Journal of Hazardous Materials 491 (2025) 138033

Contents lists available at ScienceDirect

Journal of Hazardous Materials

journal homepage: www.elsevier.com/locate/jhazmat

Review

Microplastics as drivers of carbon and nitrogen cycling alterations in aquatic ecosystems: A meta-analysis

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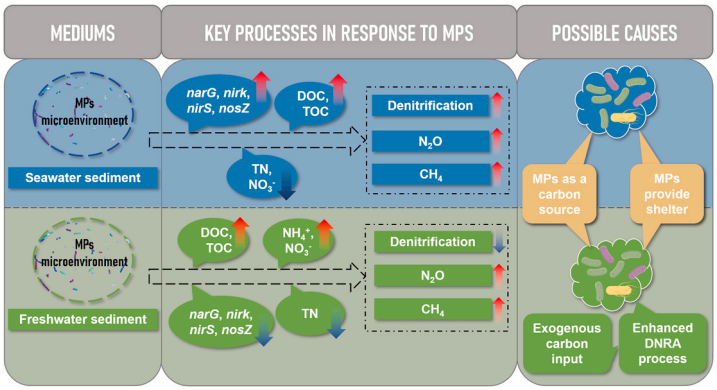
HIGHLIGHTS

- MPs enhance DOC and TOC concentrations, promoting greenhouse gas emissions.
- MPs increase denitrification in seawater sediments.
- MPs weaken denitrification but heighten DNRA in freshwater sediments.
- BMPs exert stronger effects on carbon and nitrogen cycling compared to NBMPs.

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
Microplastics
Meta-analysis
Carbon cycling
Nitrogen cycling
Aquatic ecosystem
Sediment

ABSTRACT

Microplastics (MPs) have been increasingly recognized as an emerging contaminant in aquatic ecosystems, with growing evidence of their impact on biogeochemical cycles. This study synthesizes the effects of MPs on nitrogen and carbon cycling in aquatic environments by performing a network meta-analysis. Our findings suggest that MPs enhance dissolved organic carbon and total organic carbon concentrations, promote anaerobic processes, and stimulate greenhouse gas emissions, including N_2O and CH_4. In seawater sediments, MPs significantly enhance denitrification, as evidenced by increased abundances of *narG* , *nirS*, *nirK*, and *nosZ* genes, elevated N_2O production, and reduced NO_3^- concentrations. In contrast, MP addition exhibit weaker denitrification but heightened N_2O production in freshwater sediments, likely driven by enhanced dissimilatory nitrate reduction to ammonium processes. Furthermore, biodegradable MPs exhibit stronger effects on carbon and nitrogen

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https://doi.org/10.1016/j.jhazmat.2025.138033
Received 17 January 2025; Received in revised form 3 March 2025; Accepted 20 March 2025
Available online 22 March 2025

metabolism compared to non-biodegradable MPs. These findings highlight the complex and medium-dependent role of MPs in biogeochemical cycles, emphasizing the need for interdisciplinary research to fully elucidate their environmental impacts.

1. Introduction

Microplastics (MPs), broadly defined as plastic particles smaller than 5 mm, have garnered significant attention in recent years [34,35]. These particles are generated through the mechanical fragmentation, photodegradation, and biological processes acting on larger plastic debris [5]. Additionally, some plastics are intentionally manufactured at MP sizes for products such as cosmetics [3]. It is estimated that between 10–40 million tons of MPs are released into the environment each year, with this figure steadily increasing [34]. MPs are ubiquitously distributed across soils, aquatic systems, and sediments, with their presence documented even in remote deserts, deep-sea trenches, and polar regions [1, 21,41,45]. Their concentrations are particularly elevated in aquatic systems and sediments, with studies reporting levels reaching 813 ± 185 items/L in coastal waters and 100 ± 57 items/g dry weight in sediments [13]. The pervasive presence of MPs inevitably influences microbial community dynamics within habitats, profoundly impacting biogeochemical cycling in aquatic systems.

In aquatic systems, MPs characterized by their large specific surface area and high porosity, readily facilitate microbial attachment and community development [9]. This attachment varies depending on the type of polymer and its aging state [30]. These microbial communities differ significantly from those in the surrounding water column, with the plastic-associated habitat, termed the "plastisphere", often creating anaerobic microenvironments. Such conditions can enhance estuarine denitrification and N_2O production [33]. In marine waters, these microenvironments significantly reduce surrounding NH_4^+ concentrations and facilitate nitrification [10]. Moreover, biofilm formation on MP surfaces plays a crucial role in influencing their sedimentation rates. For example, biofilm on the surfaces of polyethylene (PE) and polystyrene (PS) MPs notably accelerates the sinking rate of MPs in laboratory conditions [16]. Sediments, with their high organic carbon content, provide an environment where MPs can rapidly colonize biofilms [27]. The presence of MPs in sediments may aggregate competitive functional microorganisms, thereby influencing biogeochemical cycles. For instance, a laboratory experiment showed that PS-MPs promoted the proliferation of certain anaerobic microorganisms, which was linked to enhanced dissimilatory nitrate reduction to ammonium (DNRA) [40].

The surface structure of MPs determines their susceptibility to microbial colonization, while their aging process makes them more available as carbon sources for microorganisms. The primary progress of MP aging is UV radiation. In seawater, ultraviolet radiation accelerates the aging of PE and polypropylene (PP) MPs, producing dissolved organic carbon (DOC) that can be directly utilized by microbial communities [32]. Surprisingly, MPs-derived DOC accounts for 10 % of the total DOC in the ocean surface, which undoubtedly stimulates the activity of heterotrophic microbes, potentially impacting the entire marine ecosystem [26]. In sediment, microbial degradation of MPs assumes a more significant role compared to photodegradation. Furthermore, microbial degradation can result in the generation of oligomers, which may have additional ecological implications [44]. Studies have shown that biodegradable MPs are more readily utilized by microorganisms in nearshore sediments, which can accelerate the emission of greenhouse gases such as CH_4 [4].

As critical processes in aquatic ecosystems, the carbon and nitrogen cycles are tightly interconnected. However, the mechanisms by which MPs influence these cycles remain inadequately understood, with a lack of comprehensive and objective data synthesis. This study employs a network meta-analysis based on data extracted from 36 publications to examine the effects of MPs on carbon and nitrogen cycling in water columns and sediments. The findings aim to elucidate the current state of research on this topic, highlight critical knowledge gaps, and propose future research directions.

2. Materials and methods

2.1. Literature search

We conducted a literature search using the Web of Science (WoS) database, employing the following search strategy to ensure that no relevant studies were overlooked: ("microplastic*" OR "micro plastic*" OR "micro-plastic*") AND ("nitrogen*" OR "cycl*" OR "biogeochemi*" OR "nitrate" OR "nitrite" OR "ammonia" OR "nitrif*" OR "denitrif*" OR "anaerobic ammonia oxidation" OR "anammox" OR "DNRA" OR "carbon*" OR "organic matter" OR "CO_2" OR "inorganic matter" OR "methane*" OR "CH_4" OR "nitrous oxide" OR "N_2O" OR "dissolved organic carbon" OR "dissolved inorganic carbon" OR "greenhouse gas") AND ("aquatic" OR "marine" OR "estuar*" OR "sediment*" OR "sea*" OR "ocean" OR "freshwater" OR "lake*" OR "mangrove" OR "wetland" OR "reservoir"). By applying this query to the WoS database with "Topic" as the search field, we retrieved a total of 4135 records as of September 2024. Each record was screened individually, resulting in the identification of 155 candidate studies. Upon further review of experimental design, methodology, and results, only 36 studies met the inclusion criteria for the final analysis. These studies shared the following characteristics: a) A clear indoor experimental design. b) Control groups without MP treatments. c) Quantification of MP concentration expressed as weight-to-weight ratios (for water or sediments) or weight-to-volume ratios (for water only). d) Provision of explicit quantitative data.

2.2. Data extraction and classification

We systematically extracted detailed information and experimental data from each study, including the following: a) Basic study information: title, authors, publication year, and DOI. b) MP characteristics: polymer type, concentration, and particle size. c) Experimental parameters: sample size (n) and standard deviation (SD). d) Environmental conditions: environmental medium, incubation time, and incubation temperature. e) Nitrogen cycling indicators: concentrations of NH_4^+, NO_2^-, NO_3^-, total nitrogen, N_2O production rate, denitrification rate, and the abundances of functional genes (AOB-*amoA*, *narG* , *nirS*, *nirK*, *nosZ*). f) Carbon cycling indicators: dissolved organic carbon (DOC), total organic carbon (TOC), and emissions of CO_2 and CH_4. For studies reporting standard error (SE) instead of SD, we converted SE to SD using the formula [11]:

$$SD = SE\sqrt{n}$$

Data were preferentially extracted directly from tables. When unavailable, "Web Plot Digitizer (WPD)" was employed to extract data from figures (https://automeris.io/wpd/). Prior to data extraction, the team standardized the WPD settings to ensure consistency. During the extraction process, two researchers independently extracted data from the same figures using WPD. After extraction, the intraclass correlation coefficient (ICC) for continuous variables was evaluated, yielding an ICC > 0.90, indicating excellent agreement between the two researchers. All figures from which data were extracted in the included studies were clear and suitable for accurate extraction. All nitrogen and carbon cycling indicators were extracted from at least 5 studies. Indicators appearing in fewer than 5 studies (e.g., N_2O emissions, CO_2 production rates) were excluded from subsequent analyses to ensure statistical

robustness. Given the high heterogeneity typically observed across ecological studies, we included as many studies as possible to enhance the robustness of the meta-analysis [29].

To facilitate subgroup analyses, MPs were categorized based on the following criteria: a) Biodegradability: biodegradable MPs (BMPs) vs. non-biodegradable MPs (NBMPs). b) Concentration (% w/w): Conc. 1 (0, 0.1], Conc. 2 (0.1, 0.3], Conc. 3 (0.3, 0.5], Conc. 4 (0.5, 1], Conc. 5 (1, 2], Conc. 6 (2, 5], Conc. 7 (5, 10]. c) Particle size (μm): Size 1 (0, 100], Size 2 (100, 200], Size 3 (200, 500], Size 4 (500, 1000], Size 5 (1000, 5000]. Additionally, to streamline statistical analysis, incubation temperature (°C) and duration (days) were grouped into four and five levels, respectively: Temperature: Temp. 1 (0, 10], Temp. 2 (10, 20], Temp. 3 (20, 25], Temp. 4 (25, 30]. Incubation time: Incu. 1 (0, 10], Incu. 2 (10, 30], Incu. 3 (30, 50], Incu. 4 (50, 70], Incu. 5 (70, 120].

2.3. Statistical analysis

For each outcome, we performed a network meta-analysis (NMA) using frequency-based methods to process the data. The effect size for each study was calculated using the Hedge's g method [42,43]:

$$g^n = \frac{\overline{X}_1 - \overline{X}_2}{s_p} J$$

Where, g^n is the effect size, $\overline{X}_1$ and $\overline{X}_2$ are the means of the two groups, s_p is the pooled standard deviation of the two groups, and J is a correction factor:

$$s_p = \sqrt{\frac{(n_1 - 1)\, s_1^2 - (n_2 - 1)\, s_2^2}{(n_1 + n_2) - 2}}$$

$$J = 1 - \frac{3}{4(n_1 + n_2 - 2) - 1}$$

Where n_1, n_2 are the sample sizes, and s_1, s_2 are the variances for each group.

Network meta-analysis accounts for both direct and indirect evidence, leading to more precise estimates of network effect sizes (g^m) [22,42]. A random-effects model based on restricted likelihood estimation was used to combine the effect sizes from each study and compute the heterogeneity variance, which was then incorporated into the overall effect size estimate [37]:

$$G = \frac{\sum_{i=1}^{k} w_i g_i^m}{\sum_{i=1}^{k} w_i}$$

Where G is the overall effect size, w_i is the weight of the i-th study, adjusted for heterogeneity variance τ^2:

$$w_i = \frac{1}{v_i + \tau^2}$$

Where, v_i is the variance of the effect size for the i-th study, and τ^2 is the estimated heterogeneity variance, taking into account heterogeneity statistics and the number of studies [37]. Therefore, the standard error (SE) for the overall effect size is calculated as:

$$SE = \sqrt{\frac{1}{\sum_{i=1}^{k} w_i}}$$

The 95 % confidence interval (95 % CI) is calculated as:

$$95\%CI = 1.96SE$$

An effect is considered significant ($P < 0.05$) if the 95 % CI does not overlap with 0.

We calculated the overall effect size and subgroup effect sizes, presenting the results as effect sizes with 95 % CIs. Subgroup analyses included MP polymer type, degradation type, particle size, concentration, and environmental medium. These calculations were performed in STATA 17 using the network package, with visualizations created in R 4.3.3 using the forestploter package.

3. Results

We extracted 402 data strings from the 36 relevant studies, all of which clearly reported the polymer types, incubation medium, incubation time, and sample sizes. Additionally, the majority of studies provided clear information on MP particle size, concentration, and incubation temperature. However, two studies did not specify MP particle size, four studies did not report MP concentration, and nine studies did not provide incubation temperature details. Of the 402 data strings, 232 were from experimental groups (excluding the control group). The most frequently reported polymers were PE (41.8 %), PVC (15.5 %), and PLA (11.2 %). The most common MP particle size, concentration, incubation temperature, and incubation time categories were: Size 1 (0, 100] (30.6 %), Conc. 3 (0.3, 0.5] (24.1 %), Incu. 2 (10, 30] (41.8 %), and Temp. 3 (20, 25] (45.7 %). In terms of environmental medium, seawater sediment (42.7 %) and freshwater sediment (40.5 %) were the most common (Fig. 1).

3.1. Nitrogen nutrients

Table 1. Overall, the addition of MPs did not exert a significant influence on nitrogen nutrient concentrations (Fig. 2). However, subgroup analyses unveiled polymer-specific effects on nitrogen salts. Notably, PBAT-MPs markedly elevated NH_4^+ ($P < 0.05$, n = 54) and NO_2^- concentrations ($P < 0.05$, n = 39), whereas PE-MPs significantly enhanced NO_3^- levels ($P < 0.05$, n = 168). Furthermore, changes in NH_4^+ concentrations exhibited a pronounced correlation with MP particle size. Specifically, Size 1 MPs significantly increased NH_4^+ concentrations ($P < 0.05$, n = 114), while Sizes 3, 4, and 5 MPs led to significant reductions ($P < 0.05$). However, these effects were independent of concentration. The environmental medium also played a critical role in modulating outcomes. In seawater water and sediment, MPs significantly reduced NO_3^- concentrations ($P < 0.05$), whereas in freshwater water and sediment, they induced an increase in NO_3^- concentrations (ES: 2.09, 95 % CI [1.13, 3.05], n = 57; ES: 0.24, 95 % CI [−0.1, 0.58], n = 174).

3.2. Total nitrogen, N_2O production rate and denitrification rate

MPs were found to significantly enhance the N_2O production rate ($P < 0.05$, n = 242) while concurrently reducing TN concentration ($P < 0.05$, n = 181), with a moderate stimulation of denitrification processes (ES: 0.6, 95 % CI [−0.06, 1.26], n = 173) (Fig. 3). Subgroup analyses revealed that NBMP and four distinct particle size categories significantly diminished TN content ($P < 0.05$). Both BMP and NBMP were observed to promote N_2O production, with BMP exhibiting a more pronounced effect. Notably, these outcomes were independent of MP particle size and concentration. Furthermore, MPs significantly accelerated denitrification rates in seawater sediments ($P < 0.05$), whereas they suppressed denitrification in freshwater sediments (ES: −0.77, 95 % CI [−1.65, 0.11], n = 60).

3.3. Nitrogen cycle-related functional genes

The addition of MPs significantly elevated the abundance of AOB-*amoA* and *nirK* genes ($P < 0.05$), while also fostering increases in the abundance of *narG* , *nirS*, and *nosZ* genes (ES: 0.22, 95 % CI [−0.26, 0.7], n = 120; ES: 0.3, 95 % CI [−0.03, 0.63], n = 213; ES: 0.22, 95 % CI [−0.08, 0.52], n = 168) (Fig. 4). Subgroup analyses revealed that BMP exerted a more substantial promotional effect on these gene abundances

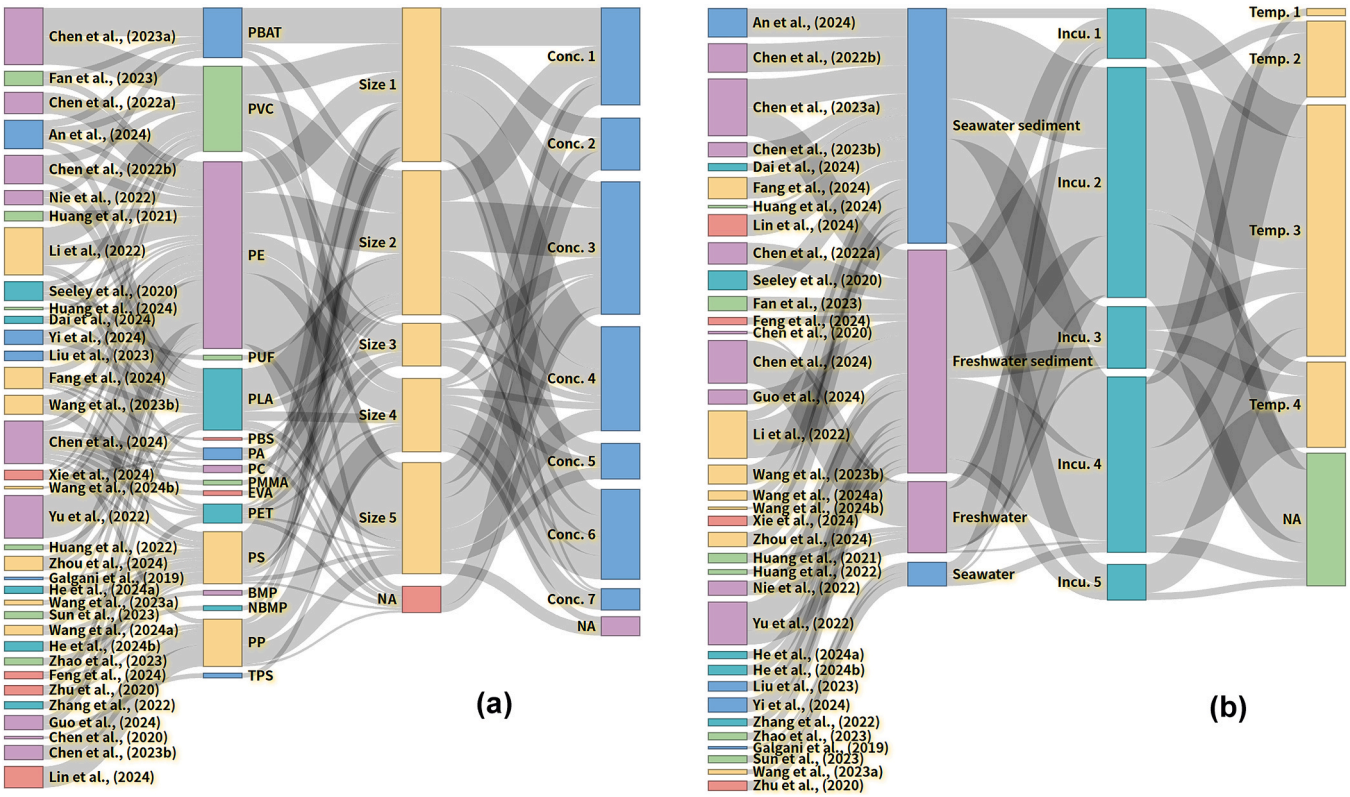

Fig. 1. Basic information of MPs (a) and other experimental parameters (b). The block size depends on the number of data strings. Abbreviations refer to Table 1.

Table 1
List of abbreviations.

Abbreviations	Full name	Abbreviations	Full name
MPs	Microplastics	PBAT	Polybutylene adipate terephthalate
PVC	Polyvinyl chloride	PUF	Polyurethane foam
PE	Polyethylene	PLA	Polylactic acid
PA	Polyamide	PBS	Polybutylene succinate
PC	Polycarbonate	PMMA	Polymethyl methacrylate
EVA	Ethylene-vinyl acetate	PET	Polyethylene terephthalate
PS	Polystyrene	BMP	Biodegradable microplastics
NBMP	Non-biodegradable microplastics	PP	Polypropylene
TPS	Tire wear particles	TN	Total nitrogen
TOC	Total organic carbon	DOC	Dissolved organic carbon
ES	Effect size	CI	Confidence interval
DNRA	Dissimilatory nitrate reduction to ammonium		

compared to NBMP. These findings were independent of MP particle size and concentration. Moreover, in seawater sediments, MPs significantly enhanced the abundance of these genes ($P < 0.05$), whereas in freshwater sediments, they displayed an inhibitory effect, particularly marked by significant reductions in the abundance of *nirS* and *nosZ* genes ($P < 0.05$).

3.4. Carbon cycle-related indicators

The MP treatment significantly elevated the concentrations of DOC, TOC, and CH_4 emissions ($P < 0.05$). Subgroup analyses demonstrated that the majority of observed effects were promotive for these parameters. Notably, BMP exhibited a marginally stronger influence on DOC, TOC concentrations, and CH_4 emissions compared to NBMP. These outcomes, however, were independent of MP particle size and concentration. Furthermore, the promotive effects remained consistent across diverse environmental matrices (Fig. 5).

4. Discussion

4.1. Impact of MPs on the nitrogen cycle

Microplastics introduced into aquatic environments predominantly accumulate in sediments, with denser particles exhibiting a more rapid settling rate. However, less dense particles may undergo density increases due to biological processes, ultimately leading to their deposition [14]. Although resuspension can reintroduce MPs into the water column, this mechanism is constrained, resulting in a substantial fraction becoming sequestered within sediments in some compartments with insufficient hydrodynamic force [2,6]. Consequently, sediments have emerged as a principal reservoir for MPs. Similar to MPs in the water column, those in sediments can facilitate biofilm formation, which selectively enriches specific microbial communities and may thereby modulate nitrogen cycling processes [27]. Notably, surface sediments represent one of the most dynamic regions for nitrogen cycling within aquatic ecosystems.

In sedimentary environments, MPs function not only as microbial habitats but also as a carbon substrate for metabolic activities. The DOC leached from aged MPs is more bioavailable, thereby stimulating heterotrophic nitrogen reduction processes [24,32]. The pronounced differences between BMP and NBMP in modulating nitrogen cycling likely stem from their distinct physicochemical properties and degradation behaviors. BMPs (e.g., PLA, PBAT) contain hydrolyzable ester bonds that facilitate enzymatic cleavage by microbial hydrolases (e.g., esterases, lipases), releasing oligomers (e.g., lactic acid, adipic acid) that serve as labile carbon sources [36,44]. This process not only fuels heterotrophic

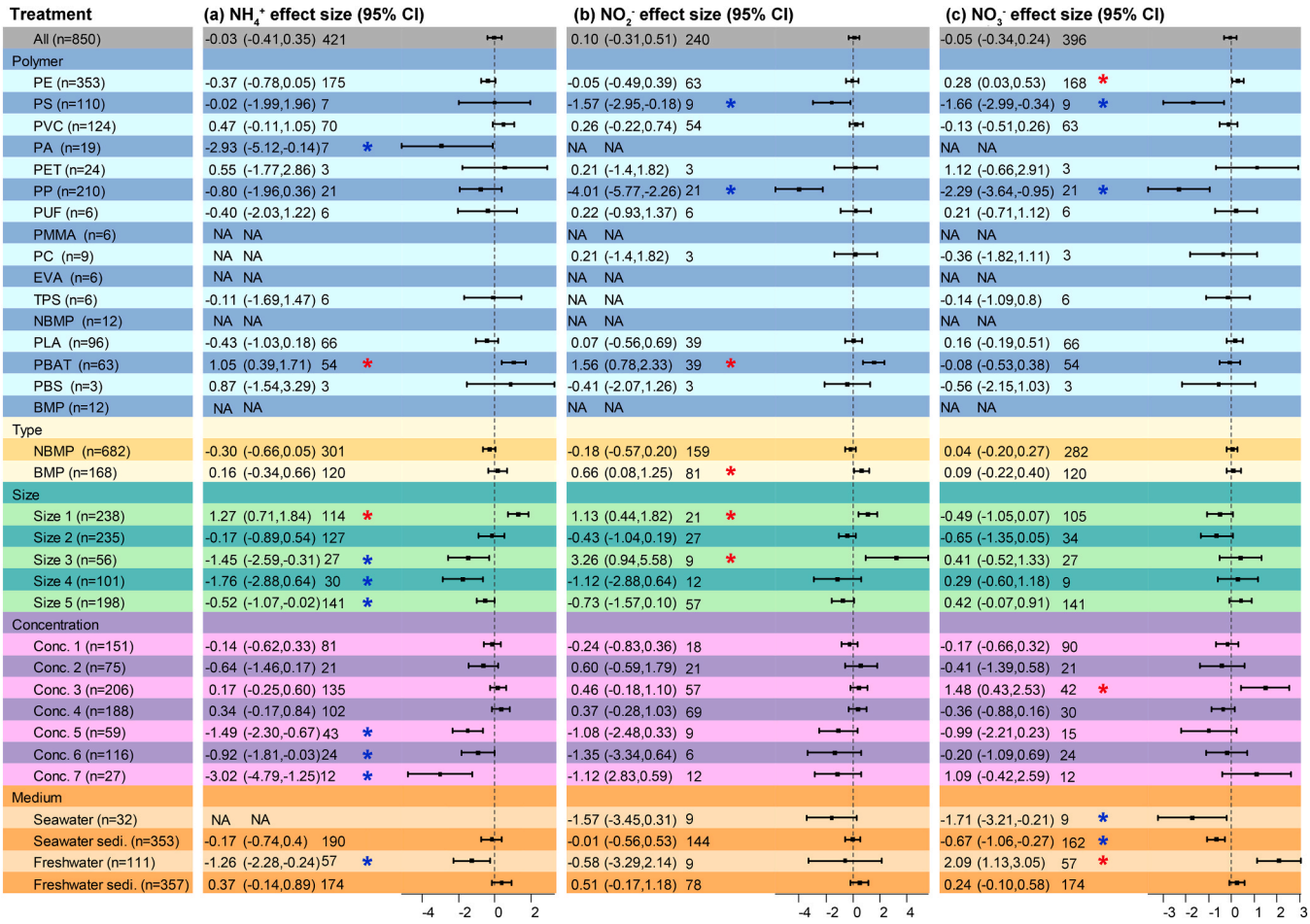

Fig. 2. The overall and subgroup effects on nitrogen nutrients are presented. Each data string is represented by the effect size, 95 % CI, sample size, and statistical significance. Red and blue asterisks indicate positive and negative effects of the treatment compared to the control group, respectively ($P < 0.05$). The error bars represent the 95 % CI. Abbreviations refer to Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

metabolism but also likely promotes localized hypoxic microenvironments due to rapid oxygen depletion. In contrast, non-biodegradable polymers (e.g., PE, PP) exhibit hydrophobic surfaces and stable C-C backbones, limiting microbial adhesion and enzymatic attack [8]. Therefore, BMP promotes nitrogen reduction processes more effectively than NBMP. Interestingly, our findings show that MPs may have different effects on nitrogen cycling in freshwater compared to seawater sediments. While the promotion of N_2O production was consistently observed across both environments, the effects on other nitrogen cycling indicators exhibited opposing trends.

As previously elucidated, MPs enhance nitrogen reduction processes in both freshwater and seawater environments, but the specific subprocesses involved differ. In seawater sediments, the observed reduction in NO_3^- concentrations following MP treatment points to a robust nitrogen reduction pathway. This is further corroborated by the increased abundance of *narG* , *nirS*, *nirK*, and *nosZ* genes, alongside elevated N_2O production and denitrification rates, collectively affirming that MPs stimulate denitrification in seawater sediments. Conversely, in freshwater sediments, elevated NO_3^- concentrations, diminished abundances of *narG* , *nirS*, *nirK*, and *nosZ* genes, and reduced denitrification rates suggest a less pronounced denitrification process. Consequently, the heightened N_2O production rate in freshwater sediments under MP treatment implies that N_2O generation likely arises from alternative pathways rather than denitrification (Fig. 6).

The majority of studies on freshwater environments in our dataset focused on lakes, which are semi-enclosed systems often situated in regions of intense anthropogenic activity. These conditions result in substantial year-round input of terrestrial organic matter and increase the susceptibility to eutrophication. Elevated organic matter levels and eutrophication can deplete dissolved oxygen in the water column, creating an environment conducive to the DNRA process [19]. The introduction of MPs may further amplify this process, as evidenced by the observed increase in NH_4^+ concentrations in freshwater sediments. Consequently, the heightened N_2O production in freshwater sediments may be attributed to an enhanced DNRA pathway [23]. Additionally, MPs were found to elevate nitrogen nutrient concentrations while reducing TN content in freshwater sediments, suggesting that MPs may accelerate the mineralization of nitrogen-containing organic matter. This pronounced mineralization process appears less prevalent in seawater sediments, underscoring the comparatively higher organic matter content in freshwater systems.

4.2. Impact of MPs on carbon storage and greenhouse gas emissions in aquatic ecosystems

Plastics are high-molecular-weight polymers characterized by C-C backbones, with C and H as their primary constituent elements. While these features are similar to those of conventional organic matter, plastics exhibit resistance to microbial degradation and utilization due to their extended carbon chains, structural stability, high crystallinity, and hydrophobic nature [15,8]. These properties historically led to the underestimation of plastics as a potential carbon storage. However,

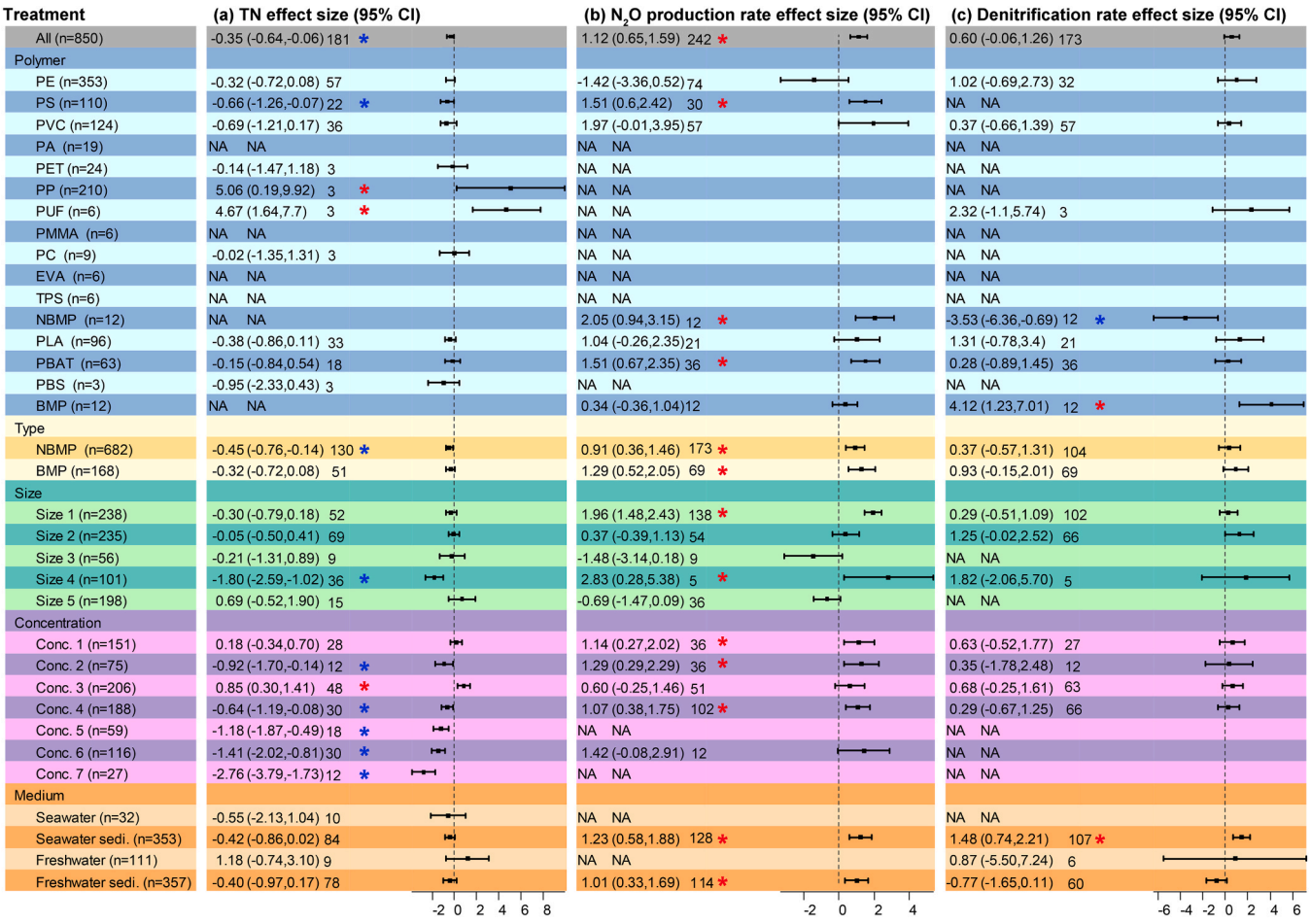

Fig. 3. The overall and subgroup effects on TN, N_2O production, and denitrification rate are presented. Each data string is represented by the effect size, 95 % CI, sample size, and statistical significance. Red and blue asterisks indicate positive and negative effects of the treatment compared to the control group, respectively ($P < 0.05$). The error bars represent the 95 % CI. Abbreviations refer to Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

recent advancements in MP research have revealed that plastics can indeed serve as a carbon source for microorganisms, thereby influencing microbial community dynamics and succession [20,28]. Although this process occurs at a slower rate compared to the degradation of other organic carbon sources, but it still exhibits bioavailable characteristics of organic matter.

The MPs addition elevates the concentrations of TOC and DOC in aquatic ecosystems. If MPs are not excluded during TOC measurements, they contribute to the final TOC values, making the increase in TOC levels predictable. More significantly, MP addition markedly raised DOC levels in aquatic environments, implying that a portion of the DOC may derive from the released carbon of MPs. This release process is likely driven by a combination of microbial degradation and photodegradation in the water column, with microbial degradation playing a dominant role in sediments. Given the slow nature of this process, MPs can persistently augment DOC concentrations in aquatic ecosystems over extended periods. It is estimated that between 7.8 ± 1.73 million tons of carbon in the form of plastic are deposited in seawater sediments annually [31]. This deposition rate has already exceeded that of other organic carbon forms in certain seawater sediments. Consequently, plastics including MPs should be re-evaluated as a substantial carbon reservoir.

A direct consequence of elevated DOC concentrations resulting from MPs in aquatic environments is the provision of an additional carbon substrate for microbial metabolism, thereby enhancing microbial respiration. Our analysis demonstrate that this includes an augmented capacity for N_2O and CH_4 production. Similar to N_2O production, CH_4 generation necessitates anaerobic conditions, but some conditions are even more stringent. From a microbial perspective, methane production is driven by the respiration of methanogenic archaea, which demand not only strictly anoxic environments but also simple molecules (e.g., H_2, CH_3COOH) as metabolic substrates [25]. These substrates may originate from the fermentation of DOC released by MPs. Consequently, the addition of MPs amplified CH_4 production. Interestingly, while MPs stimulate N_2O and CH_4 emissions, they do not significantly elevate CO_2 emissions.

In contrast to aquatic environments, studies have demonstrated that MP treatments in soil systems significantly enhance CO_2 emissions [46]. From a microbial perspective, CO_2 production is predominantly driven by aerobic microorganisms, which utilize O_2 as an electron acceptor during respiration. The soil environment, characterized by its porous structure, facilitates greater oxygen availability, thereby creating optimal conditions for aerobic microbial activity. Conversely, aquatic sediment environments are typically more compact and anaerobic [40]. This difference in oxygen availability likely suppresses the activity of aerobic microorganisms, offering a plausible explanation for the lack of CO_2 emission promotion by MPs in aquatic ecosystems. This comparison underscores that the influence of MPs on carbon cycling is contingent on environmental medium, particularly oxygen availability. It further emphasizes the necessity of accounting for such environmental variables when assessing the role of MPs in carbon storage and greenhouse gas emissions.

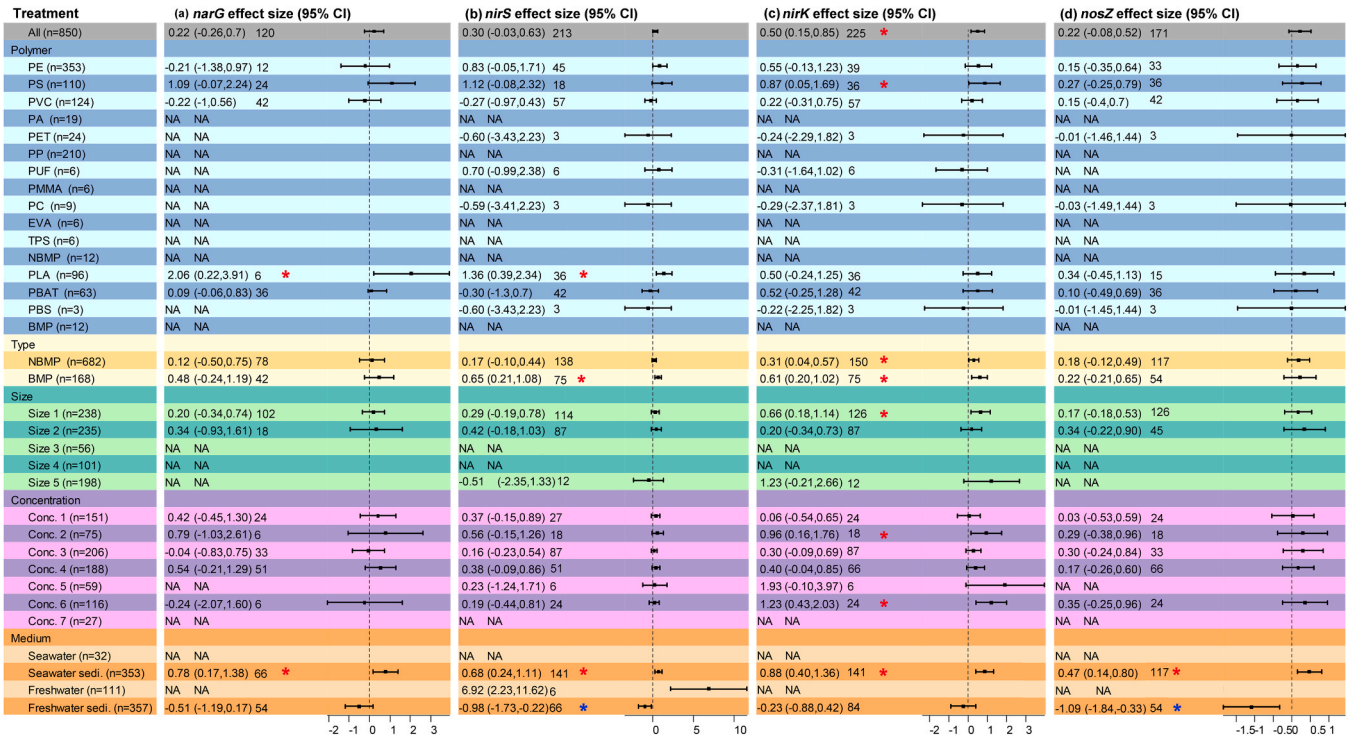

Fig. 4. The overall and subgroup effects on denitrification functional genes are presented. Each data string is represented by the effect size, 95 % CI, sample size, and statistical significance. Red and blue asterisks indicate positive and negative effects of the treatment compared to the control group, respectively ($P < 0.05$). The error bars represent the 95 % CI. Abbreviations refer to Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

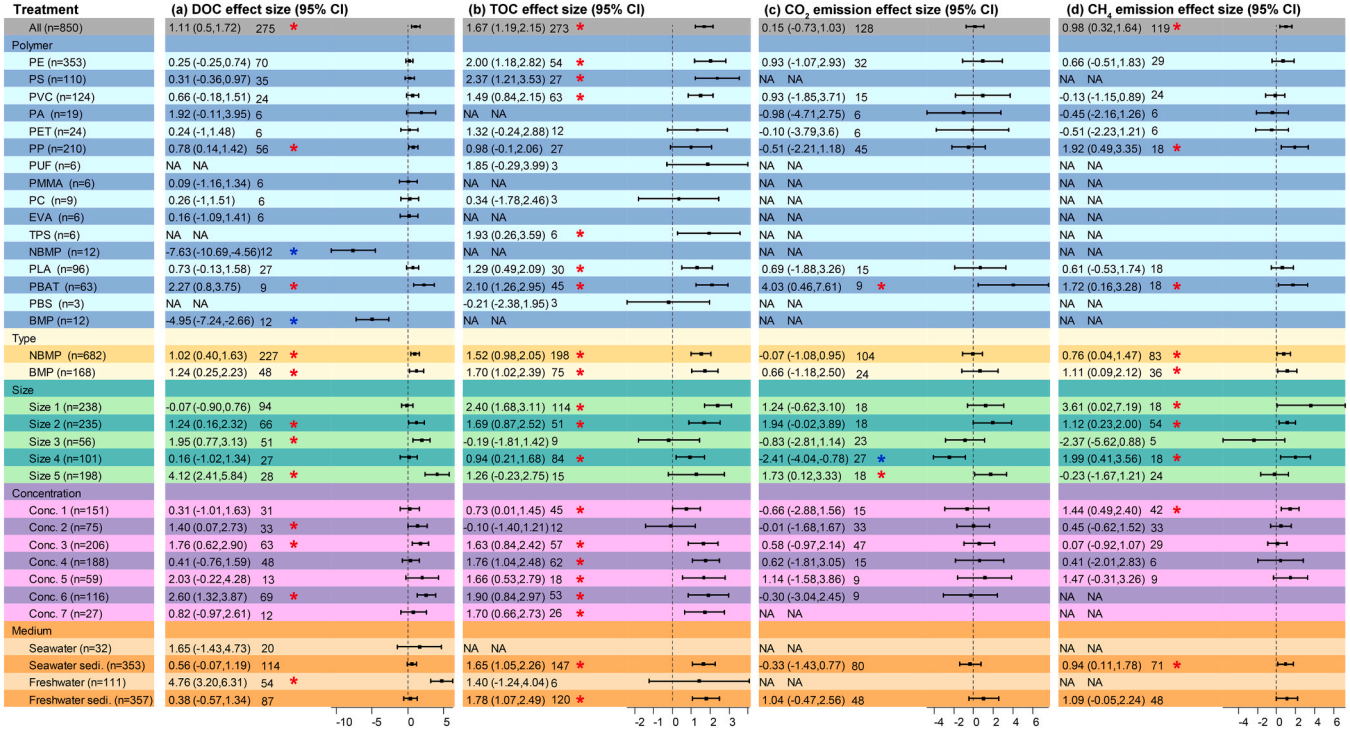

Fig. 5. The overall and subgroup effects on DOC (a), TOC (b), CO_2 (c), and CH_4 (d) are presented. Each data string is represented by the effect size, 95 % CI, sample size, and statistical significance. Red and blue asterisks indicate positive and negative effects of the treatment compared to the control group, respectively ($P < 0.05$). The error bars represent the 95 % CI. Abbreviations refer to Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

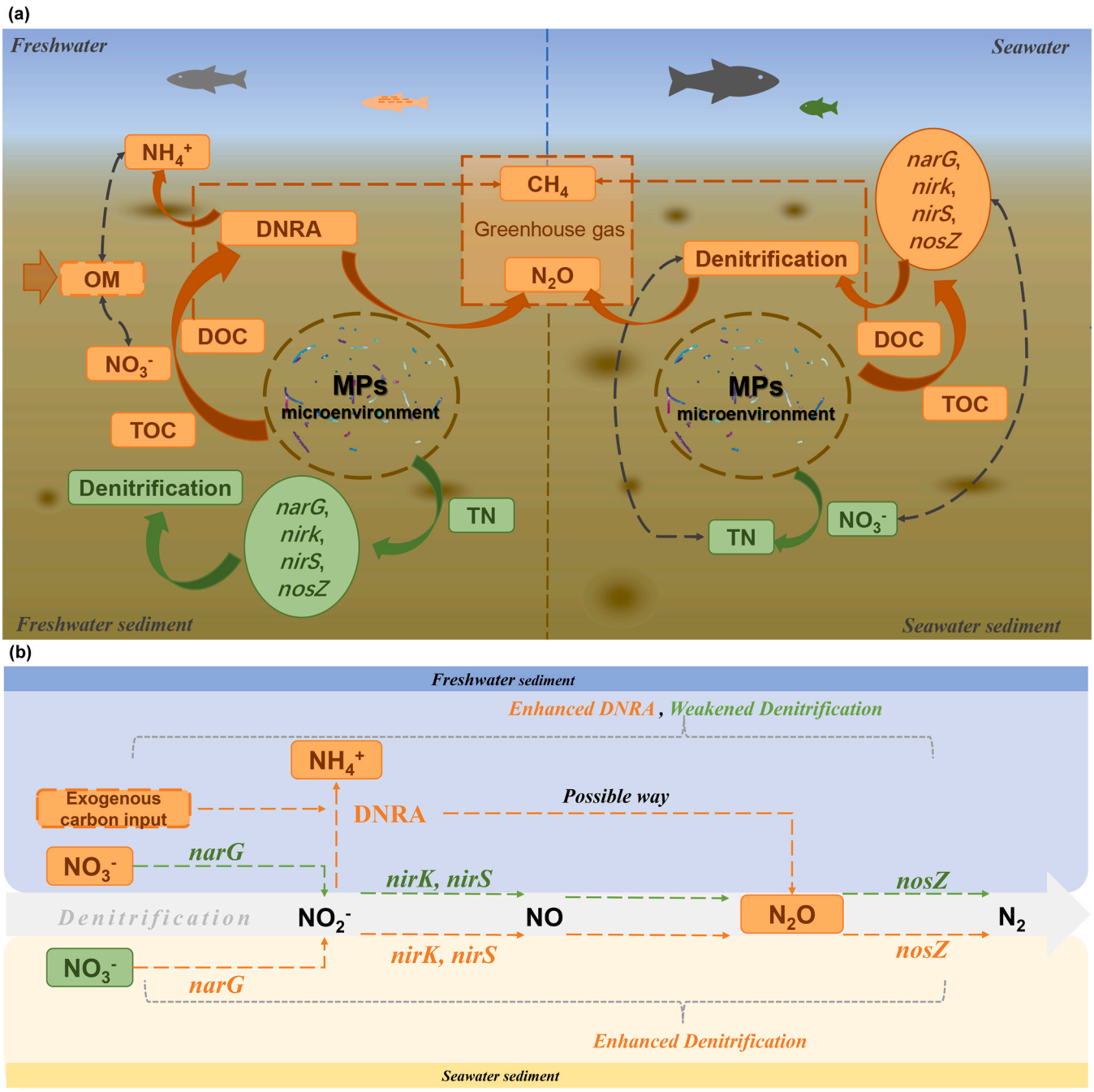

Fig. 6. The conceptual diagram illustrating the mechanisms by which MPs influence aquatic habitats (primarily sediments). Orange and green represent the promoting and inhibitory effects of MPs in specific environmental medium, respectively, while the black arrows indicate the potential close interconnections between these effects. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

4.3. Do MPs only affect carbon and nitrogen cycles?

The biogeochemical cycles of elements in aquatic ecosystems are intricately interlinked, governing the energy balance of these systems, with microorganisms serving as vital drivers of these processes. MPs in aquatic environments are demonstrably reshaping microbial community structures and functions, thereby exerting influence on both carbon and nitrogen cycles. However, a critical question emerges: do MPs solely impact carbon and nitrogen cycling, or do they extend their effects to other biogeochemical cycles as well?

The CH_4 production typically occurs in deeper sediment layers, where strictly anaerobic conditions prevail. However, CH_4 can also be generated in the water column or surface sediments, potentially originating from the gut microbiota of fish or zooplankton, or from marine snow and fecal matter, as these anaerobic microenvironments have been shown to host methanogenic archaea [25,38]. The introduction of MPs into water or sediments may foster the formation of anaerobic microenvironments, thereby promoting CH_4 production and release. This capacity of MPs to create anaerobic conditions may extend beyond CH_4 generation. For instance, studies have documented the aggregation of sulfate-reducing bacteria in seawater sediments and the enhancement of dissimilatory sulfate reduction in the presence of MPs [28,39]. Similarly, other research suggests that MPs facilitate the reduction of Fe(III) to Fe(II) [4]. These phenomena are likely attributable to the anaerobic microenvironments induced by MPs.

It is noteworthy that one of the most significant differences between

freshwater and seawater ecosystems lies in salinity. The low salinity of freshwater allows for lower SO_4^{2-} concentrations, making NO_3^- and Fe(III) critical electron acceptors for microbial respiration in anaerobic sediment environments. However, MPs increase the levels of NO_3^- and NH_4^+ in sediments, rendering Fe(III) oxidation thermodynamically less favorable, at least in surface sediments. Surprisingly, higher CH_4 production rates suggest that MPs exacerbate anaerobic conditions and increase the availability of electron donors in freshwater sediments, implying that the reduction of Fe(III) to Fe(II) may occur simultaneously. This process could enhance the release of reactive phosphorus from sediments, elevating the risk of eutrophication in freshwater [18]. On the other hand, the relatively enclosed systems of lakes, combined with greater inputs of exogenous carbon, may lead to a decrease in pH following the anaerobic metabolism of organic carbon in sediments. Under low pH conditions, iron activity is likely enhanced, potentially enabling Fe(II)/Fe(III) to participate in processes such as nitrogen reduction and sulfide oxidation [12]. MPs addition undoubtedly enhances these potential processes. In contrast, the high salinity of seawater environments typically supports elevated SO_4^{2-} concentrations. In anaerobic seawater sediments, when NO_3^- is largely depleted, SO_4^{2-} becomes a more favorable electron acceptor. Although Fe(III) is thermodynamically a superior electron acceptor, its solid-phase bioavailability and mineral heterogeneity limit its reduction efficiency [7]. Additionally, the byproduct of sulfate reduction, HS^-, can act as a reductant for Fe(III), allowing SO_4^{2-} reduction to remain active [17]. MPs accelerate denitrification and CH_4 release in seawater sediments, suggesting that SO_4^{2-} reduction is likely occurring concurrently. Studies have shown that PLA-MPs significantly enhance sulfate reduction in mangrove sediments, increasing the production of acid-volatile sulfides and elemental sulfur [39].

Nevertheless, while the involvement of MPs in sulfur, iron, and phosphorus cycling is increasingly evident, their direct or indirect mechanisms remain poorly characterized. Several critical questions remain unresolved. For instance, in freshwater sediments, the extent to which MPs promote iron reduction and the subsequent release of sediment-bound phosphorus requires further investigation. In seawater sediments, the differential effects of MPs on sulfate reduction across various sediment layers, as well as their coupling with nitrogen and iron cycling, need to be elucidated.

4.4. The limitations of this study

Although our study provides critical insights into the effects of MPs on biogeochemical cycles in aquatic ecosystems, several limitations must be acknowledged. First, publication bias may lead to an overestimation of MPs' impacts, as studies reporting non-significant or negative results are often underrepresented (Figs. S1–4). Second, the characteristics (e.g., polymer type, size, and shape) and concentrations of MPs used in laboratory experiments frequently differ from those found in natural environments, potentially creating discrepancies between experimental findings and real-world scenarios. Additionally, controlled laboratory conditions may not fully replicate the complexity of natural ecosystems, such as variations in oxygen availability, organic matter content, and microbial diversity, which could influence the true extent of MPs' effects. Finally, the microbial communities and processes studied in laboratory settings may not fully represent those in natural sediments or water columns, limiting the generalizability of the results. Future research should prioritize field studies and more realistic experimental designs to better simulate natural conditions and accurately assess the ecological risks posed by MPs.

5. Conclusion

The MPs exert profound and multifaceted effects on aquatic biogeochemical cycles, influencing carbon, nitrogen dynamics. By altering microbial community structures, MPs enhance processes such as denitrification, DNRA, methanogenesis. In seawater sediments, MPs significantly promote denitrification, while in freshwater sediments, they enhance DNRA processes, leading to increased N_2O production. The release of DOC from MPs provides an additional carbon source for microbial metabolism, further driving these processes. Notably, BMP have a more pronounced impact on carbon and nitrogen metabolism compared to NBMP. These findings underscore the importance of considering environmental medium, such as oxygen availability and organic matter content, when evaluating MPs' role in biogeochemical cycles. Future research should investigate the specific mechanisms by which MPs influence microbial activity and elemental cycling, particularly in anaerobic microenvironments. Additionally, attention should be given to the differential effects of BMP and NBMP to inform the development of sustainable alternatives. Finally, the long-term and large-scale impacts of MPs on carbon storage, nutrient cycling, and greenhouse gas emissions should be assessed.

Environmental implication

Microplastics (MPs) in aquatic environments pose significant environmental risks, impacting not only aquatic organisms but also critical biogeochemical processes like nitrogen and carbon cycling, essential for ecosystem stability and climate regulation. MPs enhance CH_4 and N_2O emissions, potentially exacerbating global warming. They also promote microbial processes in sediments, affecting nutrient dynamics and potentially compromising water quality and ecosystem health. Given the complexity of MP interactions with microbial communities and nutrient cycles, their broader ecological impact must be considered when assessing plastic pollution risks.

CRediT authorship contribution statement

Zhang Xiaoli: Writing – review & editing. **Zhang Weiwei:** Supervision, Methodology. **Xu Jilin:** Supervision, Methodology. **Zhao Jianmin:** Supervision, Conceptualization. **Wang Xiaodan:** Writing – original draft, Visualization, Software, Methodology, Formal analysis, Data curation. **Guo Shuang:** Formal analysis, Data curation. **Wang Qing:** Writing – review & editing, Validation, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This study was supported by the National Natural Science Fund of China (42476148) and the seed project of Yantai Institute of Coastal Zone Research, Chinese Academy of Sciences, Grant No. YICE351010101.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2025.138033.

Data availability

Data will be made available on request.

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