location, sampling time, characteristics of the aquatic environment (e.g., rivers, lakes, and streams), and WWTP characteristics (e.g., the nature of influent wastewater (domestic vs industrial), type of industrial activities, process configuration, and seasonal conditions). For WWTPs, particular attention was given to the treatment technologies applied and the configuration of each stage (e.g., primary, secondary, and tertiary treatment) to enable a clear analysis of PFAS behavior and seasonal variation.

Given the substantial heterogeneity among the collected studies—particularly in study design, data reporting, and analytical methods—this review did not employ statistical integration or meta-analysis, as such approaches could introduce misinterpretation. Instead, a qualitative approach was adopted, emphasizing the concentration trends observed in individual studies. PFAS concentrations in natural waters were compared based on water body type and regional characteristics, while PFAS removal performance in WWTPs was evaluated by comparing influent and effluent concentrations. When available, studies that reported concentration data for each treatment stage were also used to assess PFAS behavior throughout the treatment process. In addition, global regulatory trends were explored using official documents and technical reports published by environmental authorities with the most recent data. These materials were used

to examine current regulatory thresholds and future policy directions across countries.

3. Results

3.1. PFAS Occurrence in Surface Waters

The regional profiles of PFOA and PFOS concentrations in South Korean water systems were investigated using data collected from published peer-reviewed articles. Although the dataset does not fully include the most current monitoring results, it encompasses five major rivers and nine streams/lakes, thereby capturing the country's primary water bodies (Fig. 1, Fig. S1, and Table S3) [32-36]. Streams and lakes exhibited moderately higher concentrations than river waters, attributed to reduced dilution effects in smaller water bodies compared to large river systems. River waters consistently showed higher PFOA concentrations (2.9-19.7 ng/L) relative to PFOS concentrations (1.3-9.2 ng/L) across all sampling sites (Fig. 1(a) and Table S3). Conversely, streams and lakes demonstrated the opposite pattern, with PFOS concentrations (2.1-45.2 ng/L) generally exceeding PFOA concentrations (6.1-18.6 ng/L) (Fig. 1 and Table S3). Among the investigated water bodies—rivers (24 sampling points), streams (4 sampling points), and lakes (5 sampling points) — 29% rivers (n=7), 75% of streams (n=3), and all lakes (n=5) exceeded the USEPA NPDWR MCL of 4 ng/L for both PFOA and PFOS concentrations (Table S3) [2]. However, all sites remained below the Korean drinking water quality standard of 70 ng/L for PFOA and PFOS (established in 2018) [29].

Among river systems, the Hangang and Nakdonggang rivers exhibited the highest PFOA concentrations (14.0 ± 10.3 ng/L and 14.1 ± 12.0 ng/L, respectively), while elevated PFOS levels were observed in the Hangang and Yeongsangang rivers (8.8 ± 18.5 ng/L and 9.2 ± 7.7 ng/L, respectively). The Hangang river, traversing the Seoul metropolitan area, demonstrated a clear upstream-to downstream concentration gradient for both PFOA and PFOS compounds (Fig. 1(a) and Table S3). PFOA concentrations increased progressively from ~2.9 ng/L at upstream locations (the Namhangang river, which belongs to the Hangang river's upstream system) to ~40 ng/L at downstream locations, while PFOS was 1.3 ng/L upstream but reached 38-59 ng/L at downstream locations (Table S3). This spatial pattern suggests cumulative contamination from point sources such as WWTPs and landfill leachate along the river course [37, 38]. The Nakdonggang River, South Korea's longest waterway and a major industrial corridor, exhibited similar accumulation patterns. Middle stream site (Gumi and Waegwan) showed lower concentrations (PFOA: 6.8-9.6 ng/L; PFOS: 2.1-2.4 ng/L) compared to lower stream locations (Changwon and Busan), which recorded markedly higher levels (PFOA: 17.9-19.7 ng/L; PFOS: 6.0-6.1 ng/L). This downstream enrichment parallels Hangang river patterns, suggesting similar cumulative contamination phenomenon. These findings suggest the possibility of PFAS release to coastal areas through South Korean river water bodies, which was also observed by Ye et al. (2014), who reported PFAS concentrations increasing from 1 ng/L to 38 ng/L when approaching downstream Japanese river waters [39]. The remaining rivers (Yeongsangang, Keumgang, and Hyeungsangang) showed moderate contamination levels, with PFOA concentrations ranging from 4.9~10.2 ng/L and PFOS from 1.7~9.2 ng/L, indicating relatively lower anthropogenic inputs compared to the major urban-industrial corridors (Table S3). Among the lakes and streams, 7 out of 9 sites exhibited PFOS concentrations approximately twice as high as PFOA concentrations, with Bokha and Myeongchon lakes being the exceptions to this pattern. Geographically, streams located in the western costal area (Okgu, Gunja, Jeongwang, Singil streams) demonstrated relatively higher concentrations compared to other regions.

Detailed analysis of individual PFAS compounds was conducted at two representative sites: the Namhangang river, which is one of the two major tributaries in the upper region of Hangang river, and Sihwa lake, to examine compositional differences and temporal trends (Fig. 2) [34, 35]. The Namhangang river exhibited a distinct short-chain PFAS profile. The shortest-chain compounds examined, PFPeA and PFBS, dominated the contamination pattern with concentrations of 197.0 ng/L and 224.8 ng/L, respectively. In contrast, legacy compounds PFOA and PFOS were detected at relatively low levels (3.8 ng/L and 1.0 ng/L), while longer-chain compounds (PFNA, PFDA, and others) were below detection limits. Sihwa lake demonstrated a contrasting long-chain PFAS signature. Legacy compounds PFOA and PFOS were predominant, with concentrations of 33.6 ng/L and 47.5 ng/L, respectively, representing the highest detected levels among all compounds analyzed. The compositional differences between these two sites reveal important temporal trends in PFAS contamination. The predominance of long-chain compounds (PFOA and PFOS) in Sihwa lake samples from 2006 contrasts markedly with the short-chain compound dominance (PFPeA and PFBS) observed in Namhangang river samples from 2023-2024. The temporal shift in PFAS composition likely reflect the broader global regulatory response to PFAS contamination, although more studies are needed to strongly support this hypothesis in future research. International restrictions on PFOA and PFOS production, implemented after 2000-2002, prompted industries worldwide to transition to shorter-chain PFAS as alternatives [40, 41]. This regulatory-driven substitution has created consistent global pattern where legacy long-chain PFAS are being systematically replaced by shorter-chain alternatives in environmental samples [12, 42, 43]. This transition is exemplified in various water systems globally, with Zhao et al., (2016) documenting that short-chain PFASs comprised 88.8% of total PFAS concentrations in China's Yellow river [44]. While these shorter-chain compounds were intended to be less bioaccumulative [45, 46], their widespread detection of these compounds in recent samples demonstrates a critical oversight: their enhanced mobility properties have facilitated rapid environmental dispersion, establishing them as the dominant PFAS contaminants in contemporary water systems. Consequently, what was intended as a safer alternative has become a new generation of widespread environmental contaminants. This outcome highlights the need for comprehensive regulations targeting short-chain PFAS in South Korean water systems, with a special focus on monitoring coastal areas and downstream locations where PFAS accumulate, rather than whack-a-mole approach that simply shifts contamination from one PFAS class to another.

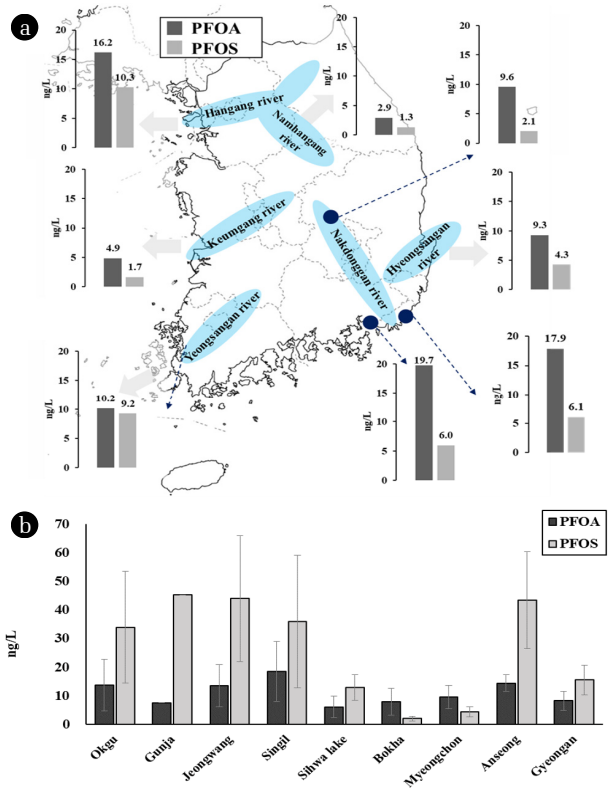


Fig. 1. PFOA and PFOS concentration in (a) river waters, (b) lakes or streams [32-36].

3.2. PFAS Profiles in South Korean WWTPs

3.2.1. Effect of wastewater type and industrial sector

To investigate point-source PFAS releases to river water, 29 WWTPs were selected based on their discharge locations within major South Korean watersheds (the Hangang, Nakdonggang, Yeongsangang, Keumgang rivers, and Jeju Island) and categorized into domestic and industrial types. PFAS concentrations were then analyzed at theses WWTPs, with eight priority compounds (PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, and PFOS) measured in both influent and effluent samples (Table 1) [47, 48]. In addition, accurately quantifying source contributions to water bodies requiring mass loading information. However, due to the lack of available literature-based data, this review only addresses concentration levels.

PFAS concentrations varied significantly by WWTP location and type, with industrial facilities consistently exhibiting higher and more variable concentrations than domestic counterparts (Table 1). Industrial WWTP influents demonstrated the greatest concentration ranges, particularly in the Hangang river watershed, where PFOS reached the highest concentration of 829.44 ± 1837.41 ng/L, followed by PFHxA at 122.97 ± 218.84 ng/L. Notably, industrial WWTPs exhibited inconsistent compound prevalence patterns that varied significantly by regional characteristics and facility types, with no consistent dominance of specific high-concentration compounds, reflecting the diverse nature of site-specific industrial activities and discharge processes. In contrast, domestic WWTPs exhibited more consistent contamination profiles, with PFOA and PFOS consistently representing the dominant compounds across all facilities, regardless of watershed location. This pattern suggests uniform sources from consumer products and household activities in domestic WWTPs, while industrial variability indicates process-specific PFAS applications and localized point-source emissions [49, 50]. Among the effluent waters investigated releasing to major rivers, all industrial WWTPs much higher concentrations than those in domestic WWTPs with in all PFAS compounds investigated, particularly industrial WWTP located in Hangang river bodies showed much higher concentration ranges than other industrial WWTPs. In domestic WWTPs, the Keumgang river facility had PFOS levels below the NPDWR (3.20 ng/L), while Jeju Island's effluent contained both compounds below the thresholds (Table 1). Although the Napo water intake, located in the lower Keumgang river basin (Napo-myeon, Gunsan City), serves as an emergency backup water source for Iksan City during extreme drought conditions when primary water resources are insufficient, given that WWTP effluent discharge levels are lower than drinking water quality standards (NPDWR), it is unlikely that treated WWTP discharge would result in meaningful contamination of downstream intake water sources

This study investigated PFAS distribution patterns across different industrial activities by collecting influent and effluent samples from four industrial sectors: paper, electronics, chemical, and metal industries (Table S4 and Fig. S2) [47]. Nine PFAS compounds were analyzed: PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFDoDA, PFHxS, and PFOS. Total PFAS concentrations

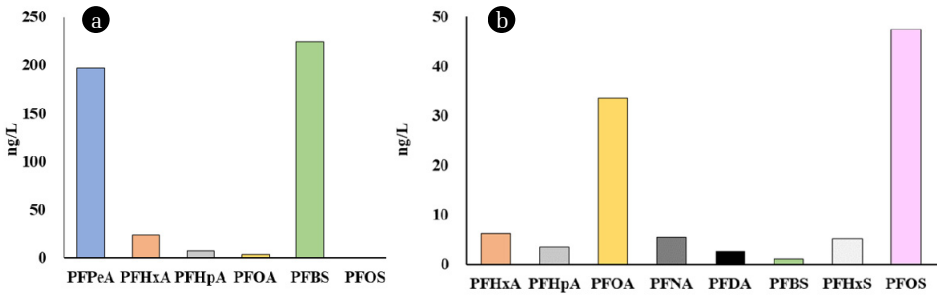


Fig. 2. Other PFAS concentrations in (a) Namhangang river, (b) Sihwa Lake (※ PFNA, PFDA, PFUnA, PFDoDA, PFTrA, PFTeDA, PFHxS, PFHpS, EtFOSAA, MeFOSAA were analyzed but not detected in Nam Han River, ※ PFOSA was analyzed but not detected in Sihwa lake) [34, 35].

in influents were highest in electronics wastewater (n=3, 136.18-991.72 ng/L), and lowest in chemical wastewater (n=1, 4.26 ng/L) (Fig. S2). Across all industries wastewater influents, PFCAs were the dominant PFAS group, with longer-chain PFCAs consistently present at lower concentrations than shorter-chain PFCAs. This pattern aligns with previous studies that reported—high predominance of PFCAs while showing minimal detection or high variability in PFSA detection [38, 51-54]. Electronics manufacturing wastewater exhibited a decreasing trend of PFCAs with increasing chain length, while PFHxS and PFOS were nearly undetected. Both paper and chemical industry influents were dominated by PFOA, but showed contrasting secondary compound profiles: paper industry contained detectable levels of PFHxS and PFOS, whereas chemical industry wastewater only contained PFHpA and PFNA with no detectable PFSAs. Metal processing wastewater displayed a unique profile with PFHxA as the predominant compound and other PFAS at minimal or undetectable concentrations. In the paper industry, the high concentration of PFOA detected reflects the widespread use of PFAS as coating materials on food-contact papers to provide grease resistance, with PFOA being particularly prevalent in the coatings for disposable products such as paper cups [5].

In effluent samples, electronics industry showed reduced concentrations for most PFAS compounds, but PFOS showed slight increase compared to influent, though without statistical difference (p=0.267). Conversely, chemical industry effluents exhibited increased PFAS concentrations, while metal processing demonstrated mixed results with elevated PFHxA levels but decreased concentrations of other compounds. Although this investigation is limited in reflecting current industrial conditions due to the lack of newly available data, the detailed analysis of PFAS source patterns remains meaningful. These findings suggest that the industry-specific PFAS characteristics in influent wastewater—including compound composition and precursor transformation potential—are critical determinants of the PFAS distribution patterns observed in final effluent discharge, underscoring the paramount importance of influent composition control. Understanding PFAS temporal trends in industrial WWTPs is important for

Table 1. Summary of PFAS concentrations in influent and effluent in Korean WWTP systems.

Refs.	Location	WWTP type	ID		PFHxA	PFHpA	PFOA	PFNA	PFDA	PFHxS	PFOS
Song et al. (2011) [47]	Hangang river	Domestic	WWTP-1 (n=6)	Influent	2.38 ± 2.42	1.15 ± 0.79	5.45 ±0 .99	0.70 ± 0.2	0.88 ± 0.47	2.43 ± 3.34	13.39 ± 8.28
				Effluent	2.26 ± 1.50	1.21 ± 0.78	6.77 ± 1.31	0.70 ± 0.30	0.60 ± 0.28	3.37±1. 38	10.89±1 1.50
Song et al. (2011) [47]		Industrial	WWTP-2 (n=5)	Influent	122.97 ± 218.84	74.05 ± 136.82	39.43 ± 43.76	10.86 ± 4.26	3.77 ± 4.93	10.74 ± 23.4	829.44 ± 1837.41
				Effluent	39.65 ± 21.69	15.32 ± 17.29	15.86 ± 8.59	4.72 ± 5.07	2.84 ± 3.65	17.18 ± 30.59	1511.33 ± 3557.98
Song et al. (2011) [47]	Nakdonggang river	Domestic	WWTP-3 (n=4)	Influent	3.25 ± 4.47	4.42 ± 6.20	13.71 ± 13.72	1.67 ± 2.11	1.30 ± 1.63	0.27 ± 0.53	3.74 ± 3.25
				Effluent	2.57 ± 4.08	4.09 ± 6.34	13.00 ± 13.62	2.01 ± 2.54	1.80 ± 2.40	0.49 ± 0.79	7.62 ± 10.30
Song et al. (2011) [47]		Industrial	WWTP-4 (n=4)	Influent	14.82 ± 24.89	2.1 ± 2.87	10.59 ± 10.93	0.88 ± 1.43	0.31 ± 0.62	4.67 ± 5.84	5.63 ± 4.28
				Effluent	23.52 ± 35.36	3.78 ± 6.23	18.60 ± 20.22	3.24 ± 4.36	0.88 ± 1.35	4.42 ± 6.11	6.25 ± 3.80
Song et al. (2011) [47]	Keumgang river	Domestic	WWTP-5 (n=1)	Influent	0.00	0.00	3.54	0.52	0.00	0.00	l.74
				Effluent	0.00	0.66	4.94	0.51	0.83	1.46	3.20
Song et al. (2011) [47]		Industrial	WWTP-6 (n=4)	Influent	20.59 ± 28.48	9.14 ± 14.20	35.31 ± 53.51	3.73 ± 4.34	1.81 ± 3.52	0.00 ± 0.00	7.76 ± 5.92
				Effluent	20.07 ± 25.66	10.47 ± 15.77	29.05 ± 21.8	4.19 ± 3.58	2.88 ± 3.77	0.29 ± 0.65	7.41 ± 9.85
Song et al. (2011) [47]	Yeongsangan g river	Industrial	WWTP-7 (n=2)	Influent	0.00 ± 0.00	11.36 ± 11.16	43.52 ± 54.57	5.79 ± 6.82	0.00 ± 0.00	4.35 ± 6.15	37.13 ± 4.70
				Effluent	19.28 ± 17.29	1.49 ± 2.11	33.42 ± 41.54	0.41 ± 0.58	0.00 ± 0.00	6.83±9. 66	203.77 ± 277.66
Kim et al. (2016) [48]	Jeju	Domestic	WWTP-8 (n=3)	Influent	2.95 ±1.80	—	3.35 ± 0.69	—	—	0.21 ± 0.37	1.83 ± 0.98
				Effluent	2.29 ± 1.20	—	3.05 ± 0.71	—	—	0.00 ± 0.00	0.98 ± 0.21

*—indicates no information available.

developing effective source control and treatment technologies. Regular watershed monitoring and continuous data updates are essential to reflect current conditions accurately, which is a key objective of this review.

3.2.2. Effect of wastewater treatment process and seasonal variation

The effect of different types of WWTP main treatment processes on the behavior of PFAS was investigated by comparing conventional treatment process and UV disinfection using data from 19 WWTPs, particularly focusing on three PFAS compounds (PFOA, PFOS, and PFHxS) that are included in the NPDWR list (Table 2) [54-56].

Conventional treatment processes in this review, incorporating primary and secondary clarifiers with conventional activated sludge (CAS) and biological treatments such as anaerobic-anoxic-oxic (A2O) or biofilter (BF) systems, demonstrate limited effectiveness for PFAS removal (removal efficiency defined as the difference between PFAS concentrations in influent and effluent divided by influents concentrations). Most conventional WWTPs showed minimal removal of total PFAS compounds, with the highest removal being only 16% for the sum of 5 PFAS compounds in WWTP-13 [56]. Instead of effective removal, negative removal efficiencies were frequently observed, ranging from -2% to -91%. These negative removal efficiencies can also be expressed as residual %, defined as the ratio of effluent concentrations to influent concentrations (Table 2). This net increase of PFAS during treatment is attributed to precursor transformation, a mechanism well-documented in previous studies [57-60]. For compound-specific differences, PFOA showed almost no removal, with the highest value being only 7% in the CAS or A2O treatment process. Most other treatments exhibited negative removal values in Denipho, A2O, and biological or UV processes, suggesting that PFOA may be readily generated from through precursor compounds during these treatments (Fig. S3). In contrast, PFOS demonstrated consistently positive removal, with 75% and 76% removal observed in the Denipho and A2O processes, respectively. Since both Denipho and A2O typically involve substantial activated sludge production, the high removal efficiency of PFOS is likely attributable to its stronger hydrophobicity and greater tendency to adsorb onto sludge. Conversely, the more hydrophilic nature of PFOA may favor its release from sludge. These findings highlight the importance of sludge adsorption-specific mechanisms, which should be further investigated in future studies.

Kim et al. (2012) investigated PFAS removal efficiency across 15 WWTPs, comprising five domestic, five industrial, and five mixed domestic-industrial facilities (Table 2) [55]. The domestic WWTP (WWTP-9) achieved 13% removal for the sum of 15 PFAS compounds, while industrial and mixed domestic-industrial WWTPs showed net PFAS increase with residuals of 102% and 191%, respectively. Most notably, the mixed domestic-industrial WWTP-11 nearly doubled total PFAS concentrations from 1,100 ng/L in influent to 2,100 ng/L in effluent, suggesting that industrial wastewater contains PFAS precursors such as fluorotelomer alcohols that transform into target compounds during treatment [1, 58]. One industrial WWTP-10 incorporating an additional BF process achieved 0% removal for PFOA, but 35% and 30% removal for PFOS and PFHxS, respectively, while overall removal remained at -2%. Given PFAS resistance to biological degradation, the positive removal observed for PFOS and PFHxS likely results from sorption onto BF materials rather than actual degradation, though further detailed studies are needed to confirm this mechanism.

Joo et al. (2021) examined PFAS concentrations in influent and secondary effluent (effluent from the biological treatment process) from two WWTPs (City I and City G), revealing markedly different removal patterns despite both employing conventional biological treatment (Table 2) [56]. City I (WWTP-12) showed predominantly negative removal efficiencies, indicating net PFAS increase in effluents, while City G (WWTP-13) achieved effective removal ranging from 10% to 78% for most compounds, with only PFOA exhibiting a 114% residual. The contrasting performance may be explained by biological treatment process differences. WWTP-12 employed the Denipho process, which intermittently introduces fermented primary sludge into the biological reactor as an organic substrate. This process may facilitate the release of PFAS previously adsorbed onto primary sludge, resulting in elevated effluent concentrations [56].

In contrast to conventional treatment, Kim et al. (2021) [54] analyzed 17 PFAS compounds in influent and effluent samples from two mixed domestic-industrial WWTPs employing conventional biological treatment with UV disinfection process (Table 2). Both facilities achieved positive removal efficiencies: WWTP-14

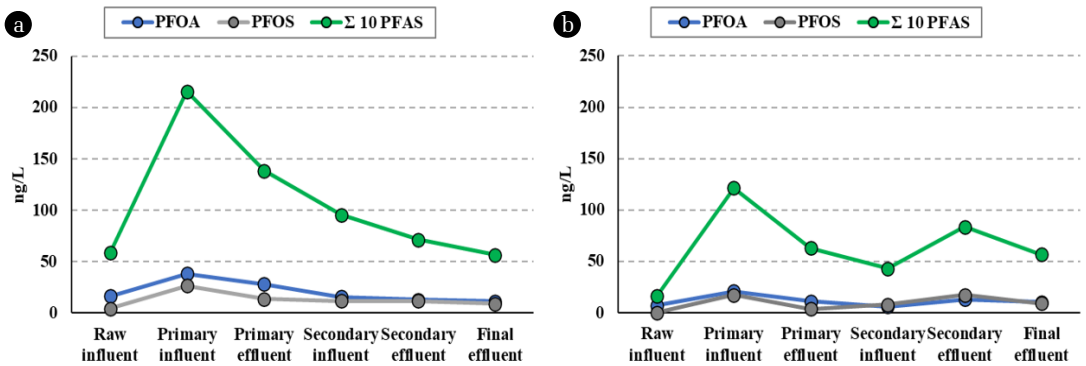


Fig. 3. Concentration changes of selected PFAS across treatment stages in industrial WWTP in Jeonju, South Korea during (a) summer (August) and (b) winter (December) (Data partially extracted and reorganized from Park et al., 2012) [65].

removed 12% and WWTP-15 removed 20% of the sum of 17 PFAS compounds. Most compounds showed positive removal efficiencies ranging from 7% to 13% in WWTP-14 and 3% to 28% in WWTP-15. PFOS was an exception, increasing from 9.15 ng/L to 10.4 ng/L in WWTP-14 while achieving 11% removal in WWTP-15. Since UV processes exhibit minimal standalone degradation efficiency of PFAS [61-63], the positive removal may result from UV activation by co-present reactive species, though this requires confirmation through controlled laboratory experiments.

Collectively, these findings indicate that conventional biological treatment processes are ineffective for PFAS degradation, with observed removal likely attributed to sorption onto sludge or microbial surfaces [64]. The negative removal values observed in some cases may result from desorption processes or precursor

Table 2. Average of PFAS concentrations in influent and effluent of different types of WWTPs in South Korea.

ID	Location	Influent type	Treatment process	Classification of WWTP	PFAS type	Influent (ng/L)	Effluent (ng/L)	Removal %[h]	Residual %[i]	Refs.
WWTP-9 (n=5)	—	Domestic	CAS[a] or A2O[b]	Conventional	PFOA	30	28	7	93	Kim et al. (2012) [55]
					PFOS	9	6.3	30	70	
					PFHxS	7.3	5.0	32	68	
					ΣPFAS (15)[d]	110	96	13	87	
WWTP-10 (n=5)	—	Industrial	CAS or A2O or BF[c]	Conventional	PFOA	62	62	0	100	Kim et al. (2012) [55]
					PFOS	110	72	35	65	
					PFHxS	98	69	30	70	
					ΣPFAS (15)	610	620	-2	102	
WWTP-11 (n=5)	—	Mixed domestic-industrial	CAS or A2O	Conventional	PFOA	550	1100	-100	200	Kim et al. (2012) [55]
					PFOS	89	110	-24	124	
					PFHxS	8.8	6.7	24	76	
					ΣPFAS (15)	1100	2100	-91	191	
WWTP-12[g] (n=1)	CITY-I	Domestic (99.6%)-industrial (0.4%)	Biological process (Denipho)	Conventional	PFHxA	7.9	9.0	-14	114	Joo et al. (2021) [56]
					PFOA	2.8	4.5	-61	161	
					PFNA	0.7	0.8	-14	114	
					PFHxS	24.1	29.2	-21	121	
					PFOS	0.8	0.2	75	25	
					ΣPFAS (5)[f]	36.3	43.7	-20	120	
WWTP-13[g] (n=1)	CITY-G	Domestic (92.5%)-industrial (7.5%)	A2O	Conventional	PFHxA	14.1	12.0	15	85	Joo et al. (2021) [56]
					PFOA	15.2	17.2	-13	113	
					PFNA	4.0	0.9	78	22	
					PFHxS	1.0	0.9	10	90	
					PFOS	3.7	0.9	76	24	
					ΣPFAS (5)	38.0	31.9	16	84	
WWTP-14 (n=1)	Nakdonggang river basin	Mixed domestic-industrial	Biological process & UV	AOP	PFOA	30.5	28.3	7	93	Kim et al. (2021) [54]
					PFOS	9.2	10.4	-13	113	
					PFHxS	5090	4450	13	87	
					ΣPFAS (17)[e]	5560	4870	12	88	
WWTP-15 (n=1)	Nakdonggang river basin	Mixed domestic-industrial	Biological process & UV	AOP	PFOA	7.8	7.6	3	97	Kim et al. (2021) [54]
					PFOS	142	127	11	89	
					PFHxS	37.3	26.7	28	72	
					ΣPFAS (17)	1300	1040	20	80	

[a]CAS: Conventional Activated Sludge
[b]A2O: Anaerobic-Anoxic-Oxic
[c]BF: Bio Filter
[d] PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, FOUEA, PFBS, PFHxS, PFOS, PFDS, EtFOSAA, MeFOSAA
[e]PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFBS, PFPeS, PFHxS, PFHpS, PFOS
[f]PFHxA, PFOA, PFNA, PFHxS, PFOS
[g]Effluent refers to secondary effluents
[h]Removal % = $\frac{(\text{Influent concentration}-\text{Effluent concentration})}{(\text{Influent concentration})}\times 100\%$, representing the percentage of PFAS removed during treatment processes relative to the influent concentration
[i]Residual % = $\left(\frac{\text{Effluent concentration}}{\text{Influent concentration}}\right)\times 100\%$, representing the percentage of PFAS remaining after treatment processes relative to the influent concentration

transformation during treatment. Therefore, detailed studies examining sorption-desorption mechanisms of individual PFAS compounds onto sludge or others, along with identification of favorable conditions for these processes, are essential to accurately understand PFAS fate in WWTPs.

Evaluating PFAS removal in WWTPs solely based on influent-effluent concentration differences may underestimate their actual behavior, as it overlooks precursor compounds and their potential transformation during treatment processes. To address these limitations, a unit-by-unit monitoring approach provides deeper insights into PFAS fate and transport mechanisms. Park et al. (2012) assessed PFAS concentration changes across multiple treatment stages in an industrial WWTP in Jeonju City, South Korea (Fig. 3) [65]. The researchers monitored 10 PFAS compounds—PFHxA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFBS, PFHxS, PFOS, and PFDS—throughout the treatment train, which included primary and secondary treatment followed by chlorination disinfection (Fig. 3). Sampling was conducted across two different seasons to assess seasonal variability in treatment performance.

Seasonal variation analysis revealed that raw influent contained higher total PFAS concentrations during summer compared to winter, which the authors attributed to rainfall during the wet season, though insufficient evidence supported this interpretation [65]. PFOA consistently dominated raw influent concentrations across both seasons, while PFCAs exhibited an inverse relationship between chain length and concentrations. PFSAs were present at substantially lower levels, with PFBS, PFHxS, and PFOS averaging 0.11 ng/L, 2.80 ng/L and 1.96 ng/L, respectively (Fig. S4) [65]. All PFAS concentrations increased substantially between raw influent and primary influent. Since grit removal and pH adjustment are unlikely to affect PFAS concentrations, this increase was attributed to return flows from sludge dewatering processes, where PFAS-containing liquors are recycled back to the influent stream [65]. During summer, total PFAS concentrations decreased slightly from 58.31 ng/L in the influent to 56.48 ng/L in the final effluent (3% reduction). Conversely, winter showed a marked increase from 16.51 ng/L in the influent to 56.61 ng/L in the final effluent—more than a 3-fold increase. Although Park et al. (2012) did not explain this winter-specific increase, it likely relates to temperature-dependent sorption-desorption behavior of PFAS in sludge matrices [65]. Temperature significantly influences PFAS adsorption onto adsorbents, as most PFAS adsorption processes are spontaneous and thermodynamically favorable [66, 67]. Additionally, Collins et al., (2024) demonstrated that elevated temperature increases PFAS leaching from municipal solid waste [68]. These seasonal variations in PFAS removal highlight the need for treatment technologies specifically designed to address PFAS persistence while maintaining consistent performance regardless of seasonal temperature fluctuation.

3.3. International Regulatory Trends for PFAS in Wastewater and Drinking Water

Globally, regulations on PFAS have primarily focused on drinking water, whereas clear, quantitative wastewater discharge standards are still absent. In the United States, the US EPA officially identified the need to develop new effluent limitation guidelines (ELGs) for major PFAS-discharging industries (e.g., metal finishing, textile manufacturing) in Effluent Guidelines Program Plan 15 [69, 70]. Additionally, a nationwide influent study is currently underway to assess PFAS concentrations in wastewater entering publicly owned treatment works (POTWs), with the aim of quantifying industrial contributions. However, most of these regulatory efforts remain in the preliminary phase, focused on planning, data collection, and building the scientific basis for future regulations. To date, there are no legally established discharge limits, industry-specific thresholds, or permitting conditions related to PFAS in wastewater. A similar situation is observed in the EU. While the EU is making progress—for instance, by promoting a broad policy to restrict PFAS use across industrial sectors and revising the Urban Wastewater Treatment Directive (2024) to include mandatory PFAS monitoring—quantitative wastewater discharge standards have not yet been established, highlighting that regulatory development in this area remains incomplete [71, 72]. Nevertheless, certain member states have begun setting PFAS-related emission limit values (ELVs): France has established 25 μg/L for PFOS in specific installations and Italy has prepared draft legislation covering several PFAS. These measures, however, are not yet harmonized at the EU level [73]. This global regulatory gap suggests that many countries are still in the early stages of developing science-based frameworks for PFAS control in wastewater and industrial effluents. These efforts typically involve source identification, exposure pathway assessment, and risk-based standard setting.

In contrast, regulatory developments for PFAS in drinking water have progressed more rapidly. Owing to its direct relevance to public health, drinking water has received higher policy priority, which has resulted in the establishment of legally enforceable standards following extensive health risk assessments and exposure analyses conducted over the past several years. For example, the US has actively developed legal frameworks for PFAS in drinking water based on years of scientific research [74]. The US EPA officially announced the NPDWR of PFAS in 2024, which established legally enforceable MCLs and non-enforceable Maximum Contaminant Level Goals (MCLGs) for five major PFAS compounds: PFOA, PFOS, PFHxS, PFNA, and HFPO-DA (Table 3) [2]. These regulatory values were derived from comprehensive human health risk assessments aimed at minimizing PFAS exposure through drinking water consumption [2]. Specifically, the MCLs for PFOA and PFOS were each set at 4 ng/L, while the MCLs for PFHxS, PFNA, and HFPO-DA were established at 10 ng/L, respectively. Moreover, the US EPA's PFAS regulatory framework incorporates the concept of dose-additive toxicity, allowing for the assessment of cumulative risk from co-exposure to multiple PFAS compounds. In addition to individual MCLs, the regulation introduces a cumulative risk metric known as the hazard index (HI), which is calculated as follows, and must remain below 1 to comply with regulatory limits (Eq. (1)) (Table 3):

$$\mathrm{HI} = \frac{\mathrm{PFHxS\,ng/L}}{10\,\mathrm{ng/L}} + \frac{\mathrm{PFNA\,ng/L}}{10\,\mathrm{ng/L}} + \frac{\mathrm{HFPO\text{-}DA\,ng/L}}{10\,\mathrm{ng/L}} + \frac{\mathrm{PFBS\,ng/L}}{2000\,\mathrm{ng/L}} < 1 \quad (1)$$

The US EPA has developed a regulatory framework that addresses both individual and mixture PFAS exposures, and incorporates risk assessments for both carcinogenic and non-carcinogenic effects. This comprehensive assessment framework serves as a model for establishing science-based regulatory limits. Building on this framework, the US EPA has continued to adjust its regulatory stance in response to emerging scientific evidence and policy needs. Recently, the US EPA announced plans to re-evaluate the regulatory validity of three compounds (i.e., PFHxS, PFNA, and HFPO-DA) by 2026, despite their inclusion in the 2024 NPDWR [75]. Furthermore, the agency is reviewing the possibility of postponing the implementation of regulations for PFOA and PFOS from the originally scheduled 2029 to 2031. These changes reflect the evolving nature of PFAS regulation in the US, shaped by the current state of best available treatment technologies, public health protection goals, and policy considerations. Therefore, continuous monitoring of regulatory changes and timely incorporation of updated scientific data may be important for adaptive and evidence-based policymaking.

The EU has also recognized the human health risks associated with PFAS and has formally initiated regulatory efforts for drinking water. Specifically, the revised Drinking Water Directive (DWD), adopted in 2020, introduced the first EU-wide legally binding drinking water standards for PFAS, which must be implemented by all member states by January 2026 [3]. The DWD outlines two types of regulatory limits (Table 3). First, a sum concentration limit of 100 ng/L has been established for 20 selected PFAS compounds, which include PFCA and PFSA with carbon chains ranging from C4 to C13. Second, total PFAS limit of 500 ng/L has been set. Total PFAS can be estimated using a total oxidizable precursor (TOP) assay [76].

In contrast to the US, the EU regulatory framework does not establish limits for individual PFAS compounds, but rather sets total and sum thresholds for groups of selected PFAS. Currently, no official documentation has been made publicly available regarding the scientific rationale behind the EU's sum concentration-based regulatory approach, and the process by which these limits were established remains unclear. As a result, some researchers have argued that the PFAS standards set by the EU DWD appear to have been determined primarily as a precautionary approach, without a fully transparent or detailed presentation of the underlying scientific risk assessment process [77]. By comparison, US EPA regulations are derived from rigorous evaluations that incorporate compound-specific toxicity values—such as reference doses (RfD) and cancer slope factors (CSF)—together with detailed exposure assessments. This contrast has led to growing concern that the EU standards should also be refined through a more comprehensive and transparent human health risk assessment framework [77].

While the US and the EU are leading the establishment of international standards for PFAS regulation, other countries are also making continuous efforts to develop their own national drinking water standards by reflecting domestic environmental conditions and technological capabilities. These countries tend to either directly adopt the scientific criteria and policy directions set by the US and EU, or modify and supplement them to fit their local context. Some nations have established legally binding regulatory limits, whereas others manage PFAS using provisional target values that do not carry legal enforceability.

Germany has adopted the EU standards but has additionally established a separate cumulative concentration limit of 20 ng/L for four PFAS compounds: PFOA, PFOS, PFHxS, and PFNA [78]. This national standard is expected to become legally binding starting in 2028. Sweden has also implemented more stringent national standards than those set by the EU. Specifically, Sweden introduced a sum concentration limit of 100 ng/L for a group of 21 PFAS compounds, which includes the 20 PFAS (i.e., C4-C13 PFCAs and PFSAs) defined by the EU, with the addition of 6:2 fluorotelomer sulfonate (6:2 FTS) [79, 80]. Furthermore, Sweden has established an exceptionally strict cumulative limit of 4 ng/L for the four key PFAS compounds: PFOA, PFOS, PFHxS, and PFNA.

Australia and Japan have primarily adopted regulatory approaches based on individual PFAS concentration. In Australia, guideline values have been established at 560 ng/L for PFOA and 70 ng/L for both PFOS and PFHxS [4]; however, these values have faced criticism for being overly lenient. In response, the Australian government has announced plans to release revised guidelines in 2025, with a current draft indicating a proposed reduction of the PFOA limit to 200 ng/L [4]. In Japan, provisional target values of 50 ng/L have been set for both PFOA and PFOS. These provisional values are expected to be upgraded to legally enforceable standards starting in 2026 [81].

South Korea currently manages PFAS under a drinking water monitoring rule, which imposes monitoring and reporting obligations on water utilities and requires compliance with values of 70 ng/L for PFOA and PFOS individually, 480 ng/L for PFHxS, and 70 ng/L for the combined PFOA and PFOS concentrations [29]. These values appear to have been established with reference to the 2016 U.S. EPA Lifetime Health Advisory (for PFOA and PFOS) and a reference dose approach informed by the Korean Ministry of Food and Drug Safety (for PFHxS). While this monitoring rule provides a degree of enforceability, it differs from legally enforceable drinking water standards, which have broader applicability (e.g., bottled water) and stricter enforcement. Thus, South Korea has not yet developed comprehensive PFAS-specific drinking water standards comparable to recent international regulatory trend.

To demonstrate the effectiveness of PFAS regulations, WWTP data from Denmark before and after regulatory implementation was compared. Denmark established stringent drinking water limits in 2021, requiring total sum of PFOA, PFOS, PFNA and PFHxS concentrations not exceed 2 ng/L—among the world's strictest standards [82]. Six municipal WWTPs in 2008 (pre-regulation) showed influent concentrations of < MDL to 19.9 ng/L (PFOA), 2.3-6.8 ng/L (PFOS), and 0.6-16.9 ng/L (PFHxS) (Table 4) [53]. Fourteen municipal WWTPs in 2021 (post-regulation implementation) exhibited reduced influent concentrations: 1.1-4.9 ng/L (PFOA), 1.2-6.4 ng/L (PFOS), and 0.2-1.7 ng/L (PFHxS) (Table 4) [83]. Although direct comparison is limited due to different sampling locations, these findings provide compelling evidence for the effectiveness of PFAS regulations in reducing source inputs of WWTPs. The lower influent concentrations suggest that drinking water regulations successfully limited PFAS usage across various applications. Denmark's experience validates the urgent need for

South Korea to develop comprehensive, legally enforceable PFAS standards aligned with international trends, moving beyond voluntary monitoring guidelines to systematic regulatory frameworks that protect public health.

4. Conclusion

This first systematic nationwide review of PFAS pollution in South Korean water systems offers an in-depth evaluation of PFAS

Table 3. Comparison of PFAS regulations in drinking water across selected countries.

Nation	Individual PFAS Limit	Cumulative PFAS Limit	Year issued	Enforceability	Reference
United States	PFOA < 4 ng/L PFOS < 4 ng/L PFHxS < 10 ng/L PFNA < 10 ng/L HFPO-DA < 10 ng/L	HI < 1 [a]	2024	O	[2]
European Union	N.A.[b]	Σ20 PFAS < 100 ng/L[c] Total PFAS < 500 ng/L[d]	2020	O	[3]
Germany	N.A.[b]	Σ4 PFAS < 20 ng/L[e]	2023	O	[77]
Sweden	N.A.[b]	Σ4 PFAS < 4 ng/L[e] Σ21 PFAS < 100 ng/L[f]	2023	O	[78, 79]
Australia	PFOA < 560 ng/L PFOS < 70 ng/L PFHxS < 70 ng/L	N.A.[b]	2020	X (Health-based guideline)	[4]
Japan	PFOA < 50 ng/L PFOS < 50 ng/L	N.A.[b]	2020	X (Provisional guideline)	[80]
South Korea	PFOA < 70 ng/L PFOS < 70 ng/L PFHxS < 480 ng/L	Sum of 2 PFAS < 70 ng/L[g]	2018	X (Monitoring guideline)	[29]

[a] Hazard index (HI)
[b]Not applicable
[c]Sum of 20 PFAS, including perfluoroalkyl carboxylic acids (PFCAs) and sulfonic acids (PFSAs) with carbon chain lengths from C4 to C13
[d]Total PFAS quantified through TOP assay (Total Oxidizable Precursor assay)
[e]Sum of 4 PFAS, including PFOA, PFOS, PFHxS, PFNA
[f]Sum of 21 PFAS, including perfluoroalkyl carboxylic acids (PFCAs) and sulfonic acids (PFSAs) with carbon chain lengths from C4 to C13, and 6:2 fluorotelomer sulfonic acid (6:2 FTS)
[g]Sum of 2 PFAS, including PFOA and PFOS

Table 4. Selected PFAS concentrations in influent and effluent of municipal WWTPs located in Denmark. (a) collected from 2008 (Data partially extracted and reorganized from Bossi et al., 2008) [53] and (b) collected from 2021 (Ministry of Environment of Denmark, 2024) [82].

(a)

	PFOA (ng/L)		PFOS (ng/L)		PFHxS	
	Influent	Effluent	Influent	Effluent	Influent	Effluent
WWTP-16	19.9	13.2	6.8	6.1	16.9	0.7
WWTP-17	< MDL[a]	15.2	2.4	12.8	14.3	1.0
WWTP-18	15.2	17.6	5.2	6.4	2.5	0.8
WWTP-19	11.7	16.3	7.1	7.9	3.7	0.6
WWTP-20	5.5	14.9	2.3	6.7	6.7	1.4
WWTP-21	18.6	< MDL[a]	3.3	< MDL[b]	0.6	< MDL[c]

[a]Method detection limit (MDL) of PFOA: 2.0 ng/L
[b]MDL of PFOS: 1.5 ng/L
[c]MDL of PFHxS: 0.2 ng/L

(b)

	PFOA (ng/L)		PFOS (ng/L)		PFHxS	
	Influent	Effluent	Influent	Effluent	Influent	Effluent
WWTP-22 (n=14)	3.1[d] (1.1-4.9)[e]	4.5 (2-14.9)	2.5 (1.2-6.4)	3.0 (0.5-21.0)	1.7 (0.2-5.0)	0.8 (0.2-1.7)

[d]Average of 14 WWTP data
[e]Range of 14 WWTP data

prevalence in surface waters and WWTPs, synthesizing existing research to establish essential baseline data for evidence-based policy development. The evidence reveals that PFAS pollution has become a pervasive environmental issue in South Korea, with industrial point sources serving as potential contributors to aquatic contamination. Conventional WWTPs demonstrate ineffective PFAS removal or even increase PFAS concentrations due to precursor transformation or desorption from sludge or biological materials. This emphasizes the urgent need for monitoring precursors alongside targeted PFAS compounds and to enhance treatment technologies, such as surface-modified activated carbon, to handle concentration fluctuations regardless of seasonal variation, as well as self-tuning systems that can adapt to different PFAS levels. Furthermore, international cases have shown that PFAS concentrations in WWTPs can vary significantly depending on whether regulatory measures are in place. This highlights that policy intervention can play a key role in effectively reducing PFAS contamination in water systems. In particular, to develop concrete and enforceable regulations for PFAS, a stepwise research approach is required. This should include regular nationwide monitoring of specific PFAS, the accumulation of updated datasets, and analysis of their occurrence patterns. Based on long-term data trends, guideline can then be established to support the formulation of legally enforceable regulations. Such an approach will minimize confusion among researchers and policymakers, promote consistent conclusions, and facilitate effective regulatory implementation. To address the growing risks of PFAS pollution, it is essential to prioritize the development of risk assessment frameworks, nationwide monitoring programs, and improved advanced treatment technologies that reflect local conditions. These efforts will help establish a foundation for science-based regulations and sustainable water quality management.

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Conflict of Interest

The authors declare that they have no conflict of interest.

Author Contributions

W.L. (Assistant Professor) conducted the literature review and drafted the manuscript; A.L. (Post-doctoral researcher) conducted the literature review; Y.C. (Professor) reviewed and revised the manuscript; Jin Chul Joo (Professor) revised the manuscript; S.P. (Professor, Corresponding author) conducted the literature review, drafted the manuscript, and supervised the overall research.

Data availability

Data will be made available on request.

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