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Distribution Characteristics and Risk Assessment of Phthalic Acid Esters: A Case Study of Water and Sediments from a Shanghai Wastewater Treatment Plant

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Abstract

Phthalic acid esters (PAEs) are ubiquitous plasticizer contaminants that pose ecological risks to aquatic organisms. However, systematic data on their occurrence and risk in both water and sediment across different treatment units of wastewater treatment plants (WWTPs) remain limited. In this study, the distribution and ecological risks of seven PAEs were investigated in water and sediments from different treatment stages of a WWTP in Shanghai, China. Total PAE concentrations in water ranged from 229.0 to 289.7 μg/L, with DBP as the dominant congener (83%~86%, up to 131.70 μg/L), and remained relatively stable throughout the treatment train. In contrast, total PAE concentrations ranged from 0.61 to 10.23 mg/kg in the sediment, with DBP again predominating (maximum 10.052 mg/kg), and showed a progressive decline along the AAO units, particularly in the oxic tank. Risk quotient (RQ) assessment revealed that DBP and DPP posed moderate risks in water (mean RQ = 1.005 and 1.227, respectively). In sediments, only DBP exhibited low ecological risk (mean RQ = 0.25), with the remaining PAEs showing no risk. These findings provide baseline data for PAE distribution and risk characterization in a WWTP with AAO process.

1 Introduction

Phthalic acid esters (PAEs) are a class of plasticizers produced by esterification of phthalic acid with alcohols, which can be readily released from plastic products and thus pose a potential exposure risk to human health (Li et al., 2024). They exhibit low polarity (log K_{ow} = 1.5 ~ 9.6) and low volatility (vapor pressure < 10^{-4} Pa). Furthermore, most PAEs feature ultra-low melting points (below − 25°C) and high boiling points (246 ~ 386°C). These properties make them widely used as plastic additives to enhance the flexibility, transparency, strength, and durability of plastics (Zhang et al., 2020). Consequently, PAEs currently represent the highest-volume organic compounds produced as plasticizers in industrial manufacturing. Globally, over three million tons of PAEs and their derivatives are used annually, with 35% serving as plasticizers in plastics and 15% as emulsifiers in pesticides (Golestanzadeh et al., 2019). China is one of the largest producers and consumers of PAEs. In China, over one million tons of plasticizers are consumed annually, with PAEs being used in 90% of PVC production (Guo and Kannan, 2011).

Based on their substituents, PAEs can be classified into two types: Short-chain (such as DMP, DEP, etc.) and long-chain (such as DEHP, DINP, etc.). The molecular weights of short-chain and long-chain PAEs differ significantly, which in turn leads to variations in their application scenarios. Higher molecular weight PAEs (HMWPs), such as BBP, DNOP, and DEHP, are primarily used to enhance polymer flexibility, accounting for approximately 80% of the total usage. Lower molecular weight PAEs (LMWPs), such as DEP, DBP, and DMP, are widely utilized in the formulations of daily chemical products.

PAEs migrate from plastics via hydrogen bonds or van der Waals forces and enter water bodies through industrial/commercial wastewater and plastic waste. As a key interception point, wastewater treatment plants (WWTPs) receive PAEs from domestic sewage, industrial wastewater, and rainwater, preventing

their direct release into aquatic environments (Pirsaheb et al., 2023; Wang et al., 2022). During treatment, PAEs are partially removed by biodegradation, while the remaining fraction requires physicochemical processes such as adsorption, sedimentation, and oxidation. The removal efficiency of PAEs varies with the treatment process employed; however, complete removal is difficult to achieve (Pirsaheb et al., 2023). Consequently, PAEs are ultimately discharged into aquatic environments via WWTP effluents. Once released, their low volatility facilitates ready migration within aquatic systems, and they are readily taken up by organisms, causing diverse toxic effects, including immunological, metabolic, endocrine, neurological, genetic, and developmental toxicity (Zhao et al., 2024). In addition to risks to aquatic organisms, PAEs also pose threats to human health. Most PAEs are endocrine disruptors (EDCs). Long-term exposure can impair reproductive development and increase the incidence of cancers, allergies, asthma, and other diseases. Recent studies have further linked PAE exposure to overweight and obesity, especially in children (Berghuis et al., 2024), and to exacerbated thyroid autoimmunity (Arash Derakhshan, 2021). If effluent concentrations of WWTPs exceed discharge standards or safety thresholds, adverse effects on aquatic organisms and human health may occur (Liu et al., 2024). Therefore, investigating the distribution characteristics of PAEs across different treatment processes and conducting corresponding risk assessments is of great significance.

Despite the high production and consumption of PAEs-containing products in China, studies on PAEs pollution levels in the country's WWTPs remain limited (Liu et al., 2023). WWTPs serve as the last barrier preventing PAEs from entering natural water bodies, thus the pollution characteristics of PAEs within WWTPs warrant attention. In this study, seven common PAEs (Table S1) were measured in different process units of a WWTP in China, Shanghai. Based on the measured concentrations, the ecological risks of PAEs in different treatment stages were assessed. The results are expected to provide a scientific basis for the selection and optimization of treatment processes to minimize PAE discharge.

2 Material and Methods

2.1 Description of the Wastewater Treatment Plant

This wastewater treatment plant (WWTP) is an enterprise in Shanghai, China, mainly engaged in the water production and supply industry. It adopts a relatively advanced sewage treatment process, namely the improved AAO + coagulation sedimentation + denitrification filter + ultraviolet disinfection. Its daily treatment capacity is as high as 200,000 cubic meters, and the effluent meets the first-class A standard of the "Discharge Standard of Pollutants for Municipal Wastewater Treatment Plants" (GB 18918 – 2002). Among them, the ammonia nitrogen and total phosphorus reach the Class IV surface water standard. In addition, the plant deeply integrates the concept of a sponge city, with a greening rate of over 90%. Through the design of vegetation buffer zones and permeable pavements, it realizes the natural infiltration and purification of rainwater, forming a small-scale ecological circulatory system.

2.2 Chemical Reagents and Experimental Instruments

All reagents used in this study were of HPLC grade, and the gases had a purity of 99.999%. Detailed information on reagents and instruments is provided in Table S2 and Table S3, respectively.

2.3 Sample Collection and Preservation

Water samples were collected from the raw influent (INF), the grit chamber effluent (GCE), the anaerobic-anoxic-oxic process effluent (AAO-E), the secondary clarifier effluent (SCE), and the final tailwater effluent (FE). Concurrently, sludge/sediment samples were retrieved from the influent channel (INF-C), the grit chamber (GC), the anaerobic tank (AT), the anoxic tank (AXT), the oxic/aerobic tank (OT), and the thickened sludge underflow (TS).

To minimize contamination during sampling and transport, all sampling materials were pre-rinsed with Milli-Q water before use, and gloves were worn to maintain clean handling. Water samples were collected using a fixed-volume method. A stainless-steel sampler was used to collect 6 L of water from approximately 30 cm below the water surface, which was then stored in 1 L glass sampling bottles. For each sampling site, two parallel sub-sites were established, located 100 m upstream and downstream, respectively. To facilitate subsequent experimental analyses, an additional 500 mL of water was collected from each site for routine water quality testing. Sediment samples were collected using a grab sampler, with approximately 1 kg of sediment obtained at each site. After collection, the sediment samples were immediately wrapped in aluminum foil and securely placed in stainless steel buckets for preservation. All samples were stored in a vehicle-mounted refrigerator during transport and then kept in the dark at 4°C in the laboratory until PAE analysis.

2.4 Sample Pre-treatment for PAEs Analysis

2.4.1 Water Sample

Each water sample was filtered under vacuum through a 0.45 μm membrane to remove suspended solids. A 10 mL aliquot of the filtrate was transferred into a separatory funnel, mixed with 25 mL of n-hexane, and shaken thoroughly at about 45°. After phase separation, the lower aqueous layer was discarded, and the upper organic layer was collected and dried over anhydrous sodium sulfate (pre-baked at 550°C for 4 h). The extract was then concentrated to near dryness under a gentle stream of nitrogen (0.2 L/min), reconstituted with n-hexane to a final volume of 1 mL, filtered through a 0.22 μm organic membrane, and transferred into a sample vial. The prepared samples were stored for subsequent GC-MS analysis.

2.4.2 Sediment Sample

Visible non-plastic particles were removed from the sediment samples. The samples were then dried in an oven at 55℃ to constant weight. For PAE extraction, a 1.0 g aliquot of dried sediment was weighed into a glass tube, mixed with 15 mL of acetone/n-hexane (1:1, v/v), and ultrasonicated at 25℃ (500 W) for 40 min. The supernatant was collected, and the residue was extracted again with 10 mL of the same solvent for 20 min. The combined extracts were centrifuged at 4000 rpm for 10 min, then concentrated

to near dryness under nitrogen. The residue was reconstituted with 200 μL of n-hexane, filtered through a 0.22 μm organic membrane, and transferred into a sample vial for subsequent GC-MS analysis.

2.5 GC-MS Analytical Method

The GC-MS method for PAE quantification was adapted from our previously established protocol (Yang et al., 2025), with some modifications. Chromatographic separation was performed on a DB-5MS fused silica capillary column (30 m × 0.25 mm × 0.25 μm). The injector temperature was set at 280℃, and splitless injection of 1.0 μL was employed. The column oven temperature was programmed as follows: initial temperature 60℃ held for 1 min, ramped to 220℃ at 20℃ /min and held for 4 min, then increased to 250℃ at 5℃/ min and held for 1 min, and finally raised to 290℃ at 20℃/min with a hold of 5 min. Helium was used as the carrier gas at a constant flow rate of 1.0 mL/min.

Mass spectrometric detection was carried out using an electron impact (EI) ion source at 70 eV. The ion source temperature was maintained at 250℃, the quadrupole temperature at 150℃, and the transfer line temperature at 300℃. Data acquisition was performed in the selected ion monitoring (SIM) mode. MS parameters for PAEs (retention time, characteristic ions, and standard curve equation) are listed in Table S4.

2.6 Ecological Risk Assessment of PAEs Pollution

The ecological risks of PAEs to aquatic environments were assessed using the risk quotient (RQ) method (Wang et al., 2026; Xu et al., 2022). The RQ approach is primarily employed to evaluate the risks posed by PAEs to aquatic organisms, based on the maximum detected concentrations of PAEs in water samples. The RQ is calculated as follows:

RQ = MEC/PNEC

RQ is defined as the ratio of the Measured Environmental Concentration (MEC) to the Predicted No-Effect Concentration (PNEC).

For compounds with available data, PNEC values can be obtained from published literature. For compounds lacking such data, PNEC can be estimated using toxicological endpoint concentrations LC_{50} and an assessment factor (RAF, f = 100), such as: $PNEC_{aqua}=LC_{50}/f$; $PNEC_{sediment}= f_{oc}\cdot K_{oc}\cdot PNEC_{aqua}$, where f_{oc} is the weight fraction of organic carbon in sediment (assumed to be 10% in this study), and K_{oc} is the sediment-water partition coefficient. The ecological structure activity relationships program (ECOSAR) developed by the United States Environmental Protection Agency (USEPA) can predict the toxic effects of chemical substances on organisms, including aquatic species, birds, and mammals (Sanderson et al., 2003). In this study, ECOSAR software was employed to calculate the toxicity thresholds of PAEs to aquatic organisms. Referring to aquatic toxicity data, the lower value between the 96-h LC_{50} for fish and the 48-h LC_{50} for *Daphnia magna* was selected as the LC_{50} for PNEC derivation (Zhang et al., 2021). Based on the calculated RQ values, the ecological risk levels of PAEs in the aquatic environment were classified (Lemly, 1996) as detailed in Table S5.

2.7 Quality Assurance and Quality Control

All plastic materials were excluded throughout the experimental procedure. Prior to use, all glassware was baked at 450°C for 12 h in a muffle furnace to eliminate any potential organic residues. A procedural blank consisting of pure hexane was injected before each sample run. The final concentrations of all extracted samples were corrected by subtracting the corresponding blank values. One out of every ten samples were selected at random for duplicate analysis, and the relative standard deviation (RSD) was maintained below 15% (Table S4). In addition, the sample recovery of water and sediment was 71.2 ~ 108.6 and 78.0 ~ 117.2 (Table S4).

2.8 Statistical Analysis

Data processing and graphing were performed using Microsoft Excel 2019 and Origin 2021 (Origin Lab).

3 Results and Discussion

3.1 Concentration and Distribution Characteristics of PAEs in Water from Different Treatment Processes

The concentration ranges of PAEs detected in water samples from different treatment processes are summarized in Table 1. Among the seven target PAEs, five were detected in 100% of samples, indicating that PAEs contamination has become ubiquitous throughout the entire facility. However, two medium- to long-chain PAEs, DBEP and DPhP, were not detected in any water sample, possibly due to their low influent concentrations or specific adsorption by the treatment processes.

Table 1
PAEs concentrations of different water treatment processes (μg/L)

PAEs	Min	Max	Median	Average	Standard Deviation	Detectable Rate (%)
DAP	4.60	4.90	4.80	4.78	0.13	100.00
DBP	100.30	131.70	102.20	110.54	13.54	100.00
DPP	3.60	3.70	3.70	3.68	0.04	100.00
DBEP	0.00	0.00	0.00	0.00	0.00	0.00
DPhP	0.00	0.00	0.00	0.00	0.00	0.00
DNOP	6.10	6.50	6.20	6.22	0.16	100.00
DNP	6.10	6.20	6.10	6.14	0.05	100.00

The concentrations of the PAEs across all treatment processes are shown in Fig. 1a. Their concentrations remained relatively stable throughout the treatment stages, indicating that conventional wastewater treatment processes achieved only very limited removal of PAEs. A 2020 study on four

WWTPs reported total PAE concentrations of 568.9 ~ 1847.5 ng/L in the effluents (sum of four congeners) (Wang et al., 2020). In comparison, the total concentration of seven PAEs detected throughout the entire treatment train in our study ranged from 229.0 to 289.7 μg/L (i.e., 229,000–289,700 ng/L), which is substantially higher. Consequently, the total PAE concentrations in WWTP effluents are rather variable across different studies, likely due to differences in influent composition, treatment processes, and the number of target congeners analyzed. As shown in Fig. 1b, DBP exhibited the highest concentration among all congeners across different treatment stages, with concentrations ranging from 100.30 to 131.70 μg/L and a mean value of 110.54 μg/L, accounting for 83%~86% of the total PAEs. This was followed by DNOP and DNP, with concentration ranges of 6.10 ~ 6.50 μg/L and 6.10 ~ 6.50 μg/L, respectively. The predominance of these three compounds is likely attributable to their widespread use in plastic products and personal care items (Zaman et al., 2025). According to *China's Environmental Quality Standards for Surface Water* and *Water Quality Standards for Drinking Water Sources*, the concentration limits for DBP is 3 μg/L. The levels detected in this study exceeded these standards, which may pose ecological risks to the aquatic environment. A previous study investigated the composition of 16 PAEs in the Yitong River, China, which receives effluent from a WWTP, and similarly found that DBP was the most abundant PAE congener (Zhang et al., 2025). Overall, among WWTP-related PAEs, DBP appears to pose a relatively high environmental risk and may therefore merit priority consideration in future regulation.

3.2 Concentration and Distribution Characteristics of PAEs in Sediment from Different Treatment Processes

The concentrations of PAEs in sediments exhibited considerable variability (Table 2). Among the nine detected PAEs, DBP showed a detection frequency of 100%, with a maximum concentration of 10.052 mg/kg and a mean value of 5.038 mg/kg, making it the dominant PAE in the WWTP sediments. A study on the northern coastal sediments of Vietnam reported that DBP and DEHP were the predominant PAEs among the 10 target congeners (Duong et al., 2026), which is similar with our finding. DNP followed, with maximum concentrations of 0.313 mg/kg, also demonstrating a 100% detection frequency, indicating that although DNP concentrations were low, it was widely present in the WWTP. DPP, DPhP, and DNOP exhibited a detection frequency of 66.7%, with mean concentrations all below 0.1 mg/kg. In addition, DAP and DBEP were not detected in any samples. These results reflect significant differences in the distribution characteristics of individual PAEs in the sediments. In addition, a previous survey reported that 50.3%~99.1% of the PAEs in the study area were distributed in the soil and sediment phases rather than in the aqueous phase (Wang et al., 2022), which is consistent with our finding that sedimentary PAE levels uniformly exceeded their aqueous counterparts.

Table 2
Concentrations of PAEs in sediments from different treatment processes (mg/kg)

PAEs	Min	Max	Median	Average	Standard Deviation	Detection Frequency (%)
DAP	0.000	0.000	0.000	0.000	0.000	0.0
DBP	0.151	10.052	0.012	5.038	3.402	100.0
DPP	0.000	0.036	0.016	0.015	0.014	66.7
DBEP	0.000	0.000	0.000	0.000	0.000	0.0
DPhP	0.000	0.143	0.009	0.043	0.062	66.7
DNOP	0.000	0.204	0.050	0.073	0.082	66.7
DNP	0.053	0.313	0.090	0.119	0.098	100.0

As shown in Fig. 2a, within the AAO process, the concentrations of Σ_7PAEs in sediments from INF-C, GC, AT, AXT, and OT progressively decreased, indicating that the biological treatment process in this WWTP has a good degradation effect on PAEs in sediments, especially the aerobic stage. In another WWTP employing the same AAO process, biodegradation accounted for 83% of total PAE removal, while adsorption contributed only 6% (Kong et al., 2015). This finding is consistent with our observation that sludge-associated PAE concentrations reached their minimum in the oxic tank. Subsequently, the TS exhibited relatively high PAEs concentrations, resulting from the microbial consumption of labile organic matter (e.g., proteins, polysaccharides) during thickening, which decreased volatile solids and thus relatively enriched the persistent, hydrophobic PAEs in the solid phase.

As shown in Fig. 2b, DBP accounted for a relatively small proportion (25%) in the OT sediment but contributed to over 92% of the total PAEs in sediments from the other treatment units. Aerobic microorganisms in the oxic tank have been demonstrated to utilize DBP as a carbon and energy source (Du et al., 2021), thereby exhibiting a strong degradation capacity for this compound. In contrast, DNP accounted for 51% of PAEs in the OT sediment, a high proportion that likely results from its resistance to aerobic degradation. The two nitro groups in DNP act as strong electron-withdrawing groups, reducing the benzene ring's electron density and consequently making DNP less prone to attack by the oxidizing enzymes of aerobic microorganisms.

3.3 Pollution Risk Assessment of PAEs

Based on this approach, PNEC values for 7 common PAEs were derived, and the relevant parameters are presented in Table 3.

Table 3
Ecological risk assessment parameters and risk levels for different types of PAEs

PAEs	LC_{50} (mg/L)	RAF	log K_{ow}	log K_{oc}	$PNEC_{aqua}$ (mg/L)	$PNEC_{sediment}$ (mg/kg)
DAP	5.32	100	3.355	3.006	0.053	5.394
DBP	1.11	100	4.610	4.259	0.011	20.152
DPP	0.327	100	5.592	5.241	0.003	56.957
DBEP	3.060	100	4.061	3.711	0.031	15.730
DPhP	2.540	100	4.097	3.747	0.025	14.185
DNOP	0.010	100	8.392	8.037	0.000	1088.930
DNP	0.002	100	9.521	9.165	0.000	2924.354

The RQ values of PAEs in water from different treatment processes are presented in Fig. 3a. The ecological risk assessment of 7 common PAEs in water from different treatment processes is summarized in Table 4. Among the seven PAEs, DPP exhibited the highest RQ values across all treatment processes, with a mean RQ value of 1.227, warranting particular attention. In addition, DBP, which has high bioaccumulation potential and documented toxicity to various organisms (Zaman et al., 2025), showed RQ values exceeding 1 (mean RQ = 1.005). Kotowska et al. (2020) reported that DBP as a PAE of high environmental concern, with significant risk to all trophic levels in both treated effluent and receiving groundwater. The discrepancies between their risk assessment and that of the present study are primarily attributable to differences in the raw wastewater characteristics and the specific treatment processes employed. At last, the remaining five PAEs showed no ecological risk. Notably, both DBP—a legacy PAE with well-documented toxicity—and DPP—a relatively novel alternative—exhibited RQ values exceeding the threshold of 1 in our study. Therefore, the PAEs transition is ongoing, yet without concomitant risk mitigation. This highlights the need for a precautionary approach when introducing novel substitutes to replace regulated legacy compounds.

Table 4
Ecological risk assessment levels of PAEs in water and sediment

Water				Sediment			
PAEs	**Mean RQ**	**Max RQ**	**Ecological Risk Level**	**PAEs**	**Mean RQ**	**Max RQ**	**Ecological Risk Level**
DAP	0.090	0.092	no	DAP	0.000	0.000	no
DBP	1.005	1.197	medium	DBP	0.250	0.499	low
DPP	1.227	1.233	medium	DPP	<0.001	0.001	no
DBEP	0.000	0.000	no	DBEP	0.000	0.000	no
DPhP	0.000	0.000	no	DPhP	0.003	0.010	no
DNOP	0.000	0.000	no	DNOP	<0.001	<0.001	no
DNP	0.000	0.000	no	DNP	<0.001	<0.001	no

The RQ values of PAEs in sediments from different treatment processes are shown in Fig. 3b. Although sedimentary PAE concentrations were higher than those in water, the corresponding RQs were close to zero. This discrepancy is mainly attributed to the low bioavailability of sediment-associated PAEs due to strong sorption onto organic matter, whereas the RQ calculation based on water-phase PNECs fails to reflect the actual exposure of benthic organisms. Among the seven PAEs, only DBP exhibited RQ values exceeding 0.1, while the RQ values of the remaining PAEs were all below 0.01, indicating relatively low ecological risk. The maximum RQ value was observed for DBP in influent sediment, reaching 0.499. The lowest RQ values for PAEs were found in sediment from the OT, suggesting that the aerobic treatment process in this WWTP effectively degrades PAEs. However, as the final treatment unit, thickened sludge exhibited a relatively high RQ value for DBP (0.225), warranting consideration of appropriate disposal methods to avoid secondary pollution.

The ecological risk assessment of seven PAEs in sediments from different treatment processes is summarized in Table 4. Only DBP posed low ecological risks, while the remaining PAEs showed no ecological risk. These results indicate that the ecological risk posed by PAEs in sediments of this WWTP is generally controllable. However, a study on the Bohai Bay and Laizhou Bay sediments reported that DBP and DIBP accounted for 73.8% of total PAEs, with RQ ≥ 10 at 33% of the sampling sites (Han et al., 2026), highlighting the site-specific nature of PAE risk assessments.

4 Conclusion

In summary, this study investigated the distribution and ecological risks of seven PAEs in water and sediments across different treatment units of a Shanghai WWTP. DBP was the dominant congener in both phases throughout the plant. Aqueous PAEs remained largely unchanged across the treatment units, suggesting limited removal by the integrated processes. In contrast, sedimentary PAEs declined

progressively along the AAO train, with the greatest attenuation observed in the oxic tank. In terms of ecological risk, DBP and DPP both exceeded the RQ threshold of 1 in water, indicating moderate risks, whereas in sediments only DBP posed a low risk. The coexistence of DBP and DPP with RQ > 1 indicates that replacing legacy PAEs with alternatives does not necessarily reduce ecological risks. DBEP was undetected in all samples including influent, suggesting limited local usage rather than removal. DPhP was absent in water but present in sediments, while DAP showed the opposite pattern, implying distinct partitioning behaviors. Given that only seven PAEs were targeted, future studies should expand the congener spectrum to include more widely used homologues such as DEHP and DIBP. For WWTP management, priority monitoring should be directed toward DBP and DPP in effluents, and operational parameters of the oxic tank should be optimized to enhance aerobic degradation of DBP. Additionally, proper disposal of thickened sludge is necessary, as PAE concentrations in this phase remained relatively high and may pose secondary contamination risks.

Declarations

Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

Author Contribution

Zhiyin Zhang: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft. Wenping Jiang: Investigation, Methodology. Mengru Fu: Investigation, Visualization, Writing – review & editing. Cheng Peng: Visualization, Investigation, Resources. Wei Zhang: Resources, Funding acquisition, Supervision, Writing – review & editing. Xiaoyong Qian: Conceptualization, Resources.

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Data availability

The data that support the findings of this study are available from the corresponding author upon reasonable request.

References

1. Arash Derakhshan, A. 2021. Association of phthalate exposure with thyroid function during pregnancy. Environment International 157, 106795.
2. Berghuis, S.A., Bocca, G., Bos, A.F., van Faassen, M., Foreman, A.B., van Vliet-Ostaptchouk, J.V. and Sauer, P.J.J. 2024. Adolescent urinary concentrations of phthalate metabolites and indices of overweight and cardiovascular risk in Dutch adolescents. Environment International 194, 109167.
3. Du, H., Hu, R.W., Zhao, H.M., Huang, H.B., Xiang, L., Liu, B.L., Feng, N.X., Li, H., Li, Y.W., Cai, Q.Y. and Mo, C.H. 2021. Mechanistic insight into esterase-catalyzed hydrolysis of phthalate esters (PAEs) based on integrated multi-spectroscopic analyses and docking simulation. Journal of Hazardous Materials 408, 124901.
4. Duong, H.T., Nguyen, H.T., Hoang, N., Pham, L.H., Van Vu, T. and Van Do, M. 2026. Pollution characteristics, spatial variation, and ecological risks of phthalate esters in sediment from northern coastal regions of Vietnam. Marine Pollution Bulletin 222, 118814.
5. Golestanzadeh, M., Riahi, R. and Kelishadi, R. 2019. Association of exposure to phthalates with cardiometabolic risk factors in children and adolescents: A systematic review and meta-analysis. Environmental Science and Pollution Research 26(35), 35670–35686.
6. Guo, Y. and Kannan, K. 2011. Comparative assessment of human exposure to phthalate esters from house dust in China and the United States. Environmental Science & Technology 45(8), 3788–3794.
7. Han, B., Zheng, L. and Liu, Y. 2026. Combined contamination of OPEs and PAEs in coastal sediments: Distribution, sources, and ecological risks in Northern China's Marginal Seas. Marine Environmental Research 216, 107859.
8. Kong, M., Song, Y., Zhang, Y., Liu, R., Wei, J. and Zheng, L. 2015. Fate of phthalate esters in municipal wastewater treatment plant and their environmental impact. Water Science and Technology 73(6), 1395–1400.
9. Kotowska, U., Kapelewska, J. and Sawczuk, R. 2020. Occurrence, removal, and environmental risk of phthalates in wastewaters, landfill leachates, and groundwater in Poland. Environmental Pollution 267, 115643.
10. Lemly, A.D. 1996. Evaluation of the hazard quotient method for risk assessment of selenium. Ecotoxicology and Environmental Safety 35(2), 156–162.
11. Li, J., Liu, B., Yu, Y. and Dong, W. 2024. A systematic review of global distribution, sources and exposure risk of phthalate esters (PAEs) in indoor dust. Journal of Hazardous Materials 471, 134423.
12. Liu, B., Lv, L., Ding, L., Gao, L., Li, J., Ma, X. and Yu, Y. 2024. Comparison of phthalate esters (PAEs) in freshwater and marine food webs: Occurrence, bioaccumulation, and trophodynamics. Journal of Hazardous Materials 466, 133534.
13. Liu, W., Li, X., Lv, H., Liang, C., Wang, Q., Yao, X., Dong, C., Zhang, W., Wang, J., Zhu, L. and Wang, J. 2023. Occurrence and health risk assessment of phthalates in a typical estuarine soil: A case study of the various functional areas of the Yellow River Delta. Science of The Total Environment 904, 166972.

14. Pirsaheb, M., Nouri, M. and Hossini, H. 2023. Advanced oxidation processes for the removal of phthalate esters (PAEs) in aqueous matrices: A review. Reviews on Environmental Health 38(2), 265–279.
15. Sanderson, H., Johnson, D.J., Wilson, C.J., Brain, R.A. and Solomon, K.R. 2003. Probabilistic hazard assessment of environmentally occurring pharmaceuticals toxicity to fish, daphnids and algae by ECOSAR screening. Toxicology Letters 144(3), 383–395.
16. Wang, C., Guo, Y., Feng, L., Pang, W., Yu, J., Wang, S., Qiu, C., Li, C. and Wang, Y. 2022. Fate of phthalates in a river receiving wastewater treatment plant effluent based on a multimedia model. Water Science and Technology 86(9), 2124–2137.
17. Wang, J., Wen, Y., Shao, Y., Schäffer, A., Rojo-Nieto, E., Schmitz, K., Jahnke, A., Zervou, S.-K., Hiskia, A. and Chen, Z. 2026. Emerging contaminants in sediments of a tributary to the Yangtze river in the Three Gorges Reservoir Area: Fate, ecological risk assessment, and prioritization for management. Journal of Hazardous Materials 502, 140894.
18. Wang, R., Ji, M., Zhai, H. and Liu, Y. 2020. Occurrence of phthalate esters and microplastics in urban secondary effluents, receiving water bodies and reclaimed water treatment processes. Science of The Total Environment 737, 140219.
19. Xu, H., Liu, Y., Xu, X., Lan, H., Qi, W., Wang, D., Liu, H. and Qu, J. 2022. Spatiotemporal variation and risk assessment of phthalate acid esters (PAEs) in surface water of the Yangtze River Basin, China. Science of The Total Environment 836, 155677.
20. Yang, Y., Han, Y., Zhao, X., Liu, X., Zhang, W., Peng, C., Kong, L. and Zhan, X. 2025. Risks of biodegradable films: The time-lagged release of phthalic acid esters and organophosphates esters under realistic agricultural environments. Environmental Pollution(377), 126467.
21. Zaman, F., Rahman, M.A., Abdullah, R.B., Anik, A.H. and Tareq, S.M. 2025. A potential threat of the phthalate acid esters (PAEs) in the different environmental matrices to public health: A systematic review. Environmental Monitoring and Assessment 197(11), 1283.
22. Zhang, W., Xiaowei, Z., Peng, G., Ning, W., Zini, L., Jian, H. and Zheng, Z. 2020. Distribution and risk assessment of phthalates in water and sediment of the Pearl River Delta. Environmental Science and Pollution Research 27(11), 12550–12565.
23. Zhang, X., Bai, M., Dong, A., Du, X., Ding, Y. and Zhao, K. 2025. Contribution to distribution and toxicity prediction of organic pollutants in receiving waters from wastewater plant tailwater: A case study of the Yitong River, China. Water 17(14), 15.
24. Zhang, Y., Jiao, Y., Li, Z., Tao, Y. and Yang, Y. 2021. Hazards of phthalates (PAEs) exposure: A review of aquatic animal toxicology studies. Science of The Total Environment 771, 145418.
25. Zhao, K., Wang, S., Bai, M., Wang, S. and Li, F. 2024. Distribution, seasonal variation and ecological risk assessment of phthalates in the Yitong River, a typical urban watercourse located in Northeast China. Science of The Total Environment 930, 172696.

Figures

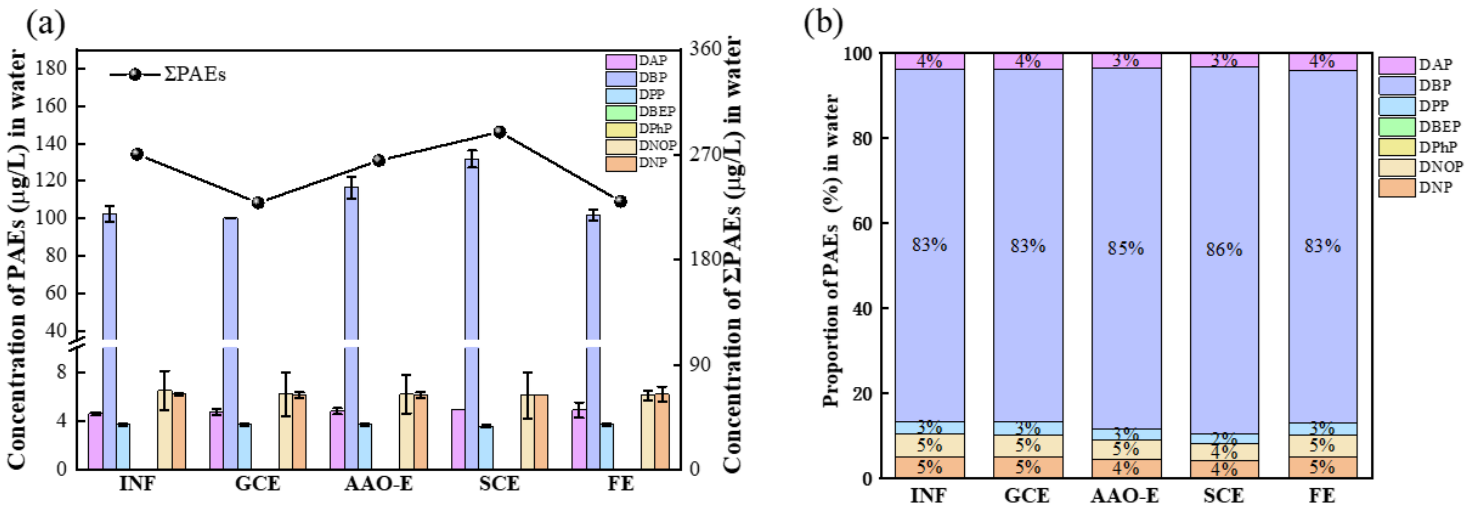


Figure 1

(a) Concentrations and (b) proportions of PAEs in water from different treatment processes.

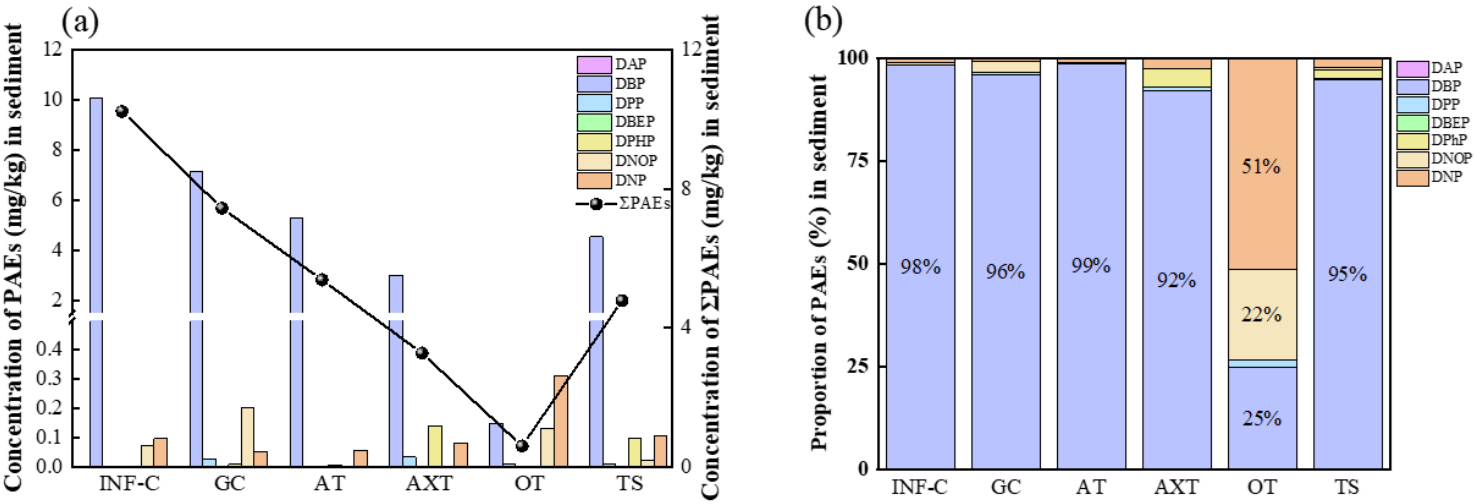


Figure 2

(a) Concentrations and (b) proportions of PAEs in sediment from different treatment processes.

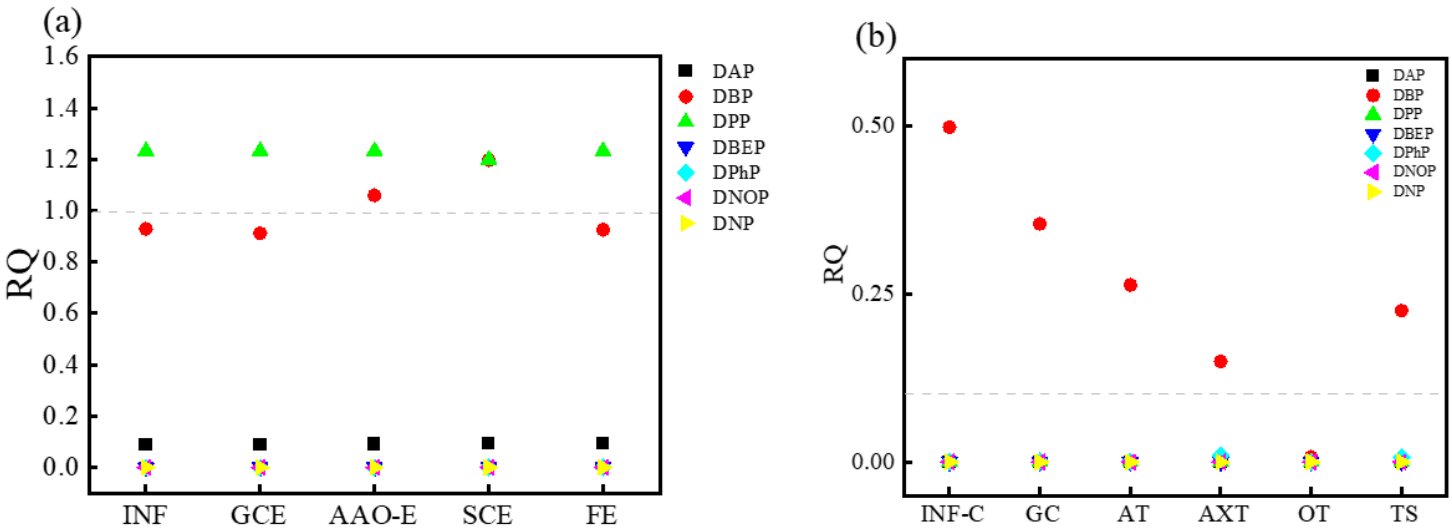


Figure 3

RQ values of PAEs in (a) water and (b) sediment from different treatment processes.

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