SEDIMENT–SEAWATER PARTITIONING OF PER- AND POLYFLUOROALKYL SUBSTANCES IN VIETNAMESE COASTAL WATERS: ROLES OF CHAIN LENGTH AND HYDROPHOBICITY

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Pham Thi Phuong Thao*, Ninh Thi Thu, Pham Thi Tra, Nguyen Tien Dat[1], Bui Quang Minh

Center for High Technology Research and Development, Vietnam Academy of Science and Technology

**Email: thaopham284@gmail.com*

TÓM TẮT

SỰ PHÂN BỐ CÁC CHẤT PER- VÀ POLYFLUOROALKYL GIỮA TRẦM TÍCH–NƯỚC BIỂN Ở VÙNG VEN BIỂN VIỆT NAM: VAI TRÒ CỦA CHIỀU DÀI MẠCH VÀ TÍNH KỴ NƯỚC

Các chất per- và polyfluoroalkyl (PFAS) đã được phân tích trong nước biển và trầm tích thu thập từ 20 địa điểm dọc theo bờ biển Việt Nam bằng tàu Viện sĩ Oparin. Nồng độ PFAS thay đổi trong khoảng từ 0,97 đến 10,55 ng/L (trung bình: 5,37 ng/L) trong nước biển và từ 0,195 đến 2,165 ng/g TLK (trung bình: 0,755 ng/g TLK) trong trầm tích. Các PFCA chuỗi ngắn (C4–C7, như PFBA, PFPeA, PFHxA) phổ biến trong nước biển, trong khi các PFCA chuỗi dài (≥ C8 như PFOA, PFNA, PFDA, PFUnDA) có xu hướng tích tụ trong trầm tích. Hệ số phân bố trầm tích-nước biển (log K_{OC}) dao động trong khoảng 3,2–4,8, cho thấy mối tương quan đáng kể với chiều dài chuỗi PFAS ($R = 0,94$, $p < 0,05$), và thể hiện xu hướng các PFAS chuỗi dài được ưu tiên hấp phụ vào trầm tích. Kết quả cho thấy ô nhiễm PFAS đáng kể ở vùng nước ven biển Việt Nam, có thể là do chất thải công nghiệp và đô thị. Những phát hiện này nhấn mạnh sự cần thiết phải tăng cường giám sát và quản lý để giải quyết tình trạng ô nhiễm PFAS trong môi trường biển nhiệt đới.

Từ khóa: *PFASs, phân chia trầm tích-nước biển, Akademik Oparin, ô nhiễm biển*

1. INTRODUCTION

Per- and polyfluoroalkyl substances (PFASs) are a category of synthetic organic pollutants widely used in industrial and consumer products owing to their chemical stability and hydrophobic properties [1]. Nonetheless, their stability, potential for bioaccumulation, and capacity for long-range transport have elicited worldwide apprehension regarding their environmental effects and toxicity [2]. PFASs have been identified in various matrices, such as water, sediment, air, and biota, underscoring their prevalence and intricate behavior in aquatic ecosystems, particularly marine ecosystems [3, 4]. The distribution of PFASs between seawater and sediment is crucial for understanding their transport, fate, and bioavailability. Their distribution is contingent upon physicochemical features, including carbon chain length and functional groups, as well as environmental parameters such as total organic carbon (TOC) and sediment grain size [5, 6]. Research indicates that long-chain PFASs concentrate in sediments, whereas short-chain PFASs persist in the aqueous phase, requiring more examination to assess potential ecological risks [7]. Despite the growing body of research on PFASs,

investigations in Vietnamese coastal environments are limited, especially concerning their partitioning between seawater and sediment. Due to Vietnam's industrialization, urbanization, and aquaculture practices, coastal regions are vulnerable to PFAS pollution via industrial discharges, municipal wastewater, and runoff [4, 8]. Previous studies have indicated substantial PFAS contamination in the East China Sea and Vietnamese coastal waters, underscoring the necessity for further investigation in Southeast Asian coastal areas [9]. Comprehending the prevalence and distribution of PFASs in Vietnamese coastal sediments and saltwater is crucial for assessing pollution levels and formulating appropriate management methods. This study, utilizing bottom seawater and sediment samples from the research vessel 'Akademik Oparin', aims to (i) measure the concentration of PFASs in these samples and (ii) evaluate the prevalence and partitioning behavior of PFASs in Vietnamese coastal waters. The results will improve comprehension of PFAS pollution in tropical marine ecosystems and enhance environmental conservation initiatives.

2. EXPERIMENT

2.1. Chemicals and reagents

The 14 PFASs (Table S1) are categorized into three distinct groups. For brevity, abbreviations are used throughout and full chemical names and physicochemical properties are provided in Table S1. For example, PFBA (perfluorobutanoic acid), PFHxS (perfluorohexanesulfonic acid), and PFOS (perfluorooctanesulfonic acid). The first group consists of short-chain PFCA (SC-PFCA), which includes PFHxA, PFPeA, and PFBA. The second group is long-chain PFCA (LC-PFCA), comprising PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFTeDA, and PFHxDA. Lastly, the PFSA group includes PFOS and PFHxS. All analytical standards were acquired from LGC Standards (purities ≥ of 97%, Germany). The internal standards (IS) for PFASs (Table S1) were acquired from Wellington Laboratories Inc. (purities ≥ 98%, ON, Canada). Operating conditions for ultra-high-performance liquid chromatography linked with high-resolution mass spectrometry were shown in Table S2. Ultra-pure water (UPW, 18.2 MΩ/cm) was acquired using a Milli-Q-Integral system from Merck Millipore (Burlington, MA, USA). Methanol (MeOH), acetonitrile (MeCN), acetic acid (AA), $MgSO_4$, and NaCl were obtained from Sigma-Aldrich (St. Louis, MO, USA). Oasis Hydrophilic-Lipophilic (HLB, 500 mg/6 mL) and Supelclean™ ENVI-Carb (100 mg/3 mL) were acquired from Waters (Milford, MA, USA) and Supelco (Bellefonte, PA, USA).

2.2. Sample collection

In May and June 2023, seawater and sediment samples were collected from the coastal regions of South Central and Southern Vietnam, specifically from Nha Trang to Ba Ria–Vung Tau. Figure 1 illustrates the regions designated for sampling on the map (20 locations). Bottom seawater samples were obtained 1 m above the seafloor, while sediment samples were gathered by utilizing a sediment grabber. All sample collection equipment consisted of stainless steel and underwent pre-cleaning with UPW and MeOH. All samples were preserved in glass bottles and vials, maintained at a temperature of 4°C. In the laboratory, seawater samples were filtered using glass microfiber filters (Whatman, diameter 47 mm). Wet sediment was freeze-dried, homogenized, and sieved through a 40-

mesh screen. The dried sediments were then covered with aluminum foil and stored at -18°C until extraction.

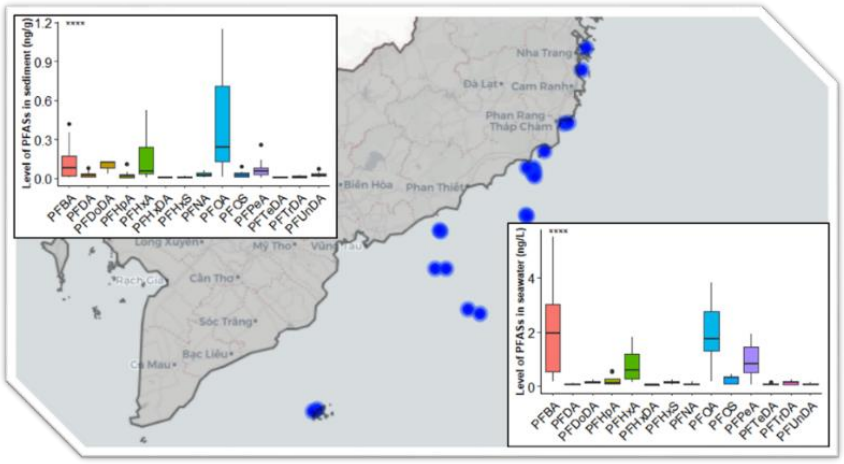

Figure 1. *Sampling sites along the Vietnamese coast. Basemap: Natural Earth, 1:10 m land and coastline. Data CRS: WGS84 (EPSG:4326); map plotted in R (sf, ggplot2, ggspatial) after projecting to a Vietnam Albers Equal-Area Conic*

2.3. Sample preparation

Seawater. In brief, 500 mL of seawater, enhanced with 50 µL of IS (1 µg/mL), was gently shaken and allowed to equilibrate for 30 min. This solution was then loaded at a rate of 2 mL/min through the Oasis WAX cartridge, which had been preconditioned with 6 mL of MeOH and 6 mL of UPW. A total of 3 mL of 5% MeOH in UPW was utilized as the elution solvent to remove impurities from the cartridge, followed by the drying. Subsequently, 3 mL of MeOH was utilized as the elution solvent and applied to the column. The eluate was evaporated to dryness using a nitrogen stream at 1°C and was then redissolved in 1 mL of MeOH:UPW (1/1, v/v). The final extract was filtered through a 0.22 µm PTFE membrane prior to analysis.

Sediment. In brief, 50 µL of IS (1 µg/mL) was added to 5 g of sediment samples, which were held in a 50-mL tube. Then, 8 mL of 1% AA in UPW and 10 mL of MeCN were introduced, followed by vigorous shaking by hand, sonication for 5 min, and vortexing for 2 min. Following this, 4 g of $MgSO_4$ and 1 g of NaCl were added into the tube, shaken vigorously to prevent clumping, vortexed for 2 min, and then centrifuged at 8,000 rpm for 5 min. Subsequently, 5 mL of the supernatant was reduced to 2 mL using N_2 gas at a temperature of 1°C. Following concentration, the extracts were purified using a Supelclean™ ENVI-Carb cartridge that was preconditioned with MeOH. The SPE cartridge was eluted with 3 × 1 mL of MeOH, followed by concentration of the eluents to a final volume of 0.5 mL. The final extract was diluted to 1 mL with UPW and filtered through a 0.22 µm PTFE membrane prior to analysis. The concentrations of PFASs in the sediment were determined based on a dry weight (dw) basis. To ascertain the total organic carbon (TOC) in sediment samples, the removal of inorganic carbon is essential. The determination of TOC was conducted utilizing a TOC analyzer (Multi N/C 2100, Analytik Jena AG, Jena, Germany). The moisture content of the sediment was assessed through the method of oven drying.

2.4. QA/QC

The results of the method validation are found in Table S3. Calibration for each PFAS covered 0.1–50 ng/mL with excellent linearity ($R^2 > 0.995$). The limits of detections (LODs) were established as the concentrations of peaks exhibiting a signal-to-noise ratio of 3, with the exception of PFOS, which was calculated by taking the mean concentration in the procedure blank and adding three times the standard deviation of the quantified concentrations. The limit of quantification (LOQ) was determined using the formula LOQ = 10 × SD. The LOD for PFASs in water and sediment sample was found to be between 0.01 and 0.05 ng/L and 0.001 and 0.005 ng/g, respectively. The spiked samples at three concentrations (0.1 ng/L, 1.0 ng/L, and 10 ng/L) were replicated six times for each concentration, conducted over three consecutive days. Repeatability (within-day) and within-laboratory reproducibility (across days) were < 15% RSD for all analytes; typical RSDs were 2.1–9.0% (seawater) and 2.1–8.5% (sediment). Recoveries by compound and matrix are reported in Table S3.

2.5. Statistical analysis

The analysis of PFASs was performed utilizing R software for statistical analysis. Maps were produced in R using sf, ggplot2, ggspatial, and rnaturalearth (Natural Earth 1:10 m). While geospatial data are stored in WGS84 (EPSG:4326), we projected all layers to a Vietnam Albers Equal-Area Conic before adding the scale bar (ggspatial::annotation_scale), ensuring metric accuracy.

The Shapiro-Wilk test ($p > 0.05$) was conducted to assess the normality of PFAS concentration. The Kruskal-Wallis H test was utilized to evaluate the variations in concentrations of PFASs. The partition behaviors of PFASs between seawater and sediment were evaluated by calculating the sediment-seawater partition coefficients (KOC, L/kg dw) of PFASs. Following the adjustment for the TOC content in the sediment, the partition coefficient is determined as outlined below:

$$K_{OC}\frac{1000 \times C_{Sediment} \times 100}{C_{Water} \times f_{oc}}$$

Where $C_{Sediment}$ is the concentration of PFASs in sediment (ng/g dw), C_{Water} is the concentration of PFASs in seawater (ng/L), and f_{oc} is the percentage of TOC in sediments. The f_{oc} of sediment from this study was 0.37–0.89% (mean of 0.63%).

In addition to regressions versus per-fluorinated carbon chain length, we assessed the relationship between compound-specific mean $\log K_{OC}$ (this study) and literature $\log K_{OW}$ values (Table S1) using Pearson's r and Spearman's ρ. Analyses were restricted to PFAS with detection frequency ≥ 40% in both matrices and with available $\log K_{OC}$, and were also repeated for PFCAs only. Statistical significance was set at $p < 0.05$.

3. RESULTS AND DISCUSSION

3.1. PFAS in seawater

All seawater samples contained PFASs, with ΣPFAS = 0.97–10.55 ng/L (mean 5.37 ng/L) (Table 1). Concentrations were lower than those in the Bohai Sea (16.9–118.0 ng/L) and Yangtze River (13.0–77.4 ng/L), but higher than the Baltic Sea (1.6–5.2 ng/L), Norwegian Sea (0.14–0.85 ng/L; 140–852.6 pg/L), and the East Sea coastal region (0.12–1.02 ng/L; 120.6–1015.1 pg/L) [10-14]. Although source apportionment was not conducted, the mid-to-high ΣPFAS observed along our South-Central to Southern transect

(Nha Trang to Ba Ria–Vung Tau; Figure 1) are consistent with settings where PFAS-relevant activities occur - e.g., textile/leather finishing, paper and packaging coatings, chromium electroplating that historically used PFOS-based mist suppressants, and the storage/use of AFFF [15, 16]. Similar associations between industrialized coasts and elevated PFAS have been reported globally, including in the Bohai Bay and Vietnamese coastal regions [9, 10, 14]. PFOA (DF = 90%), PFBA (DF = 80%), and PFNA (DF = 80%) were the most prevalent PFAS, with concentration ranges of 0.19-3.83 ng/L for PFOA, 0.20-5.50 ng/L for PFBA, and 0.04-0.18 ng/L for PFNA. Zhu *et al.* 2024 reported that PFOA (1.6-19 ng/L) and PFNA (1.7-14 ng/L) were the most abundant PFAS in East China Sea seawater [17]. In this study, PFBA and PFOA levels in seawater exceeded those found in Antarctica (PFBA: 102±26 pg/L; PFOA: 158±79 pg/L) and the Antarctic Ocean (PFOA: 15 pg/L) [18, 19]. All seawater samples contained LC-PFCA compounds, with concentrations ranging from 0.41 to 4.12 ng/L, primarily due to PFOA. SC-PFCA had the highest average concentration in seawater samples at 3.12 ng/L, followed by LC-PFCA at 2.12 ng/L and PFSA at 0.31 ng/L. PFBA was the primary contributor to the SC-PFCA group, whereas PFOS was the main contributor to the PFSA group. SC-PFCA comprised a significant portion of seawater samples (60–100%), while LC-PFCA represented only 0–20%, predominantly less than 10% (Figure S1a). PFSA represented a minimal share of the PFAS contamination profile in the analyzed seawater samples. The increased SC-PFCAs detected in seawater likely result from their high solubility in water and extensive use in industrial production [15].

Table 1. *Concentrations of PFASs in seawater, sediment, and log_{10}-transformed sediment-seawater partition coefficients from Vietnamese coastal areas.*

Compound	Seawater (ng/L)			Sediment (ng/g)			Log K_{OC}	
	DF (%)	Range	Mean	DF (%)	Range	Mean	Range	Mean
PFBA	80	0.20 - 5.50	2.15	95	0.002 - 0.416	0.120	3.2 - 4.2	3.9
PFPeA	75	0.06 - 1.94	0.95	80	0.002 - 0.255	0.069	3.5 - 4.3	3.9
PFHxA	70	0.15 - 1.80	0.76	75	0.004 - 0.521	0.137	3.7 - 4.5	4.3
PFHpA	40	0.07 - 0.57	0.23	50	0.003 - 0.110	0.027	3.6 - 4.4	4.2
PFOA	90	0.19 - 3.83	1.98	90	0.012 - 1.147	0.402	3.8 - 4.8	4.3
PFNA	80	0.04 - 0.18	0.09	85	0.002 - 0.058	0.029	4.2 - 4.8	4.6
PFDA	65	0.04 - 0.14	0.08	70	0.004 - 0.078	0.026	4.2 - 4.8	4.6
PFUnDA	35	0.05 - 0.16	0.09	40	0.006 - 0.073	0.030	NC	
PFDoDA	15	0.09 - 0.25	0.16	15	0.035 - 0.127	0.095	NC	
PFTrDA	30	0.03 - 0.25	0.13	30	0.003 - 0.025	0.011	NC	
PFTeDA	20	0.05 - 0.16	0.09	20	0.002 - 0.009	0.006	NC	
PFHxDA	15	0.03 - 0.06	0.05	15	0.002 - 0.006	0.004	NC	
PFHxS	65	0.08 - 0.26	0.15	70	0.004 - 0.016	0.008	3.6 – 4.0	3.8
PFOS	75	0.06 - 0.45	0.26	85	0.007 - 0.092	0.033	3.7 - 4.5	4.2
SC-PFCA	95	0.15 - 8.14	3.12	100	0.002 - 0.949	0.272		
LC-PFCA	100	0.41 - 4.12	2.12	100	0.075 - 1.215	0.450		
PFSA	95	0.08 - 0.60	0.31	95	0.010 - 0.092	0.035		
ΣPFSA	100	0.97 - 10.55	5.37	100	0.195 - 2.165	0.755		

DF: Detection frequency (%)

NC: not calculated because DFs in sediment or DFs in seawater are less than 40%.

3.2. PFASs in sediment

In this investigation, four out of fourteen PFAS compounds exhibited DFs < 40% in the sediments, specifically PFDoDA, PFTrDA, PFTeDA, and PFHxDA. ΣPFAS concentrations in sediment samples varied from 0.195 ng/g to 2.165 ng/g, with a mean of 0.755 ng/g. This result was lower than the ΣPFAS concentrations in sediments from the Bohai Sea (0.70-4.13 ng/g dw) and the Yellow Sea (0.21-4.74 ng/g dw), but comparable to the values in the Baltic and North Sea (0.018-2.6 ng/g dw) [12, 20, 21]. The predominant ΣPFAS content in sediments was PFOA (DF = 90%, range 0.012-1.147 ng/g dw), succeeded by PFHxA (DF = 75%, range 0.004-0.521 ng/g dw) and PFBA (DF = 95%, range 0.002-0.416 ng/g dw). The values obtained exceeded those found in sediment from the Yangtze River (PFBA: mean of 0.53 ng/g dw, PFOA: mean of 0.33 ng/g dw), but they were inferior to those recorded in Antarctica (PFBA: mean of 1.24 ng/g dw). The mean concentrations of PFAS groups diminished in the sequence of LC-PFCA (0.450 ng/g dw) > SC-PFCA (0.272 ng/g dw) > PFSA (0.035 ng/g dw). The results indicate that the LC-PFCA exhibited a high concentration in sediment, consistent with prior research findings [16, 22]. The hydrophobic LC-PFCA, characterized by elongated carbon chains, exhibited a tendency to preferentially adhere to silt particles [23]. In comparison to seawater, LC-PFCA constituted a significant concentration within the pollution profile (Figure S1b). The sample distribution predominantly occurred in the LC-PFCA region (40-100%), whereas SC-PFCA varied between 0-60%. Concurrently, all sediment samples had a PFSA distribution ratio below 20%. The restricted occurrence of PFSA in sediment samples is attributable to the gradual reduction of PFSA and international regulations [24]. Simultaneously, SC-PFCA progressively supplanted LC-PFCA in global industrial production sectors, resulting in an analogous contribution ratio for this group compared to LC-PFCA [12].

3.3. Sediment-seawater partition coefficients of PFASs

Sediment–seawater partition coefficients ($\log KOC$) for PFAS with $DF \geq 40\%$ ranged from 3.2 to 4.8 (Table 1). Across PFCAs, mean $\log KOC$ increased linearly with per-fluorinated chain length – from 3.9 (PFBA) to 4.6 (PFDA) – (Figure 2; $R = 0.94$, $p < 0.05$). These values are comparable to those in the Yangtze River (3.51–5.78) but exceed those reported for turbid Chinese coastal waters and other open/coastal settings [17, 25]. This discrepancy is likely attributable to seawater depth, salinity, temperature, and sediment characteristics [26]. The trend indicates stronger sediment affinity for long-chain PFAS; an exception is PFBA, whose $\log K_{OC}$ is similar to PFPeA. This atypical behavior likely reflects ion-specific electrostatic interactions and reduced steric hindrance, rather than hydrophobicity alone [10, 27]. Overall, molecular structure (chain length, functional group) together with environmental conditions (salinity, temperature, TOC/mineralogy) modulate PFAS partitioning and persistence [26].

Beyond the strong increase of $\log K_{OC}$ with per-fluorinated chain length ($R = 0.94$), mean $\log K_{OC}$ also showed a significant positive association with literature $\log K_{OW}$. Considering compounds with sufficient data (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFHxS, PFOS; $n = 9$), Pearson's

correlation was r = 0.826 ($p < 0.05$) and a linear fit yielded a slope of 0.18 (95% CI: 0.07–0.29) log K_{OC} units per unit log KOW (Table S4). When restricted to PFCAs (n = 7), the correlation remained significant (r = 0.860, $p < 0.05$) with a slope of 0.17 (95% CI: 0.05–0.28). These patterns indicate that hydrophobicity contributes to sediment affinity; however, the weaker K_{OC}–K_{OW} relationship relative to chain length and the behavior of short-chain PFCAs (e.g., PFBA) suggest additional controls such as electrostatic interactions, salinity-dependent speciation, and sediment organic matter/mineralogy [26, 27]. We emphasize that the literature log K_{OW} values refer to neutral species, whereas marine PFASs are predominantly anionic; hence, K_{OW} is an informative but imperfect proxy for environmental partitioning.

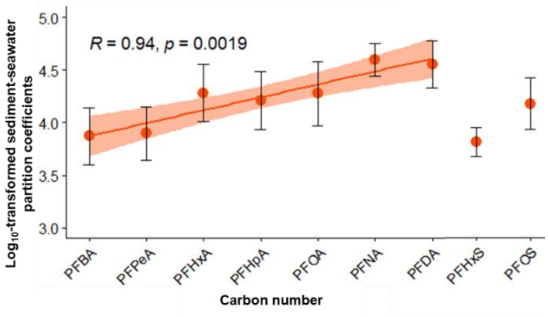

Figure 2: *Log_{10}-transformed sediment-seawater partition coefficients (log K_{OC}) of PFASs in Vietnamese coastal areas. The red line means a 95% confidence level.*

4. CONCLUSION

PFASs were ubiquitous along the Vietnamese coast, with seawater ΣPFAS = 0.97–10.55 ng/L (mean 5.37 ng/L) and sediment = 0.195–2.165 ng/g dw (mean 0.755 ng/g dw). Short-chain PFCAs (e.g., PFBA, PFPeA, PFHxA) predominated in seawater, whereas long-chain PFCAs (e.g., PFOA, PFNA, PFDA) were enriched in sediments. Sediment–seawater partition coefficients (log K_{OC} = 3.2–4.8) increased strongly with per-fluorinated chain length (R = 0.94, $p < 0.05$), indicating greater sediment affinity for long-chain homologues and a tendency for short-chain species to remain in the aqueous phase. These findings underscore the need for continued monitoring and risk-based management, focusing on SC-PFCA inputs to seawater and LC-PFCA accumulation in sediments in Vietnamese coastal environments.

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REFERENCES

[1]. Buck, RC, Franklin, J, Berger, U, Conder, JM, Cousins, IT, de Voogt, P, Jensen, AA, Kannan, K, Mabury, SA, and van Leeuwen, SPJ, (2011). Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. *Integrated Environmental Assessment and Management*, **7**, 513-541.

[2]. Mu, H, Li, J, Chen, L, Hu, H, Wang, J, Gu, C, Zhang, X-x, Ren, H-q, and Wu, B, (2022). Distribution, source and ecological risk of per- and polyfluoroalkyl substances in Chinese municipal wastewater treatment plants. *Environment International*, **167**, 107447.

[3]. Yun, X, Lewis, AJ, Stevens-King, G, Sales, CM, Spooner, DE, Kurz, MJ, Suri, R, and McKenzie, ER, (2023). Bioaccumulation of per- and polyfluoroalkyl substances by freshwater benthic macroinvertebrates: Impact of species and sediment organic carbon content. *Science of The Total Environment*, **866**, 161208.

[4]. Xiao, S-K, Wu, Q, Pan, C-G, Yin, C, Wang, Y-H, and Yu, K-F, (2021).

Distribution, partitioning behavior and potential source of legacy and alternative per- and polyfluoroalkyl substances (PFASs) in water and sediments from a subtropical Gulf, South China Sea. *Environmental Research*, **201**, 111485.

[5]. Yang, L, Zhu, L, and Liu, Z, (2011). Occurrence and partition of perfluorinated compounds in water and sediment from Liao River and Taihu Lake, China. *Chemosphere*, **83**, 806-814.

[6]. Yin, C, Pan, C-G, Xiao, S-K, Wu, Q, Tan, H-M, and Yu, K, (2022). Insights into the effects of salinity on the sorption and desorption of legacy and emerging per-and polyfluoroalkyl substances (PFASs) on marine sediments. *Environmental Pollution*, **300**, 118957.

[7]. Pan, C-G, Wang, Y-H, Yu, K-F, Zhang, W, Zhang, J, and Guo, J, (2020). Occurrence and distribution of perfluoroalkyl substances in surface riverine and coastal sediments from the Beibu Gulf, south China. *Marine Pollution Bulletin*, **150**, 110706.

[8]. Dan, SF, Lan, W, Yang, B, Han, L, Xu, C, Lu, D, Kang, Z, Huang, H, and Ning, Z, (2020). Bulk sedimentary phosphorus in relation to organic carbon, sediment textural properties and hydrodynamics in the northern Beibu Gulf, South China Sea. *Marine Pollution Bulletin*, **155**, 111176.

[9]. Muir, D and Miaz, LT, (2021). Spatial and Temporal Trends of Perfluoroalkyl Substances in Global Ocean and Coastal Waters. *Environmental Science & Technology*, **55**, 9527-9537.

[10]. Lin, K, Han, T, Wang, R, Tan, L, Yang, X, Zhao, T, Chen, Y, Wan, M, and Wang, J, (2022). Spatiotemporal distribution, ecological risk assessment and source analysis of legacy and emerging Per- and Polyfluoroalkyl Substances in the Bohai Bay, China. *Chemosphere*, **300**, 134378.

[11]. Leng, Y, Xiao, H, Li, Z, Liu, Y, Huang, K, and Wang, J, (2021). Occurrence and ecotoxicological risk assessment of perfluoroalkyl substances in water of lakes along the middle reach of Yangtze River, China. *Science of The Total Environment*, **788**, 147765.

[12]. Joerss, H, Apel, C, and Ebinghaus, R, (2019). Emerging per- and polyfluoroalkyl substances (PFASs) in surface water and sediment of the North and Baltic Seas. *Science of The Total Environment*, **686**, 360-369.

[13]. Joerss, H, Xie, Z, Wagner, CC, von Appen, W-J, Sunderland, EM, and Ebinghaus, R, (2020). Transport of Legacy Perfluoroalkyl Substances and the Replacement Compound HFPO-DA through the Atlantic Gateway to the Arctic Ocean—Is the Arctic a Sink or a Source? *Environmental Science & Technology*, **54**, 9958-9967.

[14]. Wang, Q, Tsui, MMP, Ruan, Y, Lin, H, Zhao, Z, Ku, JPH, Sun, H, and Lam, PKS, (2019). Occurrence and distribution of per- and polyfluoroalkyl substances (PFASs) in the seawater and sediment of the South China sea coastal region. *Chemosphere*, **231**, 468-477.

[15]. Glüge, J, Scheringer, M, Cousins, IT, DeWitt, JC, Goldenman, G, Herzke, D, Lohmann, R, Ng, CA, Trier, X, and Wang, Z, (2020). An overview of the uses of per- and polyfluoroalkyl substances (PFAS). *Environmental Science: Processes & Impacts*, **22**, 2345-2373.

[16]. Lam, NH, Cho, C-R, Kannan, K, and Cho, H-S, (2017). A nationwide survey of perfluorinated alkyl substances in waters, sediment and biota collected from aquatic environment in Vietnam: Distributions and bioconcentration profiles. *Journal of Hazardous Materials*, **323**, 116-127.

[17]. Zhu, W, Liu, W, and Jin, H, (2024). Sediment-seawater partitioning,

bioaccumulation, and biomagnification of perfluorobutane sulfonamide in marine environment. *Water Research*, **255**, 121466.

[18].Yamazaki, E, Taniyasu, S, Wang, X, and Yamashita, N, (2021). Per- and polyfluoroalkyl substances in surface water, gas and particle in open ocean and coastal environment. *Chemosphere*, **272**, 129869.

[19].Zhao, Z, Xie, Z, Möller, A, Sturm, R, Tang, J, Zhang, G, and Ebinghaus, R, (2012). Distribution and long-range transport of polyfluoroalkyl substances in the Arctic, Atlantic Ocean and Antarctic coast. *Environmental Pollution*, **170**, 71-77.

[20].Zhao, Z, Cheng, X, Hua, X, Jiang, B, Tian, C, Tang, J, Li, Q, Sun, H, Lin, T, Liao, Y, and Zhang, G, (2020). Emerging and legacy per- and polyfluoroalkyl substances in water, sediment, and air of the Bohai Sea and its surrounding rivers. *Environmental Pollution*, **263**, 114391.

[21].Zhong, H, Zheng, M, Liang, Y, Wang, Y, Gao, W, Wang, Y, and Jiang, G, (2021). Legacy and emerging per- and polyfluoroalkyl substances (PFAS) in sediments from the East China Sea and the Yellow Sea: Occurrence, source apportionment and environmental risk assessment. *Chemosphere*, **282**, 131042.

[22].Bai, X and Son, Y, (2021). Perfluoroalkyl substances (PFAS) in surface water and sediments from two urban watersheds in Nevada, USA. *Science of The Total Environment*, **751**, 141622.

[23].Li, J, Ai, Y, Hu, J, Xu, N, Song, R, Zhu, Y, Sun, W, and Ni, J, (2020). Polyfluoroalkyl substances in Danjiangkou Reservoir, China: Occurrence, composition, and source appointment. *Science of The Total Environment*, **725**, 138352.

[24].UNDP, (2024). The new POPs under the Stockholm Convention.

[25].Shang, M, Dong, J, Xie, H, Wang, Y, and Du, Y, (2023). Source, transport, and fate of perfluoroalkyl acids (PFAAs) in turbid bay environments: Significant roles of suspended sediment and water column stratification. *Water Research*, **243**, 120384.

[26].Li, Y, Oliver, DP, and Kookana, RS, (2018). A critical analysis of published data to discern the role of soil and sediment properties in determining sorption of per and polyfluoroalkyl substances (PFASs). *Science of The Total Environment*, **628-629**, 110-120.

[27].Nguyen, TMH, Bräunig, J, Thompson, K, Thompson, J, Kabiri, S, Navarro, DA, Kookana, RS, Grimison, C, Barnes, CM, Higgins, CP, McLaughlin, MJ, and Mueller, JF, (2020). Influences of Chemical Properties, Soil Properties, and Solution pH on Soil–Water Partitioning Coefficients of Per- and Polyfluoroalkyl Substances (PFASs). *Environmental Science & Technology*, **54**, 15883-15892.