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Life in the "Plastisphere": linking the biodiversity of microbial communities to the risk of micro-(nano-)plastics and related new contaminants

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Life in the "Plastisphere": linking the biodiversity of microbial communities to the risk of micro-(nano-)plastics and related new contaminants

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Editorial: Life in the "Plastisphere": linking the biodiversity of microbial communities to the risk of micro-(nano-)plastics and related new contaminants

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Editorial on the Research Topic
Life in the "Plastisphere": linking the biodiversity of microbial communities to the risk of micro-(nano-)plastics and related new contaminants

Introduction

Micro-(nano-) plastics (MNPs; MPs <5 mm; NPs <1 μm) are a type of emerging environmental contaminant, and they are ubiquitous in various environmental compartments worldwide (Kumar et al., 2021; Wu et al., 2024), including the oceans, intertidal zones, forests, and freshwater systems, and even extend to remote regions across the globe (Wu et al., 2022). These pathways cause severe threats to the structural integrity and functional stability of ecosystems (Wu et al., 2022). The Research Topic compiled a series of innovative research findings in the field of MNP research. Specifically, they contained three core categories of literature: critical review papers (2), perspective articles (2), and original research papers (5). The related research findings delved into the complex attributes of MNP pollution, which are reflected in multiple aspect: Distribution characteristics, migration patterns, and multiple toxicity. The publications specifically involved the diverse sources of MNPs, their environmental migration pathways, as well as the far-reaching impacts they had on the ecological environment, socioeconomics, and human health (Thompson et al., 2004; Wu et al., 2024; Zhu and Huang). These studies not only contribute to gain a deeper and more comprehensive understanding of MNP pollution, and clarifies the research priorities for future in-depth multi-dimensional studies and cross-disciplinary management.

Core consensus

The core consensus of the studies could be categorized into three aspects: First, the overall understanding of MNP pollution were deepen through multi-perspective analyses in each environmental media; Second, the key mechanisms driving MNP transport and their multiple toxic effects were tried to clarified; Third, targeting the urgency and effectiveness of MNP pollution control were discussed and proposed by a series of governance strategies.

The notable contribution firstly lied in its systematic occurrence of multiple environmental media, including marine, freshwater, intertidal zones and forests, which deepening the overall understanding the ecological specificity of MNP pollution in existing multi-perspective analyses. Bel Hassen et al. investigated that MNP with main types of polyethylene (PE), polypropylene (PP), and polystyrene (PS) could be flowed into ocean through land-based inputs, river discharge, marine activities, and atmospheric depositions, and finally accumulated both on the surface water and seafloor. They also found almost 94% of marine plastic debris persisted on the seafloor, much higher than that existing on surface waters. The finding changed the previous view that MNPs mainly contaminate the ocean surface. The threat of MNPs in freshwater systems is equally worthy of attention (Eid et al.; Schubert et al.). Discarded paper cups in the Nile River release ions, heavy metals, and MNPs. These substances can accumulate in aquatic organisms such as fish intestines (Eid et al.) and subsequently enter the food chain, posing potential risks to human health. Another study focused on freshwater bacterial communities (Schubert et al.) and explored the effects of leachates from synthetic polymers on the growth and community dynamics of freshwater bacterial communities. Shabib et al. conducted the bibliometric analysis of MNP research globally and within the coastal region, and call for more attentions on the necessity of investigating the occurrence and distribution of MNPs in specific regions, like coastal zones.

The second insight of this Research Topic concentrated on exploring the cognition of the key driving mechanisms of MNPs transportation. For example, Jingya et al. stimulated the systematic experiments on retention and remobilization behaviors of original and aged PS MNPs in unsaturated porous media. The results showed that the migration capacity of aged PS particles decreased with retention degree, especially after multiple wet-dry cycles. In contrast to the laboratory experiments, Sun et al. constructed a three-dimensional hydrodynamic-Lagrangian particle tracking coupled model to explore the migration and diffusion characteristics of MNPs in Beibu Gulf, and revealed that MNP transportations were extremely complex, which jointly controlled by ocean circulation, monsoons, vertical movement and extreme weather, finally exhibiting complex spatiotemporal variation characteristics. Similar to the above-mentioned results, Huang et al. reported that the terrestrial conditions would also affect MNP migration. Especially in the intertidal zones, MNPs exhibited complex transportation behaviors according to the intertidal types, being roughly categorized as mudflats, sandy beaches, rocky beaches, and biological beaches. Most of the MNPs were stranded in the mudflats under the conduciveness of low water velocity. While in the sandy beaches, MNPs influenced by both the external conditions and intrinsic characters, illustrating drifting, beaching, settling, burying and/or resuspending processes. During the migration process, parts of MNPs could be casted onto and then incorporated into the rocky beaches, resulting in the contamination of plastiglomerate, pyroplastic and plasticrust. Parts of MPs might be ingested by the coastal organisms directly or indirectly. Similar to MNPs in biological beaches, MNP pollution could also have the multifaceted hazards in forest ecosystems (accounting for 32% of the global land area), but with main input pathways of atmospheric deposition (Tao et al.).

The third crucial discovery of the Research Topic mainly focused on the ecological effects of MNPs and related leachates (Schubert et al.). MNP particles cause direct damage to forest ecosystems with the following steps: First, soil microporosity was reduced after the uptake of MNPs, which would disrupt the symbiotic relationships between mycorrhizae and tree roots, and finally leading to micronutrient deficiencies in trees. Through the processes, it could be seen a significant decline in the carbon sequestration capacity and biodiversity levels of forest ecosystems (Tao et al.). The presence of MNPs also inhibited the enzyme activity in forest soils, and then directly hinders critical biogeochemical cycling processes (Tao et al.). In marine environments, Bel Hassen et al. conducted a comprehensive review of the impacts of MNPs on marine organisms on various manifestations, including the reduced reproductive capacity of zooplankton due to the ingestion and accumulation of MNPs in the blubber and acoustic fat pads of cetaceans. This ingestion and accumulation impaired their key physiological functions, such as communication and thermoregulation. This research result further confirmed the disruptive effect of MNPs on ecosystem functions. Eid et al. also confirmed the bioaccumulation of MNPs in Nile River fish inducing the biological health risks through physical damage and chemical toxicity through their cross-trophic effects in the food chain. Moreover, relevant studies estimated a substantial economic losses, especially for the affected industries, fisheries and aquaculture. In addition, leachates could also cause impacts on free-living bacterial communities in freshwater environments, during which the total bacterial biomass was stable through the weathering process. Another important phenomenon that chemical composition could influence the free-living aquatic bacteria, resulting in altering microbial loop functioning and biochemical cycles of aquatic ecosystems.

The relevant studies in this Research Topic also remind a series of governance strategies for the urgency and effectiveness of MNP pollution requiring interdisciplinary collaboration, policy innovation, and public participation among other elements. Traditional treatments for plastic wastes contained landfilling and incineration, taking the proportion about 40% and 25%, respectively. Landfilling needed large space along with releasing large numbers of MNPs under the weathering processes, while incineration would also discharge abundant MNPs and other hazardous substances (e.g., dioxins) into the atmosphere (Zhu and Huang; Bel Hassen et al.). Especially for the surge of plastic waste during/after the COVID-19 pandemic, they advocated circular economy frameworks as a new pathway for whole-lifecycle supervision of plastics, covering all stages of their production, use, and disposal. Moreover, Huang et al. further explored the artificial intelligence-based MNP mapping tools to track the sources, spatiotemporal distribution and migration paths of MNPs, thereby identifying the pollution hotspots, driving mechanisms and ecological risk correlations, finally providing scientific support for precise governance. The targeted policy recommendations should be discussed and conducted for the formulation of forest-specific monitoring protocols (Tao et al.), including subsidizing biodegradable agricultural mulch films, upgrading the capture systems of submicron particles in sewage treatment plants, integrating MNP monitoring network into the protected areas, and promoting reusable alternatives. Particularly in the post-pandemic era, policies needed to balance public health and sustainability goals, such as levying plastic taxes to fund recycling infrastructure, highlighting the application potential of thermochemical recycling technologies and emphasizing the bio-based alternative materials (Zhu and Huang). Advocating the citizens to strengthen public educational efforts regarding the environmental risks of hidden plastics (Shabib et al.; Eid et al.). All studies called for the establishment of a more robust and collaborative manage system. The aforementioned measures can not only reduce MNP input but also foster a shared sense of responsibility among the public for planetary health.

Conclusion

This article systematically discussed the extensive presence characteristics, complex migration patterns and multi-dimensional hazards of MNPs in various environmental media such as the marine, freshwater and forest ecosystems. In the ocean, most plastic waste were finally found accumulating on the seabed, while the fate of MNPs together with the related leachate in freshwater generally interfere with the structure and function of free-living bacterial communities, and then threaten the human health. In contrast, MNPs in intertidal zones showed various migration patterns according types of coastlines, and caused directly or indirectly effects after being ingested. In forest ecosystems, MNPs mainly entered through atmospheric deposition, disrupting soil microbial communities and carbon cycling functions. To reduce the risk of such pollution, scientists, policymakers and the public had to work together to transform existing scientific insights into practical governance outcomes. For scientists, future efforts should be paid on developing sustainable plastic transformation techniques, including thermochemical recycling and biological degradation technologies. For governments, related environmentally friendly policies should be enacted for the treatment of waste segregation standards under environmental monitoring networks. Meanwhile, governments also had sought to minimize plastic use by raising public awareness of the. Through outreach and advocacy, the public awareness on the negative consequences of MNPs pollution should be promoted, and should be encouraged to reduce plastic consumption through daily life decisions such as giving up plastic consumer products, applying reusable materials and recycling plastic wastes for multiple purposes. The Research Topic not only contribute to gain a deeper and more comprehensive understanding on the occurrence, distribution, fate, and ecotoxicity of the MNPs in oceans, intertidal zones, forests, and freshwater systems, but also urging more in-depth multi-dimensional techniques and cross-disciplinary managements on the treatments of plastic wastes.

Author contributions

SZ: Writing – original draft, Writing – review and editing. HT: Writing – review and editing. XZ: Writing – review and editing, Writing – original draft. TY: Writing – review and editing. GC: Writing – review and editing, Supervision. PW: Writing – review and editing, Supervision.

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Microplastic research trends in the Gulf region from a global perspective

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Introduction: Microplastic (MP) pollution research has experienced significant global growth, with an exponential increase in publications since 2015. Despite this surge, research focused on the Gulf region remains limited. This gap is particularly concerning given the Gulf's dense industrial activities and substantial waste production. This study provides the first bibliometric analysis of MP research globally and within the Gulf region, examining the current state and emerging trends while identifying specific topics for future studies.

Methods: A bibliometric analysis of peer-reviewed articles published between 2000 and 2024 was conducted for global and Gulf region research. Bibliometric analysis employed several tools to identify trends, keyword networks, and research gaps, with manual refinement of keywords to enhance accuracy. The study also analyzed leading countries and institutions contributing to MP research.

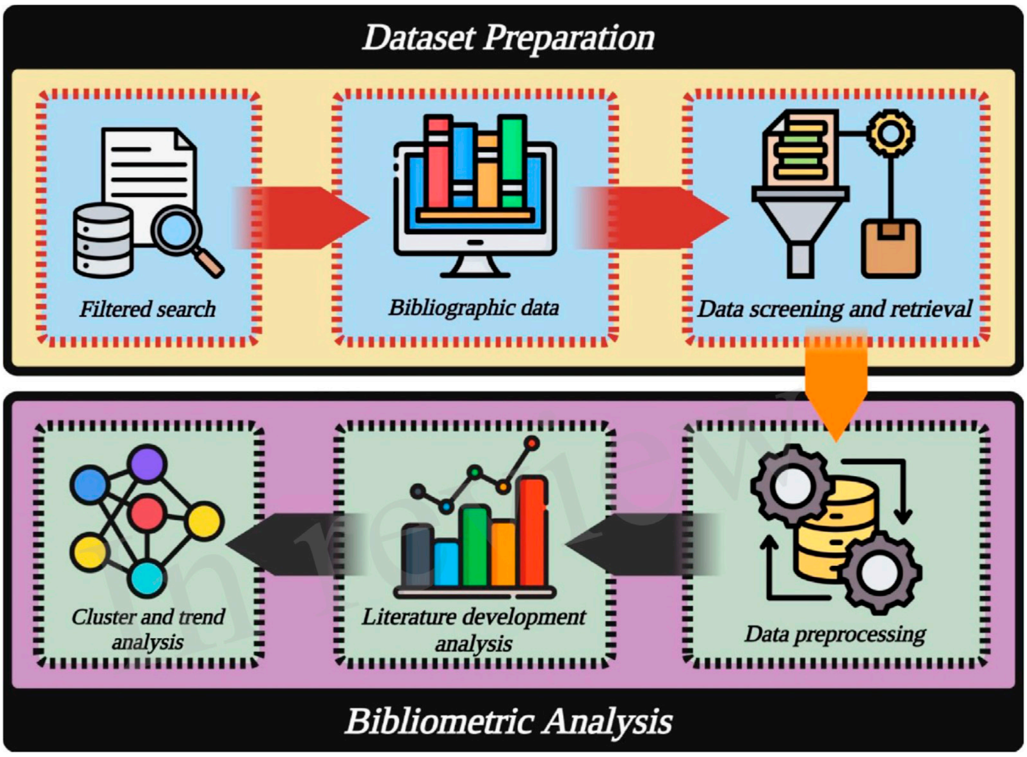
Results and Discussion: The analysis highlights significant global contributions, with China emerging as a leader in MP research globally and Iran playing a prominent role within the Gulf region. While global publications have increased exponentially since 2015, the Gulf region's research output remains underrepresented, accounting for only about 1.1% of global publications. This underscores the need for enhanced regional research to better integrate the Gulf into the global discourse on MPs. Keyword analysis reveals that certain areas of MP research remain unexplored in the Gulf region. The trend analysis shows an evolving focus globally, shifting from basic interactions of MPs with marine environments to more complex issues such as environmental health and ecosystem impacts. This progression indicates a maturing field that increasingly addresses the broader implications of MP pollution.

KEYWORDS

microplastic, Gulf region, bibliometric analysis, cluster analysis, marine pollution

1 Introduction

Pervasive occurrence and detection of microplastics (MPs) in a vast array of environmental matrices has ignited a novel, serious environmental concern around which a tremendously growing body of research has been formed in recent years (Thompson et al., 2009; Cole et al., 2011; Li et al., 2018). Studies have mentioned earlier the potential of MPs being transported to even the most distant marine



GRAPHICAL ABSTRACT

environments to pose a threat to the aquatic and human lives (Horton and Barnes, 2020), making MPs a hazard of large scale impact (Prata et al., 2019). The current state and prospects of plastic production and consumption further highlight the importance of addressing this issue with urgency. Due to the wide scale of non-point sources of MPs (He et al., 2022; Tan et al., 2022; Jiang et al., 2024), there is a need for a multidisciplinary, comprehensive understanding of the current state of MP pollution to carve out promising solutions to the problem at hand.

Although there have been numerous studies focusing on MP pollution in different types of matrices, the number of studies exclusively focused on gulf areas—particularly the region in Western Asia (commonly recognized as the Gulf region)—has not seen relatively equal attention. Due to their commonly dense areas of commercial and industrial activities such as shipping traffic, aquaculture, and petroleum activities, coastal tourism, urban runoff, and wastewater discharge, gulf areas are prone to intense loads of MP pollution, which makes them the ultimate downstream location that offers a great chance to understand the dynamics of MP pollution. Despite these motivations for further research, gulf areas remain underrepresented in MP research, which may root in the complex nature of pathways and interactions inherent in MP pollution in marine environments (Ivar do Sul and Costa, 2014). This underrepresentation may stem from factors such as the high unpredictability of environmental conditions (Alfaro-Núñez et al., 2021) and the inadequacy of sampling methods used in marine samplings (Razeghi et al., 2021). One of the main reasons for this is the relatively small size of the samples that are used and reported for representing the state of pollution in these comparatively large areas (Harris, 2020; Thushari and Senevirathna, 2020; Alfaro-Núñez et al., 2021).

This is particularly critical, specifically in the Gulf region due to rapid urbanization and population growth in coastal areas, as well as high plastic consumption. As reported by Stöfen-O′Brien et al. (2022), mismanaged plastic waste streams are a major source of MP pollution, especially in Gulf region surrounded by centers of population producing 1.33 million metric tons of plastic waste annually. All of the above-mentioned factors point out the need for comprehensive and coordinated research in the Gulf region in order to gain a better understanding of the present state of MP pollution.

Prior to taking any further steps to approach the question at hand empirically, it is essential to obtain a dependable understanding about the present literature and establish a picture of the current state of knowledge regarding MP pollution in the Gulf region. Conventional methods of achieving this task usually rely on widespread investigation of manually selected resources, which has a good chance of missing valuable information, hence yielding a partially skewed understanding. One of the most robust and accurate approaches to overcoming this is bibliometric analysis, which is a powerful tool to identify trends, patterns, the current state, and future directions of research on a particular subject (Ellegaard and Wallin, 2015). Using comprehensive scholarly literature databases, and through a complete investigation of connections between authors, researchers, countries, and keywords of publications, bibliometric analysis can reveal new information that cannot be unlocked through conventional exploration of publications.

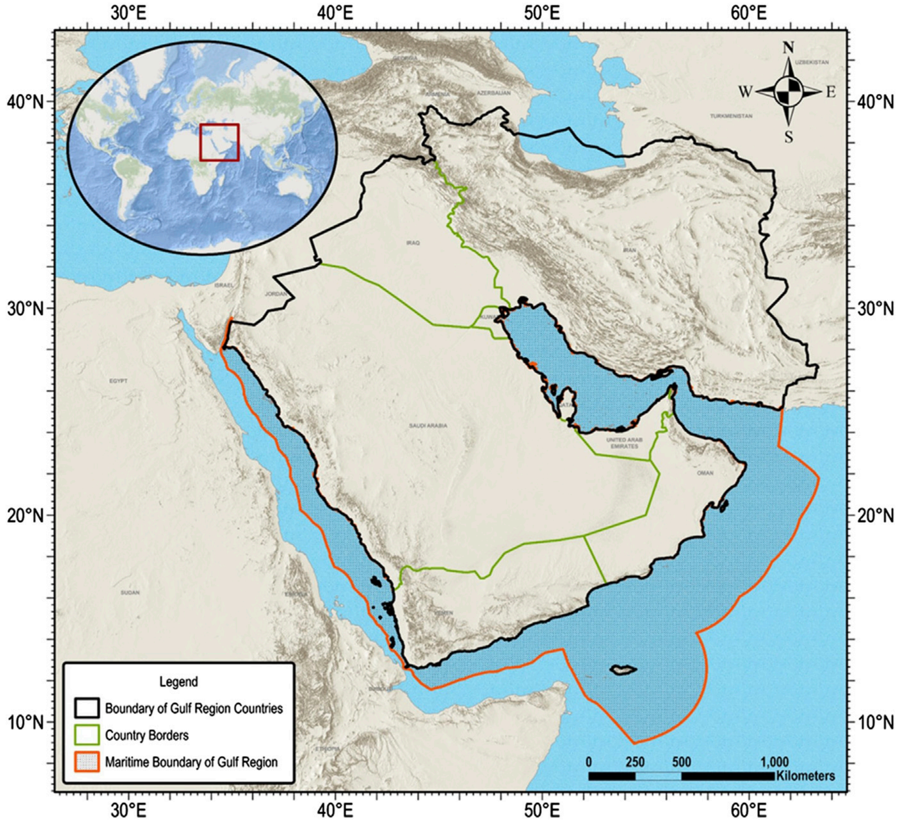


FIGURE 1
Boarders of the Gulf region studied by the present bibliometric analysis.

This study aims to systematically explore the current state of research on MPs both globally and in the Gulf region, revealing the current research status and identifying gaps to be addressed by future studies. The primary aim of incorporating global data was to provide a comparative framework that highlights the relative scarcity and scope of research specifically within the Gulf region. Based on the outputs, different aspects are analyzed for both research topics, including publication outputs, citation patterns, network analyses of authors and institutions, as well as trend analyses of topics. This research is crucial for addressing the environmental risks posed by MPs to marine biodiversity and the safety of the seafood supply, as evidenced by MP ingestion in commercial fish species and potential bioaccumulation through marine food webs that directly impact both ecosystem health and seafood security. The main objectives of this study include identifying the significant research gap present in the Gulf region regarding MPs, as well as the difference in tracks between global focus and the Gulf region's focus on MP research.

2 Materials and methods

2.1 Gulf region study area

The study area for the bibliometric analysis of MP research in the Gulf region encompasses a significant portion of Western Asia, specifically highlighted by the marine and coastal environments critical to understanding the regional impact of MPs pollution. This region, as shown in Figure 1, includes four main areas of Persian Gulf, Gulf of Oman, Gulf of Aden, and the Red Sea, along with the countries with maritime boundaries in this region: the Kingdom of Saudi Arabia (KSA), Kuwait, Bahrain, Qatar, the United Arab Emirates (UAE), Oman, Iraq, Yemen, and Iran. These countries and water bodies constitute the broadest definition of the Gulf region, including countries of the Gulf Cooperation Council (GCC) and countries along the coastline of the Persian Gulf. This geographical scope is particularly relevant given the ecological, economic, and sociopolitical importance of these areas. It should be noted that countries on the western bank of the Red Sea, such as Egypt, Sudan, Eritrea, and Djibouti, were excluded from the analysis to maintain a more definitive scope of the Gulf region.

The Gulf region is a vital corridor for global oil shipping and possesses rich marine biodiversity that is highly susceptible to pollution exports from inland areas, including MPs (Samimi-Namin and Hoeksema, 2023). The region's marine environment supports crucial economic activities that are particularly vulnerable to MP pollution. The fishing industry in the region faces direct threats from MPs through potential contamination of commercial species (Kibria, 2023). Moreover, the industrial sector in the Gulf region is heavily dependent on seawater for various processes, making it vulnerable to MP pollution. Desalination plants, which are pivotal in supplying freshwater needs in the region, include membrane water treatment technologies that concentrate MPs on the membrane and in the concentrate and their disposal channels

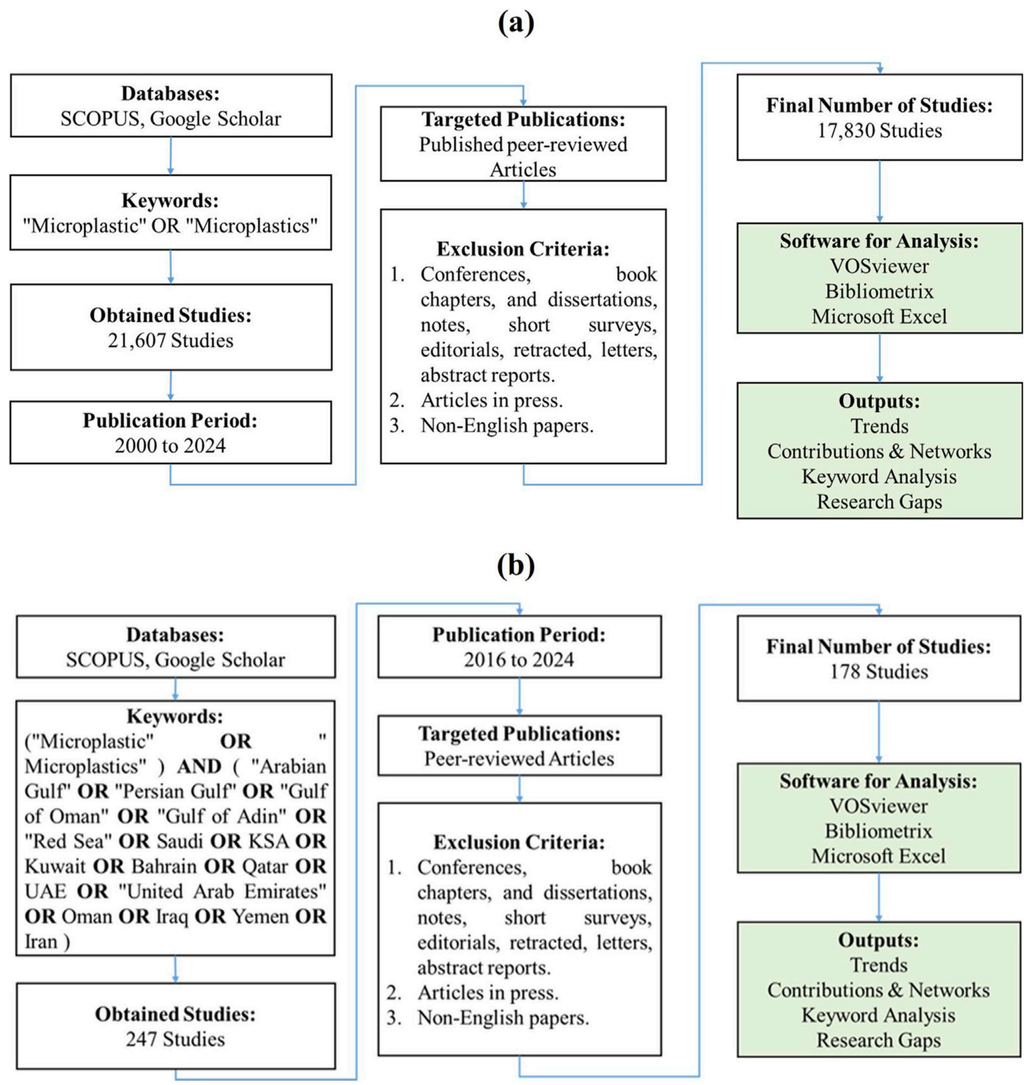


FIGURE 2
Bibliometric analysis framework of published articles on MPs globally **(A)** and in the Gulf region **(B)**.

MPs back to the environment (Tang and Hadibarata, 2021). Other marine-dependent industries, such as power generation facilities using seawater for cooling, petrochemical plants requiring process water, and coastal manufacturing units, are similarly affected by MP pollution through impacts on water intake systems and equipment deterioration. Therefore, focusing on the Gulf region for MP research addresses not only an environmental issue but also a critical economic concern affecting regional food security, industrial operations, and public health. The transboundary nature of MP pollution in these interconnected water bodies underscores the need for comprehensive regional studies and coordinated mitigation strategies.

2.2 Bibliometric analysis

Figure 2 depicts the flowchart summarizing the bibliometric analysis of MP research at both global and Gulf region scales. The bibliometric analysis framework summarizes the utilized databases, keywords, targeted publication period, exclusion criteria, filtering stages, total studies covered, and analysis tools. Two separate databases were obtained in the present study, one corresponding to the global scale analysis, while the other corresponding to the Gulf region. In the first step, for both frameworks, the articles of interest were searched for through SCOPUS and Google Scholar databases, to obtain the largest number of published papers that could relate to the present topic without applying any exclusion criterion. After obtaining the initial number of studies, conferences, book chapters, dissertations, notes, short surveys, editorials, articles in press, and non-English papers were excluded from the search results. The obtained articles were then analyzed using different software, such as VOSviewer, R-Studio (Bibliometrix), and Microsoft Excel, for trends, keywords analysis, research gaps, and different contributions.

For the first step of bibliometric analysis related to global MP research (Figure 2A), the keywords used for searching the articles, mainly through the titles, keywords, and abstract, included the terms "Microplastic" OR "Microplastics", to obtain an overview of globally conducted research regarding MPs, which returned 21,607 articles. Following, the filtering step was initiated, retaining only publications in the period of 2000–2024. Finally, after applying the exclusion criteria mentioned previously, the total number of studies was limited to 17,830 English, peer-reviewed journal studies. On the other hand, the keywords used for the second bibliometric analysis focusing on the Gulf region (Figure 2B) were included in the following search query: ("Microplastic" OR "Microplastics") AND ("Arabian Gulf" OR "Persian Gulf" OR "Gulf of Oman" OR "Gulf of Aden" OR "Red Sea" OR Saudi OR KSA OR Kuwait OR Bahrain OR Qatar OR UAE OR "United Arab Emirates" OR Oman OR Iraq OR Yemen OR Iran). Initially, 247 studies were obtained, all published during the period of 2016–2024; hence, filtering of the publication period was not necessary, and the final number of studies obtained after applying other exclusion criteria was 178 studies.

The analysis of keywords network reveals smaller subjects within a certain topic, allowing for a detailed examination of research patterns and the topical structure of the larger subject at hand. VOSviewer is a powerful tool for this purpose, establishing the keyword network based on the co-occurrence of keywords in different studies. By analyzing the frequency of co-occurrence of two keywords in the literature, smaller topical zones that indicate the general focus of their representative articles can be identified. Identifying these smaller topical zones in MP research on both global and regional scales (*e.g.,* the Gulf region) reveals relatively or completely untouched topics that have not been addressed by regional studies, thus providing invaluable suggestions for future research.

Before inputting the database into VOSviewer, several additional steps were conducted manually to ensure a more accurate depiction and analysis of the keyword network. As mentioned in previous sections, some fundamental keywords shared by almost all the extracted studies were used to obtain the data file for this analysis. Therefore, in the first manual step, these keywords were identified and omitted from the network analysis (Supplementary Table S1). Additionally, instances of differently phrased but synonymous keywords (e.g., "human" vs "humans" or "ocean pollution" vs "marine pollution") were identified (Supplementary Table S2). Since VOSviewer uses the co-occurrence of keywords as the criterion for establishing connections and depicting the keyword network, it is essential to merge these similar keywords before conducting a meaningful analysis.

3 Results and discussion

3.1 Annual publications

The first output of the bibliometric analysis is the annual number of publications, which provides a quantitative comparison of research outputs between global and regional research conducted on MPs. Figure 3A shows the year-wise distribution of publications for global and Gulf region research. The global research on MPs, spanning 2000 to 2023 (2024 was excluded as it is not complete and the publications are up to July only), illustrated an overall significant increase in publication numbers. Initially, publication numbers were mostly in single digits until 2008, but a significant rise began in 2015, with the largest increases seen after 2018. The peak of this trend occurred in 2023 with 4,826 publications, highlighting a marked surge in global interest and research activity concerning MPs. In contrast, research on MPs specifically in the Gulf region, started in 2016 and showed a steady increase in publications, albeit at considerably lower levels. The regional research peaked at 47 publications in 2023, indicating a growing, yet more contained, research interest within the Gulf region.

The Gulf region's research output, representing only 1.1% of global MP publications, appears disproportionately low given the region's vulnerability to MP pollution. This relatively limited research contribution stands in contrast to other comparable marine regions, as evidenced by subsequent analysis of leading countries' research outputs, suggesting potential opportunities for expanded research efforts in the Gulf. Both patterns demonstrate increasing trends, yet the global data reveals a more robust exponential growth. Interestingly, both scales witnessed their highest publication counts in 2023, indicating the increased interest in research regarding the topic. This comparative analysis highlights the escalating global and regional focus on MPs, showcasing varying scales of research engagement and intensities.

3.2 Average annual citations

Information regarding the influence of a publication on a scientific field can be obtained through citation pattern analysis. Furthermore, the development and growth structure within a field of research can be shown through the analysis of citation patterns (Bornmann and Daniel, 2008). Citation data and metrics derived from citations serve as a standard framework for comparing the relative effectiveness and impact of journals within a field of research. The average number of citations per year is calculated by dividing the total number of citations by the number of years the author or journal has been publishing papers. This can be a very useful metric to assess the yearly impact for a journal or author. However, since older publications have more time to receive recognition, it is essential to normalize citation scores to obtain an accurate picture. This normalization is achieved by dividing the total number of citations each publication has received by the number of years it has had to receive citations (up to 2023). Figure 3B shows variations in annual average citations per publication from 2000 to 2023 both globally and in the Gulf region. Globally, citations gradually increased until 2009, followed by a notable spike in 2011. This indicates that the work during these years was particularly impactful within the field. After 2011, yearly citations show a steady growth, suggesting sustained attraction towards the research topic at hand, peaking at 7 citations in 2022. In contrast, research on MPs in the Gulf region started in 2016, showing a swift ascent in citations, peaking sharply at 7.8 in 2018. This peak suggests that studies from this period were highly referenced, perhaps due to groundbreaking findings. From 2019, it is evident that Gulf region MP studies have received a similar level of recognition despite being at early stages of research development.

Comparatively, both patterns exhibit a peak in citations followed by a decline, reflecting a common lifecycle in academic citations where studies initially gain significant attention before stabilizing.

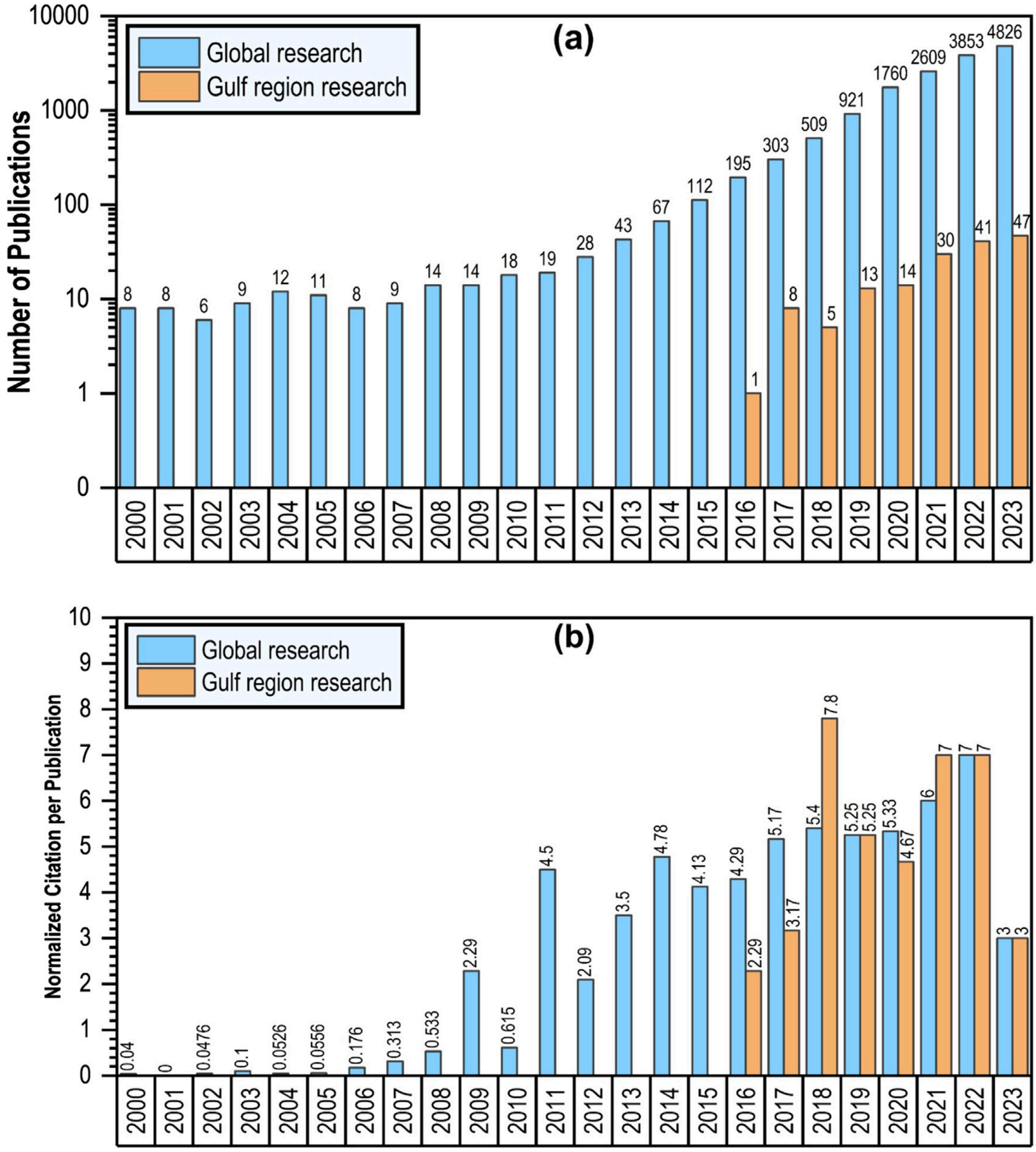


FIGURE 3
Year-wise distribution of publications on MPs globally and in the Gulf region since 2000 **(A)**, and average annual citations for MPs research globally and in the Gulf region since 2000 **(B)**.

The global research data shows a steadier increase and a longer peak, indicative of a gradual buildup of foundational research that has influenced subsequent studies over a more extended period. Conversely, MP research in the Gulf region displays a more sudden rise in number, which may indicate a rapid uptake and influence of global findings or focused impactful studies that sparked new research efforts.

3.3 Leading authors

Table 1 highlights the leading authors in publications related to MPs both globally and in the Gulf region. Globally, there are 38,506 authors with publications on MPs, which is a significantly huge number compared to the 558 authors that published on the same topic within the borders of the Gulf region. The leading global author on MPs, Wang J., accounts for only 0.5% (525 documents) of the total publications. In contrast, the leading author in research on MPs in the Gulf region, *i.e.*, Abbasi S., contributes to around 4.5% (25 documents) of the total research publications. The average number of authors per publication provides useful insight into the level of collaboration across various studies. The average number of authors per publication on the global scale was 2.16; however, in the Gulf region, this value rises to 3.13, indicating a higher level of collaboration in the field of MP research.

TABLE 1 The list of leading authors in MP research both globally and in the Gulf region.

Global research on MP			MP research in the Gulf region		
Author	Number of documents	Contribution (%)	Author	Number of documents	Contribution (%)
Wang J	525	0.50	Abbasi S	25	4.48
Zhang Y	506	0.48	Keshavarzi B	18	3.23
Wang Y	469	0.45	Moore F	18	3.23
Li Y	449	0.43	Turner A	17	3.05
Wang X	367	0.35	Dobaradaran S	10	1.79
Li J	336	0.33	Busquets R	9	1.61
Liu Y	323	0.31	Rezaei M	9	1.61
Li X	270	0.26	Akhbarizadeh R	8	1.43
Wang Z	264	0.26	Duarte CM	8	1.43
Liu X	250	0.25	Naji A	8	1.43

3.4 Leading countries

Assessing the geographical distribution of publications in a research field provides critical insights into the global research landscape, revealing both regional focuses and international collaborations. This analysis not only highlights the leading contributors but also portrays the extent of intra- and inter-country collaborations that strengthen the development of related technologies. Table 2 outlines the leading 10 countries based on the number of publications according to Scopus database for both global and regional research conducted on MPs. It is evident that counties are not confined to collaborations within the same country, as all countries have multiple-country publications (MCPs), indicating collaboration between authors from different countries. In the global research landscape on MPs, China dominates with 5,735 documents, featuring a high level of international collaboration with 1,294 MCPs. This is followed by the United States and India, with substantial but considerably lower contributions. European nations such as Italy, Germany, and Spain also show strong involvement, particularly Italy, which demonstrates significant international collaboration. Other noteworthy contributors include South Korea, the United Kingdom, Australia, and Canada, all displaying robust research outputs and collaboration with other contributing countries. In contrast, research in the Gulf region is led by Iran, with 123 documents and 59 MCPs, followed by smaller contributions from the United Kingdom and Saudi Arabia. Other regional participants like Poland, the United States, Germany, the United Arab Emirates, Kuwait, Netherlands, and Egypt show varied levels of involvement and collaboration, with overall smaller research outputs compared to the global scene.

Based on Table 2, there is a clear disparity in the volume and scale of research activities. On the global scale, China is spearheading the MP research. Conversely, the Gulf region, led by Iran, presents a more modest but still collaborative research environment. The presence of countries outside of the Gulf region indicates international interest extending into MP research in the Gulf region. This highlights not only the different scales of research activities between the two areas but also the distinct focus and emerging significance of regional environmental research in the Gulf, contrasting with the extensive, established research networks seen globally.

The leading place of China in MP research could be explained by its significant industrial activities and associated pollution concerns, given its status as a major plastic producer and consumer. The extensive maritime boundaries of the country necessitate understanding the impact of MPs on marine ecosystems, crucial for maintaining the health of its fishing, shipping, and tourism industries. Moreover, leading MP research enhances geopolitical influence of this country, allowing it to shape international environmental policies. This aligns with the broader goals of demonstrating global leadership and advancing technological and scientific capabilities, particularly in developing new pollution control methods and recycling technologies.

In the Gulf region, the significant interest of Iran in research on MPs in the Gulf region originates from the need to protect the critical marine ecosystems of the Persian Gulf, which are vital for the fishing industry and overall economic growth of the country. This research is crucial for addressing the environmental risks posed by MPs to marine biodiversity and the safety of the seafood supply due to the impacts of MP ingestion on commercial fish species and potential bioaccumulation through marine food webs that would directly impact both ecosystem health and seafood security in the Gulf region (Kibria, 2023). The interest of the Gulf countries such as Iran and Saudi Arabia in researching MPs in the Gulf region reflects a shared regional concern, as the Persian Gulf's marine ecosystems are crucial for local economies and biodiversity. Countries bordering the Persian Gulf account for approximately 50% of global desalination capacity (Ibrahim et al., 2020), which makes water quality of great importance to these countries. By leading studies in this area, Iran not only addresses public health concerns related to environmental pollutants but also positions itself as a regional leader in environmental studies.

3.5 Leading funding programs and agencies

3.5.1 Leading funding agencies on global MP research

Analysis of the performance of funding and sponsorship agencies is vital for a comprehensive investigation into a specific research topic. Studies funded through certain programs tend to be

TABLE 2 The leading 10 countries according to the number of publications on MP research globally and in the Gulf region.

Global research on MP			MP research in the Gulf region		
Country	Number of documents	MCPs[a]	Country	Number of documents	MCPs
China	5,735	1,294	Iran	123	59
United States	907	229	United Kingdom	31	26
India	803	228	Saudi Arabia	20	6
Italy	795	247	Poland	14	11
Germany	782	218	United States	13	9
South Korea	581	121	Germany	12	10
Spain	564	168	United Arab Emirates	11	3
United Kingdom	526	194	Kuwait	10	7
Australia	396	178	Netherlands	10	7
Canada	396	141	Egypt	8	2

[a]Multiple-country publications (MCPs).

framed within a more strategic and meaningful framework, often necessitated by the funding agency. Additionally, financially supported studies are more likely to receive attention from the scientific community due to their generally higher quality, made possible by access to laboratory facilities and more rigorous sampling methods. Previous studies have shown a correlation between the average number of citations a paper receives and the amount of funding it has been granted (Wang and Shapira, 2011).

A deeper look into the programs providing funding support reveals the concerns driving these expenditures. Table 3 shows the performance of top countries in terms of funding global MP research, executed through smaller funding agencies, with the top 10 agencies shown in Supplementary Table S3. The results indicate that global MP research is predominantly funded through public programs. These funding agencies have collectively sponsored 66.5% of studies, while the rest have been conducted by independent researchers. According to Table 3, China has financially dominated the international stage through 55 public funding agencies that have supported 4,835 studies (30%) of global research on MPs and is responsible for 32% of global publications, highlighting the strong financial support behind Chinese institutions to drive global MP research. The European Union ranks second, with several developed countries collectively accounting for 6.6% of funding for MP research. The significant disparity in research outputs between China and other regions, such as the European Union and the United States, reflects not only the volume of funding but also the strategic prioritization of environmental research in these regions.

The substantial leadership of Chinese institutions is attributed to the serious water pollution challenges China has long battled through policies and regulations, such as “The Water Pollution Law,” “The Water Law,” “The Soil and Water Conservation Law,” and approximately eight other initiatives focusing on marine pollution, reflecting an extensive regulatory framework that began in 1988 with the “Marine Environment Protection Law of the People’s Republic of China (Marine Law).” This regulatory environment has galvanized Chinese institutions to prioritize MP research. Policies such as China’s “Marine Environment Protection Law” and the EU’s “Plastics Strategy” play a substantial role in aligning research priorities with national and regional environmental concerns (European Commission, 2019). While these policies provide a structured framework that may attract funding and support larger-scale research initiatives, the impact of these policies varies based on regional priorities and available resources. Previous studies indicate that China’s significant contributions to global plastic pollution mitigation are shaped by both international scrutiny and domestic necessity, with its legislative efforts bolstering research capabilities and regulatory effectiveness (Liu et al., 2022).

China’s regulatory framework has evolved from addressing single issues, such as banning plastic bags in 2007, to a comprehensive system targeting the entire plastic lifecycle. This includes policies focused on collection, recycling, and reuse, which have increasingly emphasized circular economy principles. Notably, China has implemented 122 policies that explicitly target MPs, reflecting the growing recognition of MPs as a critical environmental issue (Fürst and Feng, 2022). The 2017 national program for MP pollution research marked a significant milestone, further integrating MPs into broader regulatory and environmental protection strategies.

A detailed investigation into China’s funding mechanisms reveals that policy-driven initiatives often align with increased funding allocations, though further research could clarify the specific influence on study quality and research outcomes. The participation of various departments and institutions across provinces (*e.g.,* Shandong, Jiangsu, Zhejiang, Guangdong, Fujian, Hainan, Shandong, Hubei, Hunan, Hebei, Heilongjiang, Guangxi, Jilin, and Jiangxi, respectively, in terms of the number of sponsored studies) along with national-level funding programs in the funding agency information confirms a collective national effort in MP research development. Analysis of the current state of MP research in China suggests that the regulatory environment will likely lead to even more restrictive policies on plastic waste and MP pollution in the future (Xu et al., 2021).

TABLE 3 Performance of countries in funding research on MPs both globally and in the Gulf region.

Global research on MP			MP research in the Gulf region		
Country	Number of funding agencies	Number of studies funded[a]	Country	Number of funding agencies	Number of studies funded[a]
China	55	4,835	Iran	35	78
European Union	11	1,076	Saudi Arabia	9	17
United States	11	572	Kuwait	3	6
South Korea	9	510	United States	8	4
Germany	4	463	United Arab Emirates	3	4
Spain	6	335	Israel	2	4
Brazil	4	319	United Kingdom	5	4
United Kingdom	5	297	China	3	2
Portugal	4	273	Oman	2	2
Canada	6	262	Egypt	2	2

[a]Some studies are co-funded collaboratively by several funding agencies.

On the other hand, the commitment of the European Union to financially support MP research arises from its long-standing dedication to plastic pollution prevention and control on a regional level by aiming at plastic pollution directly or indirectly through programs such as "Plastics strategy," "REACH (Registration, Evaluation, Authorization and Restriction of Chemicals) Regulation," "Marine Strategy Framework Directive (MSFD)," "Horizon 2020 and Horizon Europe Programs," "JPI Oceans MPs Projects (€18 million)," "Single-Use Plastics Directive," and "Circular Economy Action Plan." The collaborative funding approach within the European Union, fostering cross-border research initiatives, has significantly enriched the scientific literature on MPs. The unique success of this collective effort lies in the encouragement of different universities from various European countries to collaborate on their projects, which fertilizes literature enrichment through knowledge mobilization.

3.5.2 Leading funding agencies in gulf region

The sponsorship data on MP research in the Gulf region show that 74% of studies in the region are financially supported by various sponsorship programs. The Gulf region faces unique challenges in combating MP pollution due to its extensive coastline and high plastic usage, necessitating increased funding and coordinated research efforts to develop effective mitigation strategies. The financial resources required for conducting a conventional MP analysis study arise from significant costs associated with sampling, laboratory equipment, and personnel requirements. Meyers et al. (2024) estimated the costs of conducting seawater MP analysis to range from $850 for purely microscopy-based methods to $9,100 per year for more advanced spectroscopy methods, even with a sampling number as low as ten samples per year. Additionally, due to the relative novelty of the topic, the majority of scientific literature still demands more data collection to establish an adequate database before meaningful modeling efforts can reduce associated costs (Everaert et al., 2018; Koelmans et al., 2019; Lusher et al., 2021; Bäuerlein et al., 2023; Phan and Luscombe, 2023).

According to the number of funding agencies in the Gulf region, also shown in Table 3, Iranian institutes have been leading MP research in the region through 35 funding programs that have sponsored around 44% of studies on the subject. Notable projects funded by the Iran National Science Foundation have led to significant advancements in understanding MP pollution in the Persian Gulf, setting a benchmark for future research in the region. Saudi Arabia follows in second place, holding a 9.5% share of the research sponsorship. Although certain policies and action plans within the Gulf region countries focus on marine litter and, in particular, marine plastic pollution control, the majority of studies in the Gulf region are mainly funded by universities with a focus on broad research development rather than targeted environmental initiatives (Supplementary Table S4). For instance, Shiraz University alone has funded 18 studies, whereas no regionally coordinated environmental action plan appears in the sponsorship data. This distinction suggests a symptom of disconnect between the sources of research funding and the implementation of environmental policies, as the funding is predominantly allocated for academic research purposes, not directly tied to environmental action plans. This gap highlights the need for enhanced institutional support and better alignment between research efforts and environmental policies to harness the scientific findings from MP research in the Gulf region.

Globally, funding programs are incentivized through relevant environmental policies that provide financial resources to support studies addressing specific environmental issues. According to results of funding agencies' activities, of the 17 countries that have sponsored MP pollution research in the Gulf region, only five (Iran, Saudi Arabia, Kuwait, United Arab Emirates, and Oman) are geographically located within the Gulf region. Countries outside the Gulf region sponsor around 19% of MP research in the Gulf region. A part of this external support comes through international programs such as the "G20 Implementation Framework for Actions on Marine Plastic Litter," which are established within broader regional treaties. Considering the critical need for further research on MP pollution in the Gulf region, the necessity for increased financial support becomes even more evident. Future research efforts in the Gulf region should focus on comprehensive empirical studies enabled by improved financial contributions of the Gulf region countries and developing stronger policy frameworks to effectively address MP pollution.

3.6 Research keywords analysis

Figure 5 show the dynamic and complicated relationships among the keywords regarding global and regional MP research, respectively. The curved lines indicate the connections between keywords, highlighting their repetition in different studies. The size of the circles for each keyword in the co-occurrence network visualization generally signifies the frequency of occurrence of that keyword in the dataset. Larger circles represent keywords that appear more often in the literature, indicating that these topics are more frequently discussed or studied within the field. During the visualization step, for the purpose of a more focused analysis, a benchmark was used such that only keywords occurring a minimum of 15 times are visualized. Before the filtering step, the global and regional data on MP research (with an occurrence of at least 15 times) yielded 705 and 65 keywords, respectively, which were filtered to 383 and 32 keywords. Supplementary Table S1, S2 in the supplementary data list keywords omitted from the analysis due to no meaningful relevance, along with similar keywords that have been merged into a single term.

Based on the co-occurrence of different keywords within the network, the larger network was divided into smaller topical clusters, each revolving around a specific sub-topic within the general subject of MP research. This approach allows for a comprehensive, systematic analysis of research patterns, offering more accuracy than conventional literature reviews based on a random selection of studies. According to Figures 4, 5, global and regional research on MPs can be broken down into five and two distinct research clusters, respectively, sorted based on the occurrence rate of the representative keywords.

3.6.1 Cluster analysis

The first cluster in global MP research focuses on the sampling, analysis, and consequences of marine MP pollution. The emergence of this cluster as the largest hub of MP research is partly explained by the fact that the term "microplastic" was first coined with a focus on marine pollution (Thompson et al., 2004). This cluster primarily focuses on the prevalence of MPs in coastal and marine regions, including estuaries and beach sediment studies. Various analytical methods, such as FTIR, Raman Spectroscopy, and SEM imaging, have been involved in the initial studies in marine research, creating a focal point within this cluster on the characteristics of MP particles. Results showed a major focus on fibrous MP and plastic microbeads in marine water and beach sediments. Another area covered by studies in this cluster focuses on identifying MP particles in marine biota, with keywords such as "ingestion," "gastrointestinal tract," "fish," "bivalve," "mussel," "shrimps," and other types of marine organisms. A subsequent focus of this cluster has been the potential impact of seafood MP pollution on human health, with keywords such as "food safety" and "trophic transfer." According to Figure 5, similar efforts in the Gulf region (cluster 1) reveal that some key elements of marine MP pollution, such as the potential impact on human health, remain to be investigated by new studies in the region. Due to the lack of studies on MP pollution in the Gulf region, one focal point of global cluster 1 emerges as a separate research cluster (cluster 2) in the Gulf region research, focusing on characterizing MPs using similar analytical techniques.

The second cluster of global MP research primarily focuses on mitigation and removal strategies for MPs from water and wastewater. Keywords such as "wastewater," "wastewater treatment," "water quality," and "water treatment," along with treatment methods like "coagulation" and "electrocoagulation," highlight the efforts to explore "microplastic removal" from these environments. However, the second cluster also includes the distinct topic of atmospheric deposition of MPs, which has a different objective. Additionally, some studies within this cluster investigate the health impacts of indoor and outdoor MPs, indicated by keywords like "inhalation," "health," and "exposure." Compared to other topics, research on MP atmospheric deposition has not expanded as significantly, presenting opportunities for new studies to enhance understanding of the atmospheric mechanisms of MP transport. Examining MP research patterns in the Gulf region (Figure 5) reveals that this specific area remains largely unexplored, with the exception of one effort by Abbasi et al. (2024). Cluster 3 of global MP research predominantly centers on the bioaccumulation and toxicity of MPs in various aquatic and terrestrial organisms. A major theme is exploring how MPs accumulate in the bodies of organisms and their resulting toxicological impacts, encompassing a wide range of biological and cellular effects. Studies frequently investigate oxidative damage and stress mechanisms, apoptosis, autophagy, inflammation, and immune responses, shedding light on the molecular pathways affected by MP exposure. Species-specific research is prevalent, with model organisms such as algae, Daphnia magna, zebrafish, and *Caenorhabditis elegans* playing pivotal roles in toxicity testing and understanding species-specific responses. The combined effects of MPs with other pollutants and stressors are also a significant focus, highlighting the complexity of environmental exposures. Advanced molecular techniques, including proteomics, transcriptomics, and metabolomics, are extensively employed to elucidate the mechanistic underpinnings of MP toxicity, providing a detailed understanding of changes in gene expression, protein synthesis, and metabolic processes. Overall, this cluster underscores a comprehensive and multifaceted approach to studying the intricate and far-reaching impacts of MPs on biological systems.

Cluster 4 of global MP research centers on the biodegradability and environmental impact of MPs, emphasizing their degradation processes and interactions with microbial communities. Studies focus on how different types of plastics, including polyethylene, polypropylene, and biodegradable alternatives like PLA, break down under various environmental conditions (Gewert et al., 2015; Kole et al., 2017). The role of microbial biofilms and communities in enhancing or inhibiting the degradation process is also a key area of investigation (Galloway and Lewis, 2016). Additionally, research addresses the broader environmental and health implications of MPs, including their potential to carry antibiotic-resistant genes and pathogens (Wright et al., 2020). This cluster also highlights the importance of sustainable management practices, such as recycling and promoting a circular economy, to mitigate plastic pollution (Rochman et al., 2019). Overall, this cluster underscores the need for

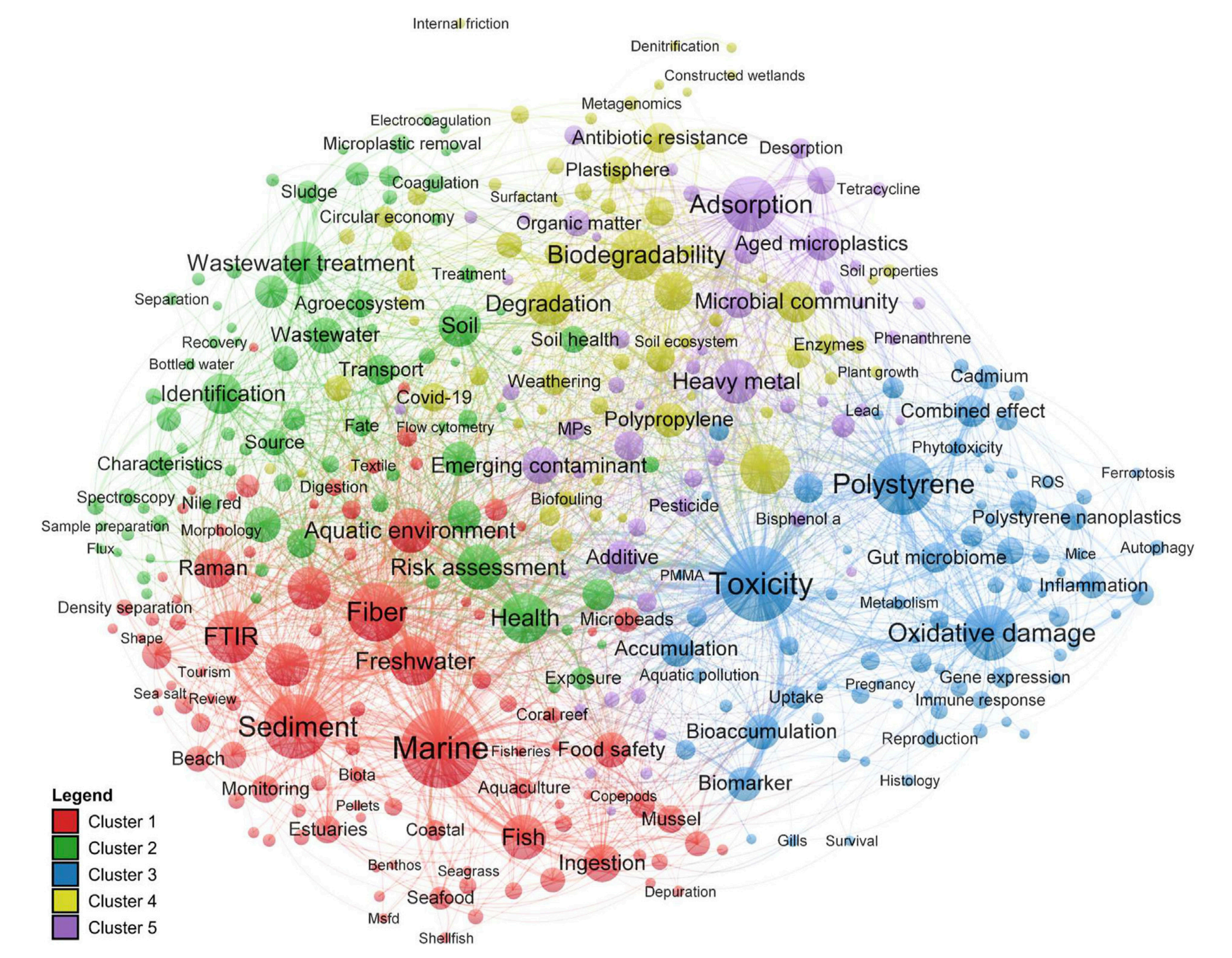


FIGURE 4
Keywords co-occurrence network of global MPs research. Clusters 1 to 5 correspond to marine MP pollution, MP removal strategies, MP bioaccumulation, MP biodegradability, and adsorption of secondary contaminants onto MPs, respectively.

comprehensive strategies to address the persistence and ecological impacts of MPs.

Cluster 5 centers on the adsorption properties of MPs and their role as vectors for various environmental contaminants, including heavy metals, pharmaceuticals, antibiotics, and persistent organic pollutants. Studies demonstrate that the physicochemical properties of MPs, such as hydrophobicity and surface characteristics, significantly influence their capacity to adsorb these substances, impacting their bioaccessibility and environmental fate (Wang et al., 2019). Environmental factors like salinity, pH, and temperature further affect the adsorption-desorption processes, altering the interactions between MPs and contaminants (Bakir et al., 2016). Research also highlights the changing interaction properties of aged MPs due to weathering, which can modify their adsorption behavior and ecological impact (Holmes et al., 2014). This cluster underscores the complexity of MP pollution, emphasizing the need for comprehensive risk assessments to understand their dual role as pollutants and carriers of hazardous substances (Yu et al., 2021). Comparison between research clusters in global and Gulf region studies reveals that of the five global topical zones, only the first cluster has been explored to a limited extent in the Gulf region, with significant gaps in other zones that require further investigation.

3.6.2 Trending topics analysis

An essential feature of VOSviewer is its overlay analysis, which allows researchers to highlight specific attributes within a bibliometric map, such as citation impact, publication year, or research topics (van Eck and Waltman, 2010). This overlay analysis provides a clear and intuitive way to identify temporal patterns of research through close examination of the emergence year of certain research keywords. The benefits of this feature include the ability to easily identify trends, emerging topics, and influential research within a field (Cobo et al., 2011). Overlay analysis of global and Gulf region studies facilitates trend comparison and identifies topics spearheading future MP research.

Supplementary Figure S1 and Table 4 highlight the temporal evolution of research keywords and the top 20 recent research keywords on both global and Gulf region studies, respectively, providing a detailed comparative view of global and Gulf region trends. Globally, recent keywords such as “Ferroptosis,” “Lettuce,”

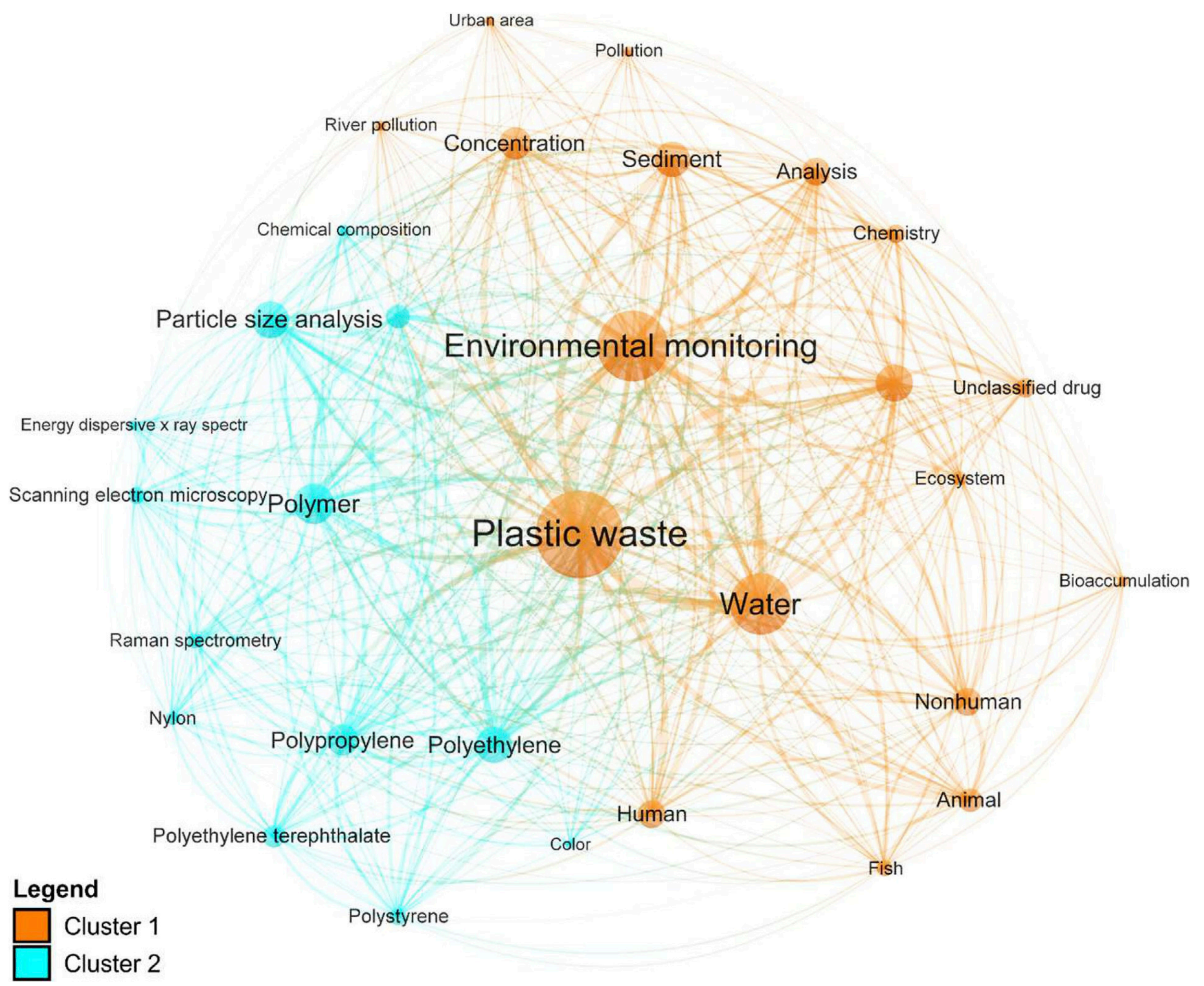


FIGURE 5
Keywords co-occurrence network of MP research in the Gulf region. Cluster 1 highlights efforts on sampling, analysis, and the consequences of marine MP pollution, with limited exploration of human health impacts. Cluster 2 focuses on characterizing MPs using advanced analytical techniques, reflecting a regional emphasis on foundational aspects of MP research.

"Photoaging," and "Spatiotemporal distribution" have emerged in 2023, reflecting a shift towards novel research areas. In 2022, keywords like "Pregnancy," "Nitrous oxide," "ROS," and "Soil microplastics," marked an expansion into environmental and health-related impacts. These emerging themes are distributed across multiple clusters, illustrating the breadth of novel research avenues being explored.

In the Gulf region, the latest research keywords from 2022, such as "Energy dispersive X-Ray spectroscopy," "Ecosystem," "River pollution," and "Raman spectrometry," highlight a focus on analytical techniques and environmental impacts. These recent topics indicate an emerging interest in understanding the analytical and ecological aspects of MP pollution. In contrast, the keywords from 2021, including "Polymer," "Scanning electron microscopy," "Pollution," "Chemical composition," and "Polypropylene," reflect an earlier stage of research primarily centered on material types, pollution measurement, and fundamental analytical methods. This temporal progression from 2021 to 2022 underscores a shift from basic material characterization and pollution metrics to more sophisticated analytical techniques and ecosystem impact assessments. The concentration of keywords in clusters 1 and 2 further suggests a focused research agenda in the Gulf region, emphasizing foundational and analytical studies in MP research. These contrasting emergence patterns highlight the maturity and diversification of global MP research, in stark contrast to the Gulf region, where efforts remain in their developmental stages. This disparity underscores the critical need for targeted research initiatives to bridge the gap and foster a more comprehensive understanding of MP pollution in the Gulf region (Supplementary Figure S1).

4 Conclusion

The bibliometric analysis conducted on MP research on both global and Gulf region scales provides a comprehensive overview of the current state and evolution of the field. Globally, the vast array of studies highlights a robust and exponentially growing interest in

TABLE 4 Temporal emergence of MP research keywords in global and Gulf region studies.

Global research on MP			MP research in the Gulf region		
Keyword	Cluster	Average publication date	Keyword	Cluster	Average publication date
Ferroptosis	3	May 2023	Energy dispersive X-Ray spectroscopy	2	Aug 2022
Lettuce	3	Jan 2023	Ecosystem	1	Feb 2022
Photoaging	3	Jan 2023	River pollution	1	Feb 2022
Spatiotemporal distribution	2	Jan 2023	Raman spectrometry	2	Jan 2022
Pregnancy	3	Nov 2022	Polymer	2	Oct 2021
Nitrous oxide	4	Nov 2022	Scanning electron microscopy	2	Oct 2021
ROS	3	Oct 2022	Pollution	1	Sep 2021
Soil properties	4	Oct 2022	Chemical composition	2	Sep 2021
LDIR	2	Oct 2022	Polypropylene	2	Aug 2021
Denitrification	4	Oct 2022	Human	1	Jul 2021
Bioremediation	4	Oct 2022	Nylon	2	Jul 2021
Soil microplastics	2	Oct 2022	Polyethylene terephthalate	2	Jul 2021
Tire wear particles	5	Sep 2022	Concentration	1	May 2021
Hepatotoxicity	3	Sep 2022	Environmental monitoring	1	May 2021
SERS	2	Sep 2022	Color	2	May 2021
Carbonyl index	1	Sep 2022	Particle size analysis	2	May 2021
Autophagy	3	Sep 2022	Plastic waste	1	May 2021
Biochar	5	Sep 2022	Polystyrene	2	Apr 2021
Oxidation	4	Aug 2022	Polyethylene	2	Apr 2021
Personal protective equipment	4	Aug 2022	Water	1	Apr 2021

MPs, evidenced by a significant surge in publications, especially post-2014, peaking in 2023. This global surge highlights a widespread recognition of the environmental and health implications posed by MPs, with a substantial body of research focusing on various aspects ranging from particle size and toxicity to broader environmental impacts. In contrast, MP research in the Gulf region, though increasing, accounts for just 1.1% of the global output, highlighting its underrepresentation. This disparity points to a critical research gap in the region despite its unique environmental challenges and reliance on marine ecosystems. Gulf region studies have shown a growing trend but on a much smaller scale, indicating a nascent but gradually acknowledging field within the local scientific community. Peaks in publications and citations within the Gulf region indicate periods of heightened research activity and interest, likely driven by regional environmental policies or growing scientific capabilities.

Moreover, the analysis of leading journals, authors, and countries further portrays the landscape of MP research. Globally, countries like China dominate the field, contributing significantly through a large number of studies and extensive international collaborations, reflecting their industrial and environmental challenges as well as their scientific ambitions. Conversely, the research in the Gulf region is predominantly led by Iran, highlighting a focused but limited regional effort to address MPs pollution, with much room for growth in terms of both scope and depth.

This study not only identifies the significant research gap present in the Gulf region but also interprets the varying scales of engagement and focus on MP research. The insights drawn highlight the necessity for increased regional research efforts, particularly in understanding the localized impacts of MPs on the marine-rich environments of the Gulf. There is a pressing need for targeted research initiatives to integrate the Gulf region into the global MP studies, fostering more effective local and global environmental health strategies.

Data availability statement

The data that support the findings of this study are openly available at the Federated Research Data Repository (FRDR) website at https://doi.org/10.20383/103.01145

Author contributions

AS: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Software, Visualization, Writing–original draft, Writing–review and editing. MM: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Software, Visualization, Writing–original draft, Writing–review and editing. MM: Conceptualization, Funding acquisition, Project administration, Resources, Supervision, Writing–review and editing. FR: Conceptualization, Funding acquisition, Project administration, Resources, Supervision, Writing–review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

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"The paper cups Nile": microplastics and other hazardous substances leached from paper cups: paper cups aquatic environmental bane in the River Nile, Egypt

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Recent studies suggest that paper cups may also contribute to environmental pollution, particularly through the release of microplastics (MPs). The Nile River, one of the world's most vital water sources, faces alarming contamination levels, raising concerns about its ecological health. This study investigated whether paper cups release MPs, ions, and heavy metals into water and assessed the potential impact of MPs on fish. In order to completely comprehend the nature and scope of the issue, 1 L of water was collected from the Nile River in Assiut, Egypt and the paper cups were ripped into tiny pieces. Paper cups were similarly soaked in similar volumes of distilled and tap water. Four months later, the leachate from each trail (three replicates for each) was analyzed to determine and compare the distribution of specific ions, heavy metals and microplastics. In order to clarify the availability of MPs in freshwater fish, the intestines of two common fish species (*Oreochromis niloticus* and *Bagrus bajad*) were collected from the River Nile in Assiut and examined. Polyethylene, polystyrene and polypropylene were the three main forms of microplastics identified in water samples from the Nile. Also, paper cups soaked in tap water leached the same three groups of MPs, but in lower amounts. Some microplastics may take longer to biodegrade in water, as evidenced by the absence of other forms of microplastics like rayon and polyvinyl chloride in any of the water samples under investigation. The present findings also indicate a noteworthy accumulation of MPs in the intestines of *O. niloticus* and *B. bajad*. In conclusion, these results indicated release of some ions, heavy metals, and microplastics from paper cups into water and the River Nile water is polluted with paper cups which have a negative effect on aquatic organisms. This study brings us one step closer to investigating and fully understanding the nature and extent of the problem posed by paper cups and their effects on the River Nile and freshwater fish, which will ultimately be reflected in human health risks.

KEYWORDS

paper cups, leachates, microplastics, River Nile, Assiut city, fish

1 Introduction

The Nile River, Mediterranean Sea, and the Red Sea have become significantly polluted with plastics and microplastics, even though they are among the most vital resources for human survival (Chaudhry and Sachdeva, 2021). MPs pollution has recently spread to freshwater environments such as lakes, rivers, wetlands, estuaries and even ground water (Du et al., 2021; Wang et al., 2020). Rivers are essential for the lives of countless people and have been demonstrated to be the primary routes by which large volumes of MPs and plastic debris are transported from land-based sources into the oceans (Lebreton et al., 2017; Schmidt et al., 2017). Consequently, microplastics have recently been detected in the water, organisms, and sediment of several major river systems worldwide, including the Thames in the United Kingdom (Horton et al., 2017), the Seine in France (Dris et al., 2015), the Rhine in Germany (Mani et al., 2015), and the Danube in Austria (Lechner and Ramler, 2015); the Amazon in South America (Andrade et al., 2019; Schmidt et al., 2017); the Yangzte in Asia China (Zhang et al., 2015); and the St. Lawrence in North America (Castañeda et al., 2014). Despite being one of the most well-known river in the world, the Nile River is left off this list. Consequently, this study tried to fill in the gap in identifying microplastics as a component of paper cups in the River Nile and freshwater fish. Li et al. (2020) found that microplastic pollution of freshwater ecosystems is increasing at an unprecedented and concerning rate, despite the fact that the concentration of microplastics in freshwater ecosystems is lower than in marine habitats. Even though the majority of MP research has focused on the marine environment, there is growing interest in identifying MPs in river systems. Thus, this study directed attention toward the examination of microplastics in some freshwater fish from the River Nile in Assiut City, due to the global production of plastic continues to increase and eventually release into the environment.

Given that plastic waste is believed to pose serious threats to wildlife, human health, and climate change, it has been reported in every region of the natural environment worldwide (Simantiris, 2024). Consequently, there has been a swift global shift in recent decades toward alternative solutions for plastic waste, prompting governments and industries to replace single-use plastics with paper products. However, several studies have highlighted the negative impacts of both single-use plastics and paper products, concluding that paper is not a sufficient solution due to various harmful effects on the environment and human health (Fidan and Ayar, 2023; Joseph et al., 2023; Ranjan et al., 2021).

It is recognized that one of the most polluting industries in the world is the paper sector (Kumar et al., 2020; Singh et al., 2022), primarily due to the use of chlorine compounds in paper processing, which have detrimental environmental effects (Kumar et al., 2012). Paper cups are often discarded in landfills or improperly disposed of, contributing to (micro) plastic waste and potentially polluting the world's oceans (Fidan and Ayar, 2023; Foteinis, 2020). Paper and paperboards account for 31% of the global packaging market and they are mostly utilized in food packaging for the purposes of containing and safeguarding food products, making them convenient to store or consume and informing customers about pertinent information, including marketing aspects (Jones and Comfort, 2017). Paper is the preferred material for the food companies due to its good standing for being environmentally friendly (Oloyede and Lignou, 2021). It is widely used in primary applications, such as those in which food products come into direct contact with it, as well as secondary applications, such as those in which primary packaging needs to be moved and stored (Khwaldia et al., 2010).

Additionally, paper and paperboard are the primary materials used in the production of beverage cups, ice cream cups, microwave popcorn bags, baking paper, milk cartons, and fast food packaging, such as pizza boxes and other similar items (Chen et al., 2023; Deshwal et al., 2019). In spite of, the paper cups are perceived as ecofriendly alternatives or biodegradable frequently due to greenwashing efforts (Viera et al., 2020). However, a thin layer of polylactic acid added to the inner face of many paper cups to create a waterproof layer between the beverage and the paper. Nevertheless, along with pesticides and other environmental pollutants, the vast and complex set of chemicals emitted from packaging material is considered a food hazardous agent (Grob et al., 2006). This could seriously endanger people's health, depending on how much of specific paper chemicals and components find their way into food products and are consumed (Joseph et al., 2023; Poças et al., 2010). According to Biedermann-Brem et al. (2016) and Joseph et al. (2023), the issue of food migrants has grown more complicated because to their wide variety and differing degrees of toxicity. Paper straws may include a variety of hazardous compounds (such as heavy metals, formaldehyde, fluorescent materials, ink, etc.) that can be harmful to people and/or promote microbial contamination, according to a study by Qiu et al. (2022). Research on the environmental toxicity of single-use items and materials that come into contact with take-out meals is necessary to fill in these knowledge gaps. A few previous studies have investigated the toxicity of leachates from particular kinds of plastic food packaging materials. According to research by Thaysen et al. (2018), leachate from expanded polystyrene cups was toxic to aquatic invertebrates. Almroth et al. (2023) indicated that, single-use take-away beverage cups of different materials can all induce toxic effects in midges and mismanaged waste has a detrimental effect on aquatic biota.

Despite the fact that paper has an environmentally safe component, there are significant problems associated with its production and recyclable nature (Deshwal et al., 2019). Less than 1% of paper coffee cups are recycled worldwide, despite major efforts and gradual improvements in wealthier economies to increase recycling rates, this estimate was determined for three reasons, according to Triantafillopoulos and Koukoulas (2020): (a) the belief that the paper and plastic combination used to make paper coffee cups is difficult to recycle; (b) the fact that many recycling facilities are unable to commit to processing waste streams contaminated with food; and (c) the fact that recycling programs within communities and venues are inconsistent and ineffective. Disposable paper cups are composed of 90%–95% paper and 5%–10% hydrophobic plastic film by weight (Arumugam et al., 2018; Constant, 2016). Thus, unfortunately, recycling these paper cups would be extremely difficult as paper cups are coated with polyethylene terephthalate (PET), which prevents them from being recycled or decomposed (Biswal et al., 2013; Foteinis, 2020).

In our daily lives, poor disposal of single use plastics (SUP) items like paper cups causes to plastic pollution in aquatic habitats as they become more prevalent (Foteinis, 2020). According to Foteinis (2020), there are two main paths for paper cup disposal, namely landfilling and recycling, assuming that each route accounts for

FIGURE 1
(A) Map of Egypt showing the catch site of *Oreochromis niloticus* and *Bagrus bajad* from the River Nile at Assiut, Egypt. **(B, C)** are two sites of plastic piles observed on the Nile shore at Assiut, from which the fish and water samples were obtained.

100% of the waste stream. In recent years, post-consumer garbage has accounted for around 35% of municipal solid trash by weight and has become a significant component in many landfill sites (Kapukotuwa et al., 2022). In recent surveys, Miarc (2020) estimated that the yearly market value of single-use OOH hot paper coffee cups is 118 billion, and by 2025, it is expected to grow at a 1.8% CAGR to reach 294 billion units. Reusing and recycling waste materials in an economical, safe, efficient and environmentally friendly way is one of the best ways to cut down on the amount of trash produced (Arumugam et al., 2018). Owing to these issues, there was a push to fully understand the nature and extent of problem of paper cups by some analysis trials to contribute to close this information gap. Further studies need to be carried out to get a better understanding of the environmental pollution by paper cups.

There are various solutions that have been suggested for limiting plastic discharges into the environment (Suzuki et al., 2024; Thaysen et al., 2018). In all over the world many countries, states, townships, and cities are making a great attention towards banning plastics from OOH items (Triantafillopoulos and Koukoulas, 2020). In India the Kerala state government has gone a step further in banning production, sale, and use of single-use plastics including paper coffee cups (Triantafillopoulos and Koukoulas, 2020). In a recent life cycle study, Foteinis (2020) showed that current methods for disposing paper cups has the global environmental footprint of 1.5 million Europeans and that recycling could decrease this impact by 40 percent. Collaborations on a national and international level should be established to carry out circular economy initiatives (such as implementing a system of fees and levies for single-use products), policy development, public and academic education (including information on the problems with paper alternatives), industrial company disposal fees and regulations, and sustainable solutions to keep the market system from collapsing (Simantiris, 2024).

2 Material and methods

2.1 Study area

The study was conducted along the River Nile in the Assiut Governorate, with two primary sampling sites selected based on

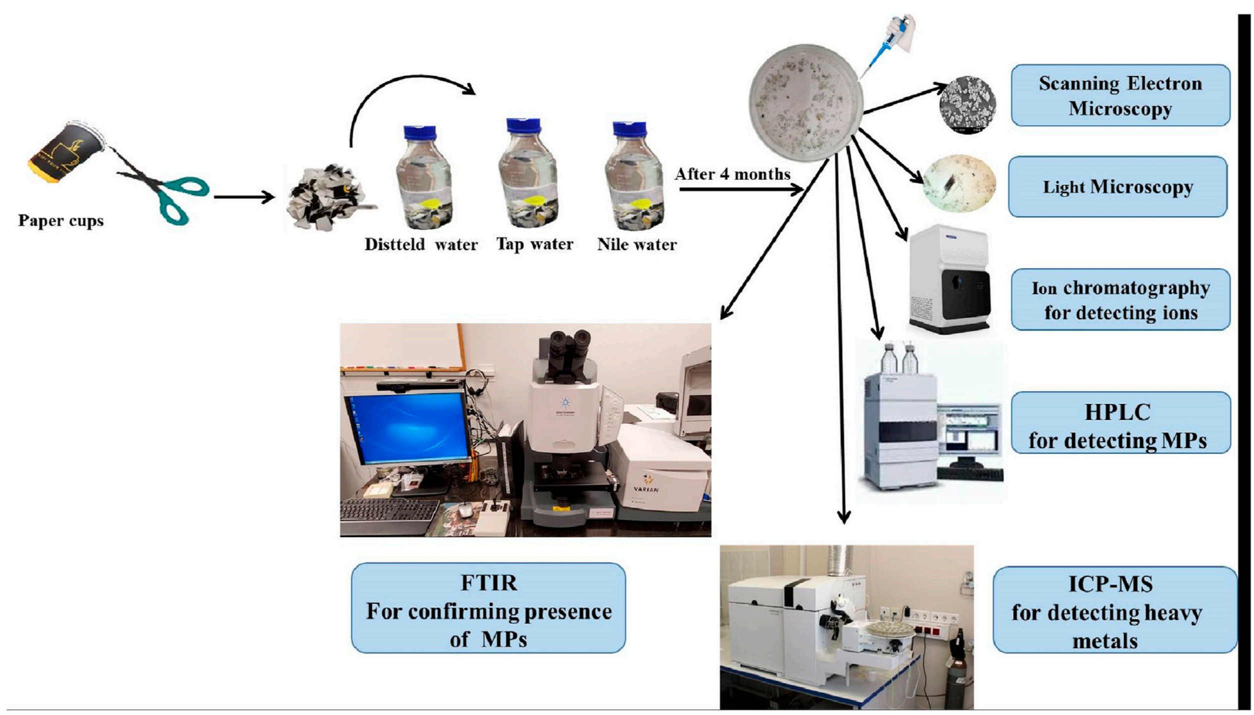

FIGURE 2
Summarized technique for the analysis of paper cups leachate following 4 months of soaking.

observed plastic and paper cups accumulation along the riverbank. These sites were chosen to obtain the fish and water samples (Figure 1). The first sampling site was located near a densely populated urban region within Assiut City (Figure 1B). This area is characterized by high human activity, including residential neighborhoods, and markets. The riverbank here showed visible signs of anthropogenic pressure, with significant plastic and waste piles consisting of disposable cups, plastic bags, and other debris. The proximity of this site to informal waste disposal points and active fishing areas raises concerns about direct pollution pathways into the River Nile. The surrounding area includes agricultural fields irrigated by river water, which may further spread contaminants to the soil and crops.

The second sampling site was situated in a more rural setting, downstream from Assiut City (Figure 1C). This site is relatively less populated but still exhibited noticeable plastic waste, particularly from agricultural runoff and domestic activities in nearby villages. The surrounding environment consists of small farms and natural vegetation, with some local fishing and livestock watering activities observed along the riverbank. Although human activity is less concentrated here, the accumulation of waste from upstream sources is evident, as plastic debris and paper cups are carried downstream by the river's flow. Both sites provide distinct settings for examining the impact of plastic and paper cups pollution in different environmental and socio-economic contexts.

2.2 Paper cups analysis trial

A number of paper cups were purchased from the commercial market. Three cups, each weighing 3.95 g (with a total weight of 11.87 g), were cut into small pieces using scissors and placed into 1 L of water collected from the River Nile in Assiut City. The same number of paper cups were also placed in an equal volume of tap and distilled water. After 4 months, the leachate from each trial (three replicates for each) and the River Nile water before adding paper cups fragments were analyzed to determine and compare the distribution of specific ions, heavy metals, and microplastics (Figure 2).

2.3 Analysis of the paper cups leachate and water of the River Nile

2.3.1 Ion chromatography

The ions were detected in samples of water from the River Nile, leachate of tap water contained paper cups and tap water without paper cups using an Ion Chromatography system according to Michalski (2006) and Ranjan et al. (2021). Water samples were prepared before analysis to ensure compatibility with the ion chromatography system. Standard solutions with known ion concentrations were also prepared, and a specific volume (10–50 μL) of the water sample was introduced into the ion chromatography system. As the sample passes through the column, the ions interact with the charged stationary phase and are separated according to their charge, size, and affinity for the column. The detector captures the ions as they exit the column, generating a chromatogram. Each peak in the chromatogram represents a specific ion, with the peak area indicating its concentration. Finally, the sample's chromatogram is compared to the calibration curve to quantify the ions.

2.3.2 High-performance liquid chromatography (HPLC) analysis for microplastics detection

The samples of water from the River Nile and leachate of tap water contained paper cups were stored in glass bottles to avoid contamination. The samples were filtered using polycarbonate membranes with a pore size of 0.45 μm to isolate microplastics. Filters were rinsed with distilled water to remove residual salts and particulate matter. The filtered samples were mixed with a saturated NaCl solution and left to settle for 24 h. The floating fraction, containing microplastics, was collected for further analysis. To remove organic contaminants, 30% hydrogen peroxide (H_2O_2) was added to the collected samples, and the solution was heated at 60°C for 24 h. The resulting microplastic residue was dried at 40°C to avoid polymer degradation. Adsorbed chemicals or monomers were extracted using methanol as an organic solvent, with ultrasonication applied for 20 min to enhance the extraction process. Finally, the solvent extract was filtered through a 0.22 μm syringe filter to prepare the sample for HPLC analysis. A high-temperature gradient HPLC system, PL XT-220 (Polymer Laboratories, Church Stretton, England), was utilized for the analysis according to Heinz and Pasch (2005). A high-temperature gradient HPLC system (PL XT-220) with a Nucleosil 500 stationary phase (25 × 0.46 cm, 5 μm particle size) was used for analysis. An ELSD detector (PL-ELS 1000) operated at 160°C nebulization, 270°C evaporation, and 1.5 L/min air velocity. The eluent flow rate was 1 mL/min. A robotic system (PL-XTR) automated sample preparation and injection, maintaining temperatures of 140°C for the column, 150°C for the injection port and transfer line, and 160°C for the sample block and robotic arm tip. Data processing was performed using "WinGPC-Software".

2.3.3 Inductively coupled plasma mass spectrophotometer (ICP-MS) for heavy metals detection

The samples of water from the River Nile, leachate of tap water contained paper cups and tap were filtered using 0.45 μm or 1.2 μm membrane filters to remove particulates, debris, and large organic matter. The filtered samples were then diluted with Milli-Q water (ultrapure water) at a ratio that ensured the analyte concentrations fell within the linear range of the ICP-MS. To the diluted samples, concentrated nitric acid (HNO_3) (trace metal grade) was added to ensure the complete dissolution of metal ions and to prevent precipitation. The concentration of heavy metals was measured using ICP-MS and compared to the calibration curve following the methodology described by Ranjan et al. (2021).

2.3.4 Post-analysis for microplastics

Following the ICP-MS analysis for heavy metals, any particulate matter remaining in the digestion residue was collected and isolated. Fourier Transform Infrared Spectroscopy (FTIR) was then utilized to confirm the presence of microplastics and determine their composition in the water samples according to Albrecht et al. (2007).

2.3.5 Scanning electron microscopy (SEM)

SEM was used for viewing and clearing up what were found in the River Nile water before putting paper cups and in the leachates of the same water after containing paper cups for 4 months at scanning electron microscopy unit, Assiut University (JEOL JEM-1200 EX II). The films were viewed under different magnifications ranging from 1,000 to 1,5000X. 5 μL of the tested water were airdried and viewed under the SEM.

2.3.6 Light microscopy

It was also used for examination the three tested water (River Nile, tap and distilled water) before and after containing paper cups under a ×10 objective with a ×1 eyepiece using Omax microscope with 14 MP USB Digital Camera (CS-M837ZFLR-C140U) (A35140U3; China).

2.4 Fish collection

The Nile Tilapia *Oreochromis niloticus* and *Bagrus bajad* (12 fish/species) were obtained from the River Nile at Assiut and transported to the fish Pollution Laboratory, Assiut University, Egypt (Figure 1).

2.5 Microplastics (MPs) quantification

Intestine (the digestive tract) samples were collected and digested in 10 mL of hydrogen peroxide (30%, v: v) at 70°C for 2 h, and 100 μL of the resultant solution was then examined under the light microscope for microplastic detection according to Deng et al. (2017) and photographed under a ×40 objective with a ×10 eyepiece using Omax microscope with 14 MP USB Digital Camera (CS-M837ZFLR-C140U) (A35140U3; China).

3 Results and discussion

3.1 Detection of ions and heavy metals in River Nile water, tap water and tap water contained in paper cups

Indeed, during the COVID-19 epidemic, there was a surge in the usage of food packaging materials and the corresponding creation of solid waste (de Oliveira et al., 2021). Many previous studies have investigated that harmful chemicals and substances can leach from paper and cardboard-based food packaging (Vandermarken et al., 2019). Owing to these issues, this study had a shed on the problem of paper cups by detection of microplastics, heavy metals and ions in samples of water from the River Nile, leachate of tap water contained paper cups and tap water only. The results of this research indicated that, the highest levels of some ions like fluoride, chloride, nitrite, sulfate and nitrate were detected in the water of the River Nile then leachate of tap water contained paper cups but only chloride ion was detected in the tap water without paper cups (Figures 3c, d). Similarly, other study has shown that, ions such as sulfate, nitrate, chloride and fluoride might be found in paper cups, all may originated from the chemical treatment of paperboards (Ranjan et al., 2021). This interpretation is also consistent with Ozaki et al. (2004), who reported that a number of chlorine-containing chemicals were used to bleach the paper pulp in order to remove lignin and give the final product a brighter white colour.

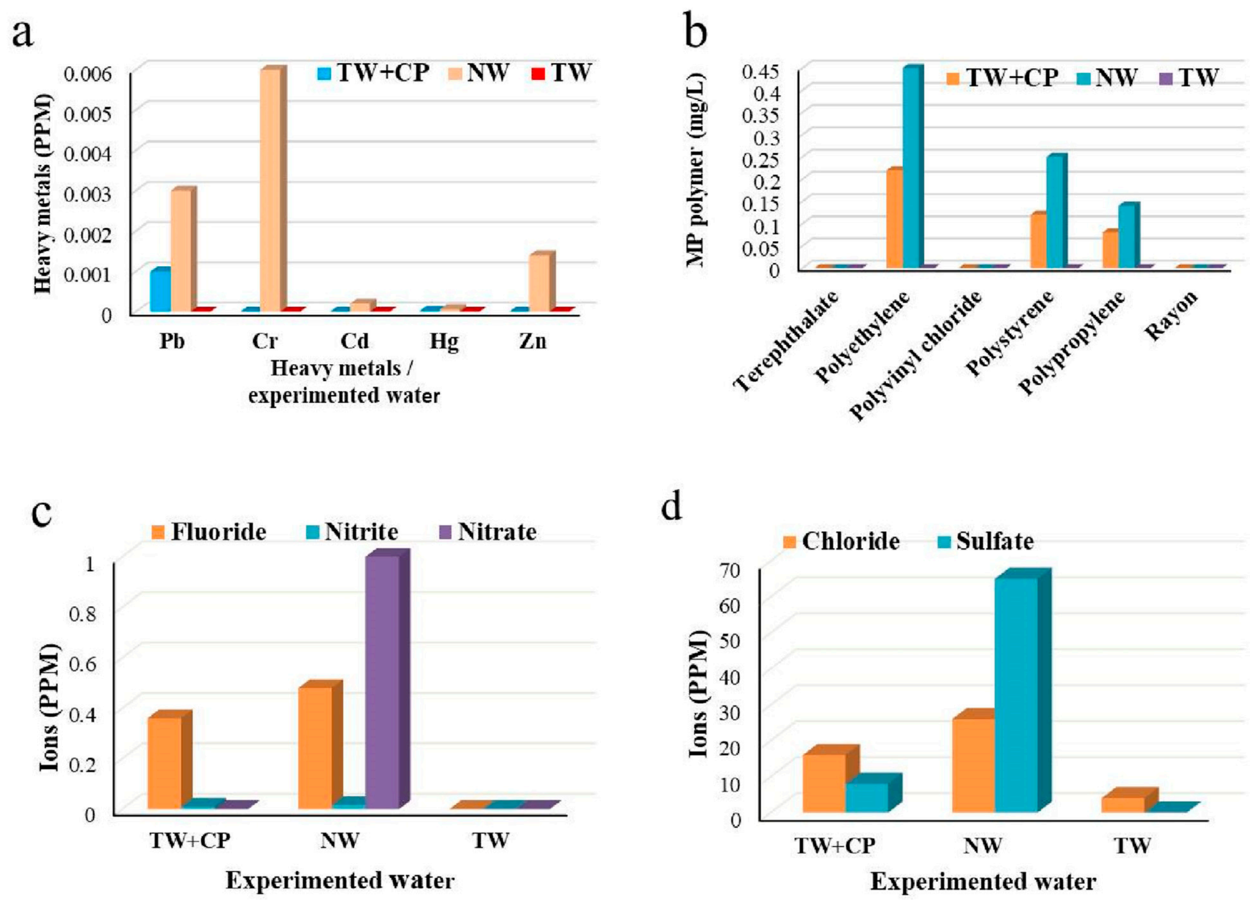

FIGURE 3
Showing heavy metals **(a)**, microplastics **(b)** and ions **(c, d)** detected in the water of the River Nile at Assiut city (NW), tap water (TW) and the leachate of tap water contained paper cups (TW + CP) (by Inductively Coupled Plasma Mass Spectrophotometer (ICP-MS), High-Performance Liquid Chromatography (HPLC) and Ion Chromatography respectively).

Additionally, paper boards used in food packaging are treated with per- and polyfluoroalkyl substances (PFASs) and other fluorinated chemicals; these compounds have the lipophobic and hydrophobic qualities that make paper containers waterproof and stain-resistant (Cantoral et al., 2019; Trier et al., 2018). Joseph et al. (2023) estimated that the average daily intake of fluoride from hot beverages in paper cups was 7.04 ± 8.8 μg/kg body weight. Rezvani Ghalhari et al. (2021) and Karami et al. (2019) had analysed fluoride and nitrate in tea and water. Health risk related with fluoride intake through tea and coffee was analysed by Satou et al. (2021).

This study revealed that the highest levels of certain heavy metals, including Pb, Cr, Cd, Hg, and Zn, were detected in the water of the River Nile. In the leachate of tap water contained paper cups, only Pb and Hg were detected, while no heavy metals were detected in tap water without paper cups (Figure 3a). Likewise, these heavy metals are well-known additives used in the manufacturing process to provide paper and paperboards specific properties (Ranjan et al., 2021). These heavy metals are not degradable, hazardous heavy and toxic even at low concentrations (Ranjan et al., 2021). According to Zeng et al. (2023), disposable food containers such paper cups, plastic cups, plastic bags, and plastic bowls offer new ways for metals and other elements to leak into drinking water, beverages and fast food in hot environments. In addition, lab experiments were conducted by Almroth et al. (2023) to examine the toxicity of paper and plastic cups and lids on aquatic midge larvae. According to the results of their investigation, paper products have a tendency to dissolve quickly in fluids, ultimately resulting in release chemicals and breaking down into tiny pieces that may impact the environment by damaging sediments, aquatic life, and the water column.

The presence of heavy metals (Pb, Cr, Cd, Hg, and Zn) in the water of the River Nile aligns with findings from other global river systems, where heavy metal pollution is often associated with microplastic (MP) contamination (Ta and Babel, 2023). For instance, studies on the Chao Phraya River in Thailand recorded high abundance of MPs in both water and sediments, with Pb and Cu adsorbed onto their surfaces (Ta and Babel, 2020). Similarly, Purwiyanto et al. (2020) observed significant adsorption of Pb and Cu on various polymer types, including PP, PES, OVC, PE, and nylon, in the Musi River, Sumatera, Indonesia. This adsorption phenomenon is particularly concerning because MPs can act as carriers for heavy metals, enhancing their persistence in aquatic environments and increasing their bioavailability to aquatic organisms (Cao et al., 2021). The toxic effects of heavy metals and MPs on biota have been widely reported (Foley et al., 2018;

Khalid et al., 2021b; Walkinshaw et al., 2020). Comparing our results with those from Poland, where studies have detected variations in Cr, Ni, Cu, Zn, Cd, and Pb concentrations in the Nida River and higher heavy metal concentrations in other rivers such as the Warta River (Bhat and Janaszek, 2024; Jaskuła et al., 2021; Sojka and Jaskuła, 2022), highlights the global scale of this issue. Whether in Africa, Asia, or Europe, rivers are becoming repositories for hazardous pollutants, which can have long-term ecological and health consequences (Gwenzi and Chaukura, 2018; Mushtaq et al., 2020; Singh et al., 2024).

3.2 Detection of microplastics in River Nile water, tap water and tap water contained in paper cups

This study revealed the presence of polyethylene, polystyrene, and polypropylene MPs in the River Nile, with a greater abundance than in tap water contained in paper cups (Figure 3b). This suggests that river systems act as major reservoirs for MP pollution from various sources, a trend that has been extensively documented in other freshwater systems. For example, studies in China have confirmed the presence of MPs in nearly all investigated water bodies, with concentrations reaching millions of particles per cubic meter (Gupta et al., 2023). The long residence time of MPs in rivers, influenced by water volume and movement, contributes to their widespread distribution (Khalid et al., 2021a). Furthermore, flood events can cause MPs and their associated heavy metals to be redistributed across different aquatic systems, including lakes and reservoirs (Harrison et al., 2018). Notably, in this study no MPs were detected in tap water without paper cups (Figure 3b), indicating that drinking water sources may initially be free from MPs but could become contaminated through contact with paper cups or plastic-based materials. In Thailand, the Chao Phraya River an essential source of drinking water and aquaculture has been found to contain MPs across various regions, from agricultural to urban and estuarine zones (Ta and Babel, 2023). Such contamination poses a direct threat to food security and public health.

According to He et al. (2022) microplastics can be categorized into six groups: polyethylene, polystyrene, polypropylene, polyurethane, polyvinyl chloride, and polyethylene terephthalate based on their molecular composition. Ranjan et al. (2021) and Joseph et al. (2023) examined how many microplastics an average person would consume while drinking hot liquids in a paper cup, such as tea or coffee. Other forms of microplastics such as Polyvinyl chloride and Rayon were not detected in any type of tested water, these indicated that microplastics need more time for biodegradation in the water. Nevertheless, there are worries that paper products may be exposed to all the drawbacks of plastic products in addition to the unfavorable effects of their chemical makeup because the majority of single-use paper products are said to include polymer coatings to keep the paper pulp from combining with the food or beverage (Ranjan et al., 2021; Simantiris, 2024). The investigation in this study indicated the release of certain ions, heavy metals, and microplastics from paper cups into water. Additionally, the River Nile is polluted with paper cups, which have a negative effects on aquatic organisms.

The FTIR analysis of water samples collected from the Nile River and the leachate of tap water containing paper cups respectively, revealed the presence of distinct peaks that can be attributed to various polymers, indicating potential contamination with microplastics (Figures 4A,B). In both spectra, the broad peak around 3,400 cm^{-1} corresponds to O-H stretching vibrations, which are likely due to water absorption but could also suggest the presence of cellulose-based polymers such as rayon (Coates, 2000; Premraj and Doble, 2005). The prominent peaks around 2,900 cm^{-1} are characteristic of C-H stretching vibrations, which are commonly observed in hydrocarbon-based polymers like polyethylene (PE), polypropylene (PP), and polystyrene (PS) (Pavia et al., 2015; Shah et al., 2008). Specifically, these peaks suggest the presence of aliphatic chains, which are a defining feature of these polymers. The appearance of peaks near 1700 cm^{-1} in the spectrum may be attributed to C=O stretching vibrations, which are a signature of ester groups present in polyethylene terephthalate (PET) (El-Azazy et al., 2022). Additionally, peaks around 1,400–1,500 cm^{-1} are indicative of CH2 and CH3 bending vibrations, further supporting the presence of polypropylene or polyethylene (Stuart, 2004). The peaks near 700 cm^{-1} are significant, as they may correspond to aromatic C-H bending, which is characteristic of polystyrene and PET, or C-Cl stretching, which would point to polyvinyl chloride (PVC) (Coates, 2000). The spectrum also displays peaks in the region of 1,000–1,300 cm^{-1}, which could be associated with C-O stretching in PET or C-Cl bonds in PVC.

Comparing the two spectra, the differences in peak intensities and sharpness suggest varying levels of polymer contamination in tap water containing paper cups and Nile water, with Nile water showing more pronounced peaks, indicative of a higher microplastic concentration. Overall, the analysis strongly indicated the presence of multiple polymers, including PET, PE, PP, PS, and potentially PVC and rayon, reflecting diverse sources of microplastic pollution in the River Nile, with paper cups playing a significant role in this contamination. These findings highlight the widespread issue of plastic and paper cup pollution in water sources and emphasize the need for further research into the environmental and health effects of microplastics and paper cups.

3.3 Microscopic analysis of microparticles in water samples before and after soaking with paper cups

In the current study, the films of the river water, both the leachate soaked with paper cups after 4 months and non-soaked water under the SEM showed particles and microparticles with different size and distribution that increased after putting the paper cups (Figure 5). Light microscopy photomicrographs of the three tested water (River, distilled and tap water) before and after soaking with paper cups showed that River Nile water was not clear and contained impurities but tap and distilled water seemed clear. After soaking with paper cups there were impurities which take different shapes such as rodlet, fibrous and spherical fragments and increased in River Nile water (Figure 6). Along with this study, various irregular shapes of the released MP particles from paper cups and the water contained in the disposable cups were investigated

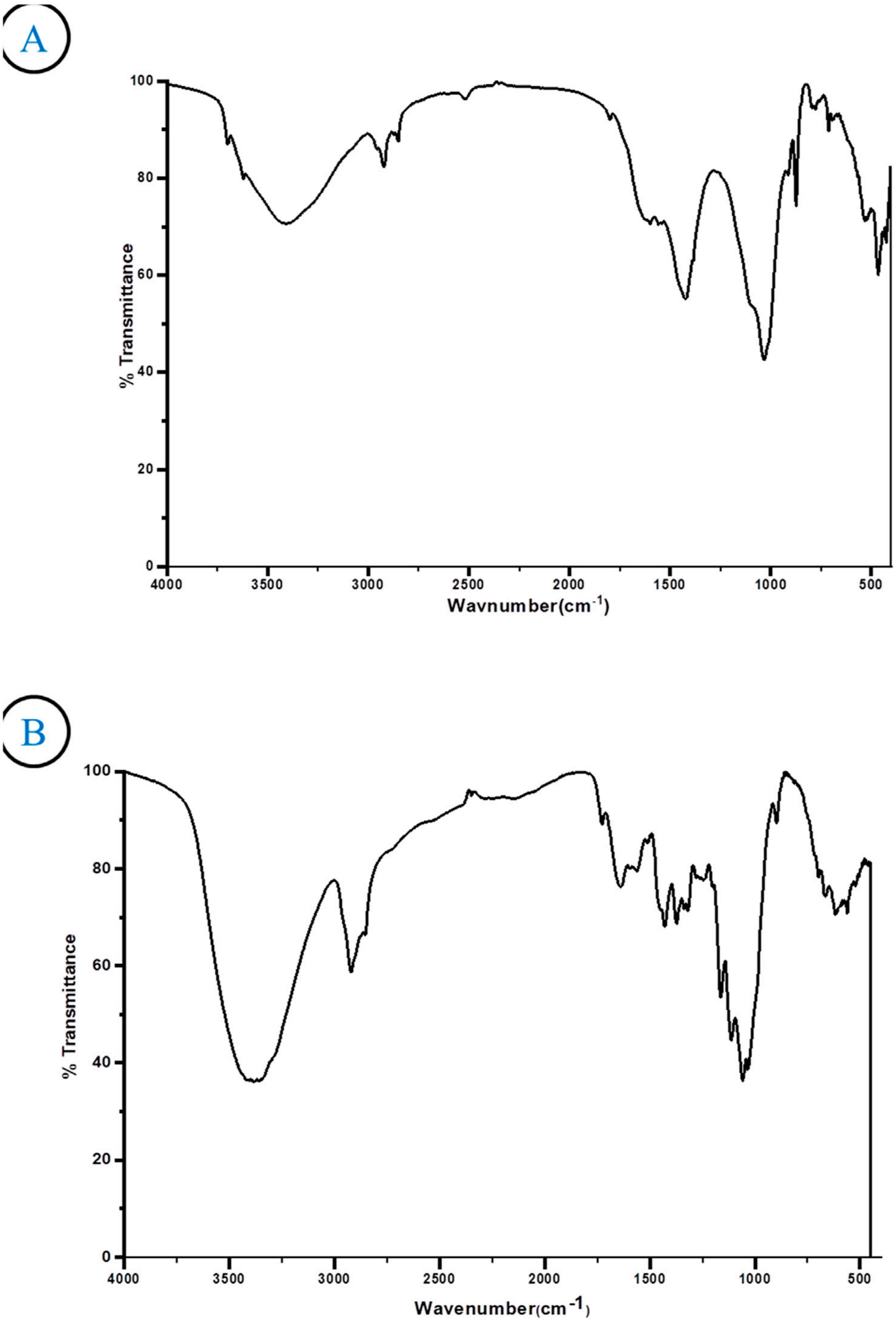

FIGURE 4
FTIR spectra of **(A)** River Nile water and **(B)** leachate of tap water containing paper cups.

in studies by Ranjan et al. (2021) and Chen et al. (2023) respectively, that could be taken up by humans and therefore pose health risks. Plastic fibers and microparticles were previously detected in tap waters (Pivokonsky et al., 2018), mineral water bottles in various countries and mineral water bottles (Koelmans et al., 2019; Schymanski et al., 2018).

According to Ranjan et al. (2021) the amount of particles that can be consumed when hot liquid is consumed in a 100 mL paper cup is (102.3 + 21.1) × 106 particles/mL. According to Batel et al. (2016), microplastics can be separated into five basic types based on their physical characteristics: fragments, fibers, films, foams, and pellets. Plastic, both whole and fragmented, has been found on beaches (Browne et al., 2011), floating on the surfaces of seas and lakes (Biginagwa et al., 2016), in the deep sea (Woodall et al., 2014) and in a wide range of species (Gall, 2015). The size and structure of microplastics are assumed to be markers of their fate and dissolution

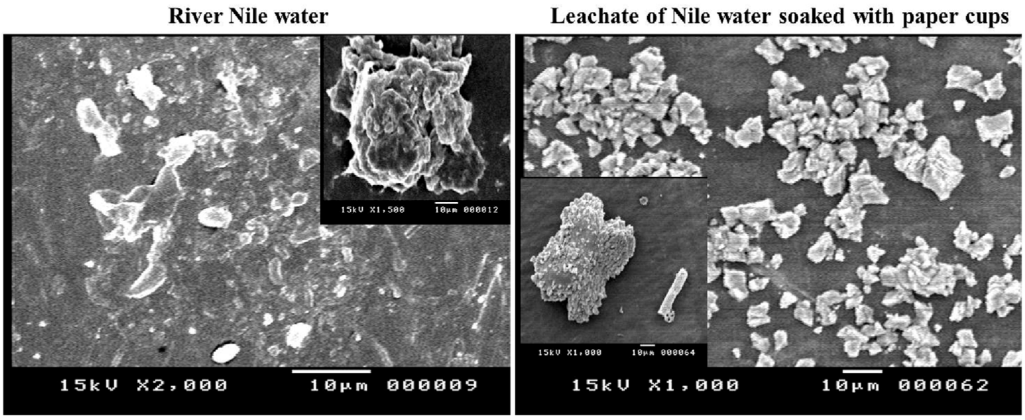

FIGURE 5
Photomicrographs of Scanning Electron Microscopy (SEM) comparing the appearance of river water, both the leachate soaked with paper cups after 4 months and non-soaked water.

in the environment (Biedermann-Brem et al., 2016). Significant volumes of microplastics infiltrate the sediment and water as the climate continues to warm and the use of plastic goods increases (Bioplastics, 2017).

3.4 Microplastic contamination in *O. niloticus* and *B. bajad*: an indicator of River Nile pollution

The results of this study revealed that, most *O. niloticus and B. bajad* that were analyzed for detection of microplastics as an indicator to microplastics pollution had contained irregular-shaped microplastics particles as showing in (Figures 7A, B) respectively. Accordingly, these findings suggested that microplastic contamination may be endangering fish in the River Nile, if environmental authorities are not drive a concerned actions toward its control. The presence of MPs in these fish could provide further information that could help in understanding the fate of MPs in freshwater bodies. Evidence of microplastic accumulation in fish, both demersal and pelagic, was revealed by the studies (Boerger et al., 2010; Lusher et al., 2013). Also, Neves et al. (2015) reported that up to one-third of the fish under study had ingested microplastics, ranging from 0.2 to 1.9 particles/fish. Polycyclic aromatic hydrocarbons (PAHs), herbicides and heavy metals are among the various contaminants that MPs can carry and absorb. As a result, they are more easily absorbed by organisms, transported throughout different fish organs and ultimately become more toxic (Bollainpastor and Agullo, 2019; Vedolin et al., 2018). Tissue-accumulated MPs influence the ability of organism to breathe, alter cell osmotic pressure, which can ultimately cause organ malfunctions and death (Ma et al., 2018). Feeding and skin absorption are among the main routes of MPs entry into organism tissues (Andrade et al., 2019). Microplastics impede an organism's digestive system, prevent animals from eating, or cause false satisfaction once they enter its body (Arumugam et al., 2018). When *Mytilus edulis* fed polyethylene microspheres (less than 80 µm) and *Carcinus maenas* fed polystyrene microspheres (10 µm) via gill respiration, intestinal damage was observed in the two species (Baker, 2013). *Daphnia magna* exposed to leachates from teabags containing microplastics showed anatomical anomalies (Hernandez et al., 2019).

Furthermore, microplastics could interfere with food, mechanically injure living things, and cause fish to lose their ability to allocate food and forage (Batel et al., 2016). Fishing productivity may drop as a result of these pollutants, according to reports that seriously endanger the growth and health of fish in Lake Amatitlán (Oliva-Hernández et al., 2021). Secondary and tertiary treatment could effectively eliminate the majority of MPs (Lares et al., 2018; Murphy et al., 2016). Whoever, Murphy et al. (2016) indicated that this technique couldn't stop the MPs effluents into aquatic systems. Some major factors, including water quality, human activity, urbanization, and wastewater treatment technology, limit the amount of microplastic pollution in freshwater systems (Zhang et al., 2022). The effects of exposure to microplastics can differ depending on whether it was received directly or indirectly (Enyoh et al., 2020). Therefore, MP pollutants have harmful effects on aquatic life, such as impairing their immune systems and digestive systems, which could cause fish, oysters, mussels and sea turtles to become extinct (Caron et al., 2018; Hipfner et al., 2018; Matsuguma et al., 2017). According to Du et al. (2020), direct exposure occurs when contaminants come into direct contact with an organism, usually leading to acute toxicity over a short period of time. Chronic organ toxicity is caused by indirect exposure, which is the incorporation of contaminants and microplastics into the food chain (Siddiqui et al., 2023). Pollutants including microplastics are incorporated into the food chain through indirect exposure, which results in long-term organ damage (Alijagic et al., 2024). The ingestion of microplastics has been associated with several hazards in a variety of aquatic organisms including oxidative stress, growth suppression, changed behaviour, cytotoxicity, reproductive

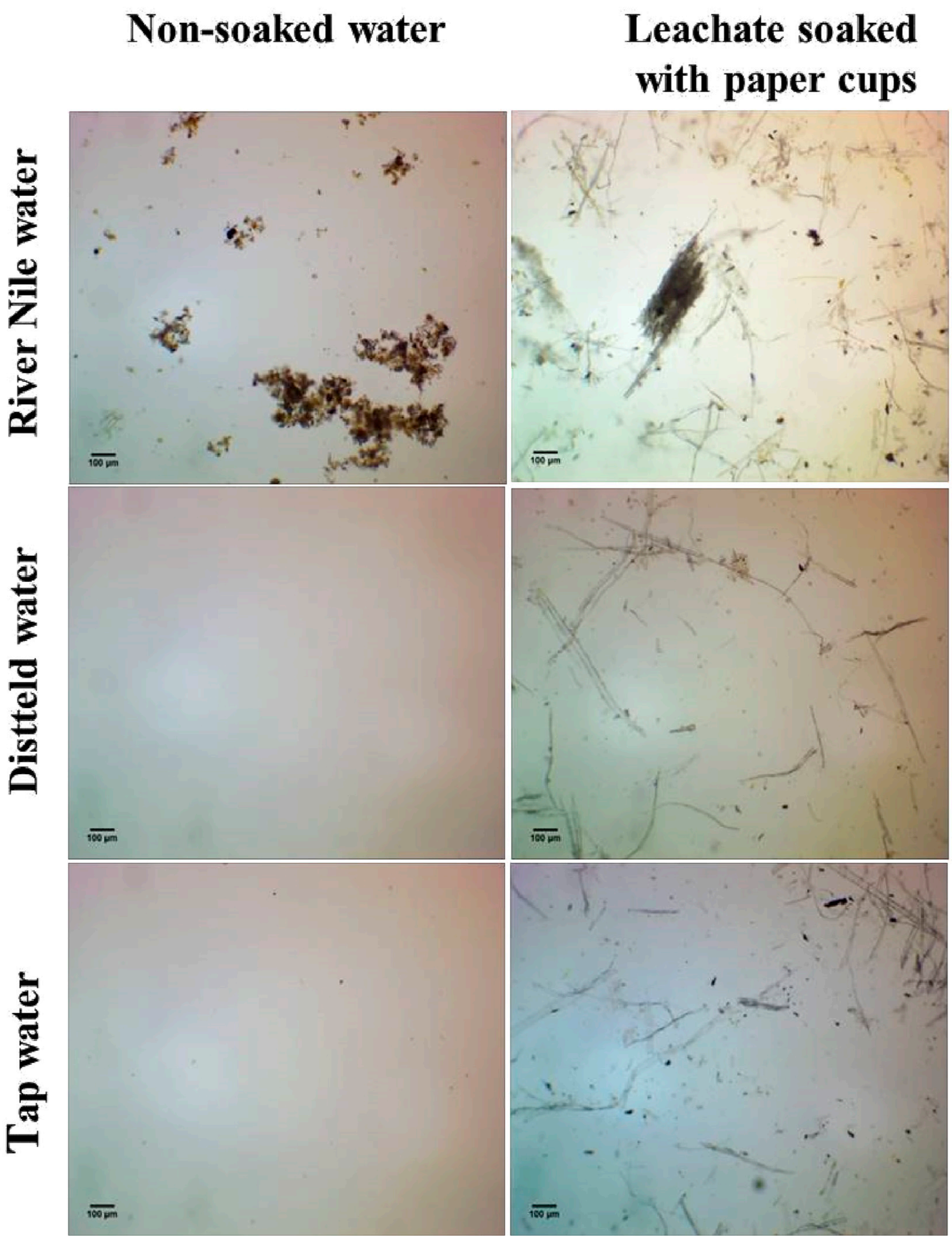

FIGURE 6
Light microscopy photomicrographs indicating the appearance of the three tested water (River, distilled and tap water) before and after soaking with paper cups.

failure, and differential gene expression (Amin et al., 2020; Hamed et al., 2019; Kedzierski et al., 2020; Meaza et al., 2021; Osman et al., 2023; Ugwu et al., 2021).

Overall, the presence of microplastics in fish from the River Nile indicates ongoing contamination, which may have an impact on the trophic network and fish consumers. Therefore, environmental and health authorities must pay close attention to limit microplastic pollution in the River Nile, particularly from widely used paper cups that often end up in the environment and persist in freshwater ecosystems.

4 Conclusion

The current findings concluded that paper cups released certain ions, heavy metals, and microplastics into the water. While some

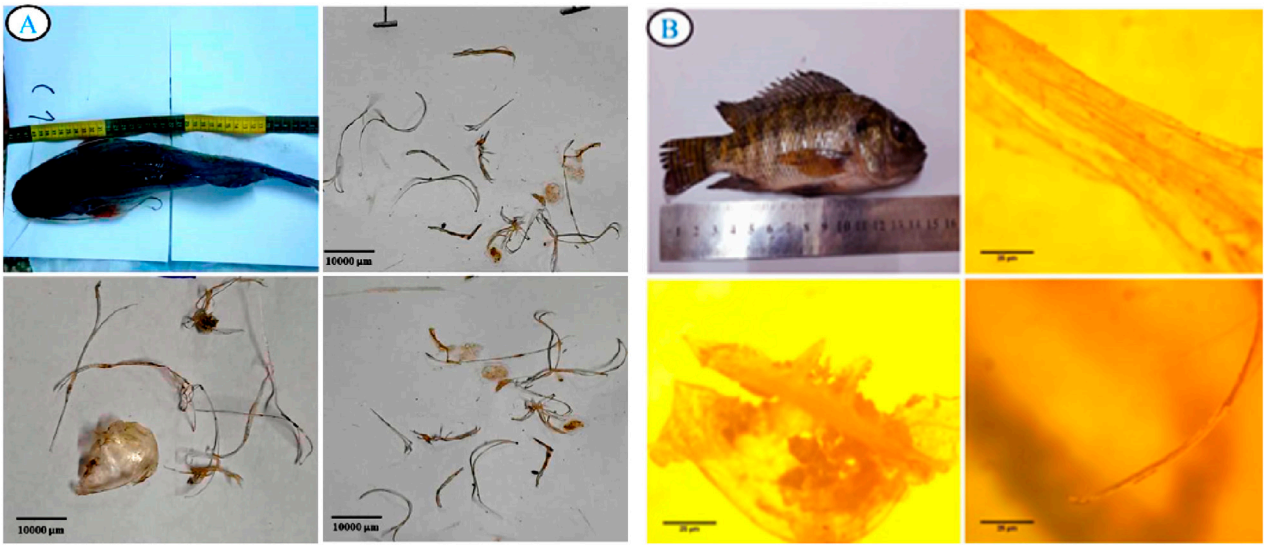

FIGURE 7
Photographs of the two investigated fish species from the River Nile, Assiut with their intestinal loads of microplastics, *Bagrus bajad* **(A)** and *Oreochromis niloticus* **(B)**.

microplastics, such as PVC, PET, and rayon, were not detected, their absence suggests they require more time to biodegrade. Our study identified microplastics in specific fish species from the River Nile at Assiut, highlighting potential threats from microplastic pollution and paper cup waste. The contamination caused by paper cups affected water quality, harming aquatic organisms and potentially endangering human health through fish consumption. While single-use paper products have harmful environmental impacts, banning them outright without providing viable alternatives is not a practical solution. Thus, this study recommends that future research should focus on developing strategies to minimize paper product pollution at its source. They should also assess the benefits of consumer education campaigns and other initiatives aimed at encouraging eco-friendly behavior, as well as the immediate and long-term environmental effects of these actions. Overall, the most promising solution to mitigate pollution from paper cups is to combine the environmental consciousness of consumers with economic measures for purchasing single-use products to reduce their usage.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material, further inquiries can be directed to the corresponding authors.

Ethics statement

The animal studies were approved by Assiut university committee, MBRSI-Research Ethics Committee. The studies were conducted in accordance with the local legislation and institutional requirements. Written informed consent was obtained from the owners for the participation of their animals in this study.

Author contributions

ZE: Conceptualization, Investigation, Methodology, Resources, Writing–original draft, Writing–review and editing. UM: Conceptualization, Supervision, Writing–original draft, Writing–review and editing. A-DS: Conceptualization, Investigation, Methodology, Resources, Supervision, Writing–original draft, Writing–review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Generative AI statement

The author(s) declare that no Gen AI was used in the creation of this manuscript.

Publisher's note

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Mind the gap: Sustainable management of the surging plastic waste in the post-COVID-19 pandemic

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The outbreak of COVID-19 inevitably boosted the global consumption of personal protective equipment (PPE), including face masks, gloves, and protective clothes. While wearing PPE could protect the public health from the transmission of infectious diseases, a concern draws attention on the environmental issues of plastic waste. This review examines the dual challenges of managing pandemic-associated plastic waste and mitigating the ecological and health risks posed by micro- and nanoplastics (MNPs). The results showed that the traditional technologies including landfilling and incineration accounted for around 40% and 25%, respectively. The incineration could reduce over 90% of the waste volume, but release MNPs or byproducts. Circular economy strategies—guided by reduce, reuse, and recycle principles—offer promising alternatives, particularly advanced thermochemical recycling that converts waste into value-added chemicals. For PPE, disinfection prior to reuse is of great necessity, including incineration (>800°C), chemical disinfection (ozone, H_2O_2), or physical methods (steam, microwaves). Although sorption and filtration strategies could remove the MNPs with over 99% efficiency, they are still in the lab-scale. Biological solutions—such as bacteria, enzyme, and worms—demonstrate potential for degrading synthetic polymers. This work underscores the urgency of integrating circular economy frameworks and tried to submit a comprehensive proposal to reduce the plastic waste, which could finally reduce the environmental burden brought by the COVID-19 pandemic.

KEYWORDS

post-pandemic period, plastic waste, personal protective equipment, health risks, management policy

1 Introduction

The outbreak of the COVID-19 pandemic has led to a significant increase in the usage of personal protective equipment (PPE). The widespread use of PPE, particularly face masks, gloves, sanitizer bottles, medical test kits, has played a critical role in blocking the transmission of the coronavirus (Wu et al., 2024a). The global plastic production increased from 370.7 million tonnes (Mt) in 2018 (pre-pandemic) to 413.8 Mt in 2023 (post-pandemic) (PlasticEurope, 2025). Even in the post-pandemic, the public perceptions and attitudes towards health and hygiene practices have changed, resulting in an annual growth rate of approximately 10% (Isobe et al., 2019). Wearing masks in public, which was once considered unusual outside of specific cultural contexts or health scenarios, has now become widely accepted and normalized. Therefore, the rapid escalation in PPE usage disrupted global supply chains and overwhelmed existing waste management systems (Zhao et al., 2024; Li et al., 2025).

Despite the concerted efforts made by the Basel Convention's Plastic Waste Partnership to advance global plastic waste management frameworks, about 79% is finally discarded in landfills or the natural environment (Geyer et al., 2017). Without appropriate treatment, the discharge of PPE would be particularly problematic since the pathogens or viruses found in the PPE could be the origins of the multiple outbreaks. PPE together with other plastic waste could also entangle the fauna and animals in the ecosystems, finally leading to injuries, impaired mobility, or even death, while contributing to long-term ecological disruption and threatening biodiversity in affected habitats. More than 267 species were reported to be influenced by plastic waste, especially 86% of turtles, 44% of birds, 43% of mammals, and various fish species (United States Environmental Protection Agency, 2022). Recent publications also revealed that the presence of plastic waste could also change the climate by disrupting carbon sequestration processes and altering the reflectivity of ice and terrestrial surfaces, thereby exacerbating global warming and climate instability (Sunil et al., 2024; Villarrubia-Gómez et al., 2024). On the other hand, climate change (e.g., higher temperature, stronger UV radiation, etc.) can further increase the degradation of plastic waste, releasing uncountable microplastics (MPs; <5 mm) or nanoplastics (NPs; <1 μm) into the environment (Wei et al., 2024).

Individuals are inevitably exposed to micro-(nano-)plastics (MNPs) through ingestion, inhalation, and dermal contact, contributing to health risks such as oxidative stress, chronic inflammation, and neurological impairments (Vethaak and Legler, 2021). For instance, exposure to MPs at environmentally relevant concentrations (e.g., 0.0125 mg mL^{-1}) induces mitochondrial dysfunction in human hepatic and lung cells, characterized by excessive mitochondrial reactive oxygen species (mtROS) production and suppressed mitochondrial respiration (Lin et al., 2022). Beyond cellular effects, MP accumulation in organs may trigger gut microbiota dysbiosis, exacerbate inflammatory responses, and impair neurobehavioral functions in model organisms (e.g., fish and mice), including altered locomotion and memory deficits (Guimaraes et al., 2021; Lee et al., 2022). The generation of MNPs could also release plasticizers, such as phthalates and bisphenol A, can interfere immune homeostasis and potentiate neurotoxicity, underscoring the systemic risks of plastic pollution through absorption, distribution, metabolism, and excretion (ADME) pathways (Wu et al., 2022).

The escalating global production of plastics, coupled with mounting evidence of the ecological toxicity of MNPs and plastic additives in ecosystems, has spurred interdisciplinary research efforts to develop targeted strategies for mitigating plastic pollution and its pervasive environmental impacts. Therefore, the study first summarized the traditional treatment of plastic waste and discussed the circular economic solution for common plastic waste. Second, the review evaluates the suitable treatments concerning the PPE generated by the pandemic. Finally, recent treatments of MNPs were also summarized and evaluated. Overall, this manuscript will be of vital importance to understand the treatments of plastic waste and MNPs, which can be conducive to reducing the overloaded pressure on the ecosystem, especially the possible risks brought by coronavirus disease prevention manners during (or even after) the pandemic.

2 Traditional treatment and circular economic solutions of plastic waste

Traditional approaches for plastics mainly include landfilling and incineration, accounting for around 40% and 25%, respectively (Vanapalli et al., 2021). Landfilling requires large space, along with releasing MNPs after gradual fragmentation under the influence of environmental processes (e.g., UV radiation and biological effects). The disposal strategy is unsustainable and contains the potential risks of contaminating soil and groundwater in the long term (Qi et al., 2020). Incineration reduces the waste volume by over 90% but with the release of MNPs and other hazardous substances (e.g., dioxins) into the atmosphere (Allen et al., 2019). Even before the pandemic, the International Solid Waste Association reported that the present waste disposal arrangements have been incapable of handling plastic waste properly (Jang et al., 2020). Coupled with the inevitably impending increase of plastic consumption during the pandemic, the situation will further overwhelm waste disposal systems worldwide (Patrício Silva et al., 2020). Thus, the search for circular economic solutions becomes more urgent than ever.

The circular economic solution requires plastic waste to be guided by the "3R" principles: reduce, reuse, and recycle (Figure 1). Governments have sought to minimize plastic use by raising public awareness of the negative consequences of MNPs pollution. Through outreach and advocacy, the public is encouraged to reduce plastic consumption through daily life decisions such as giving up plastic consumer products, applying reusable materials and recycling plastic wastes for multiple purposes. It should be noted that diminishing plastic consumption in lab work is also very critical as researchers (only accounting for 0.1% of the total population) disproportionately cause approximately 5.5 million tons (2%) of plastic waste worldwide (Urbina, M., Watts, A., Reardon, 2015). Some regulations were promulgated to ban plastic of low quality from entering the market and levy taxes on plastic consumption (Clancy et al., 2023). Meanwhile, some plastic products free of hazardous materials, such as plastic packaging (daily) and pipette-tip boxes (laboratory), could be reused after cleaning and disinfection (Wu et al., 2024b).

Recycling has emerged as a promising technology to reduce the impact of plastic wastes (Qi et al., 2020). According to plastic types and applications, different approaches including primary, secondary, tertiary and quaternary recycling can be adopted (Liu et al., 2024). Primary recycling is defined as closed-loop recycling, whereas secondary recycling referred to downgrading. These two recycling methods belong to mechanical recycling, which is the key approach employed to recycle plastic wastes (Liu et al., 2024). Mechanical recycling has several limitations, including difficulties in sorting out and high levels of contamination (Bezeraj et al., 2024). Tertiary recycling refers to completely depolymerizing the plastics into chemical constituents, which are then converted into value-added products, while quaternary recycling refers to recovering energy from waste or valorization (Wu et al., 2024b). Current studies focus on tertiary and quaternary recycling with three main directions: improving the recycling efficiency, reducing the need for sorting and expanding recycling plastic types. Two main

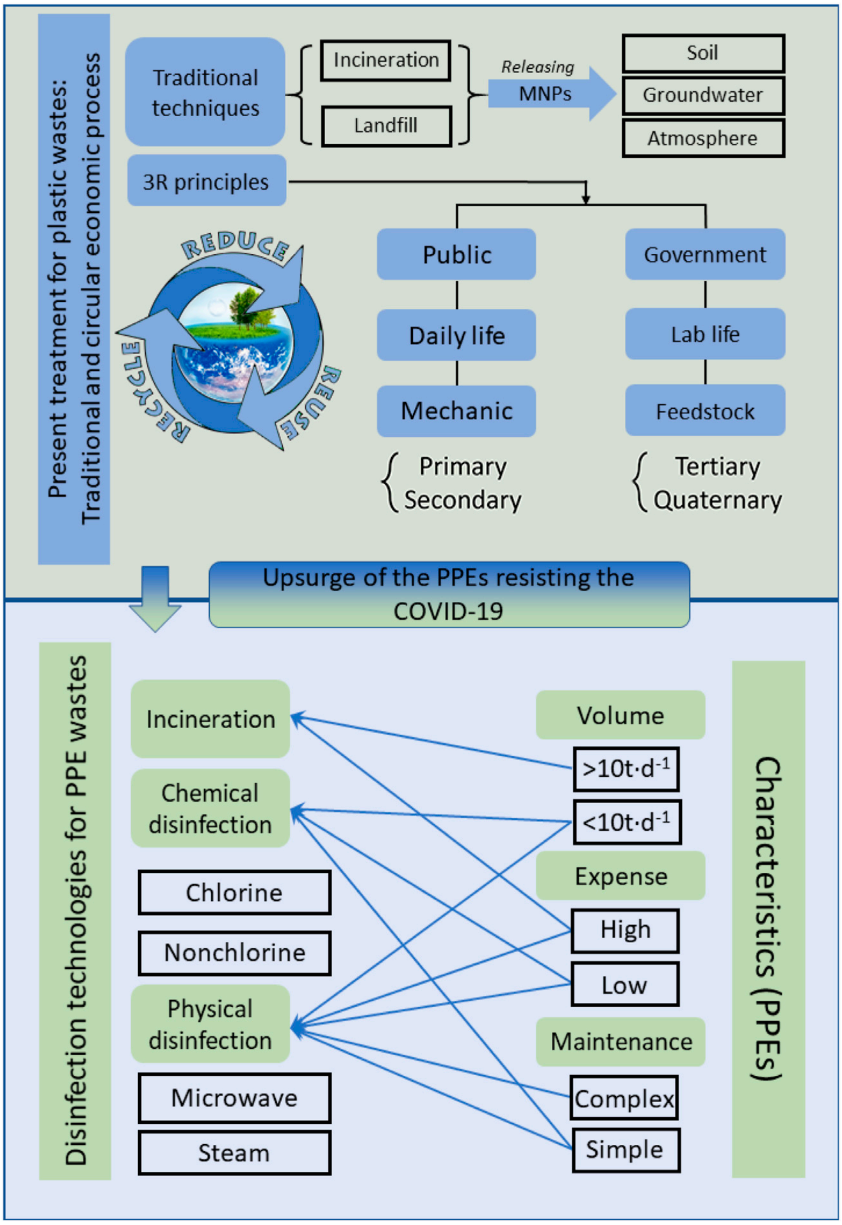

FIGURE 1
Present and advanced management for plastic wastes and the disinfection process for PPE wastes generating from resisting the COVID-19.

thermochemical routes were further proposed, namely, pyrolysis and gasification. Through pyrolysis, the single-type or mixed plastics wastes were converted into a mixture of value-added products such as hydrocarbons, hydrogen, carbon nanotubes, carbon monoxide and liquid fuels (Wu et al., 2024b). Gasification theoretically offers feedstock flexibility regarding the plastic type that can engender various types of useful gases (Choi et al., 2024). Yao et al. (2018) demonstrated a two-step process, including decomposition and catalyzation, that can reform plastic wastes to hydrogen-rich gases. Great improvement on a one-step technique was proposed by deconstructing plastics into hydrogen and carbons simultaneously (Jie et al., 2020). With the catalyzation of aluminum ferrite spinel, a yield of over 97 wt% of the hydrogen (55.6 $mmol \cdot g^{-1}$ plastic) was produced from the depolymerized plastic (Jie et al., 2020). Another one-step technique developed by Zhang et al. (2020) converts PE of various grades directly to liquid alkylaromatics at low temperatures (280°C ± 5°C), and the yield rate was kept above 80 wt% simultaneously. Although these techniques are still in the lab-scale, the prospects for treating the plastic wastes as a valuable sourced feedstock for generating value-added products seem attractive and promising.

3 Treatment for PPEs related to the COVID-19 pandemic

Eliminating the pathogens or virus is a critical procedure in the safe handling and disposal of abandoned PPE before any further treatment can be conducted. This is particularly crucial in the context of highly infectious diseases such as COVID-19, where improper disposal could lead to secondary transmission. In the COVID-19 pandemic, around 247 tons·day^{-1} of medical wastes were generated in Wuhan, approximately 5 times higher than before the outbreak (40–50 tons·day^{-1}) (You et al., 2020). A similar increase in medical waste was also found in many other cities that were affected by the pandemic. These massive discarded PPEs are far beyond the treatment capacity of hazardous wastes. Therefore, some countries try to utilize the municipal solid waste management system. According to the regulations on the administration of medical wastes, these wastes can be further handled after being decontaminated adequately through incineration, chemical and physical disinfection (Janik-Karpinska et al., 2023). Incineration treatments refer to the process of eliminating pathogens or viruses completely under a high temperature. As most of PPEs are made by PP and polyester, the incineration processes are relatively harmless, straightforward and efficient in which up to 90% organic matters could be burned up at over 800°C, thereby eliminating the hazardous components after the complete combustion. However, the remaining ash and gaseous byproducts must be carefully managed to prevent secondary pollution. The process is widely adopted when dealing with waste of large volume (>10 t·d^{-1}) with sufficient financial support (Figure 1). Despite it contained high effectiveness, incineration still requires substantial financial and infrastructural support, making it less feasible for regions with limited waste volumes (<10 tons per day) or inadequate funding. When the waste volume is smaller than 10 t·d^{-1} with limited financial resources, both chemical and physical disinfection would be preferred. Chemical disinfection is the process of killing pathogens or viruses with some typical disinfectants, including chlorine- (e.g., sodium hypochlorite) and nonchlorine-disinfectants (e.g., hydrogen peroxide, ozone, UV, etc.) (He et al., 2020). Chlorine disinfectants effectively inactivating pathogens by oxidizing the peptide bonds and inactivate the proteins. However, this process would release harmful byproducts such as dioxins and aromatic chemicals, which pose environmental and health risks. Common nonchlorine-disinfectants, like ozone, hydrogen peroxide and ultraviolet irradiation, offer safer disinfection with minimal harmful residues. They are often utilized to denature the proteins, resulting in the inactivation of the viruses. Physical disinfection means the destruction or removal of the pathogens by physical methods, including microwave and steam disinfection. Microwave with wavelengths between 915 ± 25 and 2450 ± 50 MHz can be absorbed by the substances and then generate the heat via molecules vibrating and rubbing (Wang et al., 2020).With the accumulation of the heat, the temperature would rise to the range from 177°C to 540°C and exhibit high-temperature disinfection under the inert atmosphere. This method has been confirmed to be much effective in on-site inactivate coronavirus by the Chinese Ministry of Ecology and Environment. Another common technique is steam disinfection, referring to the wet heat treatment that inactivates the proteins and kills the microorganisms under the saturated water vapor (93°C–177°C) (Ilyas et al., 2020). Under this temperature, the time needs for disinfection often around 20 min. This technique has been proved to contain low investment and operation cost (Pereira et al., 2025). Consequently, the amount of trash, available funds, and infrastructure all influence the decontamination method selection. For large-scale treatment, incineration is still the best option, but for smaller operations, chemical and physical disinfection offer good substitutes. In order to ensure sustainable medical waste management during future pandemics, future research should concentrate on increasing disinfection efficiency while reducing environmental effects.

4 Removal of MNPs using sorption and filtration methods

The ecotoxicity of MNPs calls for the development of further removal techniques, including sorption and filtration methods. These small particles are prone to being adhered onto the surface of marine algae, such as *Fucus vesiculosus, Pseudokirchneriella subcapitata*. The results found that the microalgae capabilities could be relatively high for the MNPs with positive charges as the electric attraction effects generated from the anionic polycarbohydrates (Wang et al., 2025). Through adsorption, microplastics could be captured and then removed by the filtration process. Among filtration technologies, membrane-based technologies have been identified as highly efficient filtration method that can successfully remove MNPs from aquatic environments. For instance, membrane bioreactors achieved a remarkable removal rate of 99.9%, reducing the turbidity from 195 NTU to <1 NTU within just 20 min, regardless of MNPs' shape or size, even microfibers as small as 10–100 μm in both influents and effluents (Talvitie et al., 2017). When integrated with the activated sludge technique, the removal technique could be successfully scaled up to the pilot-scale, achieving a 99.4% removal efficiency (Lares et al., 2018). However, practical challenges limit its applicability for on-site MNP removal in natural water bodies. Consequently, researchers have shifted focus toward biological interaction as a sustainable alternative to degrade the natural or synthetic plastics. Several outstanding work demonstrated that the MNPs in the environment can be degraded by a series of microorganisms in laboratory trials, such as bacteria (Yoshida et al., 2016), enzymes (Austin et al., 2018; Tournier et al., 2020), and worms (Yang et al., 2015b; 2015a). In detail, *Ideonella sakaiensis*, a bacterium discovered by Yoshida et al. (2016) broke down polyethylene terephthalate using specialized enzymes like PETase. Similarly, the gut microbiota of *Galleria mellonella* waxworms enables polyethylene degradation, revealing insect-associated microbial communities as novel biodegradation agents (Yang et al., 2015b; 2015a). Furthermore, marine microbial consortia have been shown to metabolize polyethylene and polypropylene, emphasizing the role of diverse microbial ecosystems in addressing plastic pollution. These studies collectively underscore the growing focus on harnessing natural biological interactions—spanning bacteria, insects, and marine microbes—to develop sustainable solutions for plastic waste remediation.

5 Conclusion and perspectives

The COVID-19 pandemic has exacerbated global plastic pollution, driven by unprecedented PPE consumption and systemic gaps in waste management. Traditional methods remain dominant but release hazardous byproducts, while 3R principles offer a framework for mitigating plastic pollution. However, challenges persist in scaling these technologies and improving sorting efficiency. PPE disposal requires pathogen decontamination prior to treatment, with incineration being effective for large volumes but limited by cost and infrastructure. Chemical/physical disinfection provides viable alternatives for smaller-scale operations, though environmental trade-offs (e.g., disinfectant byproducts) must be managed. Meanwhile, MNPs pose escalating ecological and health risks, necessitating innovative removal strategies. Emerging biological solutions demonstrate promise for degrading synthetic plastics, highlighting nature-inspired pathways for MNP remediation.

Future efforts should drive sustainable plastic transformation through multidimensional innovation. Thermochemical recycling technologies need scale up while enhancing energy efficiency and feedstock adaptability to valorize plastic waste. Concurrently, biological degradation requires advancement to address the environmental persistence of recalcitrant polymers. Globally harmonized policies should enforce waste segregation standards under frameworks with assisted by the managements of environmental monitoring networks. Moreover, the cross-disciplinary research should be strengthened to unify materials innovation, microbiome engineering, and climate-resilient governance, creating a closed-loop nexus that simultaneously tackles plastic pollution and ecosystem health. The study tried to submit a comprehensive proposal to reduce the plastic waste to reduce the environmental burden brought by the COVID-19 pandemic.

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Conflict of interest

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Nutrient availability modulates the effects of plastic leachates on the growth and community dynamics of free-living freshwater bacteria

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Introduction: Plastic pollution poses a significant and increasing threat to aquatic ecosystems, i.e. contaminating water resources and posing health risks for humans and the environment. Yet, plastic leachates can also stimulate microbial growth and activities, impacting biochemical cycles in aquatic ecosystems. Synthetic polymers and their leachates vary in their chemical composition and thus differently impact micro- and higher organisms. This study aims to assess: i) how different synthetic polymer leachates affect free-living aquatic bacteria, and ii) how these effects vary at high vs. low nutrient conditions.

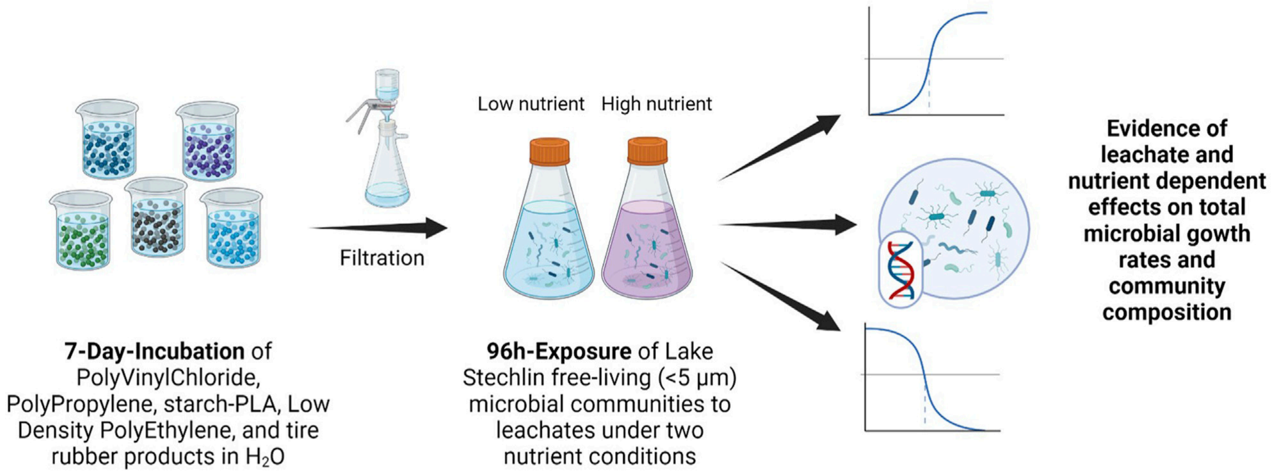
Methods: Leachates were extracted from five synthetic polymers, i.e. low-density polyethylene (LDPE), polypropylene (PP), polyvinyl chloride (PVC), starch-polylactic acid (starch-PLA), and tire rubber via incubation in ultrapure water under UV radiation. Free-living (<5.0 μm) microbial communities of Lake Stechlin, Germany, were exposed to these leachates, and changes in total microbial growth and community composition were analysed using dose response models.

Results: Nutrient availability resulted in different effects on total microbial growth and community composition of the tested synthetic polymers. For instance, PP leachates caused significant community shifts with increased total microbial growth rates at low nutrient conditions, but not at high nutrient conditions, whilst starch-PLA leachates led to community shifts at both nutrient conditions, but didn't impact total microbial growth.

Discussion: These results highlight the importance of leachate quality and nutrient availability for understanding the effects of leachates on microbial growth and community dynamics. Our findings reveal that synthetic polymer pollution has the potential to alter microbial loop functioning and hence biochemical cycles of aquatic ecosystems.

KEYWORDS

lake bacterial communities, synthetic polymers, biodegradable plastics, nutrient enrichment, bacterial growth and community composition



GRAPHICAL ABSTRACT

Graphical Abstract created in BioRender. Schubert, M. (2025) https://BioRender.com/w84p641.

Highlights

- Leachates of different materials showed varying effects on microbial growth rates and community composition
- Differences in effects of various leachates at low nutrient conditions were dampened at high nutrient conditions
- Inhibitory effects of leachates were maintained at both nutrient conditions, causing significant community shifts

1 Introduction

Plastic production has steadily increased worldwide, reaching >450 million tons in 2019 (OECD, 2022). Plastics have become increasingly diverse in chemical compositions (Chalmin, 2018), and, presently, numerous often unknown additives are added to plastics to modify their mechanical and chemical properties such as plasticizers, flame retardants, impact modifiers, antioxidants, UV-stabilizers and antimicrobials (Hahladakis et al., 2018). These additives are distributed unequally between plastic types and usages. For instance, in 2014, 90% of plasticizers produced were reportedly used for polyvinyl chloride (PVC) plastic, which is the fourth most common plastic-type to date (Czogała et al., 2021).

These chemicals have been reported to leach from plastic into the environment due to ageing processes (Bandow et al., 2017). Plastic aging is the process by which plastics undergo physical and chemical changes over time due to environmental factors such as UV radiation, temperature, mechanical stress, and microbial activity. Microbial activity may also alter the structure of plastics by breaking the bonds and using the polymers as an energy source (Salinas et al., 2023). For example, low density polyethylene (LDPE) is one of the most produced plastic types and releases mainly monoterpenes and fatty acids among the "high priority chemicals in plastics" according to ToxCast data (Zimmermann et al., 2019). Starch-polylactic acid (starch-PLA), a supposedly biodegradable alternative to petrol-based plastics requires a mixture of plasticizers, softeners, and elastomer tougheners to be used for plastic bags (Koh et al., 2018). In particular tire rubber granulates release various chemical cocktails composed of PAHs, bisphenols, phenols, and heavy metals into the environment (Halsband et al., 2020).

Leachates pose variable risks to organisms and the environment (Gunaalan et al., 2020). For instance, plastics leachates, to a large extent, are responsible for the observed adverse effects of plastics on aquatic (micro)organisms such as growth inhibition of the microalgae *Dunaliella teriolecta* when exposed to polyethene (PE), polystyrene (PS) or polypropylene (PP) (Schiavo et al., 2021). Additionally, plastic leachates of LDPE, PS, and PLA have been shown to cause shifts in the community composition of exposed marine bacteria (Birnstiel et al., 2022). As plastic leachate compositions and quantities can greatly vary between plastic types, we hypothesized that leachates of various synthetic polymers result in variable effects on the growth and community composition of free-living bacteria in freshwater.

Plastics, primarily composed of organic carbon, can release leachates that serve as a carbon source for various aquatic bacteria. For instance, in lake ecosystems, leachates from low-density polyethylene (LDPE) have been shown to increase bacterial biomass by 2.29 times when added at environmentally relevant concentrations (Sheridan et al., 2022). The study also found that bacterial growth efficiency improved by an average of 1.72 times, as the organic matter released from plastic leachates was more accessible than the natural organic matter in boreal lakes of Northern Europe. This has significant ecological implications. According to the microbial loop concept (Azam et al., 1983), bacterial uptake of dissolved organic carbon (DOC) is critical for channelling otherwise inaccessible carbon into aquatic food webs. When leachate DOC, derived from primary and secondary plastic constituents, contributes to bacterial biomass production, it enters nutrient cycles primarily through the consumption of bacteria by protists, potentially altering food web dynamics. Consequently, plastic pollution may coincide with carbon and nutrient enrichment in aquatic environments (Abreo et al., 2015). The ecological impact of plastic leachates likely varies depending on the type of plastic and its influence on microbial growth, potentially reshaping aquatic food web structures and ecosystem functions.

The objectives of this study were twofold: 1) to compare the effects of LDPE, PP, PVC, starch-PLA, and tire rubber leachates on growth rates and community composition of pelagic bacteria of a lake ecosystem, and 2) to investigate how the observed adverse effects change with nutrient availability result from the leaches from plastic. First, plastic leachates were extracted from five commercial synthetic polymer products (LDPE, PP, PVC, compostable starch-PLA, tire crumb rubber). LDPE, PP, PVC, and tire crumb rubber were selected for their high relative occurrences in aquatic environments (Uurasjärvi et al., 2020; Yuan et al., 2019; Capolupo et al., 2020) as well as for their documented effects on aquatic microorganisms (Sheridan et al., 2022; Schiavo et al., 2021; Sarker et al., 2020; Halsband et al., 2020). The starch-PLA bag was selected for comparison with a form of "biodegradable" plastics. Then, complex, free-living bacterial communities from Lake Stechlin were exposed to the extracted leachates at different nutrient conditions.

2 Materials and methods

2.1 Materials

Commercially available polymers were used to generate leachates differing in chemical quality and quantity. LDPE plastic bags ("Beste Wahl Gefrier-Beutel", Rewe, Cologne, Germany), PP rope ("PP-seil", Toom, Cologne, Germany), PVC floor ("Grauer PVC-Boden", Toom, Cologne, Germany), compostable plastic bags, made of starch-PLA (Gut und Günstig Kompostierbare Bio-Beutel), and weathered tire (Bridgestone Europe, Saventem, Belgium) were used for leachate production as described in the section below.

2.2 Preparation of plastic leachates

The extraction of the synthetic polymer leachate was performed following the method of Sheridan et al. (2022). Briefly, before the incubation, all plastic items were treated with 70% ethanol to avoid interferences (through carryover of microorganisms or organics) in leachate release during incubations. Plastic items were cut into approximately 1 cm^2 pieces before weighing. For tire rubber, tire crumbs of variable sizes were produced using a hardened steel file on the outer-most layer of a weathered tire (Bridgestone). The tire crumbs were cleaned and disinfected in 70% ethanol, which evaporated entirely before incubation. Plastics were added to 500 mL Milli Q Water in glass beakers at concentrations of 8.88 g/L. The Milli-Q water used in the study was taken from a Millipore® filtration system (RiOs™ Essential 16 Water Purification System). All samples were incubated in triplicates for 7 days at 20°C–25°C on shakers (100 rpm) using a 12 h:12 h cycle of UV-light illumination to increase the weathering of all plastic pieces (Klein, et al., 2021). UVC irradiation was performed by a TNN 15/35 lamp that emits a single narrow line at 254 nm. Here, we applied the spectral irradiance of about 2.6 W cm^{-2} for an irradiating distance of 50 cm from the lamp to the surface of the water. All samples were kept under cling film to avoid organic carbon contamination from air. In addition, Milli Q water without any addition of plastics (control H_2O) was incubated alongside all plastic-containing samples. After incubation, plastic leachate samples were filtered using 3x pre-rinsed, sterile 0.22 µm polycarbonate filters (Nucleopore) to remove any plastic particles as well as any microbial contamination. Our initial results showed that 1x filtration through polycarbonate filters resulted in a number of contaminated samples, in our final protocol, all leachates were filtered twice. Leachate samples were analysed using the Total Organic Carbon analyser TOC-L (Shimazdu CPH-CSN, Japan) for DOC concentrations using the EN 1484 method guidelines with oxidation. This metric was selected as a good proxy for amount of leachates produced since organic carbon constitutes the largest component of all plastic leachates.

2.2.1 Statistical analysis

Standard t-tests (Student) were used for statistical analysis. The experiment was repeated five times. For each repetition, an average leaching rate was determined based on the average of triplicates, subtracting the DOC concentration in the respective controls. Samples were statistically compared to the LDPE leaching rates.

2.3 Complex community exposure

2.3.1 Bacterial community incubations

Complex communities of lake microorganisms were sampled from the well-mixed epilimnion of Lake Stechlin in Germany (53: 10N, 13:02E) at a depth of 2 m. Lake water was filtered through pre-rinsed, sterile 5.0 µm polycarbonate filters (Nucleopore, Whatman, Cytiva, Kent, United Kingdom) to retain the free-living fraction of the microbial communities and avoid contamination by particulate organic matter. Filtered lake water was transferred into sterilized Erlenmeyer flasks and the microbial communities thereof were exposed to a concentration of 0.167 mL Leachate.mL^{-1}.

As such, the final concentrations of DOC from plastic leachates were: 46.9 mg DOC L^{-1} for starch-PLA, 0.4 mg DOC L^{-1} for LDPE, 9.7 mg DOC L^{-1} for PP, 4.2 mg DOC L^{-1} for PVC, and 3.5 mg DOC L^{-1} for tire rubber exposure groups. To control groups were also incubated, one exposed to the control water of the plastic leachate preparation (control H_2O), the other exposed to sterile Milli-Q water (0-control). For LB medium incubations, LB medium was added to a final concentration of 1:200 (ca. 60 mg DOC L^{-1}). The DOC concentration of the sampled lake water was at approximately 6 mg L^{-1}. A large headspace in the Erlenmeyer flasks ensured oxic conditions during the entire incubation. Triplicates were used for each experimental condition. All flasks were incubated on a shaker at 50 rpm in continuous light for 72 h (OECD, 2011). The ambient temperature was 17°C to avoid any influence of a sudden temperature change. Samples, taken from each flask before and after incubation, were fixed with formaldehyde (3% fin. conc.). The rest of the samples was filtered through 0.22 µm Durapore filters and kept frozen at −70°C for later DNA extraction.

2.3.2 Bacterial growth rate determination

Microbial growth rate was calculated using change in bacterial cell counts over time. Cell numbers were counted before and after incubation using a flow cytometer (BD FACS Aria). The samples were stained by DAPI (Sigma-Aldrich, Burlington, United States) (fin. conc. 5 μg mL^{-1}) according to the protocol of Porter and Feig (1980). To determine the correct threshold, a sample was filtered sterile to exclusively measure the background noise to be subtracted as a blank. Additionally, certain samples were randomly selected to be counted manually by fluorescence microscopy to ensure that flow cytometer readings were correct. As the filtration method in the leachate isolation experiment revealed contaminated samples in this experiment, certain expositions and their respective controls were repeated to avoid any bacterial contamination.

2.3.3 Community composition analysis

DNA was extracted from frozen filters of the complex community exposition experiment using the phenol-chloroform method (Montero-Pau et al., 2008). Extracted DNA was quantified through Qubit and the relevant marker genes were PCR-amplified before sending the DNA for 16S rRNA gene amplicon sequencing. The V4 region (515F–806R primers) of the 16S rRNA gene (2 x ~250 bp) was sequenced on a MiSeq platform (Illumina, San Diego, United States). The sequence data were processed using the DADA2 pipeline (Callahan et al., 2016). Adapters were removed before further analysis. Quality profile analysis revealed a significant decline in quality beyond 270 bases for forward reads and 220 bases for reverse reads. Consequently, the reads were trimmed at these points. Ambiguous bases were removed, bases with a quality score below 2 were truncated, and reads with an expected error rate greater than 2 were discarded.

The DADA2 machine learning algorithm was employed to train error models based on the filtered sequences. The sequences were then dereplicated using the dereplication function. Forward and reverse reads were merged, and chimeric sequences were removed. Finally, merged reads were assigned to taxonomy using the IDTAXA algorithm (Murali et al., 2018) with the Silva database v138.1. Taxonomic classification was performed using the RDP Bayesian classifier algorithm within DADA2.

2.3.4 Statistical analysis

Total abundances between samples groups were compared using t-tests (Student) or Mann-Whitney when the group did not follow a normal distribution.

Total reads per sample were deemed sufficient, ranging between approximately 41,000 and 75,000 reads. Alpha diversity was estimated using the Shannon index. Either regular t-tests (Student) or the Mann-Whitney U test were used to calculate significant differences in comparison to control samples. Bray-Curtis dissimilarity matrices were established for β-diversity. The statistical significance of differences was checked using PERMANOVA. The "vegan" package (Oksanen et al., 2022) was used for visualization of β-diversity using an NMDS analysis. Relative read abundances were used as a proxy for relative abundances of genera present in the sample to compare proportional changes of different genera between leachate exposure groups. Percentages of reads of a genus were compared between exposure groups using standard t-tests (Student).

2.4 Exposure to different concentrations

Leachates of PP, PVC and tire rubber were selected for exposure experiment as they have yielded significant differences in total abundance and growth rates of bacteria compared to the control groups in our previous experiment (2.2.1).

2.4.1 Communities exposure

Complex communities of free-living bacteria from Lake Stechlin were used, and leachates were diluted with sterile Milli-Q water to create four exposure concentrations in the treatments: 0.00167, 0.0167, 0.083, and 0.167 mL leachate per mL liquid. This resulted in final DOC concentrations of 0.09, 0.85, 4.26, and 8.52 mg DOC per L for PP; 0.03, 0.31, 1.57, and 3.14 mg DOC per L for PVC; and 0.04, 0.43, 2.16, and 4.32 mg DOC per L for tire rubber. Control samples were exposed to the same concentration series using control water (control H_2O) from the preparation of plastic leachates (2.2), with two additional controls added to account for potential dilution effects: one using double-filtered (0.22 μm) and autoclaved Lake Stechlin water and the other using sterile Milli-Q water (0-control). To increase nutrient and organic matter concentrations, exposure groups were also repeated with the addition of LB medium at a final concentration of 1:200 (ca. 60 mg DOC L^{-1}). The DOC concentration of the sampled lake water was at approximately 6 mg L^{-1}. Triplicates were prepared for all experimental conditions to ensure reproducibility.

2.4.2 Growth rate estimation

Samples of each culture were taken before and after 72 h of incubation and fixed with formaldehyde (3% fin. conc.). Cell abundance was estimated as described above for the complex community experiment.

2.4.3 Statistical analysis

Dose-response models were applied using the R-packages drc (Ritz et al., 2015) and minpack.lm (Elzhov et al., 2023) for regression analysis and determining the EC_{50} values.

3 Results

3.1 Leachate concentrations based on DOC

The compostable, biodegradable plastic bag (starch-PLA) yielded the highest leaching rates (17.8 mg DOC $week^{-1}$ g $polymer^{-1}$, p = 2.128e^{-07}), followed by the PP rope (4.3 mg DOC $week^{-1}$ g $polymer^{-1}$, p = 6.118e^{-09}) (Figure 1). PVC floor parts and the tire crumb rubber also leached significant amounts of DOC (1.5 mg DOC $week^{-1}$ g $polymer^{-1}$ and 1.1 mg DOC $week^{-1}$ g $polymer^{-1}$, respectively, p = 8.467e^{-05} and p = 0.003643), though these leaching rates were consistently lower than for the PP rope (4.3 mg DOC $week^{-1}$ g $polymer^{-1}$, p = 6.118e^{-09}). LDPE leached the lowest amount of DOC as compared to all other synthetic polymer products, and the amount of DOC in the LPDE leachates were close or occasionally even lower than in the control sample (Figure 1). Though leaching rates of LDPE were on average 0.15 mg DOC $week^{-1}$ g $polymer^{-1}$ (whilst subtracting the controls), the leaching rates were not significantly different from the controls.

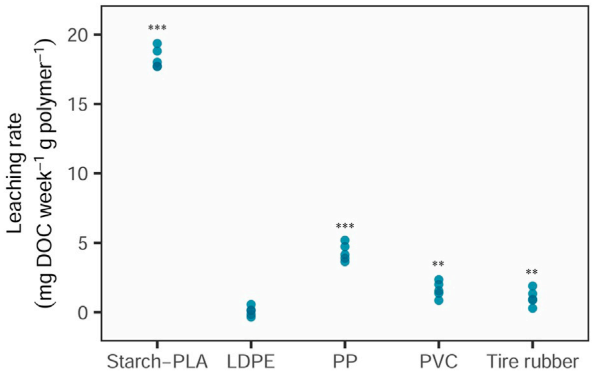


FIGURE 1
Leaching rates of synthetic polymer products. Leaching rate is calculated as DOC amount released by a given amount of synthetic polymer (mg DOC week^{-1} g polymer^{-1}). Synthetic polymers included: Starch-PLA: Starch Polylactic Acid, LDPE: Low Density PolyEthylene, PP: PolyPropylene, PVC: PolyVinylChloride, and Tire rubber: Tire crumb rubber. Statistical analysis consisted of t-tests (Student) ** represents significance of $p < 0.01$; *** represents significance of $p < 0.001$.

3.2 Microbial growth rate changes

No significant differences in microbial growth rates occurred between the 0-control and control H_2O samples at low nutrient conditions ($p = 0.11$ and $p = 0.48$ at low nutrient conditions; $p = 0.07$ and $p = 0.34$ at high nutrient conditions). As there was an insignificant difference, control H_2O samples were used for comparison with exposure samples. At low nutrient conditions (Figure 2), strong growth stimulation was noticed for PP ($p = 0.00742$) and PVC leachate samples ($p = 0.0039$). A statistically significant inhibition was noticed for the tire rubber leachate when compared with either the 0-control or control H_2O samples ($p = 0.006217$). However, neither LDPE nor compostable bag leachates showed any significant difference in bacterial growth to control H_2O samples. At high nutrient conditions (Figure 2), however, only PVC leachates revealed a significant increase in specific growth rate when compared to control H_2O ($p = 0.003904$). When comparing the normalized growth rates, there was a significant difference for PP ($p = 0.006$) and tire rubber leachates ($p = 0.002$) between the two nutrient conditions. For both leachate types, however, the addition of nutrients caused insignificant differences to the control H_2O samples in total abundance and growth rates.

3.3 Compositional changes

Overall, there were stronger differences in α-diversity between plastic leachate types at low nutrient conditions than at high nutrient conditions (Figure 3). Starch-PLA, PP, and PVC exposure groups showed a significantly stronger decrease in α-diversity than the LDPE exposure group ($p = 0.002$, $p = 0.03$, $p = 0.001$ respectively). This decrease in α-diversity was reflected by the average Shannon indices, which ranged between 0.3 and 0.5 units lower than control samples. Yet, the LDPE exposure group also showed a significant decrease in α-diversity when compared to control H_2O samples. The tire rubber exposure also tended toward a decrease in α-diversity though a high variability rendered it insignificant.

At high nutrient conditions, the only significant decrease in α-diversity occurred in the PVC exposure ($p = 0.006$). Though the Shannon index for the PP exposure tended to decrease as well, it was insignificant due to the high variance between samples. The starch-PLA exposure was the only exposure group which had significantly different α-diversities between both nutrient conditions ($p = 0.003$).

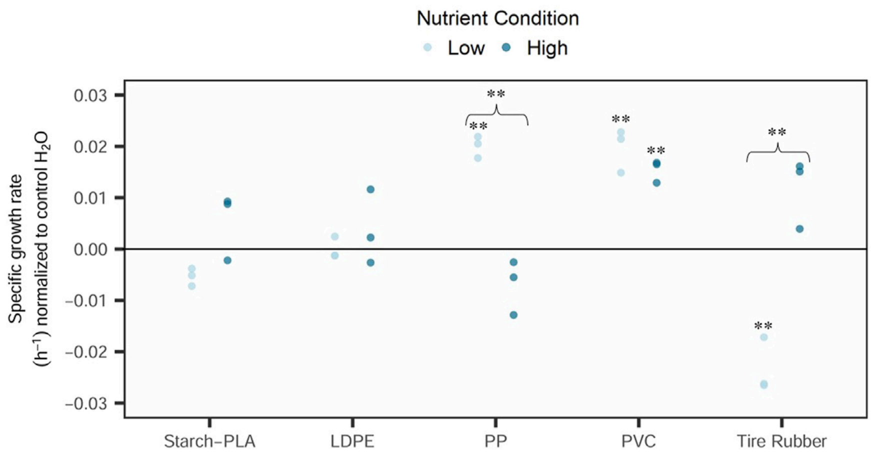


FIGURE 2
Microbial growth of natural free-living bacteria communities (<5.0 μm) in a 72-h *in vitro* exposition to synthetic polymer leachates at high vs. low nutrient conditions. Synthetic polymers included: Starch-PLA: Starch Polylactic Acid, LDPE: Low Density PolyEthylene, PP: PolyPropylene, PVC: PolyVinylChloride, and Tire rubber: Tire crumb rubber. Control H_2O samples were exposed to the control sample of the leachate extraction experiment. Leachate exposure samples were normalized to the control H_2O group by subtracting the median of this group to all values. All samples were run in triplicate. Statistical analysis consisted of t-tests (Student) or Mann-Whitney U test when the respective group does not follow a normal distribution. ** represents a statistical significance of $p < 0.01$. The curly brace indicates a significant difference between the same leachate types at different nutrient conditions. Both were compared by using the normalized values for each condition.

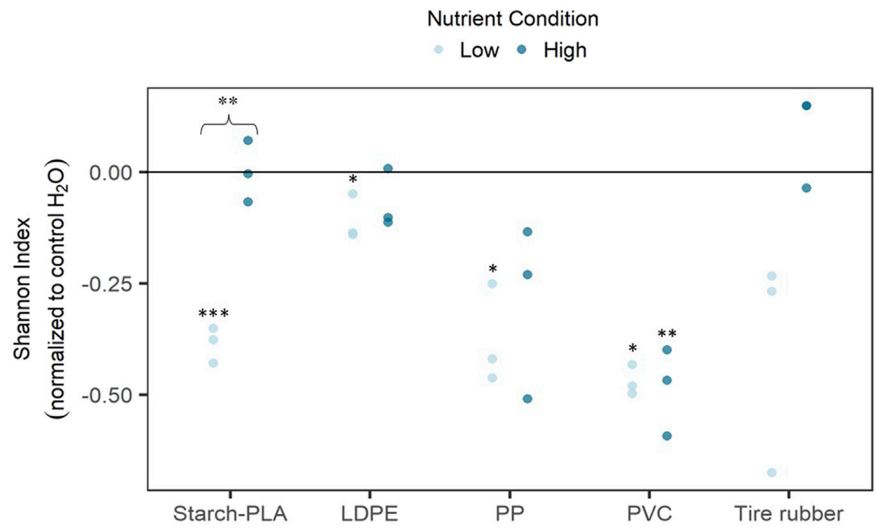


FIGURE 3
Alpha diversity (Shannon index normalized to the control H_2O sample) of natural free-living bacteria (<5.0 μm) after a 72-h *in vitro* exposition to synthetic polymer leachates at high and low nutrient conditions. Synthetic polymers included: Starch-PLA: Starch Polylactic Acid, LDPE: Low Density PolyEthylene, PP: PolyPropylene, PVC: PolyVinylChloride, and Tire rubber: Tire crumb rubber. Control H_2O samples were exposed to the control sample of the leachate extraction experiment. Leachate exposure samples were normalized to the control H_2O group by subtracting the median of this group to all values. All samples were run in triplicate. * represents a statistical significance of $p < 0.05$. ** represents a statistical significance of $p < 0.01$. Statistical analysis consisted of t-tests (Student) or Mann-Whitney U test when the respective group does not follow a normal distribution. *** represents a statistical significance of $p < 0.001$. The curly brace indicates a significant difference between the same leachate types at different nutrient conditions. Both were compared by using the normalized values for each condition.

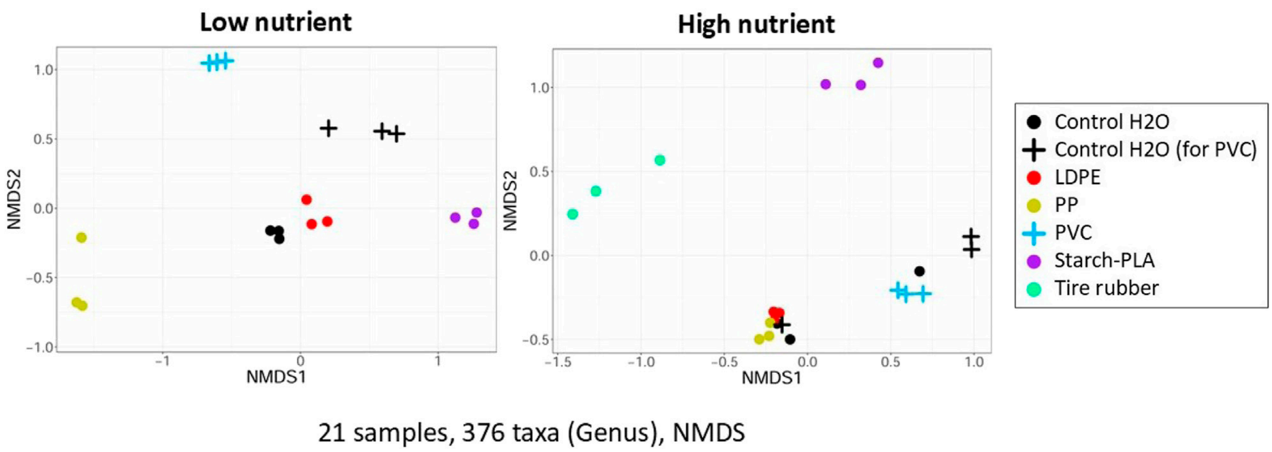


FIGURE 4
NMDS of Bray Curtis distances of natural free-living bacteria (<5.0 μm) after a 72 h *in vitro* exposition to synthetic polymer leachates at high and low nutrient conditions. Synthetic polymers included: Starch-PLA: Starch Polylactic Acid, LDPE: Low Density PolyEthylene, PP: PolyPropylene, PVC: PolyVinylChloride, and Tire rubber: Tire crumb rubber. Control H_2O samples were exposed to the control sample of the leachate extraction experiment. Leachate exposure samples were exposed to the leachates of the plastic type listed. As the experiment was repeated for the PVC exposure group, this exposure group and its respective control are represented with a plus sign rather than dots.

For tire rubber exposure, this difference was insignificant due to high variability between samples of the same exposure groups.

At high nutrient conditions, the only significant decrease in α-diversity occurred in the PVC exposure group ($p = 0.006$). Though the Shannon index for the PP exposure group tended to decrease as well, it was insignificant due to the high variance between samples. The starch-PLA exposure was the only exposure group which had significantly different α-diversities between both nutrient conditions ($p = 0.003$). For tire rubber exposure, this difference was insignificant due to the high variability between samples of the same exposure groups.

NMDS analysis revealed that LDPE exposure was most similar to control H_2O samples (Figure 4). Additionally, there were clear differences between all synthetic polymer exposure groups.

PERMANOVA using Bray-Curtis distances showed significant differences in community composition between each leachate exposure group and the respective control H_2O groups under low nutrient conditions (starch-PLA: p = 0.005, LDPE: p = 0.004, PP: p = 0.005, PVC: p = 0.013, tire rubber: p = 0.004). However, it is evident on the NMDS (Figure 4), that the community composition of the LDPE exposure group was the most similar to the control groups compared to all other exposure groups.

At high nutrient conditions, only bacterial communities of the tire rubber and starch-PLA exposures were significantly different from the respective control H_2O (starch-PLA: p = 0.036, tire rubber: p = 0.037). In contrast, control groups, PP, and LDPE exposures were overlapping. The PVC exposure was also not significantly different according to PERMANOVA. Note that PP exposure at low nutrient conditions was significantly different from the controls, yet at high nutrient conditions, they did not differ from the control H_2O exposure (p = 0.435). Similarly, the PVC exposure at high nutrient conditions was also not significantly different from the control H_2O exposure at high nutrient conditions (p = 0.318), despite clear differences at low nutrient conditions.

At low nutrient conditions, significant differences occurred between communities exposed to different synthetic polymer types (Figure 5). After exposure to starch-PLA, the hgcI-clade of *Actinomycetota* and *Pseudorhizobium* dominated these samples (on average 10.03% ± 2.19% and 5.4% ± 0.79), but Vogesella and *Acinetobacter* were present in only low proportions (0.09% ± 0.03% and 0.14% ± 0.12, respectively). Additionally, *Rhodoferax* was entirely absent from these samples. For the tire rubber exposure, the hglc-clade (19.3% ± 0.28) and *Pseudomonas* (4.8% ± 0.02) formed the dominant groups in these samples. Similar to the starch-PLA exposure, the proportion of *Vogesella* was also reduced (0.09% ± 0.03). On the other hand, *Vogesella* (26.2% ± 1.48) and *Acinetobacter* (25.3% ± 4.35) were by far the two most dominant genera in the PP exposures.

In the PVC exposure, *Acinetobacter* (36.73% ± 1.15) and *Flavobacterium* (17.7% ± 0.22) were the most dominant genera. The LDPE exposure group was most similar to control H_2O samples, with *Rhodoferax* (23.5% ± 3.44) and hgcI-clade of *Actinomycetota* (5.37% ± 2.16) forming, on average, the most dominant groups. Though some bacterial genera seem to be common in several exposures, individual leachate exposures differed in their overall composition (see Figure 4), especially the most dominant bacterial genera.

At high nutrient conditions, also at genus level resolution, there were smaller differences between leachate exposures and controls. The only pronounced compositional changes compared to the control H_2O samples occurred in starch-PLA and tire rubber exposures. For starch-PLA exposure, the dominant genera were *Acinetobacter* and *Massilia* (21.8% ± 4.32% and 15.0% ± 7.10, respectively), though there was some variability between replicates. On the other hand, *Pseudomonas* and *Massilia* were dominant after tire rubber leachate exposure (19.2% ± 1.32% and 30.8% ± 5.48, respectively). For all other exposures, i.e., LDPE, PP and PVC, differences in relative read abundance were insignificant between exposure groups and the respective controls.

Concerning effects on community composition, PVC was the only leachate type for which significant effects on total abundance and growth rates were observed at high nutrient conditions (Figure 2). Our bacterial community composition analysis indicated that this may be the result of a strong proliferation of *Acinetobacter* and *Vogesella*, the two dominant genera in the control H_2O samples at high nutrient conditions (Figure 5). This may also explain why there is a significant decrease in α-diversity at high nutrient conditions (Figure 3), whilst β-diversity was insignificantly different to control H_2O samples. Starch-PLA, despite it had no effect on total bacterial abundance, it resulted in significant community shifts at both nutrient conditions (Figure 4).

For the PP leachate exposure, however, addition of nutrients seemingly dampened the stimulatory effects on total bacterial abundance and growth rate as well as community shifts (Figures 2–5). For the tire rubber exposure groups, however, slight inhibitions across microbial communities occurred at low nutrient conditions. At high nutrient conditions, there was an apparent stimulation of growth rates for certain bacterial genera, which was seemingly equalled by stronger inhibitions, for instance of *Vogesella* (Figure 5). This may explain the lack of differences in overall growth rates compared to control H_2O samples (Figure 2), whilst there were significant differences in β-diversity when compared to the control H_2O at both nutrient conditions (Figure 4).

3.4 Concentration series exposure

At low nutrient conditions, there was a significant concentration effect ($p = 5.0e^{-04}$) for PP leachates exposures (Supplementary Figure 1a). The sigmoidal dose-response model was fitted to the PP leachate concentration series with an R^2 value of 0.868. Growth rates yielded an EC_{50} value of 0.43 mg DOC L^{-1} (Figure 6a). For PVC leachates (Supplementary Figure 1a), there was also a significant concentration effect ($p = 3.0e^{-05}$), but only at low nutrient concentrations. The sigmoidal dose-response model was fitted to the PVC leachate concentration series with an R^2 value of 0.761. These revealed a higher EC_{50} 4.74 mg DOC L^{-1} (Figure 6b). Though there is a disparity in the EC_{50} values for PP and PVC, it is important to note that the specific growth rate effects at these concentrations for PP and PVC exposures at the highest tested concentrations was on average 0.01 h^{-1} for both exposure groups.

The tire rubber leachates (Figure 6c), on the other hand, did not show data befitting a sigmoidal function thus no EC_{50} could be calculated. The specific growth rate for the second highest concentration of tire leachate (2.16 mg DOC L^{-1}) was significantly (p = 0.04459, 0.02138) higher than in the samples of 0.00167- and 0.0167 mL Leachate mL^{-1}. However, a linearized model was not statistically significant (p = 0.22). Since the R^2 value of this model was only 0.11, a linearized model is likely not ideal to represent this data. Therefore, a LOESS function was used to describe this dataset, which yielded an R^2 value of 0.812.

At high nutrient conditions, there was a significant increase in growth rate (Figure 6d) for samples which were exposed to PVC leachates when compared to control samples (p = 0.04). A dose response model was created for PVC at high nutrient conditions as the linearized model was statistically significant. The sigmoidal function yielded an R^2 value of 0.465. The EC_{50} value for the PVC exposure groups at high nutrient conditions was estimated at 0.04 mg DOC L^{-1}. Despite this lowered EC_{50} value at high compared to low nutrient conditions, the estimated change in

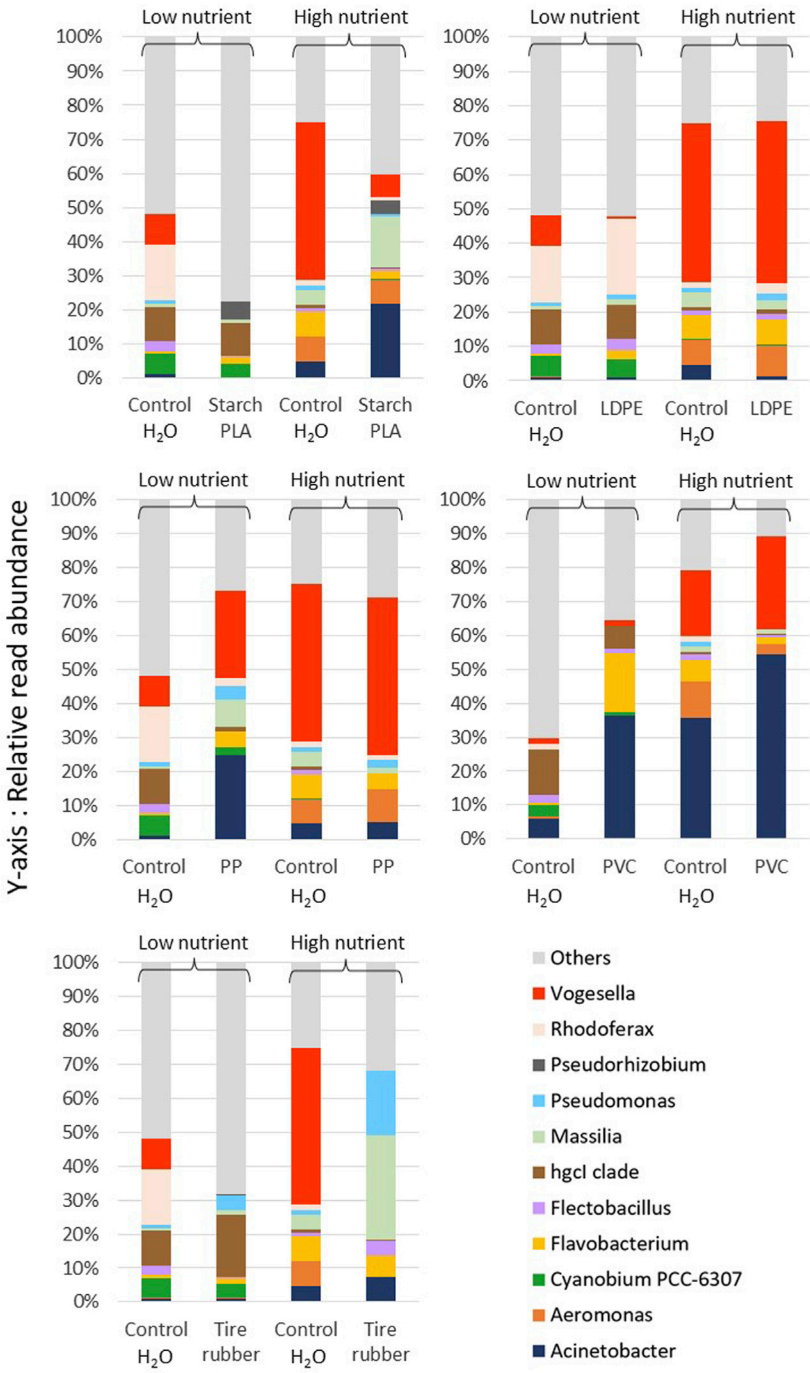


FIGURE 5
Average relative abundances of reads assigned to the most common genera of free-living freshwater bacteria (<5 μm) in each sample exposed to synthetic polymer leachates at high and low nutrient conditions. Synthetic polymers included: Starch-PLA: Starch Polylactic Acid, LDPE: Low Density PolyEthylene, PP: PolyPropylene, PVC: PolyVinylChloride, and Tire rubber: Tire crumb rubber. Control H_2O samples were exposed to the control sample of the leachate extraction experiment. Leachate exposure samples were exposed to the leachates of the plastic type listed. N = 3 for all sample groups except "Control H_2O" at high nutrient conditions and tire rubber leachates at low nutrient conditions for which N = 2, due to significant variation in relative abundance of *Acinetobacter* which skewed the average.

specific growth rate at the EC_{50} concentration was much lower at 0.0001 h^{-1} (compared to 0.027 h^{-1} at low nutrient conditions). The linearized models did not show any other significant changes in growth for any other sample (Supplementary Figure 1b).

Overall, changes in total specific growth rates at low nutrient conditions were weaker or disappeared at high nutrient conditions. This was most obvious for PP, though inhibitory effects of PVC were also weaker at high nutrient conditions.

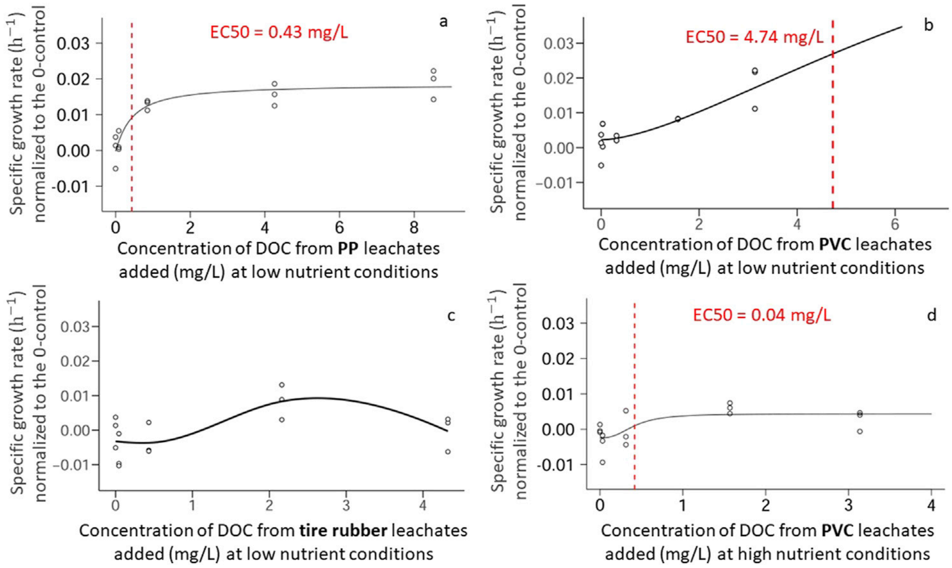


FIGURE 6
Dose-response models for different synthetic polymer leachates regarding microbial growth rates of natural free-living bacteria (<5.0 µm) from Lake Stechlin over a 72-h exposure period at either high or low nutrient conditions. **(a)** Exposures to PolyPropylene (PP) at low nutrient conditions. **(b)** Exposures to PolyVinyl Chloride (PVC) at low nutrient conditions. **(c)** Exposures to tire crumb rubber at low nutrient conditions. **(d)** Exposures to PolyVinyl Chloride (PVC) at high nutrient conditions. Concentration is expressed in volume per volume as this better represents the differences in leaching rates between plastic types.

4 Discussion

4.1 Effects of synthetic polymer leachates

Our experiments showed that plastic leachates of different synthetic polymer types have varying effects on microbial communities. A part of these variations can be explained by the differences in leaching rates among the various plastic products used in our study. Another factor is DOC quality of the different leachates and hence bacterial substrate availability (Romera-Castillo et al., 2022). The starch-PLA bag showed leaching rates which were 4–5 times higher than those of the PP rope. However, when natural free-living bacterial communities from Lake Stechlin were exposed to starch-PLA leachates, no significant differences in total growth rates were observed (Figure 2). In contrast, significant community shifts occurred at both high and low nutrient conditions in exposures to starch-PLA leachates, whilst no significant changes in growth rates were recorded (Figure 4). This indicates that starch-PLA leachates are interacting with the microbial communities, causing stimulation as well as inhibition, depending on the respective bacterial genera. This is reinforced by results of our exposure experiment of selected bacterial isolates from Lake Stechlin (Supplementary Figure 2). Thereafter, at high nutrient conditions, increased growth rates were observed for *Flavobacterium*, whereas a trend towards growth inhibition was observed for *Sphingomonas*. As starch-PLA leached the highest amounts of organic carbon, the lack of a strong stimulation of any genus implies that the main component of the organic carbon leached is little available for the free-living bacterial community of the lake. Starch-PLA bags, though sold as compostable, have been found to be biodegradable by maximal 90% in industrial compost systems, even at 60°C as per the European norm laboratory test methods (Ciriminna and Pagliaro, 2020, EN 14046). In our study, the biodegradability of this leachate type in natural aquatic settings seems to be very low, as has been found earlier for certain PLA types in the ocean (Egea et al., 2024).

The relative composition of leachates is important in determining effects on microbial communities (Gunaalan et al., 2020). Tire rubber, which is known to harbour a large variety of additives, showed inhibitory effects on growth rates of natural free-living bacterial communities at low nutrient conditions (Figure 2) - though released DOC amounts were similar to those of PVC. Moreover, our concentrations exposure experiment revealed that effects of PP, PVC and tire rubber leachates on microbial dynamics largely depend on exposure concentration, as has been previously reported in the literature (Li et al., 2016). At low nutrient conditions (Figure 6), PP and PVC leachates seem to well fit a typical sigmoidal model. The accuracy of the model can be put into question for PVC, as it seems questionable (from the four applied leachate concentrations) whether near-maximum effects have been reached. The tire rubber leachates do not fit a sigmoidal model at all, but due to statistically significant changes in growth rates for an intermediate concentration and the growth rate inhibition observed of natural free-living bacteria (Figure 2), the data can be represented by a biphasic dose-response model (Figure 6c). It is possible that at intermediate concentrations, the inhibitory chemicals are not concentrated enough to reveal a strong inhibitory effect on lake bacteria, and thus there seems to be additional available organic carbon for the bacterial community.

Though no biphasic effects of tire rubber leachates on microbes have been reported in the literature, there is evidence that tire rubber granules can both stimulate growth of certain microorganisms and inhibit growth of others. Among the organisms growing on tire leachates were members of the *Pseudomonas* genus (Leff et al., 2007), which were also dominant in our experiment at high nutrient conditions (Figure 5), and isolates had increased growth rates at high nutrient concentrations in tire rubber exposures (Supplementary Figure 2b). On the other hand, the apparent inhibition of *Vogesella* by tire rubber leachates at high nutrient conditions, one of the dominant genera in the control H_2O samples, also illustrates the inhibitory potential of tire rubber leachates. In sediment microbial communities, zinc and benzothiazole were identified as the main potential factors of tire rubber additives leading to pronounced community shifts and these chemicals are proven to be toxic for several microorganisms (Ding et al., 2022). As a consequence, tire particles were also shown to inhibit nutrient cycles in coastal sediments (Ding et al., 2022). The combination of these effects could explain the observed dose-responses in our concentrations experiment.

4.2 Environmental relevance

Exposures to different leachate concentrations also allowed us to compare the measured effects to environmentally realistic concentrations, using the same method as Sheridan et al. (2022) to calculate realistic LDPE leachate concentrations in freshwaters. This was assisted by bibliographical data on the relative occurrence of various plastic types (Uurasjärvi et al., 2020; Yuan et al., 2019; Capolupo et al., 2020).

According to our calculations, PP leachates can result in environmental concentrations of 4.88 mg DOC L^{-1}, taking the high leaching rates of this study into account and the fact that PP microplastics often dominates the entire plastics pollution in aquatic environments. Considering that the EC_{50} for PP leachates reached 0.43 mg DOC L^{-1}, significant environmental effects can be expected. For PVC, the realistic environmental concentration was found to be much lower, i.e., only 0.39 mg L^{-1}, due to its less frequent occurrence in freshwater and much lower leaching rates (according to the measured leaching rates in our experimental model system). According to our dose-response model, no significant effects on the specific growth rate of the free-living bacterial community can be expected at this concentration. For tire rubber, it is more difficult to estimate its relative contribution, particularly as these tend to be much denser and can rapidly sink onto the sediment. For this reason, the tire rubber particle abundance in the sediment was used to estimate its environmental concentrations. A realistic environmental concentration seems to be 0.32 mg DOC L^{-1}. For this concentration, according to the dose-response model fitted to the data, no significant effect on the bacterial community might be expected.

It must be mentioned that these are rough estimates to provide an idea on possible effect size of the environmentally relevant concentrations of each tested synthetic polymer leachate. Yet, in certain instances of heavily polluted water bodies, these might be much higher or else be significantly lower in more preserved water bodies. Additionally, the use of strong UV-lamps likely disproportionately affects leaching rates of certain products over others (Romera-Castillo et al., 2022), and, as mentioned previously, leaching rates and chemical make-ups are not equally distributed between products of the same synthetic polymer type. Furthermore, as seen in the mixed leachate sample of the bacterial isolates exposure (Supplementary Figure 2), the generally mixed and heterogenous plastic pollution of aquatic systems is likely to have effects which differ from those of leachates of a single plastic type or product. Nevertheless, these values can offer an important orientation for the extent of aquatic pollution by plastic leachates and their potential effect size on aquatic (micro)organisms.

It is also noteworthy to mention that these experiments may be influenced by the "bottle effect" caused by shifting the natural bacterial communities in small and closed containers (Hammes et al., 2010). It is known that this bottle effect can potentially alter the measured effects. Therefore, this incubation may be not entirely representative for natural ecosystems. Yet, microcosms with their highly controlled environmental factors can give important insights into the response of microbial communities to synthetic polymer leachates and thus provide valuable information on the effects of synthetic polymer leachate effects on complex microbial communities (Russo et al., 2016).

4.3 Effects of nutrient availability

Leachate exposures of natural bacterial communities (Figures 2, 6) indicate that effects of nutrient availability are not negligible, as has been found for pesticides (DeLorenzo et al., 2001). Furthermore, the effects of nutrient availability varied depending on the plastic type and for specific bacterial genera of natural free-living lake communities.

For the most part, stimulatory effects on bacterial communities via PP and PVC leachates are greatly dampened at high nutrient conditions compared to their respective controls as can be seen for the entire lake community (Figures 2, 6) and also for *Pseudomonas* when exposed to PVC leachates (Supplementary Figure 2). This notion can be explained by a more efficient uptake of specific forms of dissolved organic carbon over others. DOC from the LB medium is much more bioavailable than leachates from these synthetic polymers, e.g., represented by differences in bacterial carbon uptake genes (Poretsky et al., 2010). *Acinetobacter* and *Vogesella* represent an exception to this notion. Studies have shown that members of both *Acinetobacter* and *Vogesella* genera have PAH degrading capabilities (Czarny et al., 2020; Li et al., 2016), a common additive of PP and PVC plastic products. PAH degradation has also been shown to be enhanced when nutrient availability is high (Premnath et al., 2021). Further, bacterial co-metabolism can explain the simultaneous growth stimulation of *Acinetobacter* and *Vogesella* via PP and PVC even at low nutrient conditions. Nutrient enrichment has been shown to functionally influence microbial communities, especially via the expression of ABC transporters, which is downregulated when nutrients are enriched (Russo et al., 2016). Additionally, ABC-transporters have also been identified as important for microbial uptake of aromatic compounds, e.g., for an *Acinetobacter* strain (Mutanda et al., 2022). As aromatic compounds likely constitute an important component of both PP and PVC leachates, it is plausible that bacteria of the complex lake microbial community expressed less ABC transporters at high nutrient conditions, and thus were less impacted by aromatic (or other) leachate compounds.

Conversely, leachates which show inhibitory or no stimulatory effects, e.g., starch-PLA and tire rubber leachates, can cause important bacterial community shifts at high nutrient conditions. ABC-transporters have also been shown to be important in resistance to heavy metals, which are abundant in tire rubber leachates (Nies, 2003; Halsband et al., 2020). Thereby, downregulation of ABC transporters expression at high nutrient conditions can also impede the microbe's ability to resist to heavy metals. Nevertheless, the addition of nutrients nullified the effects on total bacterial abundance and growth rates. A potential explanation could be that, for instance, *Pseudomonas* is able to use tire rubber leachates as a growth substrate (Leff et al., 2007), as is suggested by increased growth rates at high nutrient conditions compared to control H_2O samples. This effect was absent for the tested *Pseudomonas* isolate at low nutrient conditions (Supplementary Figure 2b). As for PAHs (Premnath et al., 2021), tire rubber might be more bioavailable at nutrient enriched conditions, which also promote co-metabolism of highly polymeric carbon compounds. Differences in growth rates between exposures, for instance of *Rhodoferax* and *Pseudomonas* exposed to tire rubber leachates (Figure 5), indicate that the nature of these shifts varies with bacterial nutrient availability.

Our findings illustrate that plastic pollution in freshwaters leads to leachate effects on bacterial dynamics, which differ in dependence of the present environmental features. Unproductive, nutrient-poor freshwater bodies tend to have DOC concentrations ranging from 1 to 50 mg DOC L^{-1} (Menzel et al., 2005), though, for instance, concentrations as high as 300 mg DOC L^{-1} have been found in Canadian wetlands (Blodau et al., 2004). In our exposure experiments, organic carbon concentrations were approximately 6 mg and 60 mg DOC L^{-1}, representing both ends of the trophic spectrum, i.e., unproductive vs. productive freshwater bodies, respectively. Yet, our study does not provide the entire picture of what could happen in other more eutrophic aquatic systems. The addition of nutrients together with the exposure to synthetic polymer leachates is also likely more akin to anthropogenic nutrient enrichment than the leachate pollution of naturally nutrient-poor lake ecosystems. Our results illustrate a variety of effects of leachate pollution on bacterial dynamics depending on the specific environmental settings of complex and chemically diverse freshwater ecosystems.

5 Conclusion

Our results shed a new light on the complexity of effects exerted by synthetic polymer leachates on aquatic microbial communities, which extend beyond previous findings, i.e., the stimulation of microbial growth through LDPE leachates (Sheridan et al., 2022). The extent of these effects ranged from bacterial community shifts to changes in total bacterial growth rates. Our results revealed variable effects between synthetic polymer leachate types and nutrient conditions. This highlights the importance of taking the trophic states and presumably other specific environmental features of aquatic ecosystems into account when evaluating the effects of plastic leachate pollution under real world conditions. In a regulatory context, our findings emphasize the need for prioritizing use-reduction of plastics of higher leaching rates and toxic effects, such as PP, PVC and tire rubber and highlight the risks associated with replacing currently used plastics with starch-PLA based products.

To further proceed, we suggest to not solely analyse community shifts at the 16S rRNA gene level, but also use transcriptomics analyses of bacterial communities exposed to various types of plastic leachates, to highlight functional changes via up- or downregulation of specific functional genes. Transcriptomics and other OMICS approaches will increase our understanding of the mechanisms underlying our findings. Future experiments should use more diverse freshwater bacterial communities and include the particle-attached fractions to elucidate the full extent of effects, exerted by various types of plastic leachate pollution, on microbial diversity and metabolism and thus overall ecosystem functioning.

Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found in the article/Supplementary Material.

Author contributions

MS: Data curation, Formal Analysis, Investigation, Software, Validation, Visualization, Writing – original draft, Writing – review and editing. FM: Methodology, Software, Supervision, Validation, Writing – original draft, Writing – review and editing. YY: Writing – original draft, Writing – review and editing. H-PG: Conceptualization, Funding acquisition, Methodology, Project administration, Resources, Supervision, Writing – original draft, Writing – review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

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Retention and remobilization of aged polystyrene (PS) microplastics in a porous medium under wet-dry cycling

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Understanding the aging processes of microplastics (MPs) and their behavior under dynamic wet-dry cycles is critical for accurate environmental risk assessment, particularly in unsaturated zones. This study investigates the impact of aging on the retention and remobilization dynamics of polystyrene (PS) microplastics in unsaturated porous media. We employed light-aged PS MPs and conducted column experiments using sea sand under multiple wetting-drying cycles. Key physicochemical properties of the MPs were characterized, and their retention and remobilization were quantified by monitoring effluent concentrations.Aging significantly altered MP properties, increasing their surface negative charge and hydrophilicity. Consequently, aging suppressed the remobilization of retained MPs during drying phases. After five cycles, the total retained fraction of aged MPs (76.88%) was consistently higher than that of pristine MPs (72.83%). Remobilization was primarily driven by mobile air-water interfaces (AWI) once water saturation fell below a critical threshold of approximately 0.6. Although aging increased electrostatic repulsion (which hinders retention), it also increased hydrophilicity, which significantly weakened AWI-driven remobilization during drying. This retention-promoting mechanism (reduced AWI remobilization) outweighed the opposing effect of electrostatic repulsion, leading to greater overall retention of aged MPs. This study highlights the complex regulatory role of aging on MP fate and provides critical insights for assessing the environmental risks of aged microplastics in dynamic unsaturated systems.

KEYWORDS

microplastics, aging, drying-wetting cycle, unsaturated porous media, air-water interface

1 Introduction

Microplastics (MPs), defined as synthetic polymer particles, fragments, or fibers smaller than 5 mm, constitute an emerging global contaminant (Li et al., 2023). Fueled by escalating plastic consumption and continuous environmental degradation of plastic waste, global MP concentrations are projected to reach ~10 million metric tons by 2040 (Lau et al., 2020).

Pervasive across terrestrial, aquatic, and atmospheric compartments, MPs pose significant ecological threats (Koelmans et al., 2022). Unsaturated zones (e.g., soil vadose layers, intertidal sediments, seasonally arid soils) serve as pivotal pathways for MP transit from surface environments to groundwater while simultaneously acting as long-term retention sites where aging, co-contaminant associations, and episodic remobilization may occur (Feng et al., 2024). However, predicting their behavior in these systems is complicated by the interplay of two crucial factors: the inevitable aging processes that fundamentally alter their physicochemical properties, and the dynamic wetting-drying cycles driven by precipitation, evaporation, or tides, which govern transport. Understanding the combined influence of these co-occurring processes on MP fate remains limited, representing a critical knowledge gap for accurate environmental risk assessment.

Once released into the environment, plastics undergo inevitable aging processes that fundamentally alter their physicochemical properties (Lu et al., 2023). Aged microplastics can develop enhanced ecotoxicity and modified environmental behaviors compared to pristine counterparts (Abaroa-Pérez et al., 2022). Microplastic aging constitutes a complex transformation driven by multifaceted environmental drivers, including: Photo-oxidative degradation (UV radiation, oxygen), Thermal fluctuations, Aqueous chemistry (pH, salinity), Mechanical abrasion, and Biological activity (Lu et al., 2023). Sunlight, particularly ultraviolet (UV) radiation, serves as the primary driver of plastic photo-oxidative degradation in environmental systems. UV exposure initiates polymer chain scission, incorporating oxygen atoms and generating surface functional groups (e.g., carbonyls) on microplastics (Yu et al., 2024). This process increases material embrittlement, ultimately causing fragmentation and particle size reduction (Wang et al., 2023). These aging-induced transformations profoundly govern microplastic environmental mobility. Critically, aging typically increases the surface negative charge and hydrophilicity of MPs. This wettability shift alters transport behavior in porous media, modifies interfacial interactions at air-water boundaries and increases adsorption affinity for hydrophilic contaminants (Yu et al., 2024). Aging exerts dual competing influences on microplastic transport in porous media: increased surface roughness promotes retention by expanding attachment sites, while fragmentation into smaller particles enhances mobility (Ren et al., 2021). Smaller particles exhibit greater bioavailability—facilitating organismal ingestion, crossing biological barriers, and amplifying trophic transfer risks (Akhatova et al., 2022). Collectively, these aging-induced transformations govern microplastic environmental mobility, pollutant carrier potential, and ecotoxicological impacts. Critically, studies restricted to pristine microplastics suffer inherent limitations in representing environmental realism. Research on aged particles is therefore imperative to accurately resolve environmental dynamics, assess long-term fate, and quantify ecosystem risks.

Microplastic (MP) transport through porous media constitutes a dynamically complex process regulated by multifactorial interactions. The environmental behavior and ultimate fate of MPs are determined by interdependent properties of the particles themselves, the porous matrix, water saturationand the resident porewater chemistry (Gao et al., 2021). Current unsaturated soil studies primarily examine how MP properties (size, density, hydrophobicity) and media characteristics (grain size, pH, ionic strength [IS], cations, dissolved organic matter [DOM]) govern retention (Wang et al., 2022). Experimental approaches typically employ one-dimensional packed column experiments under controlled unsaturated conditions, achieved by regulating water content or pre-drying. Ling et al. documented substantially decreased MP recovery in breakthrough curves as saturation declined from 100% to 50%, demonstrating enhanced retention due to air-water interface (AWI) formation and thinner water films (Ling et al., 2022). Flow velocity effects exhibit greater complexity in unsaturated media. Ionic strength exerts a stronger influence on transport than in saturated systems, largely because reduced pore-water flow areas decrease separation distances between MPs and solid matrices (Dong et al., 2022). MP hydrophobicity—determining AWI affinity—critically regulates unsaturated transport. Mitropoulou et al. observed higher retention for larger MPs via AWI attachment (Mitropoulou et al., 2013), while Morales et al. reported DOM-enhanced retention at the AWI through altered surface hydrophobicity (Morales et al., 2011). Wang et al. systematically compared ionic strength and cation effects on aged versus pristine MPs in unsaturated sand, consistently finding enhanced retention via electrical double layer compression and reduced electrostatic repulsion at higher IS or with polyvalent cations. Aged MPs showed weaker IS sensitivity due to increased surface negative charge (Wang et al., 2022). Feng et al. observed reduced MP release from unsaturated versus saturated columns, with aging enhancing mobility of hydrophilic MPs in unsaturated media (Feng et al., 2024), though other studies note aging-induced surface roughness increases retention (Feng et al., 2022). Most studies investigate MPs under static unsaturated conditions or single infiltration events, despite natural vadose zones experiencing highly dynamic moisture regimes driven by rainfall, evaporation, root uptake, and water table fluctuations. Wetting-drying cycles significantly enhance vertical MP migration, likely through soil structural modification, pore-water pressure shifts, and altered hydrodynamic forces. These cycles also accelerate MP aging (O'connor et al., 2019); Ranjan et al. (2023) demonstrated they promote mechanical weathering and fragmentation into secondary particles, significantly increasing penetration depth in porous media. Using tidal simulations, Feng et al. demonstrated capillary fringe fluctuations effectively remobilize retained MPs and accelerate groundwater transport, highlighting hydrological dynamics' critical role (Feng et al., 2023). Research on microplastic (MP) transport in unsaturated porous media has identified key influencing factors, yet understanding of long-term behavior under co-occurring hydrological dynamics and aging processes remains limited. The complex air-water interface (AWI) interactions inherent to unsaturated systems introduce significant transport complexity.

In unsaturated porous media, where solid, liquid, and discontinuous/continuous gas phases coexist, air-water interfaces (AWIs) form within pore spaces (Crist et al., 2005; Dong et al., 2022). These interfaces introduce three primary retention mechanisms: AWI attachment, air-water-solid (AWS) contact line retention, and capture within thin water films. Since water saturation governs AWI area and water film thickness, capillary retention mechanisms exhibit strong moisture dependence (Huang et al., 2024). Infiltration, drying cycles, or capillary fringe fluctuations mobilize AWIs, exerting substantial forces on retained colloids (Bradford and Torkzaban, 2008). The core remobilization mechanism involves dynamic capillary forces from moving interfaces (Bradford and Torkzaban, 2008). As advancing/receding AWIs pass surface-attached colloids, dynamically changing surface tension generates normal detachment forces (Flury and Aramrak, 2017). These potent interfacial detachment forces constitute the primary remobilization mechanism in unsaturated media, modulated by hydrological dynamics, colloid properties, and fluid chemistry (Wu et al., 2021; He et al., 2023). For instance, during the drying phase, the advancing AWI generates significant detachment forces that can mobilize attached particles; Conversely, during wetting, the re-introduction of water disrupts pre-existing AWIs, also potentially remobilizing particles (Bradford and Torkzaban, 2008). Thus, the creation, movement, and destruction of AWIs during wetting-drying cycles represent the dominant physical processes governing colloid fate. The efficiency of these AWI-driven mechanisms, however, is highly sensitive to the surface properties of the colloids themselves, creating a complex scenario when considering environmentally aged microplastics.

A systematic understanding of how environmental aging modifies MP retention and release dynamics in porous media under realistic hydrologic fluctuations remains critically lacking. While current knowledge predominantly derives from studies under saturated or steady-state unsaturated conditions, these approaches do not capture the ubiquitous, dynamic wetting-drying cycles found in environments such as vadose and intertidal zones. This knowledge gap impedes our ability to predict the transport potential and ultimate fate of aged MPs, hindering reliable environmental risk assessments. Therefore, this study investigates the combined effects of aging on polystyrene (PS) microplastics, specifically examining their retention and remobilization behavior in unsaturated porous media under dynamic wetting-drying cycles. It aims to identify dominant controls governing aged MP retention and release under dynamic hydrologic conditions and quantify their relative contributions. This research is crucial for advancing fundamental understanding of MP behavior in natural surface environments.

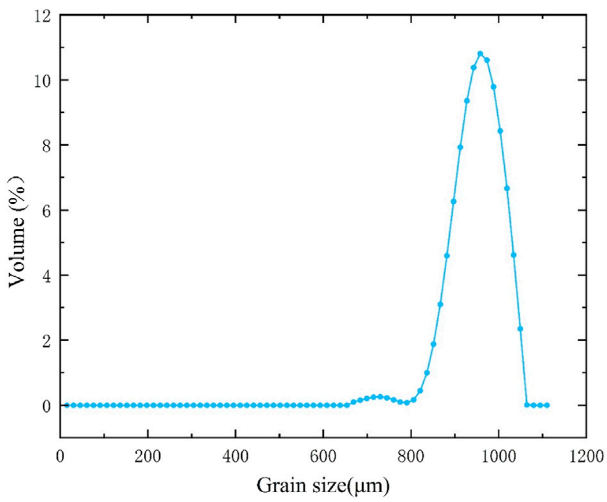

FIGURE 1
Grain size distribution of sand.

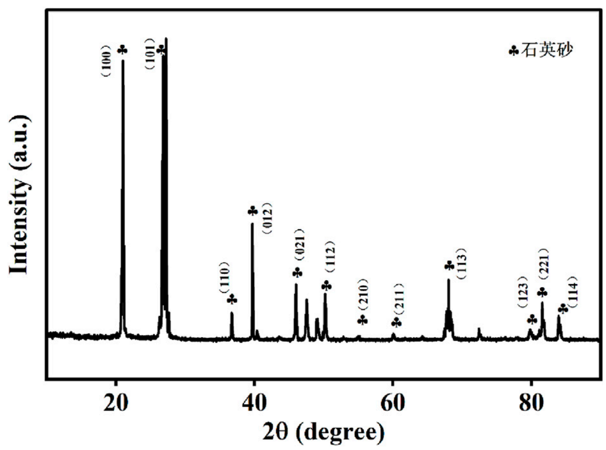

FIGURE 2
X-ray diffraction (XRD) pattern of sand.

2 Materials and methods

2.1 Chemical and porous media preparation

Sodium chloride (NaCl, >99.8% purity, Aladdin) were used. Natural sea sand from Qi'ao Island, Zhuhai, Guangdong Province served as the porous medium. The sand was rinsed repeatedly with ultrapure water until conductivity stabilized, then dried at 50°C and sieved (200-mesh). Laser diffraction analysis Laser diffraction analysis (Malvern Mastersizer 2000, Malvern Istruments, UK) showed particle diameters ranging 141–482 μm with median size (d_{50}) of 275 μm, indicating uniform size distribution (Figure 1). XRD analysis (Figure 2) identified quartz as the dominant phase, evidenced by intense peaks at $2\theta = 22°$ (100), 27° (101), and 40° (012) with high crystallinity. Minor peaks indicated accessory minerals (e.g., feldspar, mica). A broad split peak at $2\theta \approx 68°$ suggested minor poorly crystalline phases with small crystallite size.

2.2 Aged microplastic preparation

Spherical polystyrene microplastics (PS; 3 μm diameter; Jiangsu Zhichuan Technology Co., Ltd.) were artificially aged according to the method described by Xi et al. (2022). Briefly, 0.5 g of PS particles were ultrasonically dispersed in 200 mL of ultrapure water within a 250 mL quartz tube for 30 min. The resulting suspension was then subjected to photo-aging under a 1000 W mercury lamp at 25°C for 32 days. Following aging, all samples were freeze-dried and stored in the dark. Three parallel samples and one control sample were established for each experimental group. The relative errors among parallel samples were required to be maintained below 10%. The control group was subjected to identical conditions as the experimental groups except for the light treatment, thereby excluding potential interference from evaporation, biodegradation, and other processes. All glassware was thoroughly cleaned with ultrapure water and oven-dried at 50°C prior to use to eliminate potential contaminants. Throughout all experimental procedures, researchers wore cotton laboratory coats and appropriate personal protective equipment (PPE) to prevent sample contamination.

2.3 Porous media and microplastic characterization

Natural sea sand was ground to ~10 μm and analyzed using Cu-Kα radiation (2θ range: 5°–80°, step size: 0.02°, accuracy: ±0.001°). Surface morphology of pristine/aged MPs was examined by field-emission SEM at 3 kV (10k × magnification). Samples were sputter-coated with Pt (110 kV, 60 s) to prevent charging. FTIR spectra (4,000–400 cm^{-1}) were acquired at 4 cm^{-1} resolution with 16 scans to identify surface functional groups. Hydrophilicity changes were quantified via contact angle measurements. Deionized water droplets (3 μL) were deposited on compressed MP pellets. Contact angles were averaged from >3 replicates per sample, avoiding edge effects. Zeta potentials (reflecting colloidal stability) of MPs and quartz sand were determined in background solutions using a Zetasizer Nano ZS90 (Malvern). Measurements (25°C) involved 21 scans (0–500 Hz) in a 10 mm cuvette. Values were calculated from electrophoretic mobility via the Smoluchowski equation.

2.4 Column experiments: simulating cumulative retention under wetting-drying cycles

2.4.1 Remobilization of microplastics induced by wetting-drying cycles

This study investigated microplastic remobilization from porous media under wetting-drying cycles by subjecting saturated columns to repeated cycles and monitoring microplastic release. Columns were packed using the wet method to ensure saturation and homogeneity. Degassed ultrapure water was pumped upwards from the column bottom at 9 mL/min, maintaining a 5-cm hydraulic head above the sand surface throughout packing to ensure complete saturation. Packing paused at a height of 15 cm to install pre-calibrated time domain reflectometry (TDR) probes and tensiometers into access ports; these were secured with silicone sealant. Packing then resumed in uniform, saturated layers to the target height while continuously infusing degassed ultrapure water. Following the methodology of Li et al. (2013), both TDR probes and tensiometers were calibrated for each experimental column prior to testing to ensure accurate real-time monitoring of water saturation and capillary pressure.

Following packing, columns were stabilized for at least 24 h. TDR probes and tensiometers monitored water saturation and capillary pressure, confirming stability when readings exhibited no significant fluctuations. Prior to microplastic injection, degassed ultrapure water was infused upwards at 9 mL/min to remove sand debris until effluent showed no visible impurities. The water was then replaced with background electrolyte solution (10 mM NaCl), which was infused at the same flow rate for 10 pore volumes (PV), calculated as described in Equation 1 and Equation 2, until effluent absorbance matched the input solution, establishing consistent physicochemical conditions.

PS microplastic suspensions (100 mg/L in 10 mM NaCl) representing different aging durations (0 days, 32 days) were subsequently introduced into the column at a rate equivalent to 2 PV. Continuous sonication maintained homogeneous suspension concentration during injection. Immediately following microplastic injection, background solution (without microplastics) was flushed through the column until effluent microplastic concentrations became undetectable. Effluent absorbance, measured continuously by UV-Vis spectrophotometry during injection and flushing, enabled determination of microplastic concentrations and calculation of the mass retained in the porous media during the saturated phase.

After achieving stable microplastic retention, the initial drying phase commenced by draining the background solution from the column bottom at 9 mL/min. Drying concluded when effluent flow ceased and water saturation stabilized (Xie et al., 2012). Effluent microplastic concentration was continuously monitored via UV-Vis spectrophotometry during drying, enabling calculation of remobilized mass combined with flow rate data. Immediately following this initial drying, subsequent wetting-drying cycles were performed to simulate hydrological fluctuations remobilizing retained microplastics. Each cycle comprised a wetting phase and a drying phase. During wetting, background electrolyte solution (without microplastics) was slowly pumped into the column at 9 mL/min until the liquid level reached the overflow port at the column top, simulating moisture recharge and enabling remobilization upon contact with the moving aqueous phase.

A stabilization period followed each wetting phase to allow moisture redistribution before initiating the drying phase. Drying proceeded identically to the initial phase, with continuous effluent microplastic concentration monitoring enabling calculation of remobilized mass per cycle. Following the initial drying, two complete additional wetting-drying cycles were performed. Effluent microplastic concentration and remobilized mass were continuously monitored and calculated during all drying phases (including the initial and subsequent cycles). For each experimental batch, blank control tests were conducted. This involved preparing control columns packed solely with sea sand and leaching them with background electrolyte solutions free of microplastics under identical experimental conditions. The effluent from these blank columns was collected and analyzed using ultraviolet-visible (UV-Vis) spectrophotometry. No microplastics were detected in any blank samples, confirming that our experimental setup and operational procedures introduced no contamination.

2.4.2 Cumulative retention under wetting-drying cycles

To investigate microplastic transport under complex hydrodynamic conditions representative of realistic unsaturated porous media, including continuous inputs, this study designed cumulative retention experiments driven by wetting-drying cycles, systematically examining the effects of aging and material type on long-term retention and spatial distribution. Columns were packed, stabilized, and flushed with degassed ultrapure water at a constant flow rate (9 mL/min) to remove fine particles or soluble impurities, preventing interference with microplastic quantification.

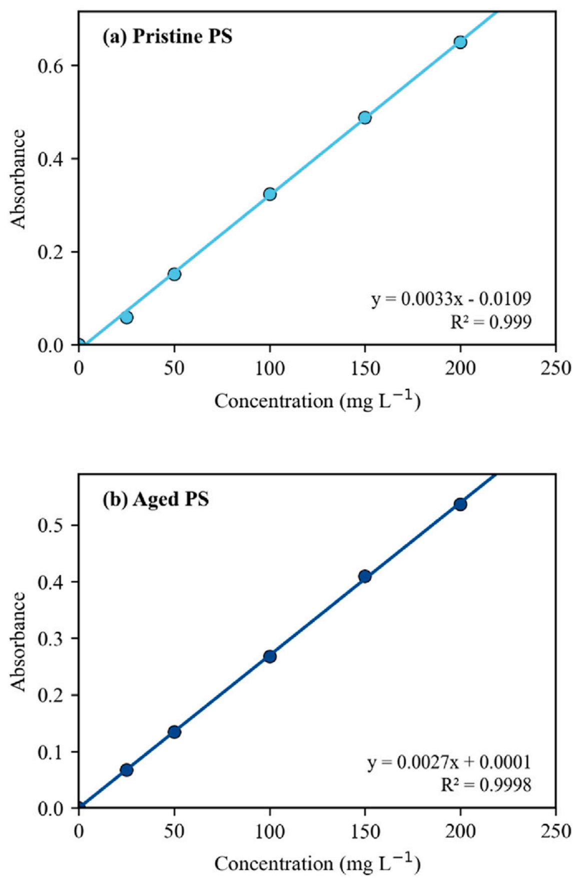

FIGURE 3
Microplastic quantification standards: **(a)** Pristine PS; **(b)** Aged PS.

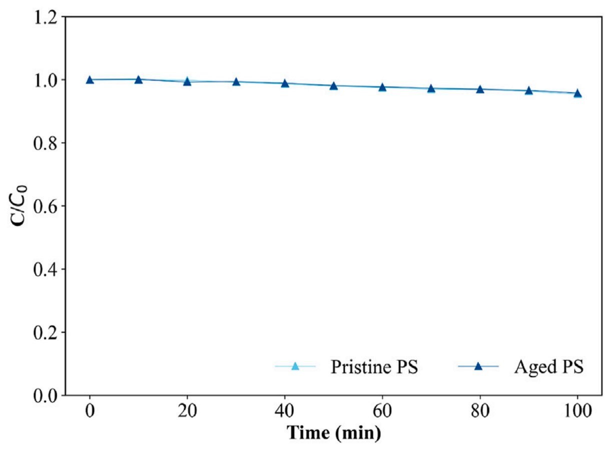

FIGURE 4
Microplastic colloidal stability: **(a)** Pristine PS; **(b)** Aged PS.

Subsequently, the columns were equilibrated with at least 10 pore volumes (PV) of background electrolyte solution (10 mM NaCl) to achieve hydraulic equilibrium. Following equilibration, an initial drainage phase established uniform unsaturated initial conditions: solution was drained from the column bottom at 9 mL/min until residual water saturation was reached (effluent flow ceased) for all columns.

Following establishment of the initial unsaturated state, wetting-drying cycles with continuous microplastic input commenced. Each cycle consisted of two phases: (1) Wetting phase: Polystyrene (PS) microplastic suspensions (100 mg/L in 10 mM NaCl; aged 0 days or 32 days) were continuously injected upward through the column bottom at 9 mL/min until the liquid level reached the overflow port, achieving near-saturation without ponding. (2) Drying phase: Solution was drained downward at 9 mL/min. Effluent containing remobilized microplastics was collected, returned to the original suspension reservoir, and homogenized via sonication to maintain constant influent concentration, simulating persistent pollution. Drainage continued until effluent flow ceased. This sequence constituted one cycle. Consecutive repetitions yielded cumulative retention data after 1, 2, and 5 uninterrupted cycles.

Post-experiment, the sand column was sectioned into nine 4-cm segments. Each segment was transferred to a clean 500-mL beaker, oven-dried at 70°C, and subjected to microplastic extraction via density flotation. A total of 150 mL of saturated NaCl solution (density 1.20 g/mL) was added to each dried sample, followed by 20-min sonication to release microplastics. The supernatant containing microplastics was transferred to 50-mL centrifuge tubes and centrifuged at 4,500 rpm for 10 min. The final supernatant was diluted to volume in 100-mL colorimetric tubes, and microplastic mass per layer was quantified colorimetrically using a standard curve. Following the experiment, triplicate measurements were performed for each subsample obtained through stratified sampling to quantify retained microplastics in the sand profile, ensuring the reliability of quantitative results.

2.5 Microplastic quantification methods

The effectiveness of ultraviolet-visible (UV-Vis) spectrophotometric quantification was validated through calibration curves established for both pristine and differentially aged polystyrene (PS) microplastics. Microplastic (MP) concentrations in effluent were quantified using UV-Vis spectrophotometry (UV 2800 S) by measuring optical density at characteristic wavelengths (238 nm for PS). Calibration curves exhibited excellent linearity ($R^2 > 0.999$) within the experimental concentration range (Figure 3).

MP suspension stability was verified by monitoring absorbance at 238 nm over 100 min at 10-min intervals. No significant settling occurred (Figure 4), with normalized concentration ratios (C/C_0) consistently >0.95. During column experiments, suspensions were continuously stirred at 1,000 rpm to maintain homogeneity and minimize error. To maintain suspension homogeneity during column experiments, continuous agitation was implemented using an ultrasonic cleaner and magnetic stirrer.

2.6 Data processing and calculations

To quantitatively evaluate the retention and remobilization dynamics of microplastics under the experimental conditions, several key metrics were calculated from the collected data. This section outlines the definitions and formulas used for this analysis.

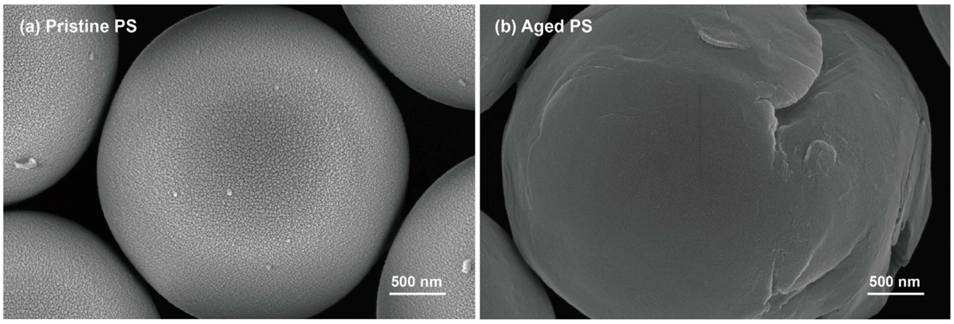

FIGURE 5
SEM surface morphology: **(a)** Pristine PS; **(b)** Aged PS.

2.6.1 Remobilization and retention metrics

The efficiency of MP remobilization during the drying phases was assessed using the Relative Release Recovery (%), a metric adapted from Bradford and Torkzaban (2008) and defined as: (Mass of MPs released in a drying phase/Mass of MPs retained in the column at the start of that cycle) × 100.

To evaluate the long-term fate of MPs under continuous input and cyclic wetting-drying, the Cumulative Retention (%) was calculated after 1, 2, and 5 cycles. This metric represents the overall retention efficiency of the porous medium and is defined as: (Total MP mass retained in the column/Total MP mass injected) × 100. Furthermore, to assess the mass balance and potential particle loss during the post-experiment extraction process, the Packing Material Recovery (%) was determined by flotation extraction using the following formula: (Total MP mass recovered/Total MP mass injected) × 100. The loss rate (%) was derived from the difference: Cumulative retention recovery - Packing material recovery.

2.6.2 Physical properties of the porous medium

The fundamental physical properties of the packed column, which influence hydraulic and transport behavior, were determined as follows. Porosity (φ), the ratio of void volume to total volume, was calculated using measured particle density (ρ_p = 2.62 g/cm^3) and bulk density (ρ_b = 1.64 g/cm^3):

$$\varphi = 1 - \left(\rho_b / \rho_p\right) = 0.374 \quad (1)$$

Based on the calculated porosity and the column dimensions (radius r, length L), the Pore Volume (PV, mL), a critical parameter for standardizing the experimental flow, was calculated as:

$$PV = \pi r^2 L \times \varphi \quad (2)$$

where r is column radius (cm) and L is length (cm). Calculated PV = 550 mL.

3 Results and discussion

3.1 Physicochemical properties of aged microplastics

The physicochemical properties of aged microplastics are presented in Figure 5. Pristine PS microspheres maintained their initial spherical morphology with smooth surfaces. After 32 days of aging, the microspheres exhibited characteristic surface flaking and distinct cracking. FTIR spectra (Figure 6) demonstrated significant chemical alterations during aging. A new peak emerged at 1738 cm^{-1}, attributed to C=O stretching vibrations, with intensity increasing over time, indicating the introduction of oxygen-containing functional groups (Xie et al., 2012). Concurrently, the C-H bending vibration peak at 1,449 cm^{-1} (associated with -CH_2-) weakened, suggesting alkyl chain scission, collectively confirming surface oxidation and polymer chain degradation. Contact angle measurements (Figure 7) revealed a significant decrease for aged PS, indicating enhanced hydrophilicity (θc < 90° signifies hydrophilicity (Ouyang et al., 2022)). This shift is primarily attributed to the generation of polar oxygen-containing groups during aging. Increased surface roughness (cracks, folds, pores) further enhanced hydrophilicity by increasing the effective surface area for water interaction. Aging for 32 days substantially increased the absolute value of the PS zeta potential from 29.76 mV to 51.43 mV, indicating enhanced surface electronegativity. This increase is primarily caused by the dissociation of introduced

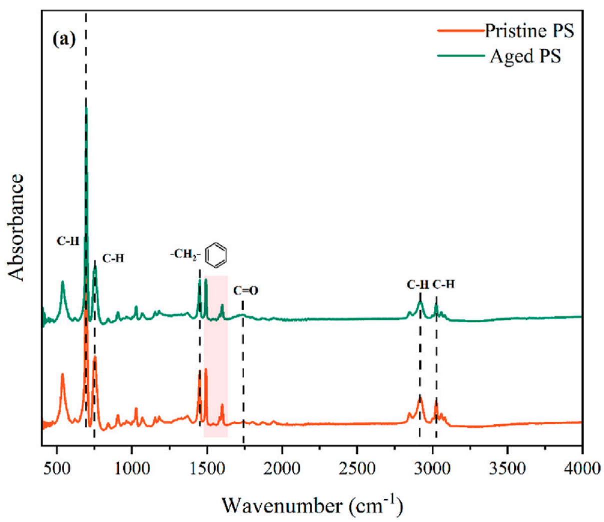

FIGURE 6
FT-IR spectra: **(a)** Pristine PS; **(b)** Aged PS.

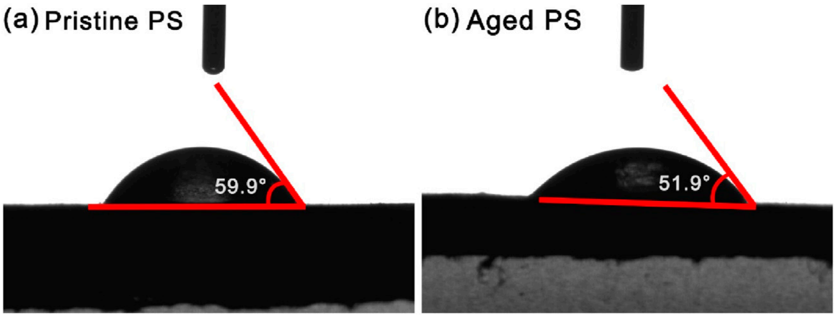

FIGURE 7
Surface contact angles: **(a)** Pristine PS; **(b)** Aged PS.

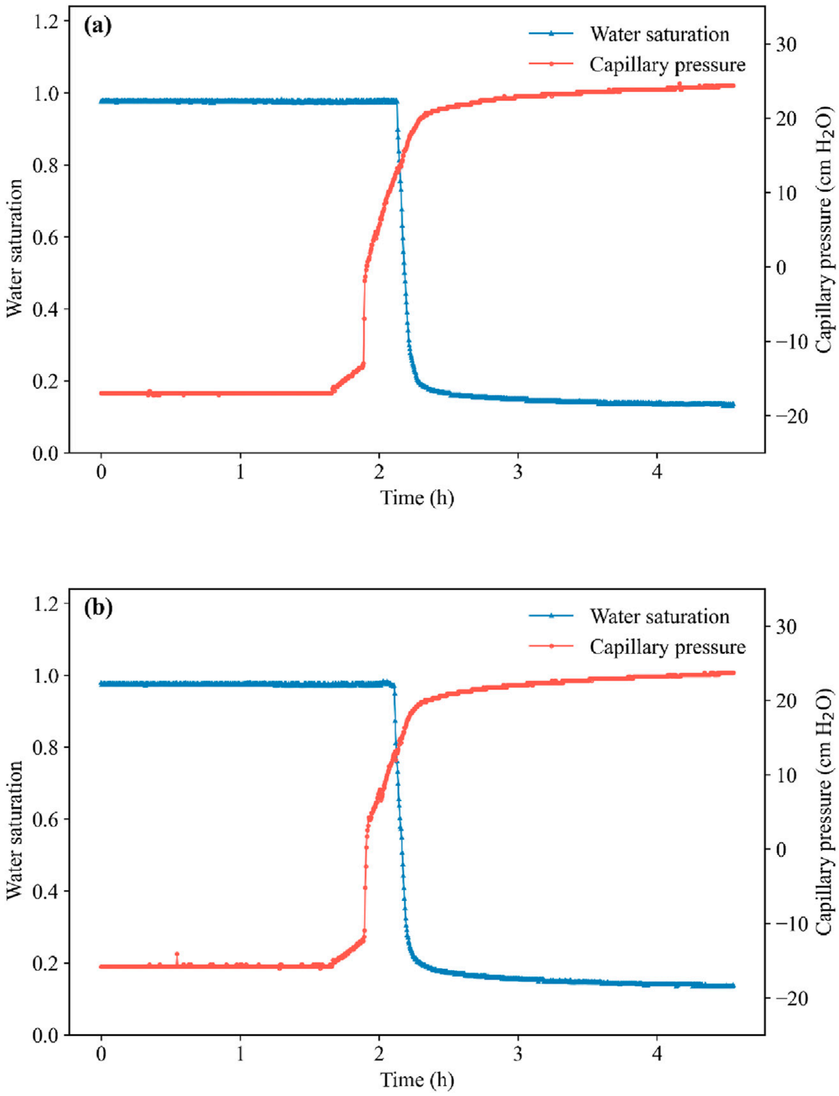

FIGURE 8
Primary drainage dynamics: Water saturation and capillary pressure for **(a)** Pristine PS; **(b)** Aged PS.

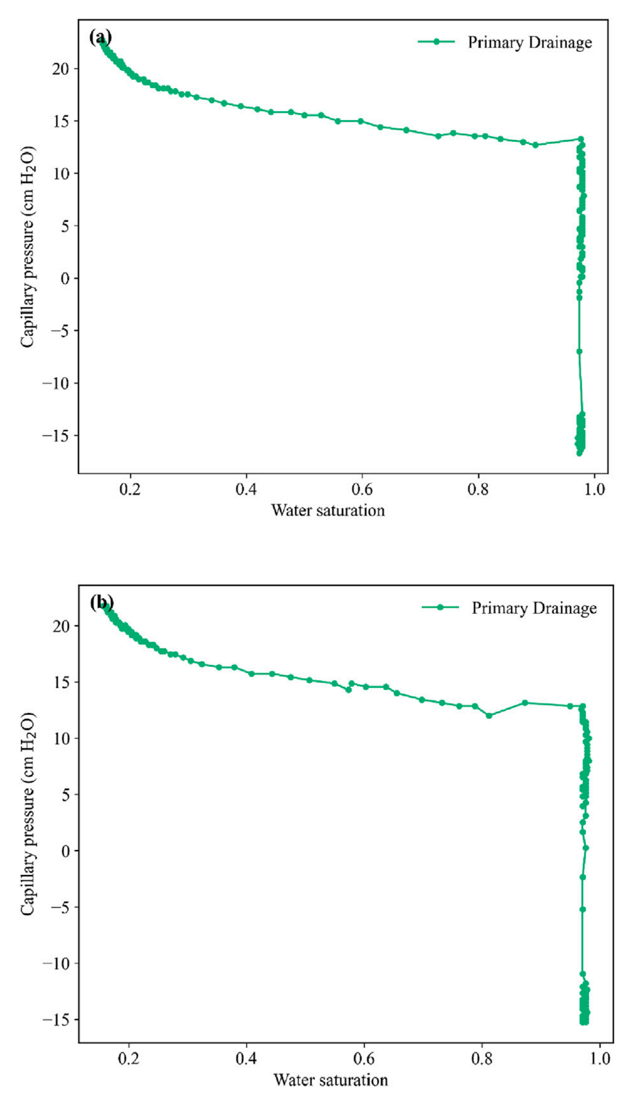

FIGURE 9
Primary drainage water retention curves: **(a)** Pristine PS; **(b)** Aged PS.

oxygen-containing functional groups, generating negative surface charges. Furthermore, aging-induced surface cracks increased the effective surface area, exposing more charged sites and enhancing negative charge density, consequently elevating the zeta potential magnitude.

3.2 Re-release behavior of aged microplastics in unsaturated media

3.2.1 Hydrodynamic characteristics of aged microplastics during wetting-drying cycles

Cumulative microplastic retention in unsaturated porous media is governed by complex interfacial processes and hydrological dynamics. To assess the long-term impact of aging on microplastic mobility under realistic unsaturated conditions, wetting-drying cycles were employed to examine remobilization mechanisms under complex hydrodynamics. Real-time monitoring of water saturation (Sw) and capillary pressure (Pc) dynamics enabled in-depth analysis of hydrodynamic characteristics during these cycles, revealing Sw/Pc evolution patterns and their influence on air-water interface (AWI) behavior.

Following saturated-phase retention, the initial drainage phase commenced. Figure 8a,b depict Sw and Pc evolution during initial drainage for pristine PS and aged PS. As drainage progressed, Sw decreased continuously from near-saturation (~1.0) while Pc increased, tracing the soil water characteristic curve. Water initially drains from larger pores with relatively flat AWIs. As drainage advances, water retreats into smaller pores where increased AWI curvature necessitates greater Pc. Rates of Sw and Pc change decelerate until stabilization, with rapid initial shifts indicating swift AWI formation and water film entrapment.

Figure 9a,b show capillary pressure (Pc) versus water saturation (Sw) curves during initial drainage for pristine PS and aged PS. During drying (drainage), the advancing air-water interface (AWI) generated significant detachment forces, acting as the key dynamic remobilization mechanism for retained microplastics, particularly hydrophobic ones. Post-drying, capillary forces retained residual water as relatively stable water films or pendular rings (Li et al., 2013), creating potential static retention sites via associated AWIs.

During subsequent wetting phases (Figure 10), Pc and Sw transitioned rapidly before stabilizing. The AWI receded and destabilized, accompanied by water film thickening. While Sw/Pc trends remained largely consistent across cycles, subsequent wetting phases never re-attained complete saturation (Sw = 1.0) due to incomplete air displacement and persistent gas entrapment within pores after multiple cycles (Li et al., 2014), generating new AWIs and water films.

Consequently, AWIs exhibit dual functionality: enabling dynamic remobilization during phase transitions (drainage) and providing static retention potential via stabilized interfaces. This interplay introduces significant complexity to microplastic retention and release dynamics.

Figure 11 shows capillary pressure (Pc) versus water saturation (Sw) curves during 1, 2, and 5 wetting-drying cycles. Similar Pc-Sw trajectories across cycles indicate that repeated cycling did not significantly alter the porous media's pore size distribution. Distinct hysteresis loops formed between drainage and wetting curves (Zhou et al., 2014), arising from pore geometry, contact angle hysteresis at the three-phase contact line, and air entrapment.

At identical Pc values, Sw was lower during drainage than wetting. This phenomenon arises because drainage exhibits more extensive AWIs and thinner water films, providing greater potential for static retention via capillary pinning at the solid-water interface (SWI). Conversely, higher Sw during wetting corresponds to reduced AWI area and thicker water films. This hysteresis demonstrates differing retention/release efficiencies between phases: drainage governs microplastic mobility through competition between enhanced static retention potential and strong dynamic detachment forces, while wetting phases exhibit comparatively weaker interfacial retention. The ultimate fate of microplastics depends on their intrinsic properties interacting with these dynamic environmental factors.

3.2.2 Remobilization behavior of aged microplastics in unsaturated porous media

Figure 12 demonstrates that microplastic release ratios (C/C_0) increase as water saturation (Sw) decreases below ~0.6 during

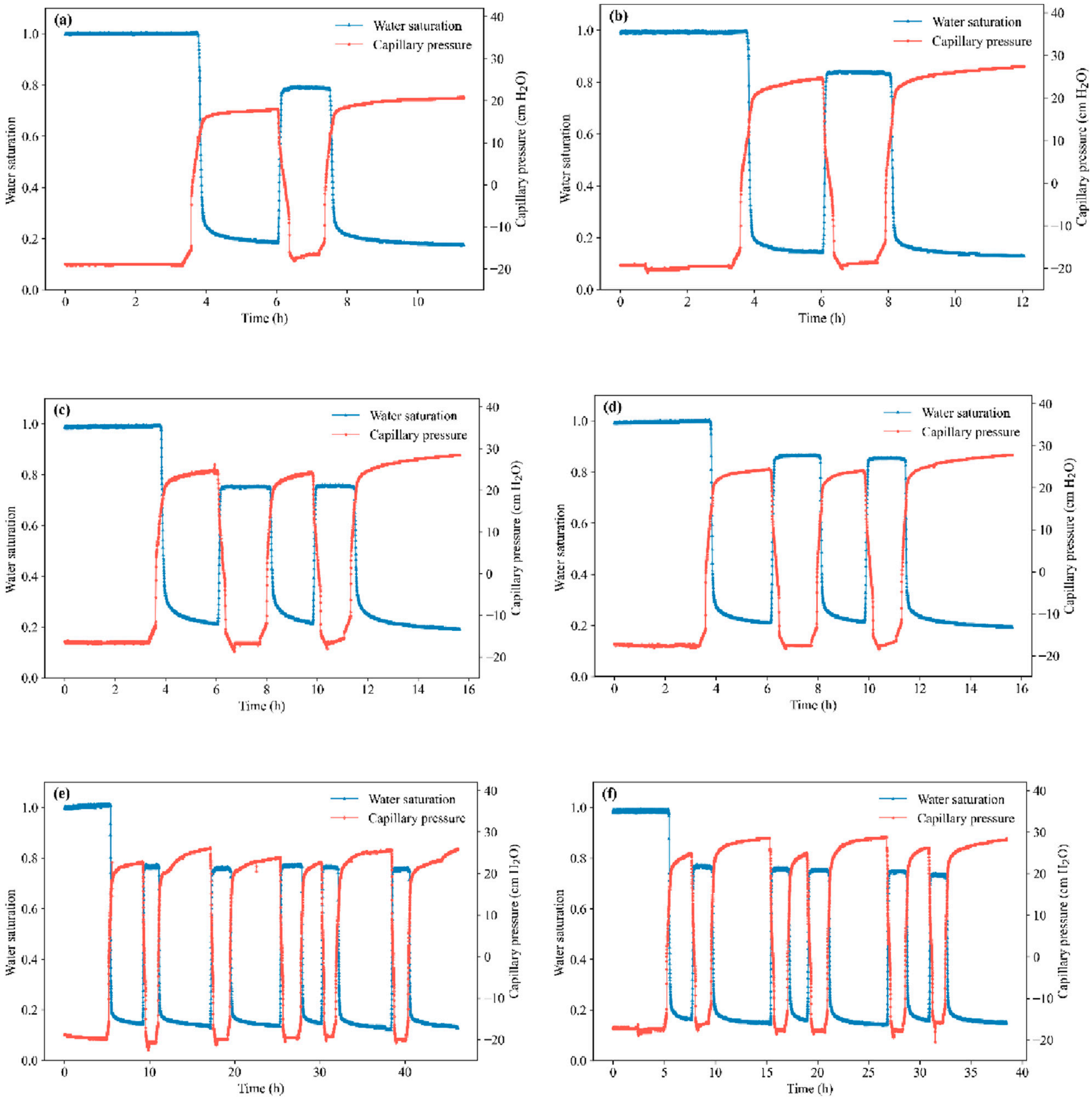

FIGURE 10
Cyclic wetting-drying dynamics: Water saturation and capillary pressure for Cycle 1 **(a,b)**, Cycle 2 **(c,d)**, Cycle 5 **(e,f)**. (Pristine PS: **(a,c,e)** Aged PS: **(b,d,f)**).

unsaturated drainage. This occurs because the advancing air-water interface (AWI) thins water films, enabling capillary menisci to exert interfacial detachment forces on attached microplastics along the air-water-solid contact line. Release initiates when these detachment forces overcome microplastic adhesion to the solid surface (Sharma et al., 2008). The release threshold at Sw ≈ 0.6 reflects the critical saturation where water films achieve sufficient thinness for the moving AWI to exert significant detachment forces at the solid-water interface (SWI) (Zhang et al., 2012).

This phenomenon parallels nanomaterial transport studies, where *E. coli* mobilization similarly peaks below Sw = 0.6 as thinning films detach cells from the SWI and transfer them to mobile phases (bulk water or AWI) (Bradford and Torkzaban, 2013). This confirms the AWI's critical role in colloid mobilization, providing a mechanistic basis for analyzing aging's impact on microplastic mobility. Crucially, aged PS exhibited lower C/C_0 than pristine PS at identical Sw during initial drainage, indicating enhanced detachment resistance.

Figure 13 further shows reduced release recovery for aged PS during initial drainage (12.05% → 7.96%), confirming suppressed remobilization. Although aging enhanced hydrophilicity and electronegativity—potentially weakening adhesion via electrostatic

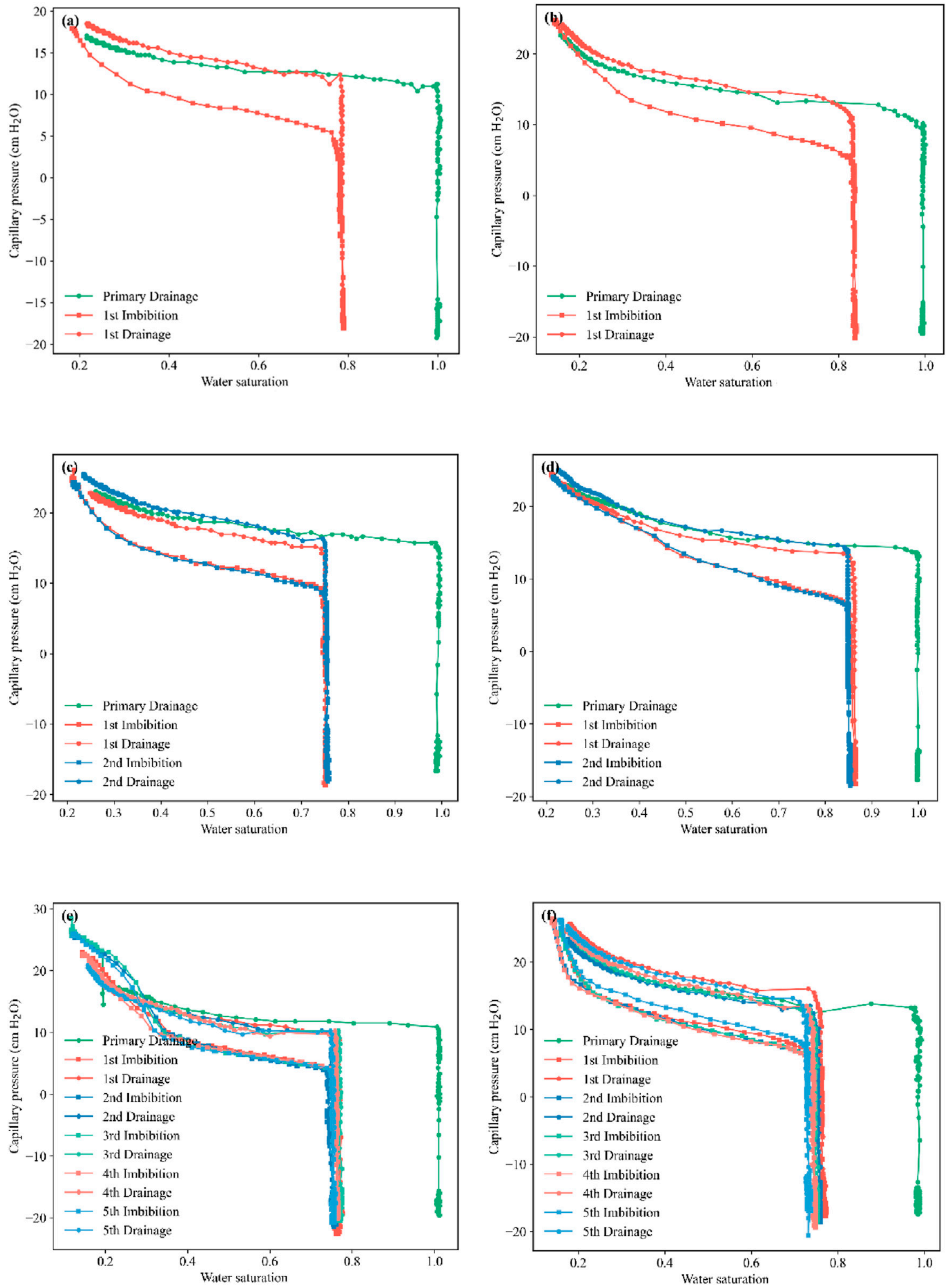

FIGURE 11
Cyclic wetting-drying water retention curves: Cycle 1 **(a,b)**, Cycle 2 **(c,d)**, Cycle 5 **(e,f)**. (Pristine PS: **(a,c,e)** Aged PS: **(b,d,f)**).

repulsion—unsaturated drainage is dominated by AWI detachment forces. These forces are critically dependent on contact angle: lower contact angles (higher hydrophilicity) reduce capillary pinning at the air-water-solid (AWS) interface, thereby weakening detachment forces (Scheludko et al., 1976). Given that AWI detachment forces exceed electrostatic and hydrodynamic forces by orders of

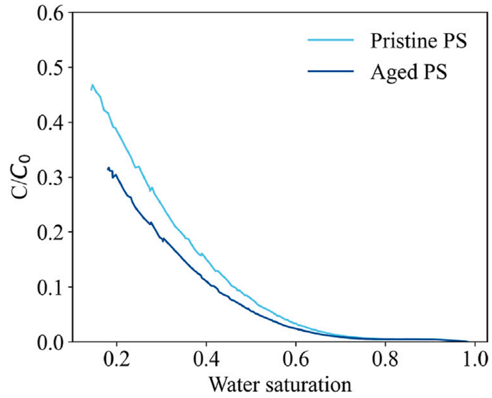

FIGURE 12
Microplastic release vs water saturation during primary drainage.

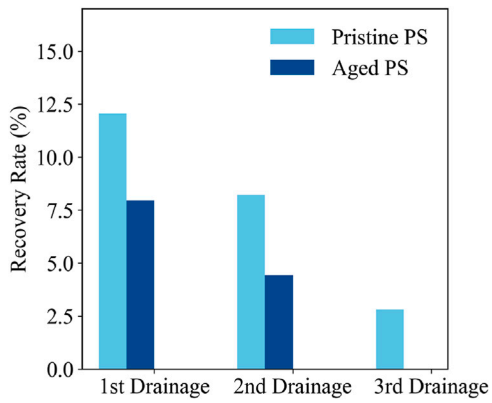

FIGURE 13
Microplastic release recovery rates.

magnitude (Hou et al., 2020), hydrophilicity becomes the governing factor, overriding any mobility promotion from increased electronegativity.

Additionally, aging-induced surface cracks (Figure 5) increase roughness, which enhances adhesion energy and impedes AWI movement across the microplastic surface (Singh and Joseph, 2005). Collectively, hydrophilicity-driven reduction in detachment forces and roughness-enhanced retention significantly reduce aged microplastic mobility in unsaturated porous media.

3.2.3 Effect of cycle number on Re-release behavior

To assess the remobilization potential of retained microplastics, wetting-drying cycles were implemented following initial drainage. Unlike initial drainage, secondary and tertiary drainage phases were preceded by wetting phases (Figure 14). During wetting, water displaced pore air, increasing water saturation (Sw) while decreasing capillary pressure (Pc), with water preferentially invading smaller pores under higher Pc.

Throughout cycling, the Pc-Sw relationship exhibited consistent hysteresis (Figure 15), demonstrating path dependence between wetting and drying phases (Topp et al., 1967). Each drainage phase reduced Sw below the critical threshold established during initial drainage, indicating that re-established mobile air-water interfaces (AWIs) regained potential to remobilize microplastics near the solid-water interface (SWI). Crucially, the wetting phase reintroduced water, disrupting pre-existing AWIs and water films, thereby remobilizing microplastics attached to these interfaces. The distinct hydrodynamic conditions between phases consequently led to phase-dependent remobilization behaviors.

Release recovery rates progressively decreased with increasing cycle number (Figure 13). This decline was more pronounced for aged microplastics: aged PS recovery decreased from 7.96% (initial drainage) to 4.43% (secondary drainage), becoming undetectable during tertiary drainage. Despite similar hydraulic conditions conducive to release in subsequent drainage phases, observed release behavior differed significantly. Both pristine and aged PS exhibited substantially lower recovery in later cycles, indicating markedly reduced mobility in unsaturated porous media after multiple cycles. Residual microplastics demonstrated long-term retention characteristics. This persistent low recovery occurred because each release event selectively mobilized weakly adhering particles (Bradford and Torkzaban, 2013). Consequently, residual microplastics after multiple cycles represent a strongly adhering, less mobile fraction with higher retention stability (Chen et al., 2015).

Aging critically altered microplastic surface properties: enhanced hydrophilicity substantially weakened AWI detachment forces. Additionally, under low saturation conditions, pore water existed as isolated films or pendular rings, trapping microplastics and impeding AWI-mediated release (Feng et al., 2024). These factors collectively rendered aged microplastics highly resistant to release after multiple cycles.

3.3 Cumulative retention behavior of aged microplastics in unsaturated media

3.3.1 Effect of aging on microplastic retention and accumulation in unsaturated media

Figure 16 shows cumulative retention rates for pristine and aged microplastics after 1, 2, and 5 wetting-drying cycles. Aged microplastics consistently exhibited higher retention than their pristine counterparts at each cycle stage, demonstrating that aging promotes long-term retention under repeated cycling.

Although aging enhanced microplastic electronegativity—increasing electrostatic repulsion with porous media—the dominant unsaturated removal mechanism, detachment at the air-water interface (AWI) during drainage, was significantly weakened by aging-induced hydrophilicity. Concurrently, the hysteresis-driven drying path created thinner water films and potentially stronger capillary pinning forces, enhancing static retention potential. Consequently, aged microplastics showed substantially lower removal efficiency than pristine particles during each drying phase, characterized by specific moisture distribution under hysteresis effects.

Following 32-day aging, all microplastics exhibited increased surface roughness. When the colloid-to-grain size ratio (dp/d50) exceeds ~0.5–1%, physical retention mechanisms (pore straining, mechanical interlocking) become significant alongside chemical

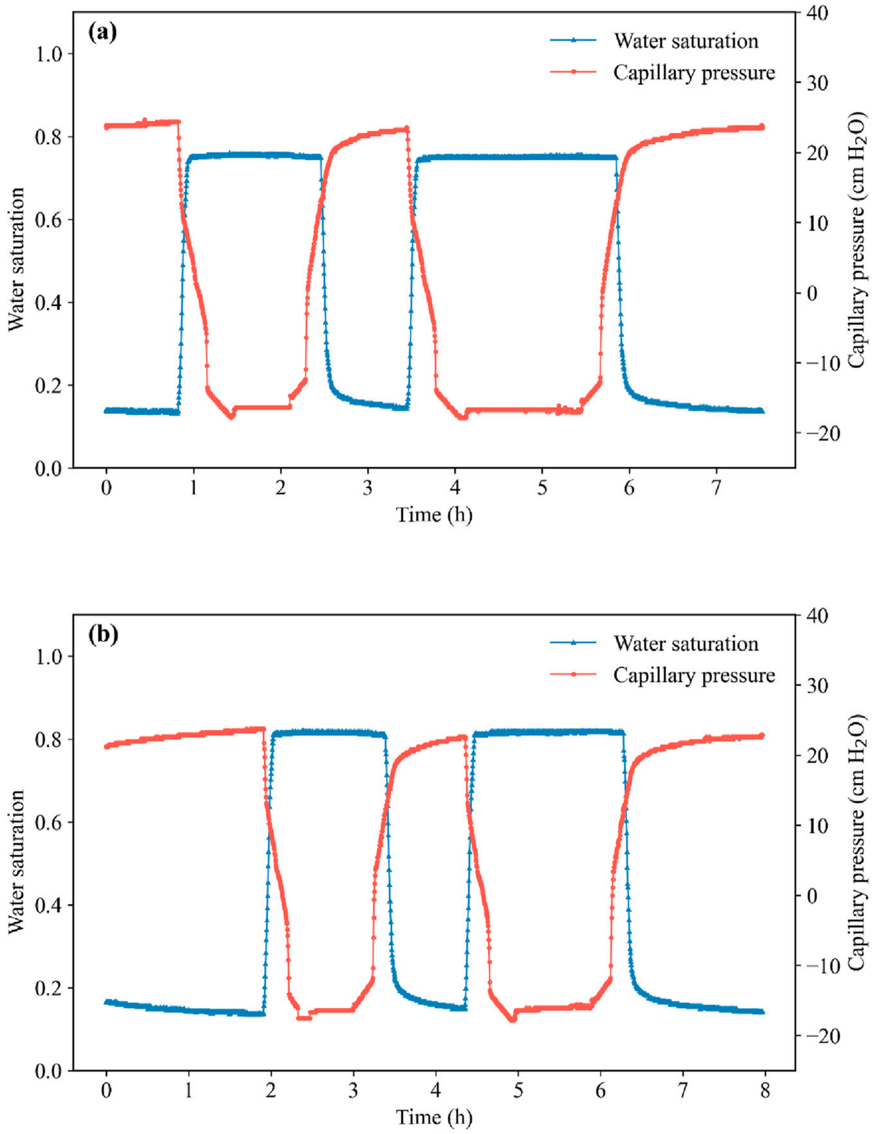

FIGURE 14
Multi-cycle wet-dry dynamics: Water saturation and capillary pressure for **(a)** Pristine PS; **(b)** Aged PS.

attachment (Bradford et al., 2007). In unsaturated media, air occupying pore space narrows effective pathways, likely enhancing physical retention of aged microplastics. Repeated injections during cycles further increased retention sites and physical straining efficiency.

Given the ~1% size ratio in this system and the complex geometry of unsaturated pores, physical retention emerged as a key mechanism. Increased roughness promotes aged microplastic retention during wetting-drying cycles (Fei et al., 2022), making them more susceptible to entrapment during flow perturbations.

Although aging-enhanced electrostatic repulsion initially inhibited chemical attachment during wetting, this effect was ultimately overridden by drastically reduced removal efficiency during the drainage phase over long-term cycling. Consequently, under the dominance of physical retention and diminished removal efficiency, cumulative retention of aged microplastics exceeded that of pristine particles.

After 5 cycles, retention rates for pristine and aged PS were 11.05% vs 9.38% (bottom) and 3.76% vs 5.48% (surface). Aging produced a more uniform vertical distribution due to enhanced hydrophilicity and electronegativity, which stabilized dispersion and prevented localized accumulation at the inlet or interfaces.

Physical retention occurred throughout the migration path as pore-straining structures (dp/d50 > 0.5–1%) were widely distributed (Chen et al., 2008), resulting in random-depth retention of aged microplastics via physical mechanisms. While pristine microplastics underwent AWI-driven re-concentration, aged particles demonstrated weaker attachment, enabling short-distance detachment/re-attachment during flow and exhibiting limited AWI-driven transport (Wu et al., 2019). Combined with repeated flushing and retention, this yielded a more homogeneous spatial distribution of aged microplastics. Aging thus simultaneously enhanced cumulative retention—through reduced removal and

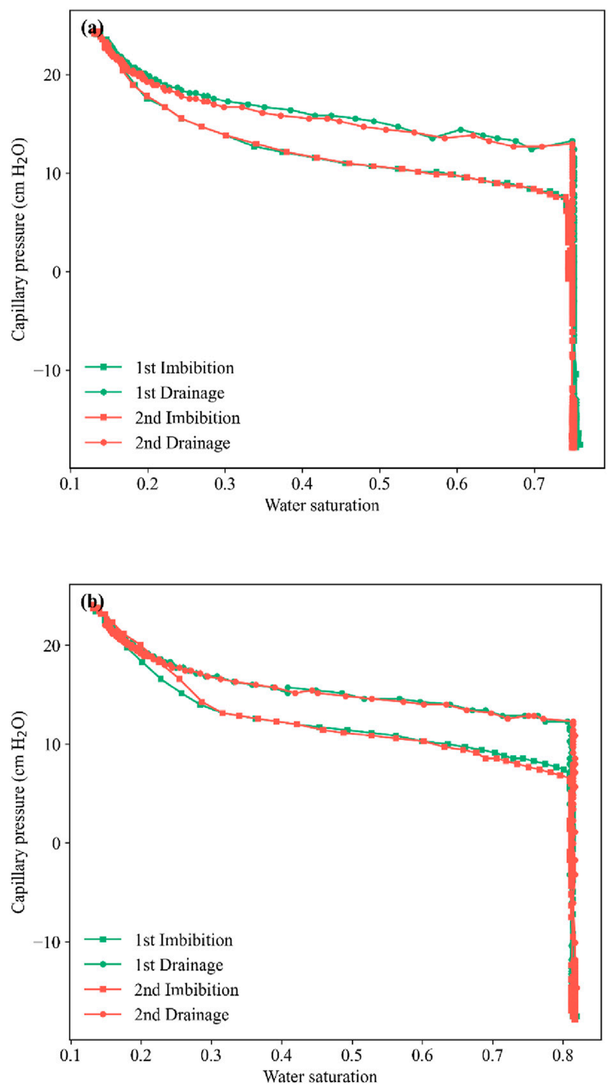

FIGURE 15
Multi-cycle wet-dry water retention curves: **(a)** Pristine PS; **(b)** Aged PS.

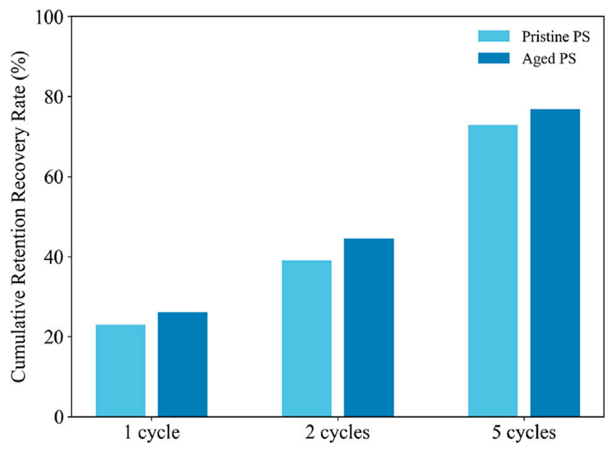

FIGURE 16
Cumulative microplastic retention recovery rates.

increased physical trapping—and distribution uniformity via weak attachment and low remobilization.

3.3.2 Effect of cycle number on cumulative retention behavior

As presented in Table 1, the cumulative retention rates of pristine PS microplastics increased progressively to 22.94%, 38.97%, and 72.83% after 1, 2, and 5 wetting-drying cycles, respectively. Aged PS exhibited even higher retention, reaching 76.88% after 5 cycles. Retention significantly increased with cycle number for both microplastic types, although the accumulation rate gradually declined. This demonstrates that repeated cycling enhanced microplastic retention within the porous media by progressively strengthening retention mechanisms while weakening removal processes.

For all microplastics, particles exhibiting weaker adhesion—those more readily mobilized by air-water interfaces (AWIs)—were likely removed during initial cycles, leaving behind a population of increasingly stable particles. Furthermore, repeated microplastic injections during wetting phases continuously introduced new particles to occupy available retention sites. Physical straining mechanisms also intensified over successive cycles: newly injected particles became strained at pore throats or adhered to rough surfaces, while previously retained particles formed "filter layers" that captured incoming particles. These cumulative effects amplified retention with increasing cycle number.

Figure 17 illustrates the vertical distribution of pristine and aged microplastics after 1, 2, and 5 cycles. Retention was consistently higher in the bottom layer (36–40.5 cm depth: pristine PS 6.20%–11.05%; aged PS 6.60%–9.38%) compared to the surface layer (0–4.5 cm: pristine PS 0.75%–3.76%; aged PS 0.75%–5.48%). However, distributions became increasingly uniform with progressive cycling.

In porous media where pore throat diameters exceed microplastic particle sizes, particles can theoretically migrate upward (Gibson et al., 2009). The observed elevated bottom retention likely resulted from the formation of "filter layers" near the inlet, where embedded particles captured subsequently arriving microplastics.

During cycling, mobile AWIs remobilized retained particles, including those within filter layers. These remobilized particles were transported upward with pore water flow and subsequently re-retained at various depths due to hydrodynamic shifts or interface collapse. This cyclic capture-transport-re-retention process continuously redistributed particles vertically.

Concurrently, flow perturbations enabled physical straining throughout the column. Under the combined influence of AWI remobilization, flow-driven retention, and inherent transport properties, repeated wetting-drying cycles progressively homogenized the spatial distribution of microplastics.

4 Conclusion

This study systematically investigated the regulatory mechanisms governing retention and release dynamics of

TABLE 1 Microplastic recovery rate in unsaturated porous media after wet-dry cycles.

Aging time (days)	Cycle number	Cumulative retention recovery (%)	Packing material recovery rate (%)	Mass loss (%)
Pristine PS	1	22.94	18.58	4.36
	2	38.97	32.34	6.62
	5	72.83	61.18	11.65
Aged PS	1	26.14	21.9	4.18
	2	44.59	35.23	9.36
	5	76.88	67.66	9.23

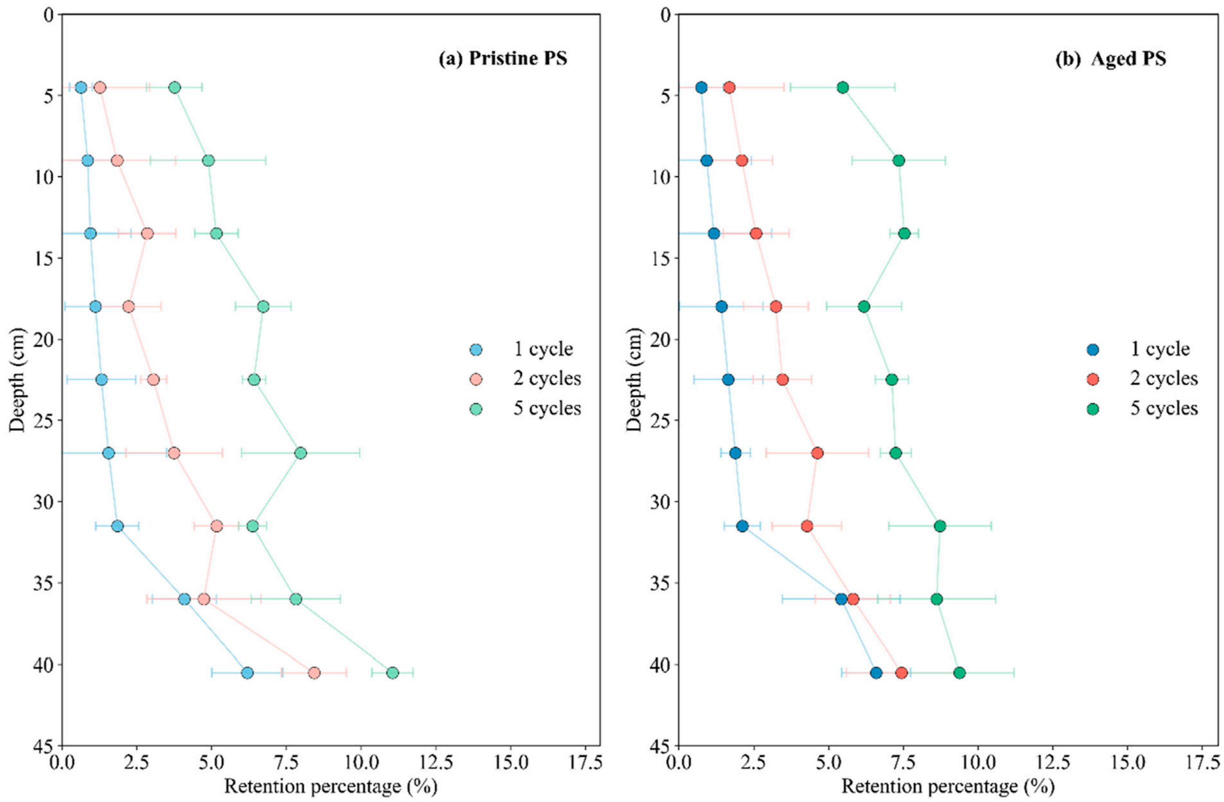

FIGURE 17
Vertical distribution of retained microplastics after wet-dry cycles: Pristine PS vs Aged PS (Cycles 1, 2, 5).

aged microplastics (MPs) in unsaturated porous media under cyclic wetting-drying conditions, revealing dominant transport mechanisms and key hydrodynamic drivers. Key findings include:

(1) Aging fundamentally modified MPs' physicochemical properties, significantly increasing surface electronegativity and hydrophilicity.
(2) During unsaturated drying phases, aging suppressed remobilization of retained MPs. Aged MPs exhibited substantially lower release efficiency than pristine MPs, with release dominated by the moving air-water interface (AWI) where hydrophobicity governs detachment. MPs undergo remobilization via capillary separation forces at the advancing AWI when water saturation falls below a critical threshold (~0.6).
(3) Under cyclic wetting-drying with continuous MP influx, aging significantly enhanced long-term retention while promoting vertical distribution homogeneity. After 5 cycles, cumulative retention of aged MPs (76.88%) consistently exceeded that of pristine particles (72.83%). This enhanced retention is attributed primarily to aging-induced surface roughness enhancing physical straining mechanisms. Critically, the cumulative effect of these retention-promoting factors outweighed aging-enhanced electrostatic repulsion during wetting phases.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding authors.

Author contributions

XJ: Data curation, Formal Analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Validation, Writing – original draft. XH: Formal Analysis, Investigation, Methodology, Validation, Writing – original draft. YL: Formal Analysis, Investigation, Methodology, Writing – original draft. YS: Data curation, Formal Analysis, Resources, Validation, Writing – original draft. ZS: Writing – original draft. HQ: Writing – original draft, Writing – review and editing, Conceptualization, Data curation, Formal Analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Supervision, Validation, Visualization. LY: Conceptualization, Data curation, Formal Analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Supervision, Validation, Visualization, Writing – original draft, Writing – review and editing.

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Conflict of interest

Author SY was employed by Zhejiang Zhouhuan Environmental Engineering Design Co., Ltd.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Getting occurrence, distribution, fate and detrimental effects of microplastic in forests into focus: expectations and challenges

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1 Introduction

Micro-(nano-) plastics (MNPs; MPs <5 mm; NPs <1 μm) have been widely distributed in the environmental compartments, and potentially threatened the ecosystems (Wu et al., 2022a; Wu et al., 2024a). Since they were firstly reported in 2004, large quantities of studies have investigated their occurrence, distribution, and potential risks in aquatic systems (Thompson et al., 2004; Wu et al., 2022b). A agreement achieved that the contained MNPs are mainly originated from the terrestrial inputs, including the water runoff, groundwater exchange, etc (Kumar et al., 2023). Thereafter, a growing body of studies concentrated on investigating the content, fate and detrimental effects of MNPs in terrestrial environments (Zhu and Huang, 2025). Newer data suggest that MNPs are moving from the soil to the plant, as demonstrated by their presence in lettuce (Li et al., 2020), carrot (Dong et al., 2021), wheat (Li et al., 2020), and rice seedlings (Liu et al., 2022), raising significant concerns on their negative consequences on the farmland. Therefore, more and more attention has been paid on the potential detrimental effects of MNPs on farm products. As a crucial part of agroforestry economics, the safety of forestry products in contrast is often ignored, which caused by not enough information on MNPs occurrence has been reported in recent studies.

Forestry land is also of great importance in ecosystem, covering over 32% of global land surface (Winkler et al., 2021), yet forests and their soils are rarely considered in MNP research. Forests act as the sink of carbon, helps regulating the climate, and provides habitats for diverse for a and fauna. The MNP contamination of forestry land could affect the health of the forestry ecosystem (Weber et al., 2023). Recent study investigate MNPs contamination in 8 forests in Korea and found they were between 20 and 720 particles kg^{-1} in three single forests worldwide (Choi et al., 2021). They further reported that the presence of MNPs can alter soil structure and decrease the fertility, having the potential influences on the growth of trees and the health of forest. Although the related information is still infancy, more and more publications have reported that the uptake of MNPs could biomagnify through the food web and pose risks to wildlifes. Thus, it is of great importance to understand the occurrence, distribution, fate and detrimental effects of MNPs in forests.

2 Sources and pathways of MPs in forestry ecosystems

2.1 Atmospheric deposition

Atmospheric transport was regarded as a critical route for MNPs contamination in forests (Allen et al., 2019). Anthropogenically generated MNPs from urban/industrial emission sources undergo complex aerodynamic transport processes. These lightweight particles are entrained in atmospheric circulation patterns, with field measurements confirming their presence in precipitation samples from remote alpine regions (e.g., Pyrenees, 2,877 m ASL) at concentrations up to 365 particles/m^2/day (Allen et al., 2019). The deposition dynamics encompass both dry (gravitational settling) and wet (precipitation scavenging) mechanisms, with particle form (sphericity, aspect ratio) markedly affecting residence times (Huang et al., 2021). Studies on canopy interception effectiveness indicate that coniferous forests catch more atmospheric MNPs than deciduous trees, attributed to increased surface roughness and wax-mediated adhesive effects (Weber et al., 2023).

2.2 Anthropogenic edge inputs

The permeability thresholds in forest buffer zones are increasingly influenced by long-term MNP inputs from adjacent anthropogenic systems. Agricultural matrices contribute large numbers of MNPs annually to various environmental compartments through composite pathways: (1) Degradation of low-density polyethylene mulch films releasing MNPs into the forests (Sintim et al., 2020); (2) Biosolid-amended fertilizers containing up to 286 particles/g dry weight (Naderi Beni et al., 2023); (3) Wastewater irrigation delivering 1.0–2.4 × 10^4 particles/kg effluent (Wu et al., 2024b). These inputs induce measurable pedological alterations, including the reduction in saturated hydraulic conductivity and the decrease of pH value in topsoil horizons. Field experiments demonstrate MP-soil interactions promote aggregate destabilization through weak hydrogen bonding with clay minerals, exacerbating nutrient leaching losses (Elmholt et al., 2008).

2.3 *In situ* generation

In situ MP generation in forest ecosystems follows quantifiable weathering trajectories governed by Arrhenius kinetics. Polymeric materials from tourism debris (PET bottles, LDPE wrappers) and logging residues (PP rope fragments, HDPE fuel containers) undergo sequential degradation: Photo-oxidative cleavage, hydrolytic depolymerization, and mechanical embrittlement (Wu et al., 2024c). Accelerated aging tests indicate high-density plastics (e.g., PET) require 8.3 ± 1.2 years for 50% mass loss under temperate forest conditions, versus 2.1 ± 0.7 years for low-density films (LDPE). Secondary MP generation rates peak with particle size distributions skewing towards environmentally persistent 10–100 μm fractions (Cózar et al., 2014).

3 Ecotoxicological effects on forestry ecosystems

3.1 Soil physicochemical impacts

MNPs systematically compromise forest ecosystems through multidimensional pathways. In soil systems, MNPs have been certified to reduce macroporosity by 15%–30% and water-holding capacity by 18%–25% via structural disruption, while their hydrophobic surfaces amplify contaminant bioavailability (Schefer et al., 2025). Wang et al. (2024) determined the concentration dependent effects of PVC MNPs on affecting the physicochemical characters of soils with various soil textures. The soil texture should be a key issue affecting their pore connectivity, especially for sandy and sandy loam soils. The results may be attributed to the following mechanisms: First, the MNPs could occupy the pore spaces of soil and create disconnected voids, thereby reducing the pore connectivity and effective porosity (Schefer et al., 2025). Second, many types of MNPs are hydrophobic, which could decrease the soil water affinity, thereby disrupting capillary forces that stabilize pore networks (Shafea et al., 2023). Third, the fate of MNPs in soil could also influence its connectivity by inhibiting the soil fauna and adsorbing other chemicals to form MNP-aggregate complexes (Schefer et al., 2025).

3.2 Microbial functional changes

Concurrently, sublethal MP exposure suppresses microbial functions: nitrogen-fixing bacteria exhibit 40%–60% reduced nifH expression, correlating with 22%–35% declines in nitrogen mineralization, while arbuscular mycorrhizal fungi show 30%–50% fewer root-colonizing hyphae, impairing phosphorus uptake (Aralappanavar et al., 2024). Metagenomic shifts further reduce extracellular enzyme activity.

3.3 Plant physiological responses

Plant physiology is equally compromised—MPs (10–300 μm) adhere to roots via electrostatic forces, reducing absorptive surface area and hydraulic conductivity (Sun et al., 2020), while sequestered Fe^{3+}/Zn^{2+} induces micronutrient deficiencies. Leached additives like Di (2-ethylhexyl) phthalate trigger root ROS surges (H_2O_2), causing mitochondrial damage and stunting conifer growth (Wu et al., 2022a). Soil invertebrates face acute toxicity: earthworms ingesting 5–20 μm MPs endure gut epithelial damage, reducing cast production by 35%–60%, while collembola suffer 50% mortality and 65%–80% egg reduction at 100 mg/kg MPs (Guo et al., 2023).

3.4 Other potential effects

Trophic transfer escalates risks—birds accumulate 12–45 particles/g in gizzards, and tertiary consumers like martens ingest DDT-laden MPs. Critically, MNPs synergize with climate stressors: MNP-contaminated soils desiccate faster during droughts, advancing tree wilting, while suppressed jasmonic acid signaling in oaks increases herbivory. MNPs also lower lignocellulose ignition temperatures, accelerating crown fires, and release toxic acrylonitrile upon combustion, compounding post-fire ecotoxicity (Jin et al., 2024). This cascade of impacts underscores MPs as a keystone stressor in Anthropocene forest decline.

4 Challenges in assessing forestry MP ecotoxicity

Comprehensive evaluation of MNP consequences in forest ecosystems is hampered by important knowledge gaps. Current methodological constraints, especially spectroscopic and chromatographic approaches, lack sufficient resolution to quantify nanoplastics (<1 μm) inside organic-rich soil matrix, where humic acids and lignocellulosic chemicals interact with polymer identification. Lack of longitudinal studies—few datasets span decadal durations required to assess MNP accumulation rates against tree lifespans (typically >50 years for temperate species) or legacy soil processes like humification cycles—adds to this analytical limit. Moreover, the complex linkages among MNPs, soil biogeochemistry (e.g., pH-dependent polymer breakdown), and biotic networks (e.g., microbiome-plastic interactions) still cause fragmented knowledge at the level of ecosystems. Addressing these challenges calls for long-term ecological monitoring systems, multidisciplinary integration of advanced characterization tools (such as pyrolysis-GC/MS with isotopic tracing), mechanistic models bridging microbial ecology, polymer science, and forest hydrology to untangle emergent hazards in these complicated systems.

5 Recommendations for policy and research

5.1 Research priorities

As for MNPs in forest ecosystems, several actions for decreasing MNP threats to forest ecosystems could be discussed as follows: First, it is of great importance to develop standardize microplastic collection process, consistent monitoring protocols specific to forest environments including harmonized methodologies for soil core sampling, canopy particle collecting, and hyperspectral detection of MNPs in organic matrix to produce comparable baseline data. Meanwhile, the ecotoxicological thresholds and species specific resistance can be determined by controlled mesocosm experiments, such as the gradient MNP concentrations from (0.1%–10% w/w), exploring the nutrient or contaminants transferring. In addition, critical research gaps requiring urgent attention on nanoparticle mobility in root systems, polymer-specific adsorption dynamics in humic soils, and co-stressor interactions (e.g., drought or heavy metals).

5.2 Policy and community engagement

Scientific insights must inform policy reforms, such as phasing out non-degradable agricultural mulch films (with subsidies for biodegradable alternatives), mandating wastewater treatment upgrades to capture sub-micron particles (prioritizing plants near old-growth forests), and integrating MNP surveillance into protected area management plans (e.g., IUCN Category II forests). Plastic influx can be decreased while promoting stewardship through parallel public engagement projects aimed to communities near forests, such as bilingual workshops on reusable alternatives to single-use plastics and citizen science initiatives for litter audits. Parallel public engagement campaigns targeting forest-adjacent communities—via multilingual workshops on reusable alternatives to single-use plastics and citizen science programs for litter auditing—can reduce plastic influx while fostering stewardship. By synergizing robust science, evidence-based governance, and grassroots behavioral change, this multipronged strategy offers a pathway toward mitigating MNP-driven forest degradation.

6 Conclusion

As the pervasive contaminants, MNPs also pose emerging threats to forests despite receiving less attention than agricultural systems. Covering 32% of Earth's land surface, forests act as vital carbon sinks, biodiversity refuges as well as receiving MNPs via atmospheric deposition, anthropogenic edge inputs from agriculture, and *in situ* generation from tourism debris. These MNPs significantly degrade forest health through multiple pathways: they alter soil physicochemical characters, reducing the pore connectivity by occupying pore spaces and increasing hydrophobicity; impair microbial functions; change plant physiology; release hazardous additives; and biomagnify risks through trophic transfer. Synergistic interactions with climate stressors exacerbate impacts, such as accelerated drought-induced wilting or altered fire behavior. The main research gaps still exist as the method limitations in calculating MNPs in organisms and a lack of long-term studies. For over these obstacles, the standardized forest-specific monitoring protocols and mesocosm experiments are required to determine ecotoxicological thresholds for keystone species, policy reforms, and community engagement initiatives to mitigate MNP-driven forest degradation.

Author contributions

KT: Writing – original draft, Data curation, Writing – review and editing, Validation, Conceptualization. SP: Writing – original draft, Conceptualization, Resources, Investigation. YM: Conceptualization, Investigation, Writing – original draft. PY: Writing – original draft, Investigation. QH: Project administration, Formal Analysis, Conceptualization, Methodology, Supervision, Data curation, Writing – original draft, Software, Visualization, Resources, Writing – review and editing, Funding acquisition, Validation, Investigation. YL: Writing – review and editing, Investigation, Writing – original draft, Supervision, Data curation, Software, Methodology, Resources, Funding acquisition, Visualization, Conceptualization, Formal Analysis, Project administration, Validation.

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Conflict of interest

Author PY was employed by Suining Runqi Investment Co., Ltd.

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Revealing the migration behaviors of microplastics in the intertidal environments

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1 Introduction

In January 2020, the novel coronavirus pneumonia (COVID-19) rapidly spread throughout the world. As of the post-pandemic (July 2023), the World Health Organization (WHO) reports 767 million confirmed cases and 6.9 million deaths worldwide, with new variants like Omicron still causing regional outbreaks (WHO, 2025). The WHO recommended taking strict interventions decisively to prevent further spread of COVID-19, such as wearing face masks, washing hands, and staying at home (WHO, 2025). The measures brought about an upsurge in the demand for plastics, especially face masks, single-use dishware, and disinfection supplies. Despite vaccination efforts, the pandemic has generated over 8.4 million tons of plastic waste globally, according to 2022 satellite estimates, primarily from medical equipment and packaging (Peng et al., 2021). This surge disrupted the 2025 global plastic reduction targets outlined in the UN Environment Assembly resolution (UNEA-5.2), as nations prioritized single-use plastics for infection control.

For better management of plastic waste, the Basel Convention on the Plastic Waste Partnership also urged progressive plastic management during the COVID-19 pandemic. However, some plastic wastes are still not disposed of properly and enter the ocean surroundings. The debris can be fragmented and degraded into microplastics (MPs; <5 mm) and nanoplastics (NPs; <1 μm) under the effects of solar radiation, temperature variation, wave slap, and biological interactions (Peeken et al., 2018). With the effect of ocean currents and tides, these micro-(nano-)plastics (MNPs) migrate and accumulate in the intertidal zone, a place with strong land–sea interactions (Lv et al., 2020).

Intertidal zones can be roughly classified as mudflats, sandy beaches, rocky beaches, vegetated marshes, and mangrove swamps (Murray et al., 2019). They are extraordinarily important as they not only provide habitat but also feed organisms. During the pandemic, the OceansAsia organization recorded that masses of various types of face masks washed up on Hong Kong Soko beaches (Figure 1) (Reuters, 2020). These MNPs contain many severe ecological hazards: They adsorb surrounding pollutants, disrupt microbial and invertebrate physiology, and accumulate in food webs, ultimately threatening biodiversity and ecosystem resilience. However, a summary of the interaction between MNPs and the intertidal environment is still very limited and not yet fully understood. Thus, the aim of the study is to determine the interactions between MNPs and intertidal zones with different geological conditions. The study also calls attention to reducing the quantity of mismanaged plastic induced by the COVID-19 pandemic in the future. Overall, this study will be of vital importance during the special period to predict the possible environmental risks brought by coronavirus disease prevention manners.

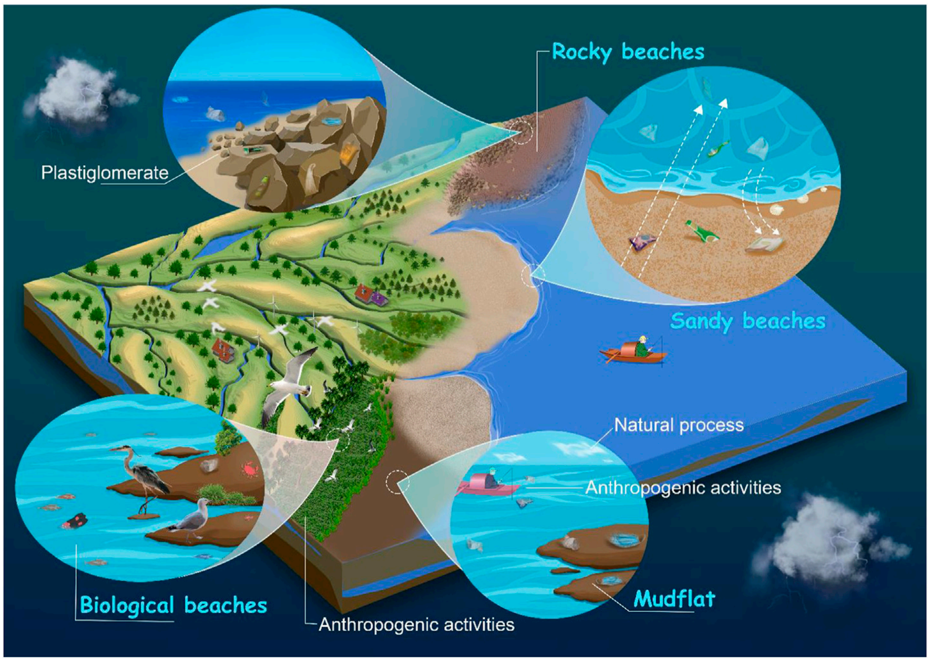

FIGURE 1
Migration of micro-(nano-)plastics (MNPs) in the various intertidal types, including mudflats, biological beaches, rocky zones, and sandy beaches.

2 Common migration of MNPs in various types of intertidal zone

Mudflats are often composed of the deposition and accumulation of suspended mineral/organic particulate matter and soluble nutrients in lacustrine, riverine, and estuarine environments (Figure 1). With the exponential usage of plastic during the pandemic, it could be speculated that a large number of MNPs will be discovered due to severe fragmentation that is the result of unique hydrodynamics, strong solar radiation, and human intervention (Wu et al., 2024). Two main routes for the convergence of plastics into the mudflats include natural processes and anthropogenic activities. Natural processes include atmospheric deposition, tidal action, surface runoff, and river transportation (Wei et al., 2024). Anthropogenic activities would be another major route, as mudflats are prime recreational sites. The domestic sewage generated from such activities would increase the secondary outbreak risks by propagating pathogens (Isobe et al., 2019).

As mudflats can supply abundant minerals and nutrients, they serve as a place for numerous plants and are a shelter and nursery ground for many fish and birds (Figure 1) (Li et al., 2020). Biological beaches have also been regarded as hotspots of MNP accumulation after convergence, generating an overlap between MNPs and some species (e.g., zooplankton, invertebrates, crustaceans, fish, seabirds, and even marine mammals) (Wu et al., 2022b). After ingestion, most large microplastics exist in the gastrointestinal tract for several days. However, some of the small MNPs (<100 μm) translocate and induce deterioration in the various systems, such as tissues, digestive, lymphatic, and even the neurological systems (Wu et al., 2022a; Huang et al., 2024). During the whole process, MNPs could release harmful chemicals like additives and even pathogens, such as the release of *Halofolliculina* on plastics, inducing disease outbreaks in organisms (Lamb et al., 2018). It could be hypothesized that the coronavirus attached to such plastics might be the source of multiple outbreaks of the COVID-19 epidemic.

Rocky beaches are where large rock predominates in the coastal zone. Recently, some rocks on beaches have been contaminated by the sink of plastics, defined as “plastiglomerates” (Figure 1) (Das, 2023; Corcoran and Jazvac, 2020). From the geologists’ perspectives, plastiglomerates might be a new indicator of the Anthropocene due to their omnipresence and persistence characteristics. Thus, more and more research works have been performed on plastic pollution on rocky beaches (Turner et al., 2019; Das, 2023). Recently, a new effect named “pyroplastic” was reported; it is generated when burnt plastic debris becomes part of an agglomerate with rocks after cooling (Figure 1). The phenomenon has been documented in many beaches, such as Hawaii beaches in the United States and white sand beaches in the United Kingdom. Meanwhile, another contamination, called “plasticrust,” was reported to form when the rock surface was encrusted by plastics and formed variable crusts (Gestoso et al., 2019). It has been observed on the mid-upper rocky shore of Madeira Island of Portugal and Giglio Island of Italy. The continuous hydrodynamics induced the crusting between MNPs and rocks in these places. Moreover, the plasticrust phenomenon becomes more severe with increased coverage as time passes (mean ± error bar: 9.46% ± 1.77% in 20 cm × 20 cm quadrants; n = 10) (Gestoso et al., 2019). Unlike “pyroplastic,” which is mainly made of polyethylene terephthalate (PET), polyethylene (PE) with blue and white colors is the predominant type of plasticrust contamination. As both PET and PE are mainly used in domestic products, the origins of the two plastic contaminations can be narrowed down to domestic wastes from their corresponding countries.

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The latest studies reported that global plastic production increased from 370.7 million tons (Mt) in 2018 (pre-pandemic) to 413.8 Mt in 2023 (post-pandemic) (Zhu and Huang, 2025; PlasticEurope, 2025). Even in the post-pandemic, the public perceptions and attitudes toward health and hygiene practices have changed. Wearing masks in public, which was once considered unusual outside of specific cultural contexts or health scenarios, has now become widely accepted and normalized, resulting in an annual growth rate of approximately 10% (Zhu and Huang, 2025). Thus, coastal marine debris monitoring and action plans might consider plastiglomerate contamination in their marine environmental monitoring guidelines.

Sandy beaches are landforms alongside the sea composed of loose particles. MNP transportation is easily affected by changing ecological conditions, such as tides, winds, waves, and thermohaline gradients (Figure 1). Drifting behavior is very common and hypothesized as a comprehensive response to tide and wind speed. Thus, the three-dimensional oil spill model was found able to well simulate and predict the drifting track of MNPs (Isobe et al., 2014). Some low-density MNPs (<1.00 g·cm^{-3}) are easily carried into the marine environment due to wind- and wave-induced currents and roll structures. The repeated MNP migration continues according to their sizes (Hurley et al., 2018). Small-size MNPs have the opportunity to escape from the offshore trapping and enter the open sea, while large-plastic debris is selectively conveyed onshore again by a combination of Stokes drift and terminal velocity (Seeley et al., 2020). Meanwhile, the migration of MNPs according to their density (1.00–1.03 g·cm^{-3}) could be influenced by the thermohaline gradients. Some MPs with higher density (>1.03 g·cm^{-3}) tend to settle and concentrate in benthic environments (Seeley et al., 2020). Most MNPs are gradually buried by turbulence and persist in the deepest depth in the sandy sediments (Wu et al., 2022b). However, the mechanisms for MNP burial are still unclear and also need more numerical simulation and experimental results. Nowadays, the numerical parameters set in simulating the migration of MNPs are mainly based on experience or publications. Therefore, future studies are needed to enhance the accuracy of the parameter setting by combining laboratory studies and field investigations.

3 Conclusion and perspectives

The intertidal zone, especially the mudflats, rocky shores, and sandy beaches, has been severely contaminated by the upsurge of plastic waste. Mudflats as "plastic sinks" concentrate MNPs up to 10 times more than sandy beaches due to their nutrient-rich sediments and dynamic hydrodynamics (Lo et al., 2020). Notably, virus-laden MNPs could be adsorbed onto organic matter, creating pathogen reservoirs. Conversely, rocky shores show concerning geochemical changes: "plasticrusts" (dominated by PE/PET) and pyroplastics (derived from PET) on coastal rock surfaces, enshrining pandemic plastics into geological archives with half-lives exceeding centuries (Das, 2023). Sandy beaches serve as "sorting engines" because low-density MNPs (<1.0 g/cm^3) re-enter marine circulation via Stokes drift, while denser particles infiltrate benthic food webs, with 22% of crustaceans showing neurotoxic amounts of nanoplastics (Huang et al., 2024). The findings underscore an urgent change: intertidal zones are active mediators of plastic–pathogen feedback loops. To address what we call the "Anthropocene Boomerang Effect"—plastics are now endangering global health through ecological feedback loops—it is necessary to redefine coastal monitoring frameworks by incorporating artificial intelligence-driven techniques for creating plastiglomerate mapping and tidal hydrodynamic models.

Author contributions

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Plastics pollution: pathways, impacts, and regulatory challenges in marine environments

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This review synthesizes existing literature on microplastics in marine ecosystems from various oceanic regions. Microplastics in marine environment originate from a range of sources, including land-based activities, rivers inputs and oceanic-based sources such as fishing, aquaculture, tourism and extreme oceanic events. Methodological and technical limitations, like sampling, identification and quantification, as well as data reporting and analysis, are key constraints in microplastics research, making it difficult to evaluate plastic debris volume in different marine environments. Microplastics have colonized diverse oceans, even polar areas. Their spatial distribution is influenced by their physicochemical properties as well as factors influencing their transport including wind driven waves, current and colonization by microorganisms. The most prevalent polymers in various oceanic systems are PE, PP, and PS, accounting for more than 60% of recovered microplastics. Microplastics affect both unicellular and multicellular marine organisms at various structural levels, causing significant disruptions that negatively impact their ecological and biological functions as well as their social behavior. This threatens both human and ecosystem health. Microplastics significantly impact marine ecosystem services, with total potential losses estimated to be between 1.18 and 2.16 trillion USD, accounting for about 2% of global GDP. Microplastics impair blue carbon ecosystems, reducing their carbon sequestration capacity and exacerbating the economic costs associated with climate regulation and coastal protection. The existing regulatory frameworks addressing plastic pollution are synthesized to identify gaps and highlight opportunities for enhancing and implementing more effective, evidence-based regulations that promote environmental sustainability.

KEYWORDS

microplastics, pathways, accumulation, impacts, ecosystem services, policies and regulations, sustainability

1 Introduction

Global plastics production reached 390.7 million tons at the end of 2021, with nearly 80% of these plastics are likely to end up in natural environment (Plastics Europe, 2020). In 2019, approximately 40% of plastics in the EU-28 were used for packaging, a significant portion of which was designated for food-related applications including single-use plastic products for food and beverage containers (Plastics Europe, 2021). Global greenhouse gas emissions from the production, use and End-of-Life (EoL) treatment of conventional plastic were estimated at 1.7 Gt CO_2-eq. Without changes in plastic use strategies, this figure is expected to increase to 6.5 Gt CO_2-eq by 2050 (Zheng and Suh, 2019).

Beyond their carbon footprint, plastics generated on land ultimately accumulate in the oceans, which act as the final sinks for this persistent pollution and pose a significant sustainability challenge. While the mechanism by which they reach the ocean could be different, the main pathway is through a transit of urban rivers network (Lebreton et al., 2017) (Meijer et al., 2021), facilitated by various means such as atmospheric inputs, population size and quality of waste management systems (Dris et al., 2016) (Jambeck et al., 2015) (Murphy et al., 2016). Numerous studies have reviewed the origin of plastic and their main pathways from land to marine ecosystems (Habumugisha et al., 2024) (Belli et al., 2024). There is a broad consensus that 70%–80% of ocean plastics originate from land-based sources, while 20%–30% come from marine sources (UNEP, 2022), with fisheries being a direct contributor (Lebreton, 2022). Other sources, like the atmospheric input, has also been reported (Liss, 2020) with a microplastic (MPs) residence time in the atmosphere ranging from minutes to days (Evangeliou et al., 2022).

Plastics are subject to degradation and continuous fragmentation due to factors such as oxidation, UV radiation, and biological effects (Andrady et al., 2015) (Zbyszewski et al., 2014) (Zettler et al., 2013). These degradation processes result in the continuous fragmentation of plastics, breaking larger macroplastics (MAPs) (>25 mm) into (MPs) (<5 mm) and eventually into nanoplastics (NPs) (<1 µm) (Gigault et al., 2021) (Zhao K. et al., 2022). The small sized fractions, i.e., the micro and nano sized plastics have been considered as emerging pollutants, threatening aquatic life and human health (Hamd et al., 2022) (Eerkes-Medrano et al., 2015). Microplastics, the most extensively studied of these small particles, appear in various shapes in the environment. They originate either as primary MPs, intentionally manufactured for products such as personal care items, fertilizers, paints, detergents, and cleaning agents, or as secondary MPs resulting from the breakdown of larger plastics (Dris et al., 2016) (Jambeck et al., 2015) (Andrady et al., 2015) (Uheida et al., 2021) (Dris et al., 2018; Cole et al., 2011). Despite the growing awareness of MPs as significant emergent pollutants, there remains a substantial knowledge gap regarding their specific sources, characteristics and complex factors influencing their behavior, and fate in the marine ecosystem. The quantities of plastic amount reaching the ocean has been estimated in several studies (Lebreton et al., 2017) (Jambeck et al., 2015) (Schmidt et al., 2017). Overall, estimates suggest a maximum value of 14 Mt/year, accounting for about 3% of all plastic production (Jambeck et al., 2015). Nevertheless, lack of standardization in research methodologies and findings strongly complicates the assessment of MPs abundance and distribution (Jolaosho et al., 2025).

Additionally, MPs have detrimental effects on natural ecosystems. It endangers marine species ranging from top predators to invertebrates as well as microorganisms, causing injuries, impaired mobility, and even death. Furthermore, when ingested, it causes internal injuries, malnutrition, and toxicity (Kühn et al., 2015) (Li et al., 2016) (Abouda et al., 2022) (Zitouni et al., 2021) (Missawi et al., 2021). Therefore, MPs pollution can negatively impact ecosystem structure and functions, thereby impairing ecosystem's ability to provide essential services to human societies. These services include provisioning services such as food supply (e.g., fish and shellfish) and raw materials (e.g., seaweed); regulating services like water purification and climate regulation (e.g., disruption of carbon sequestration); supporting services like habitat provision, biodiversity conservation, and nutrient cycles; and cultural services particularly recreation and education (Reid et al., 2005).

Building on the detrimental effects of MPs on marine life and ecosystems, their proliferation poses a significant threat particularly to valuable Blue Carbon Ecosystems, such as mangroves, seagrasses, and salt marshes (Yu H. et al., 2021) (Ogbuagu et al., 2022). These coastal and marine habitats play a crucial role in mitigating climate change as they capture and store significant amounts of carbon dioxide from the atmosphere (Macreadie et al., 2017) (Mcleod et al., 2011). The infiltration of MPs into these blue carbon environments disrupts their health and function (Lau et al., 2020) (Zhou et al., 2023) by affecting not only the organisms residing within these habitats but also their abilities to sequester carbon, ultimately reducing the ecosystems' capacity to provide essential services.

Despite the accumulated knowledge on MPs' effects on marine ecosystems, there is a lack of comprehensive analyses integrating and synthesizing current literature on the various impacts of MPs on both marine life and ecosystem services. This gap limits our understanding and assessment of the actual threats that MPs present to marine environments. Therefore, this review intends to fill these gaps by providing a holistic analysis of MPs impacts, with a focus on the losses that this pollution causes to marine ecosystem services. Furthermore, the review examines the socioeconomic and sociocultural implications of plastic pollution and provides an evaluation of existing policies and regulations with a sustainability perspective. It aims to consolidate existing knowledge and offer a new insight into the full spectrum of MPs pollution's impacts on environmental and socio-economic aspects, including the regulatory framework, by providing a detailed overview to enhance understanding and guide future environmental management and policy efforts.

2 Microplastics sources and pathways to marine environment

The main sources, fate, transport pathways and routes of MPs must be well explored and detailed to reasonably comprehend and prevent contamination in the marine environment. In this section, we investigate MPs' main origins and how they get into the marine environment. The fate and transport of plastics particles to the

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marine environment depend on several land-based sources, rivers inputs and marine and maritime activities (Lebreton and Borrero, 2013). The ratio of cities to-rivers-to-shipping lanes was established at 40:40:20% (Lebreton et al., 2012). This ratio could be adjusted to 50:30:20% (Liubartseva et al., 2018) depending on the city' capacities in terms of population density, level of urbanization and number of rivers.

2.1 Land-based sources

Land-based MPs sources are diverse and include landfills, wastewater solids and effluents, industrial facility losses, plastic agricultural mulch, polymer paints, and vehicle tire abrasion (Chae and An, 2018). Other sources have also been identified, such as MPs from washing textiles, burning plastics and atmospheric deposition (Hale et al., 2020). In most countries, plastics are dumped in landfills, either in closed or open-air facilities. Environmental problems arise when there are offsite losses or mismanaged waste including the release of plastics during transportation and disposal, as well as from municipal solid waste collection and processing. In addition, dumps installation close to coastal areas are subject to sea level rises, flooding and erosion, contributing to additional release of plastic debris in the environment (Hale et al., 2020).

Furthermore, synthetic fibers released during washing textiles have been identified as useful particles that could be used as markers of wastewater outfalls and land application of biosolids (Habib et al., 1998). Given the differences in sampling techniques and analytical methodologies, it has not been feasible to achieve consensus regarding the most prevalent types of MPs in the environment. However, the IUCN has identified synthetic textile releases from laundry as the leading source, contributing 35% of the ocean MPs load (Boucher and Friot, 2017). These MPs often escape treatment facilities in developing countries and may directly enter streams and reach the marine environment, where they undergo a long biodegradation process, indeed only 4% of polyester show biodegradation after 243 days of exposure (Hale et al., 2020).

Additionally, tire wear in vehicles is another source of MPs from terrestrial environments. Modern tires incorporate fillers such as carbon black, metallic fibers, additives, and polymeric materials combined with rubber, predominantly composed of butadiene and styrene-butadiene polymers. Together, these components serve as significant sources of secondary MPs. According to Kole et al. (2017), Americans generate on average approximately 4.7 kg of tire-wear MPs annually, which amounts to approximately 1.8 million metric tons. They posited that tire degradation may contribute 5%–10% of MPs entering global oceans. These particles infiltrate waterways via surface runoff, enhanced by the impermeability of road surfaces, or through entry into sewer systems, followed by further processing in wastewater treatment facilities (Hale et al., 2020); (Jolaosho et al., 2025). Sewage treatment facilities are major accumulators of MPs. It has been indicated that the annual input of MPs from sewage sludge to cropland in Europe and North America could exceed the overall concentration of MPs in the ocean surface water (Nizzetto et al., 2016). In sewage treatment, MPs cannot be completely removed. The removal rate can reach up to 90% after a complete treatment process (Pivokonský et al., 2020). However, depending on the treatment efficiency, some nano and/or micro-plastic residues remain discharged in wastewater effluents (Ziajahromi et al., 2017). Particularly, treatment may be less effective for MPs <100 µm. Murphy et al. (2016) noted that most of the buoyant MPs, including the majority of microbeads from personal care products, were entrained in the floating grease fraction. As consequence, the exclusion of oil and grease and primary sludge from land applied materials would thus reduce the quantity of MPs transferred to soils and this ending up into the aquatic ecosystem.

Microplastics particles can adhere to soil when irrigated with treated wastewaters, making the soil a sink for MPs that accumulate overtime (Boughattas et al., 2022) (Hattab et al., 2024a) (Zhang et al., 2022). They can move downward with percolating water and finally reach groundwater reservoirs or surface water bodies, contaminating the whole ecosystem (Dube and Okuthe, 2023). Soil-based water infiltration and cultivations practices were also identified as potential sources of MPs (Habumugisha et al., 2024), due to the use of mulching films in agricultural soils which are very common but they are difficult and expensive to recycle (Petersen and Hubbart, 2021). With less than 60% of plastic film being recovered (Zhaorong et al., 2020), residue from these films has become a major source of soil MPs (Boughattas et al., 2022), leading to the accumulation of plastic particles in or on the soil. Additionally, compost application, could also be a source of MPs to the soil (Zhang et al., 2022). It has been reported that inefficient waste and manure management practices can lead to plastic contamination of composts (Vithanage et al., 2021). MPs contaminated compost can adversely affect soil living organisms such as the earthworm *Eisenia andrei* causing cytotoxicity, genotoxicity, and neurotoxicity (Hattab et al., 2024b).

To date, little research has examined the atmospheric transport of MPs. The airborne MPs can be transported to ocean surface air and remote areas. Indeed, many airborne MPs (365 items m^{-2} day^{-1}) were recorded in the pristine mountainous area (French Pyrenees) (Allen et al., 2019). Brahney et al. (2020) show that even the most isolated areas in the United States (national parks and national wilderness areas) accumulate MPs particles after being transported by wind and rain. They estimate that more than 1,000 metric tons per year fall within south and central western U.S. protected areas. Most of these plastic particles are synthetic microfibers used for making clothes. Airborne MPs can be ingested and inhaled by humans, thus allowing MPs to enter the digestive and respiratory systems. Inhalation of MPs, especially through indoor air, contributes to higher human exposure to MPs compared to other routes of exposure imposing a potential health risk. According to Wang et al. (2019), there was a moderate increase in MPs atmospheric levels in Shanghai, where fibers accounted for 67% of the particles. As a result, the population was predicted to inhale about 21 MPs particles per day.

The investigations of the main potential sources of airborne MPs, indicate the degradation of large plastics, industrial emissions and dust resuspension (Dris et al., 2016) (Liu et al., 2019a). The use of *in situ* observations of MPs deposition combined with an atmospheric transport model to identify the most likely sources of atmospheric plastic over the land regions of the western United States, suggests that atmospheric MPs are primarily derived from secondary re-emission sources including roads (84%), ocean (11%), and agricultural soil dust (5%) (Brahney et al., 2021). Once in the atmosphere, suspended MPs are

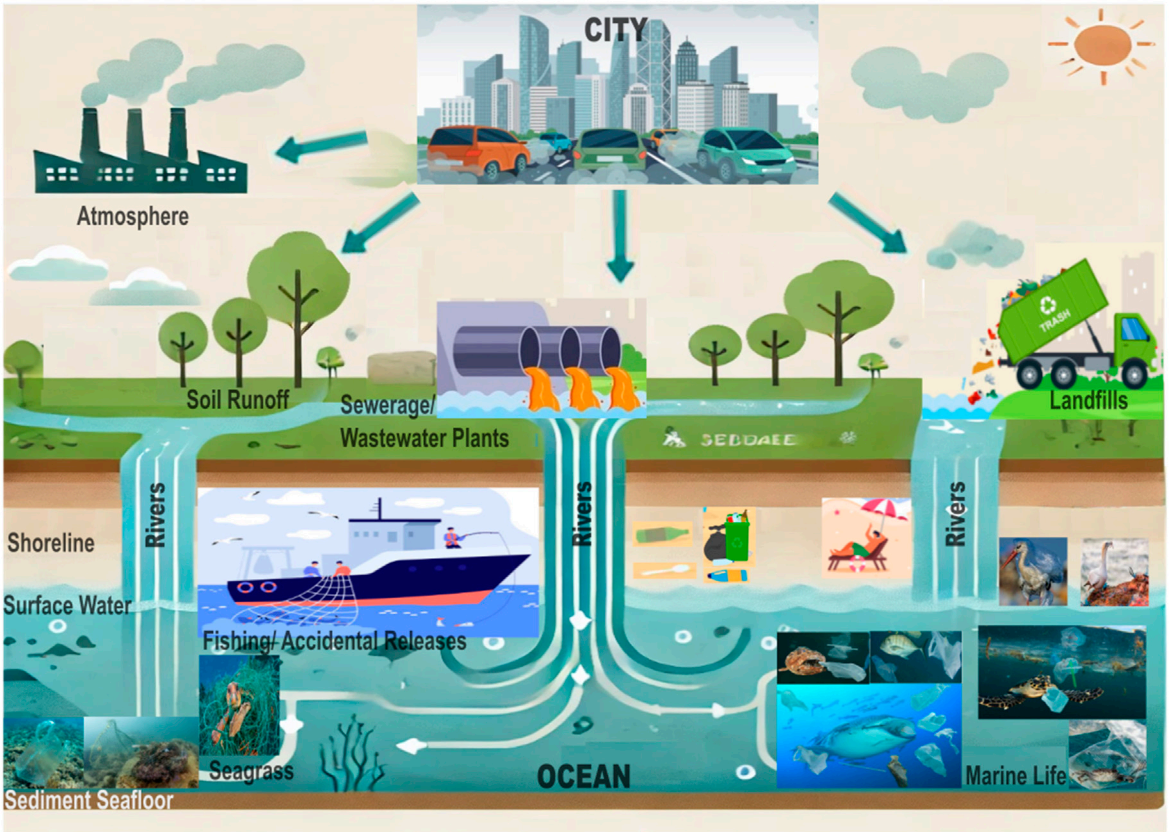

FIGURE 1
Diagram of plastic waste main sources and pathways to the ocean final sink.

transported passively by wind effects, their transmission distance has been approximately estimated to 95.0 Km in 5 months (Allen et al., 2019). Some oceanic areas can be highly impacted by suspended atmospheric MPs deposition such as the West Pacific Ocean (Liu et al., 2019b), however due to the lack of assessment initiatives, little is known about the quantities deposited in other oceanic regions.

2.2 Rivers input

Terrestrial inputs are assumed to enter the ocean primarily at coastal release points corresponding to major rivers as shown in Figure 1.

Surprisingly, research on plastic pollution in freshwater ecosystems, such as rivers, is far less common than that in marine ones. Particularly River sampling is especially inconsistent due to a lack of dedicated efforts, particularly in extremely polluted rivers such as those in Asia (Blettler et al., 2018). As a result, most studies have often used several approximations to estimate riverine input fluxes. For instance, Jambeck et al. (2015) and Lebreton et al. (2017), predominantly based their global estimates of plastic emission fluxes on the volume of mismanaged plastic waste and the likelihood of its entry into aquatic ecosystems (Lebreton et al., 2017) (Meijer et al., 2021). More recently, Meijer et al. (2021) considered both climate and geographical factors to calculate the probability for plastics entering rivers and oceans globally. According to their estimation, over 1,000 rivers generate between 0.8 and 2.7 million metric tons of plastics emission flux per year. Zhang et al. (2023) applied a top-down methodology to estimate the global discharge of plastic using ocean transport models and a dataset of sea surface plastic concentrations. Their results indicate a yearly plastic emission flow between 0.13 million to 3.8 million metric tons.

The majority of primary MPs in European rivers were from the industry sector, personal care items and cleaning products, whereas secondary MPs were fibers from synthetic textiles (Gao et al., 2024). In Asian rivers, a significant increase in MPs concentrations was pointed out near large cities, but not all studies were able to clearly link MPs concentration to factors such as population density or industrial activities (Lin et al., 2024). In this region of the world, there is a need to shift the focus from merely reporting MPs concentrations towards investigating the relationships between anthropogenic causes and MPs riverine concentrations to identify potential pollution sources.

2.3 Ocean-based sources

Commercial fishing and recreational, tourism, aquaculture and maritime activities, such as oil platforms, can all be direct causes of MPs pollution in the ocean, endangering both marine life and vegetation (Ziani et al., 2023).

Fishing gear used for direct capture contributes to local and regional marine pollution. Hence, damaged, undesired, or no longer usable plastic fishing gear (e.g., nets, lines, buoys) often end up in the ocean through accidental loss or disposal. This abandoned, lost, or otherwise discarded fishing gear is considered the major contributor to sea-based sources of marine plastic debris (UNEP, 2022), accounting for 10% of marine plastic waste and containing the highest proportion of MAPs and mega-plastics (>50 cm) floating in the ocean. This is in line with the findings of Galgani et al. (2015) affirming that fishing nets, as well as small unidentifiable pieces of plastic represent among the largest proportion of marine litter. Investigations into the origin of the debris based on label recognition in the North Pacific Ocean, which is considered as a hotspot of MPs accumulation, reveal that most of the objects come from major fishing nations (Lebreton et al., 2022). Additionally, untreated sewage dumping in areas more than 12 nautical miles from the nearest land, which is officially authorized by the International Maritime Organization, contributes to MPs enrichment, as previously reported for wastewater.

Extreme events like tsunamis can inject considerable amounts of debris into the oceans. For instance, the 2011 Great Japan Tsunami washed away an estimated five million tons of debris into the Pacific (Murray et al., 2018). This was more than 3,000 times the average annual amount of land-based litter contributed by all of Japan (Lebreton and Borrero, 2013). The bulk of this tsunami debris will eventually accumulate in the North Pacific Ocean subtropical gyre, increasing the concentration of debris in the so-called 'Great Pacific Garbage Patch', where some of which will breakdown into tiny particles and be consumed by marine organisms.

3 Microplastics characterization and distribution

3.1 Sampling identification and quantification

The accurate quantification of MPs prevalence is greatly hampered by the current methodological and technical limitations, as well as data reporting and analysis. Across the accumulated literature the most commonly used sampling methods are manta nets, neuston nets, plankton nets, and underway pump systems (Belli et al., 2024) (Mutuku et al., 2024). The mesh sizes generally ranged from 30 to 700 μm, with 330 μm being the most common among all the sampling methods (Jambeck et al., 2015) (Belli et al., 2024) (Mutuku et al., 2024). More than 80% of field studies only sampled MPs larger than 300 μm (Conke and Nascimento, 2018); therefore, MPs smaller than this size, including 95% of cosmetic microbeads, synthetic microfibers, and secondary MPs with diameters less than 300 μm, are absent from MPs datasets.

However, microplastics smaller than 300 μm are more easily absorbed by organisms. This bias could have implications on toxicity assessments since in the small micron range, MPs could penetrate body barriers and cell membranes, potentially inducing molecular perturbations (Jani et al., 1992).

Moreover, the complexity in comparing studies on MPs in surface water, is primarily attributed to the wide diversity of sampling methodologies employed, particularly in terms of mesh size and reported particle size (Strafella et al., 2022). Hale et al. (2020) have shown a massive distortion in the size distribution of MPs in surface water when comparing two datasets using different sampling techniques, i.e., pumped surface water through a 10 μm filter vs. thawing a 200 μm net. Furthermore, even after elimination of all technical sampling constraints, the MPs surface water distribution, which is justified historically and practically by a visual appreciation of the phenomenon, is challenged by the sample's representativeness, since most of the plastic pool in the ocean is either bio-fouled or sinking to the sea bottom. Indeed, 94% of plastic waste in marine environment is deposited on the seafloor, with only 1% is found on the ocean surface (Sherrington, 2016).

Numerous analytical methods have been employed for the quantification and identification of MPs in the environment (Lu et al., 2021). Early approaches often relied on optical microscopy. However, this technique cannot determine chemical composition and is therefore of limited use in differentiating effective synthetic polymers from sample interferences.

For polymer identification, Fourier transform infrared spectroscopy (FTIR) has been widely used and has significantly improved capabilities for MPs characterization. Currently, the FTIR combined with microscopy are predominantly utilized in marine environment studies (Belli et al., 2024) (Mutuku et al., 2024). However conventional FTIR is generally limited to particles larger than 10 μm, due to diffraction constraints. Raman micro spectroscopy, which allows a detection size of less than 1μm, is increasingly used (Schymanski et al., 2018). Some other cutting-edge techniques for MPs identification and characterization have also been used such as pyrolysis gas chromatography and liquid chromatography mass spectrometry (Schymanski et al., 2018). Generally, studies opted for a combination of techniques to broaden the spectrum of their findings, acknowledging the complementary strengths of each method (Da Costa et al., 2020).

In summary, although analytical approaches have rapidly advanced in the past decade, yet they still fail to fully meet the challenges presented by MPs (Lu et al., 2021). The main reported constraints are related to plastic particle size, namely, the nano-sized fraction as well as polymer types and chemical alterations.

3.2 Chemical composition

The major plastic polymers used in the manufacturing industry are polypropylene (PP), polyethylene terephthalate (PET), polyethylene (PE), polystyrene (PS), polyurethane (PUR), and polyvinyl chloride (PVC) (Lamichhane et al., 2023). These polymers constitute nearly 90% of all plastics produced worldwide. Globally, studies have identified PE, PP, and PS as being predominant in surface waters (Table 1) due to their lower densities of 0.90–0.97, 0.91–0.92, - 1.04 and 1.10 g/cm^3, respectively (Li Y. et al., 2024) (Jolaosho et al., 2025). A consistent prevalence of five polymers PE, PP, PET, PS, and Polyphthalamide (PPA) in comparable proportions across all oceans have been reported (Mutuku et al., 2024), with PP, PE and PS accounting for more than 67% of recovered polymers. The crystallinity of these plastic polymers enhances their rigidity by improving structural integrity and resistance to deformation. More specifically, the crystallinity of PE ranges from 60% to 70% justifying its used in the production of

TABLE 1 Overview of observed and modeled MPs counts in different marine ecosystems: Item density, plastics format and predominant chemical characteristics are indicated. The following abbreviations are used PE: Polyethylene; PET: polyethylene terephthalate; PP: Polypropylene; PA: Polyamide; PPA Polyphthalamide; PS: Polystyrene; PU: Polyurethane.

Location	Sampling method	Mesh size (μm)	Unit	Items density	Particle size (mm)	Format	Polymers	References
Atlantic Ocean								
Argentinean continental shelf	Manta net	350	items.m^{-3}	0.14 ± 0.08	<1	Fibers	-	Ronda et al. (2019)
Continental shelf off the south coast of Brazil	Manta net	330	items Km^{-2}	4,461 ± 3,914	-	Fragments	PA, PU	Ronda et al. (2019)
Amazon Continental Shelf, Brazil	Bucket and filtered by plankton net	64	items.m^{-3}	3,593 ± 2,264	<0.5	Fibers	PA, PU	Queiroz et al. (2022)
North Atlantic subtropical gyre	Manta net	335	items.m^{-3}	0.62 ± 0.52	1–4.74	Fragments	PE, PP, Acrylic	Courtene-Jones et al. (2022)
Mean Atlantic Ocean*			items.m^{-3}	4.98		Fragments	PE, PET, PP	Mutuku et al. (2024)
Arctic								
Northeast Greenland Sea	underway pump systems		items.m^{-3}	2.4	0.5–5	-	PP, PA, PE, PVC, Acrylic	Morgana et al. (2018)
Arctic polar water	Manta net		items Km^{-2}	28,000	<5	-		Lusher et al. (2015a)
Pacific Ocean								
Great Pacific Garbage Patch (North Pacific subtropical gyre)	Manta net	500	items Km^{-2}	678,000–2,400,000	0.5–5	-		Lebreton et al. (2018)
North Pacific subtropical gyre	Plankton net		items Km^{-2}	10^5–10^7	<5	-		Van Sebille et al. (2015)
Northeastern Pacific Ocean	underway pump systems		items.m^{-3}	8–9,200	0-1-1	Fiber/filament, fragment, pellets and thin films	-	Desforges et al. (2014)
Northwestern Pacific Ocean	Manta net	330	items Km^{-2}	6.2 × 10^4	<1	Granule, Sheet, film and line	PE, PP, PA, PVC, PS	Pan et al. (2019)
Tropical North Pacific	Bongo zooplankton net	200	items.m^{-3}	<0.018	0.2–0.35		PE, PET, PP, PVC, Nylon	Yuan et al. (2023)
Mid-West Pacific	Manta net	333	items Km^{-2}	6,028-98,335	0.3–2.5	Fiber/filament, fragment, film granule	PP, PE, PS, PET	Xiong et al. (2025)
Mean Pacific Ocean*			items.m^{-3}	1.49		Fragments	PPA, PS, PET	Mutuku et al. (2024)
Indian Ocean								
Reunion Island (southwest part of the Indian Ocean)	Manta net	500	items Km^{-2}	4,025 ± 4,760 (East) and 10,693 ± 11,275 (West)		-	PE, PP	Sababadichetty et al. (2024)
North Indian Ocean (Bay of Bengal and Arabian sea)	Manta net	330	items Km^{-2}	15,200 -72,381	0.5–5	Fibers	PE, PP	Janakiram et al. (2023)
Eastern Arabian Sea	Bango net	333	items.m^{-3}	0.002–0.046	0.5–5	Fibers, Fragments, Film	PP, LDEP, NY	Naidu et al. (2021)
Indian Ocean	Plankton net		items Km^{-2}	10^2–10^5	-			Van Sebille et al. (2015)

(Continued on following page)

TABLE 1 (*Continued*) Overview of observed and modeled MPs counts in different marine ecosystems: Item density, plastics format and predominant chemical characteristics are indicated. The following abbreviations are used PE: Polyethylene; PET: polyethylene terephthalate; PP: Polypropylene; PA: Polyamide; PPA Polyphthalamide; PS: Polystyrene; PU: Polyurethane.

Location	Sampling method	Mesh size (µm)	Unit	Items density	Particle size (mm)	Format	Polymers	References
Tropical Indian Ocean	underway pump systems		items.m^{-3}	8–132	-	Fibers	Acrylates/PU	Hildebrandt et al. (2022)
Mean Indian Ocean*			items.m^{-3}	3.17	-	Fragments	PP, PPA, PE, PS	Mutuku et al. (2024)
Mediterranean Sea								
Gulf of Gabes	Manta Net	200	items Km^{-2}	312,887 and 77,110	<1-3	Fragments	PE, PP	Ben et al. (2022)
Gulf of Lion	Manta Net	780	items Km^{-2}	6×10^3–1×10^6	1.48 ± 0.88			Schmidt et al. (2018)
W Mediterranean	Manta Net	335	items.m^{-3}	3.52 ± 8.81	-	Fragments	PE	Fagiano et al. (2022)
Central W Mediterranean	Neuston Net	200	g Km^{-2}	6.72 ± 1.5×10^4	0.2–0.5	Fragments	PE, PP	Suaria et al. (2016)
Southern Mediterranean/Bizerte lagoon	Niskin bottles		items.m^{-3}	453 ± 335	-	Fibers	PE, PP	Wakkaf et al. (2020)
Medium Mediterranean Sea*			items Km^{-2}	8.48 10^4				Simon-Sánchez et al. (2022)

food wraps, vehicle fuel tanks, and industrial pipes (Hadiyanto et al., 2021). The chemical composition of PP and PE made of polyolefins with considerably long linear hydrocarbon chains can take decades to centuries to naturally disintegrate. In addition, their chemical and physical properties, such as resistance to heat, weather, fatigue, durability, and toughness, all contribute to their limited degradability (Jolaosho et al., 2025). Consequently, this plastic pollution will persist in the environment giving the slow disintegration processes. Polymers with lower density than seawater (mostly as PP and PE) will remain floating in the surface water, carried out by the surface currents and accumulated in central zones, particular gyres and convergent zones (Van Sebille et al., 2020). The polymers with high densities compared to ocean water such as PET, PVC and PC, once introduced into the marine ecosystem will directly sink to the ocean seafloor (Engler, 2012) thus become less accessible and almost impossible to eliminate suggesting that physical elimination of MPs from the marine environment is both technically and economically not feasible.

3.3 Prevalence and distribution

A comparison of MP concentration, prevalence and distribution was carried out in various marine ecosystems (Table 1). It is worth noting the disparities in sampling techniques between the multiple studies, namely, the differences in net mesh sizes and reported units. Thus, the absolute volumes of plastic debris across different marine environments remain largely underestimated or unknown due to a lack of standardization and homogenization efforts to address the discrepancies in the present appreciation of MPs spatial distribution.

In addition, most studied collected a one-time sample from the ocean waters, and since the surface distribution of MPs is highly influenced by the ocean circulation and the atmospheric forcings (Courtene-Jones et al., 2022), this may constrain the results reproducibility and comparability (Cowger et al., 2020). Nevertheless, through the reviewed literature, some technical compromises have been proposed to homogenize the MPs concentration units, such as converting the MPs concentration unit from items/Km^2 to items/m^3 when the net height and submersion depth of the sampling net mouth were provided, and the net had a rectangular shape (Lu et al., 2021) (Mutuku et al., 2024). Li C. et al. (2020) proposed that the varying concentrations of MPs can be converted into uniform results with similar units by exploring the size and number of MPs particles per volume (estimated average of plastic particles of 1 g/mL).

A standardized dataset established to compare the abundance and distribution of MPs in surface water across different oceans waters for the period 2010 and 2023 (Mutuku et al., 2024), revealed that mean MPs concentrations ranged from 0.002 to 62.50 items m^{-3}, with an average abundance of 2.76 items/m^3. The Atlantic Ocean, with a mean concentration of 4.98 items m^{-3} (Table 1), exhibited the highest average MPs concentration, followed by the Indian and the Pacific Ocean, while the lowest concentration was recorded in the Southern Ocean (0.04 items m^{-3}). A different pattern was reported for the period between 1971 and 2013 (Van Sebille et al., 2015), based on a separate standardized dataset coupled with statistical modeling. The Pacific Ocean showed the highest accumulated number of MPs particles, followed by the Indian and the Atlantic Oceans. These discrepancies, illustrating difficulties in conducting comparatives studies, may be explained by the substantial increase in the number of observations over the last decade, mainly in the south Atlantic (Belli et al., 2024), as well as by differences in the standardization procedures, which corrected variability associated with factors affecting both plastic concentration and samples representativeness, such as sampling year and wind speed (Van Sebille et al., 2015). The Pacific Ocean, where several studies on the spatial distribution of MPs have been conducted in various regions of this vast oceanic ecosystem (Table1), has revealed differences in the spatial distribution of MPs. The Great Pacific Garbage Patch, located in the North Pacific subtropical gyre, is considered one of the densest accumulation area worldwide, likely due to its vast area covering 1.6 million Km^2, with concentrations of millimeter-sized plastic debris frequently reaching up to 10^6 pieces Km^{-2} (Lebreton, 2022), and also the large inputs of plastic waste from the coastlines of Asia and the United States (Jambeck et al., 2015). This is higher than the Mid-West Pacific, where concentrations of less than 10^5 items Km^{-2} have been reported (Wang et al., 2020). Few studies on MPs distribution have been conducted in the Indian Ocean, mainly around the Arabian Seas, with concentrations in some regions reaching up to 10^6 particles Km^{-2} (Abayomi et al., 2017). The most common forms of plastics are fibers, fragments and films (Table 1). In the Atlantic Ocean, the numerical quantity of plastic fragments was significantly higher in the North Atlantic gyre than in the open ocean or inshore areas, which can be explained by the hydrodynamics characteristics of these features which can retain debris for extended periods (Cózar et al., 2014) (Courtene-Jones et al., 2022). In the southern Atlantic, a recent review (Belli et al., 2024) pointed out that the most polluted area in the region is the Bahía Blanca Estuary in Argentina, where MPs concentrations ranged from 5,900 to 782,000 items. m^{-3} (Fernández Severini et al., 2019), a level comparable to those found in more polluted oceanic areas (Meijer et al., 2021). As summarized in Table 1, fragments and fibers are the most common morphologies, whereas PE, PP and PA are the most abundant polymers.

The Mediterranean Sea has also been identified as one of the major accumulation zones of marine plastic waste, largely due to its semi-enclosed geography, limited water circulation, and the continuous influx of plastic waste from urban and industrial activities in the surrounding regions (Van Sebille et al., 2015). In this region, plastic concentrations in sea surface waters exhibit extremely high spatial and temporal variability, as confirmed by many field surveys (Gajšt et al., 2016) (Schmidt et al., 2017) (Suaria and Aliani, 2014). High variability in MPs levels in sea surface waters has been reported, with the Northwestern Mediterranean Sea exhibiting the lowest concentration (6.25×10^3 items Km^{-2}) (Collignon et al., 2014). In contrast, the highest values were found in the Levantine Basin, with coastal waters in Lebanon measuring up to 2.2410^6 items Km^{-2} (Kazour et al., 2019) and Turkey measuring 1.15 10^6 items Km^{-2} (Gündoğdu et al., 2018). According to a dataset compiled by (Simon-Sánchez et al., 2022), the median concentration of MPs in the surface waters of the Mediterranean Sea is approximately 8.48 10^4 items Km^{-2}, and the main long-term accumulation of plastic debris in the Mediterranean occurs along the coastlines and on the sea bottom. The most common forms of plastics are fragments and fibers, with PP and PE are the prevailing polymers (Table 1).

Despite the preconceived idea that Polar Regions are exempt from plastic pollution, Zarfi and Matthies (2010) have reported that between 6.2×10^4 and 1.05×10^5 tons of plastics flow annually into the Arctic Ocean. The evidence of plastics ingestion by several Arctic Seabirds (Baak et al., 2020), along with the considerable quantities of MPs found in the deep sea floor and in organisms at low trophic level (e.g., zooplankton) (Bergmann and Klages, 2012), suggests that MPs are transported to the Artic region via oceanic and/or atmospheric actions. Recent investigations have demonstrated that MPs are present in several regions of the Artic, both in the surface water and on the seafloor (Tekman et al., 2020), and their concentrations are even higher than previously reported.

3.4 Horizontal and vertical transport

The proper identification and parameterization schemes for MPs behaviors and factors influencing their transport are essential for a better understanding of MPs distribution, underlying quantifying methods (Kukulka et al., 2012) and guiding numerical modeling research of marine MPs (Cai et al., 2023).

Buoyant plastic debris with density lower than the ocean, is subjected to a wide range of physical and biological transport processes (Figure 2). The main physical forces driving the movement of MPs identified across the literature are wind-driven waves, tides, and currents (Van Sebille et al., 2020).

At large scale, circulation is driven by surface winds generating so-called Ekman drift. A sea surface under the influence of the Earth's rotation, the Ekman transport creates areas of convergences where floating plastic items will accumulate on a large scale, corresponding to the subtropical gyres (Law and Thompson, 2014). Buoyant debris, lacking sustainable degradability, can become trapped and circulate for years in subtropical oceanic gyres, contributing to persistent marine pollution. Five main garbage patches have been identified in the North Pacific, North Atlantic, South Pacific, South Atlantic and South Indian Oceans, yielding estimations of plastic mass accumulating at the ocean's surface (Van Sebille et al., 2015).

The velocity of the wind is highly related to the quantity of plastic materials recovered from the sea surface, because the wind mixing influences the vertical distribution of plastics. Based on surface and subsurface observations and a one-dimensional column model, Kukulka et al. (2012) demonstrated that plastic concentrations measured using surface tow measurements depend on wind speed because plastic pieces are vertically distributed in the mixed layer due to wind-induced mixing, leading to the conclusion that surface tow measurements significantly underestimate the total plastic content even for moderate wind conditions. Furthermore, the vertical mixing in the ocean is induced by several processes acting at different temporal and spatial scales. It can exist as coherent structures such as upwelling and down welling, fronts and turbulence-induced structures (Van Sebille et al., 2020). Particularly, the vertical turbulent mixing in the water column under the influence of the winds and the resulting wave action, are shown to be responsible of plastics debris resuspension from the seafloor (Cai et al., 2023).

Tidal currents play a crucial role in MPs redistribution within the continental shelf systems, generating turbulence near the bottom (Trowbridge and Lentz, 2018). The turbulent flows caused by tides or waves are primarily responsible for benthic particle resuspension (Li W. et al., 2022), influencing the plastic particle positioning on the

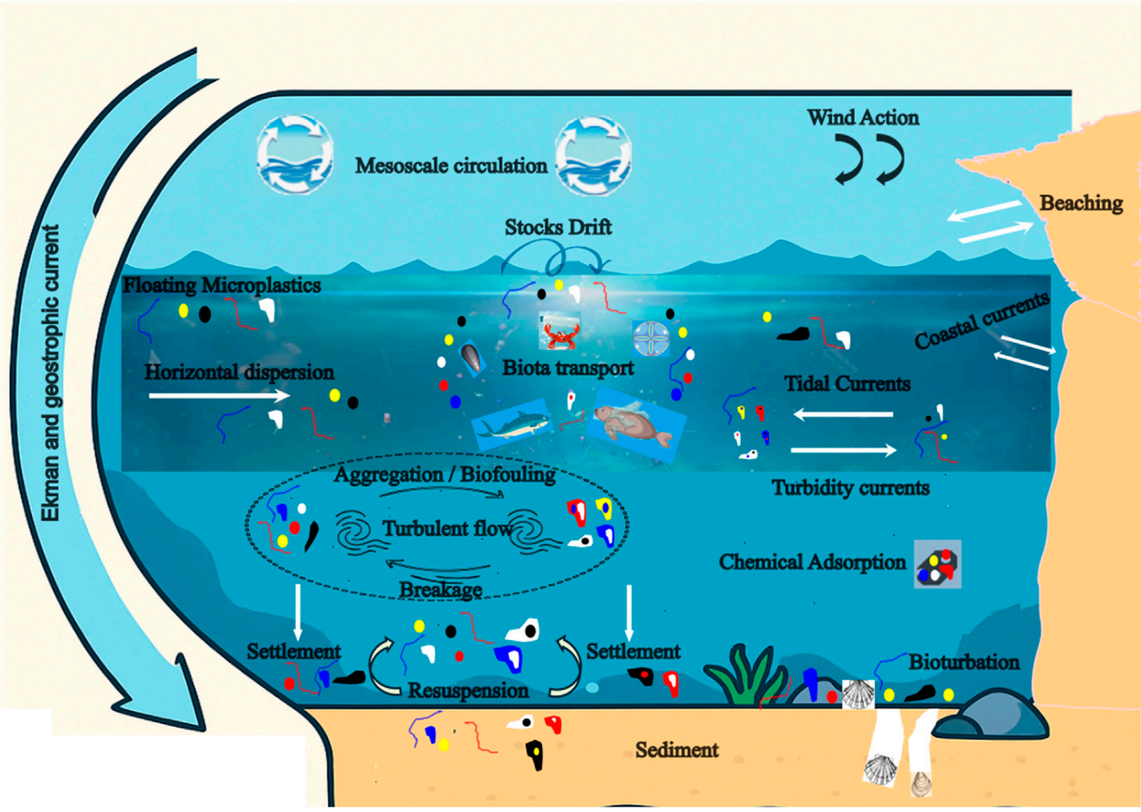

FIGURE 2
Factors influencing microplastics transport in the marine environment.

seafloor. In estuaries, tides and density fields also interact in complex ways, resulting in converging fronts or particle trapping (MacCready and Geyer, 2010), which frequently create high-concentration zones with strong tidal flows.

The fate of buoyant plastic debris in the ocean, is largely dominated by beaching onto coastlines, which removes a large fraction of floating plastic from the ocean surface (Lebreton and Andrady, 2019) (Isobe and Iwasaki, 2022). Plastic debris can also undergo changes in buoyancy due to biofouling through colonization by microorganisms such as bacteria, algae, and small invertebrates. Adhering to MPs surface, they significantly influence their transport and dispersion in aquatic environments (Anwaruzzaman et al., 2022) (Carlotti et al., 2023) (Bandini et al., 2021). MPs can also aggregate detrital materials and organic matter and concentrate in the densest planktonic layers of the water column, close to the chlorophyll maximum, thus impacting MPs settlement and accumulation (Carlotti et al., 2023). Additionally, MPs can adsorb a variety of pollutants that adhere to the surface of MPs particles through diverse processes such as ion exchange and electrostatic attraction (Yu Y. et al., 2021).

Plastic debris with a density higher than that of seawater sinks toward the seabed, considered a major sink for MPs (Woodall et al., 2014), where it can subsequently be redistributed horizontally by deep-sea circulation (Kane et al., 2020). Small MPs can also form heterogeneous aggregates favoring their settlement, owing to turbulent shear flow acting on sediments, they can resuspend thus contributing to a secondary pollution (Li W. et al., 2022). The bioturbation caused by marine organisms such as the clam behaviors, including burrowing, movement, and ingestion can provoke rapid transport of MPs to deeper layers (six to eight cm below the surface sediment) (Zhang et al., 2025).

In summary, a comprehensive and sufficient understanding of oceanographic processes and geophysical activities in surface waters is essential to apprehend aggregate abundances or concentrations of plastic in the ocean ecosystem. In addition, the MPs characteristics could be modified during the transportation processes such as transformation from micro to nano sized particles and the subsequent degradation. Therefore, extensive modeling research is needed to assess long-term changes in the transport and distribution dynamics of these particles.

4 Effects of plastic debris on organisms' biodiversity and functions

Plastics contaminate marine ecosystems across multiple trophic levels, impacting organisms ranging from phytoplankton (Hitchcock, 2022), zooplankton (Cole et al., 2013) (Alfonso et al., 2023), bivalves (Van Cauwenberghe and Janssen, 2014) (Chen et al., 2023), fish (Li J. et al., 2021), turtles (Thibault et al., 2023), marine mammals (Panti et al., 2019); and, ultimately, humans (Waring et al., 2018). Due to their ubiquity and small dimensions, MPs are easily ingested by many marine biota and transferred through food chain, which may result in bioaccumulation, referring to the

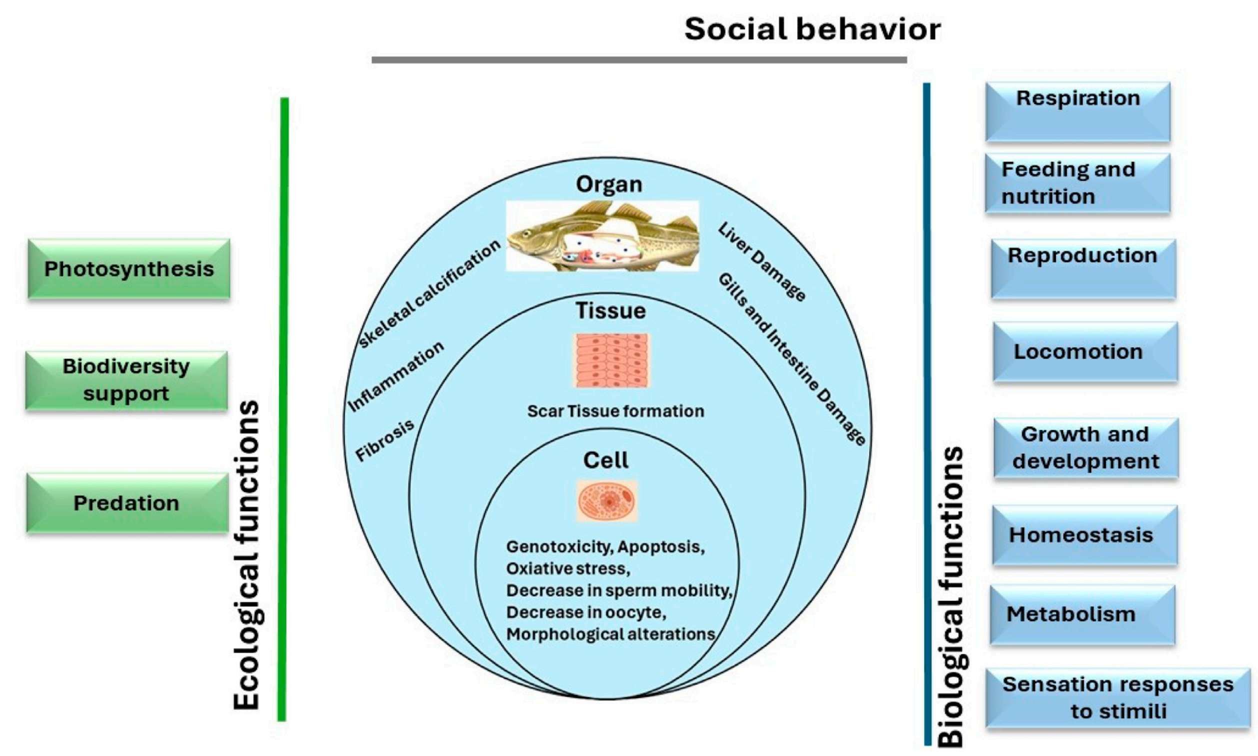

FIGURE 3
Microplastics impacts on different structural levels from cellular, tissue, organ and organ systems and the most affected functions.

accumulation of plastic particles in an organism over time (Thompson et al., 2015), and biomagnification, referring to their increase at every level of the food web (Mouat et al., 2010), leading to increased exposure of organisms at the top of the food web until they reach humans. This transfer has been experimentally demonstrated from lower to higher trophic levels with the possibility of accumulation in predators, raising concerns about the ecological and health impacts of these plastics, especially once they reach human diet via seafood intake (Farrell and Nelson, 2013) (Nelms et al., 2018) (Zhang et al., 2019).

Plastics affect both unicellular and multicellular marine organisms at multiple structural levels, including cellular, tissue, organ and organ systems, causing therefore profound disruptions like cellular alterations, tissue lesions, organ inflammation, and even physical injuries (Figure 3). In addition, plastics also disrupt the immune systems and gut microbiota. These disruptions have negative effects on biological functions like respiration, ingestion, reproduction, locomotion, and growth, ecological functions like photosynthesis and predation, and even on the scholastic behavior of these organisms. (Figure 3). In marine ecosystems, the distribution and composition of microbial communities have also been impacted by MPs, which has an effect on human and marine fauna health as well as ecosystem resilience and function. Assessing the impacts of plastics and identifying the extent of bioaccumulation and biomagnification at each stage of the food chain link up to humans will shed light on how these particles may impact humans, animals and ecosystems. Table 2 gives a summary of how plastic affects different marine organisms that are impacted by plastic contamination.

4.1 Marine organisms

The small size of MPs makes them easily mistaken for prey, leading to ingestion either through passive water filtration or feeding activity (Luís et al., 2015) (FDA, 2020). Therefore, for a number of marine species (Zitouni et al., 2021) (De Sá et al., 2015), including zooplankton (Cole et al., 2013) (Manríquez-Guzmán et al., 2023) (Malinowski et al., 2023), barnacles (Goldstein and Goodwin, 2013), bivalves such as mussels and oysters (Romdhani et al., 2022) (Cole and Galloway, 2015), as well as bigger organisms like pelagic fish (Lusher et al., 2013) and whales (Besseling et al., 2014) (Lusher et al., 2015b), ingestion is considered the main route of exposure to MPs. This widespread ingestion highlights the profound effect that MPs have on marine life. By finding different polymer particles in the gastrointestinal system and in the tissues of marine species, including fish (Pappoe et al., 2022), sea worms (Missawi et al., 2020), mussels (Romdhani et al., 2022) (González-Soto et al., 2019) (Romdhani et al., 2024), and seabirds (Fackelmann et al., 2023), the ingestion of MPs has been confirmed. According to (Santos et al., 2021), 1,288 marine species have been found to consume plastics. Approximately 400 species of fish, 54% of which are commercially relevant, ingest MPs ranging in size from 1μm to 5 mm (Djekoun et al., 2024) (Savoca et al., 2021). Furthermore, it has been reported that at least fifty cetaceans' species (56% of the infraorder) have consumed marine litter (Fossi et al., 2018) (Pereira et al., 2023). MPs have also been detected in various shellfish species, raising concerns about human health implications through dietary exposure (Li J. et al., 2021). They have been found in diverse

TABLE 2 Effects of plastics on marine organisms following a trophic gradient from primary producers to the top of the trophic chain. The following abbreviations were used: PVC: Polyvinyl chloride; PE: Polyethylene; LDPE: Low-Density Polyethylene; HDPE: High-Density Polyethylene; PP: Polypropylene; PA: Polyamide; PAC: Polyacetylene; PS: Polystyrene; UPVC Unplasticized polyvinylchloride; PMMA: Polymethylmethacrylate; PET: Polyethylene-terephtalate; PBAT: Polybutylene Adipate Terephthalate; BPA: Bisphenol A.

Target organisms	Plastic particles concentration	Plastic particles type/size	Effects	References
Cyanobacteria				
Microcystis aeruginosa	10–100 mg L^{-1}	PA, PE and PVC	Impairing Chlorophyll-*a*, photosynthetic activity, and growth rate	Kiki et al. (2023)
Limnospira (Arthrospira) maxima	5–80 mg L^{-1}	PET (4.7µm ± 0.5 µm)	Cell damage and an increase in carbon and nitrogen content	Pencik et al. (2023)
Algae				
Chlorella vulgaris	10–100 mg L^{-1}	PA, PVC and PE	Impairing Chlorophyll-*a*, photosynthetic activity and growth rate	Kiki et al. (2023)
	5–80 mg L^{-1}	PET (4.7 µm ± 0.5 µm)	Cell damage, changes in chlorophyll *a* composition and inhibitory effect on growth	Pencik et al. (2023)
Chlamydomonas reinhardtii	5–80 mg L^{-1}	PET (4.7 µm ± 0.5 µm)	Cell damage, changes in chlorophyll *a* composition and inhibitory effect on growth	Pencik et al. (2023)
Chlorella sp.	1.8–6.5 mg L^{-1}	PS beads (20 nm and 2.5 × 10^6 cm^2/g)	Decrease in photosynthetic activity	Bhattacharya et al. (2010)
Skeletonema costatum	50 mg L^{-1}	PVC (average diameter 1 µm)	Inhibition of maximum growth ratio (IR) Negative effects on algal photosynthesis (chlorophyll content and photosynthetic efficiency (ΦPSII)	Zhang et al. (2017)
Microalgae		MPs < 5 mm	Inhibition of growth, decrease in nutritional availability, decrease in chlorophyll and photosynthesis activity. Induction of oxidative stress, changes in morphology, reduction and promotion of hetero aggregates	Prata et al. (2019)
Phytoplankton		High MPs concentration	Significant changes in the phytoplankton community structure	Hitchcock (2022)
Scenedesmus obliquus	1 g L^{-1}	PS beads (70 nm)	Reduction of population growth and chlorophyll concentrations	Besseling et al. (2014)
Thalassiosira pseudonana (CCMP 1335) *Skeletonema grethae* (CCMP775) *Phaeodactylum tricornutum* (UTEX646) *Dunaliella tertiolecta* (UTEX999)	0–250 mg L^{-1}	PS NPs- and MPs (55 nm nanoparticles; 1 and 6 µm microparticles)	Inhibition of growth and induced production of exopolymeric substances with high protein-to-carbohydrate ratios	Shiu et al. (2020)
Cnidarians/Corals				
P. cf. damicornis	2.28 ± 0.12 particles g^{-1}	Nylon, PAC (101–200 µm)	Shift the coral-reef community assemblages and affect resilience	Jandang et al. (2024)
P. lutea	1.58 ± 0.25 particles g^{-1}			
Lobophyllia sp.	0.70 ± 0.12 particles g^{-1}			
P. sinensis	1.12 ± 0.25 particles g^{-1}			
Lophelia pertusa	350 spheres L^{-1}	PE beads (500 µm)	Reduction of skeletal growth rates and of septal growth	Mouchi et al. (2019) Chapron et al. (2018)
Pseudodiploria clivosa	10 mg L^{-1} per size class	PE (212–250 µm, 425–500 µm, and 850–1,000 µm)	Reduction in growth rate, impaired skeletal calcification, reduction in tissue surface area	Hankins et al. (2021)
Acropora cervicornis				

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TABLE 2 (*Continued*) Effects of plastics on marine organisms following a trophic gradient from primary producers to the top of the trophic chain. The following abbreviations were used: PVC: Polyvinyl chloride; PE: Polyethylene; LDPE: Low-Density Polyethylene; HDPE: High-Density Polyethylene; PP: Polypropylene; PA: Polyamide; PAC: Polyacetylene; PS: Polystyrene; UPVC Unplasticized polyvinylchloride; PMMA: Polymethylmethacrylate; PET: Polyethylene-terephtalate; PBAT: Polybutylene Adipate Terephthalate; BPA: Bisphenol A.

Target organisms	Plastic particles concentration	Plastic particles type/size	Effects	References
Crustacea				
Eurytemora affinis	300 µg L^{-1}	LDPE, PBAT (1–10 µm [Az1])	Dysbiosis of gut microbiota	Thery et al. (2023)
Artemia parthenogenetica	100 mg L^{-1} (1.10 and 1.26 10^6 items m^{-3})	PE, PS	Diversity increases and dysbiosis of gut microbiota, and reduction of the growth rate [Az2]	Li et al. (2021c)
Litopenaeus vannamei	10 and 20 mg mL^{-1}	PS (75 nm)	Reduction in intestinal fold height, intestinal structural damage, dysbiosis of gut microbiota, oxidative stress and metabolic disorders	Zhu et al. (2024)
Zooplankton				
Centropages typicus	>4,000 mL^{-1}	PS beads (7.3 µm)	Decrease in algal ingestion rate	Cole et al. (2013)
Acartia clausi		PS beads (15.7 µm ESD)	Decrease in algal ingestion rate	Ayukai (1987)
Calanus pacificus		PS beads (15 µm)	Decrease in algal ingestion rate	Fernández (1979)
Daphnia magna	0.22–103 mg nano-PS L^{-1}	Nano-PS aged, Pristine nano-PS	Reduction of body size and severe alterations in reproduction and neonatal malformations	Besseling et al. (2014)
Calanus helgolandicus	75 mL^{-1}	PS beads (20 µm)	Decrease in ingestion, fecundity and survival	Cole et al. (2015)
Bivalves				
Cerastoderma glaucum and	0.1% and 0.5% sediment dwt	PE microspheres (63–75 µm, 150–180 µm and 250–300 µm)	Decrease in emergence frequency of near-surface-dwelling	Urban-Malinga et al. (2021)
Limecola balthica				
Crassostrea gigas	0.023 mg L^{-1}	PS microspheres (2 and 6 µm)	Decrease in sperm motility, oocyte numbers (fecundity) and size (energetic investment per oocyte), larval yield and growth	Sussarellu et al. (2016)
Mytilus galloprovincialis		PE	Genotoxicity, tissue damage of gills and Digestive system	Bråte et al. (2018)
Mytilus edulis	0.2 mg L^{-1} (~1,170 MPs mL^{-1}) and 20 mg L^{-1} (~117,000 MP mL^{-1})	HDPE (4–6 µm and 20–25 µm)	Alteration of gut microbiota and an increase in the abundance of potential human pathogens	Li et al. (2020b)
Worms				
Arenicola marina	0%–5% by weight (sediment)	UPVC (130 µm)	Decrease in feeding activity and energy reserves and increase of phagocytic activity of immune cells	Wright et al. (2013)
Hediste diversicolor	10 and 50 mg kg^{-1} sediment	MPs (<30 µm)	Increase of acidic mucus production in seaworm tissues	Abouda et al. (2024)
Namalycastis jaya		PS MPs	DNA and oxidative damages	Saikumar et al. (2024)
Sea Urchins				
Paracentrotus lividus		PE, PP	Impaired feeding, reduced reproductive success, and physical damage to internal organs	Galloway et al. (2017)
Arbacia lixula	26 µg L^{-1}	PE MPs	Negative effects on physiology and histology. Decrease in the viabilities of coelomocyte subpopulations	Şahin et al. (2024)
Paracentrotus lividus	10, 50, 10^3, 10^4 L^{-1}	PS microbeads	Affecting immune cell proteome in a concentration-dependent response, alterations in minor morphological immune cell types, severe alterations of metabolism and cellular processes	Murano et al. (2023)

(Continued on following page)

TABLE 2 (*Continued*) Effects of plastics on marine organisms following a trophic gradient from primary producers to the top of the trophic chain. The following abbreviations were used: PVC: Polyvinyl chloride; PE: Polyethylene; LDPE: Low-Density Polyethylene; HDPE: High-Density Polyethylene; PP: Polypropylene; PA: Polyamide; PAC: Polyacetylene; PS: Polystyrene; UPVC Unplasticized polyvinylchloride; PMMA: Polymethylmethacrylate; PET: Polyethylene-terephtalate; PBAT: Polybutylene Adipate Terephthalate; BPA: Bisphenol A.

Target organisms	Plastic particles concentration	Plastic particles type/size	Effects	References
Fish				
Dicentrarchus labrax		PMMA (45 nm), PVC	Decrease in esterase and alkaline phosphatases levels in blood plasma and skin mucus respectively. Increase expression of genes and receptors involved in lipid metabolism. Severe histological changes in distal intestine	(Brandts et al., 2018) (Pedà et al., 2016)
Danio rerio (zebrafish)	100 and 1,000 µg L^{-1}	PS, PP (≤12 µm)	Decrease in larvae swimming competence, survival and hatching rates. Upregulation of inflammation and oxidative stress related genes and inhibition or increasing of acetylcholinesterase activity. Apoptosis in blood cells and liver oxidative damage	(Pedà et al., 2016) (Santos et al., 2021) (Priyadharshini et al., 2024)
Oryzias melastigma	10 mg L^{-1}	PS (2 µm, 10 µm, 200 µm)	Diversity alteration and dysbiosis of gut microbiota, hepatic inflammation and little fibrosis, and lipid metabolic disorders	Zhang et al. (2021b)
Oryzias javanicus	100, 500 and 1,000 µg L^{-1}	PS (0.5 µm)	Decrease in gut microbial richness and diversity and metabolic disorder	Usman et al. (2022)
Oryzias latipes	0.1 mg L^{-1} (2.5 10^7 particles L^{-1})	PS (2 µm)	Dysbiosis of gut microbiota, reduction of the functional bacterial species, reduction of the shoaling behavior	Tamura et al. (2024)
Marine Mammals				
Marine mammals		65.5–436 µm in the acoustic fat pads	Inhibition of acoustic fat pads critical functions	Merrill et al. (2023)
Marine mammals		24.4–1,387 µm in Blubber tissue	Inhibition of blubber critical functions	Merrill et al. (2023)
Balaenoptera musculus (Blue whale)		PE, PP (1 mm to several cm)	Digestive blockages, exposure to toxins, and impact on prey species	Desforges et al. (2015)
Sea Turtles				
Chelonia mydas		PE, PP (1 mm to several cm)	Digestive blockages, malnutrition, decrease in growth rates, and increase of mortality	Wright et al. (2013)
Caretta caretta		BPA, PTA, PET and PC in abdominal fat and liver tissues (0.45 µm - 1 mm)	Gastrointestinal impairment and an important level of contamination in tissues	Di Renzo et al. (2021)
Seabirds				
Fulmarus glacialis		PE, PP (1 mm to several cm)	False sense of satiation, malnutrition, physical injury, and exposure to toxic chemicals	Galloway et al. (2017)
Ardenna carneipes		MPs and MAPs	Extensive scar tissue formation, fibrosis in seabird stomach tissues, *"Plasticosis"*	Charlton-Howard et al. (2023)
Calonectris borealis	0–57 pieces, 0–0.1016 g	MPs	Alteration of the composition of the gastrointestinal tract (GIT) and increase in microbial diversity	Fackelmann et al. (2023)
Fulmarus glacialis	0–57 pieces, 0–0.1016 g	MPs	Decrease in commensal microbiota, and increase in pathogens and antibiotic-resistant and plastic-degrading microbes of the GIT	Fackelmann et al. (2023)

bivalve species (Djekoun et al., 2024) (Cho et al., 2019) (Hermabessiere et al., 2019) (Baechler et al., 2020) (Aung et al., 2022), such as mussels, especially *Mytilus edulis*.; and the common cockle *Cerastoderma edule* (Hermabessiere et al., 2019) (Botelho et al., 2023). Moreover, MPs have been found in farmed oyster shells (*Crassostrea angulata*) off Taiwan's coast; the quantities were higher in smaller shells where inorganic fractions containing fibrous MPs and rayon were predominant (Chen et al., 2023).

Small plastic fragments may have distinct effects due to their increased surface area, their ability to cross tissue or cell boundaries,

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or their interactions with other chemicals present in the environment (Worm et al., 2017). The small size of MPs promotes their translocation across gastrointestinal membranes via endocytosis-type mechanisms and their distribution in tissues and organs (Alimba and Faggio, 2019). The presence of MPs outside the gastrointestinal tracts of marine mammals (dolphin, gray whale, bearded seal) was first revealed by Merrill et al. (2023), marking the first identification of translocation and deposition of MPs into various tissues including blubber, melon, acoustic mandibular jaw fat, and caudal lung. This research indicated negative impacts on these tissues, potentially inhibiting critical functions or contributing to the cumulative effects of multiple anthropogenic stressors on marine mammals, ultimately reducing their fitness (Pirotta et al., 2022).

Microplastics have physical effects, on aquatic organisms mainly through mechanisms such as entanglement, suffocation and ingestion (Pegado et al., 2018), with ingestion and entanglement can lead to death, reduced growth rate, reproductive complications and hepatitis (Auta et al., 2017). On the other hand, according to Alimba and Faggio (2019), MPs increase the disruption of the gene expression necessary for controlling oxidative stress in marine vertebrates and invertebrates, leading to genomic instability, endocrine disruption, neurotoxicity, reproductive abnormalities, embryotoxicity and transgenerational toxicity (Romdhani et al., 2022) (Romdhani et al., 2024) (Missawi et al., 2022). Reproduction is particularly sensitive, with energy depletion resulting from exposure to MPs affecting fecundity and fertility (Sussarellu et al., 2016). Moreover, NPs and MPs have been reported to interfere with feeding and reproduction, negatively impacting fertility and offspring quality, which are key components of an organism's fitness (Romdhani et al., 2024) (Worm et al., 2017). For the Zebrafish, sentinel species in MPs research, the exposure to different types of MPs leads to several effects including disruption of metabolism, chronic inflammation, intestinal toxicity and damage to the intestinal lining and structure, decreased reproductive ability, and possible effects on growth and development (Marana et al., 2022) (Qiao et al., 2019a) (Jin et al., 2017). Hepatic cellular damage partly due to the downregulation of genes and pathways linked to DNA damage repair and cell proliferation regulation has also been reported (Tian et al., 2024).

Concerns have been expressed over MPs' effects on marine fauna's gut microbiota because it is essential to metabolic regulation in all organisms and coordinates most metabolic pathways. Several marine organisms, such as mollusks (Li L-L. et al., 2020), crustaceans (Li Q. et al., 2021) (Thery et al., 2023) (Zhu et al., 2024), fish (Zhang C. et al., 2021) (Usman et al., 2022) (Tamura et al., 2024), and seabirds (Fackelmann et al., 2023), can develop dysbiosis, or an imbalance of microbial species, in their gut microbiota as a result of MP exposure. In general, the relative abundance of the main bacterial groups—Proteobacteria, Firmicutes, Bacteroidetes, Actinobacteria, and Fusobacteria—increases (Fackelmann et al., 2023) (Zhu et al., 2024) (Li Q. et al., 2021) (Pamanji et al., 2024) or decreases (Zhang C. et al., 2021) (Li J. et al., 2021) (Zhu et al., 2024). Furthermore, marine taxa exposed to MPs exhibited alterations in the composition of their gut microbiota, with a decline in the abundance of beneficial and commensal bacteria such as Sulfitobacter and Pseudoalteromonas genera (Zhu et al., 2024) and an increase in opportunistic and potentially pathogenic bacteria like *Pseudomonas*, *Aeromonas*, *Streptococcus* (Usman et al., 2022) and *vibrio* (Tamura et al., 2024) (Zhu et al., 2024), as well as antibiotic-resistant and plastic-degrading microbes. Some fish and shrimp studies also report a reduction in gut microbial richness and diversity (Zhang C. et al., 2021) (Usman et al., 2022). However, an increase in gut microbial diversity and richness has been observed in the Brine Shrimp *Artemia parthenogenetica* and the Zebrafish *Danio rerio* (Li R. et al., 2023) (Tian et al., 2024), respectively, indicating that the response may differ by species. In addition, the extent of gut microbiota shifts depends on the exposure duration (Li L-L. et al., 2020) well as on plastic size (Zhang C. et al., 2021), type and concentrations (Thery et al., 2023), even though the concentrations used in the experimental studies can be realistic, much like those in the environment, or nonreal, much lower or higher.

Microplastic-induced dysbiosis has been associated with negative health effects, including lower growth (Li Q. et al., 2021), oxidative stress (Zhu et al., 2024) (Pamanji et al., 2024), hepatic inflammation (Zhang C. et al., 2021) and dysfunction (Tian et al., 2024), intestine structural damage (Zhu et al., 2024), and reduced shoaling behavior (Tamura et al., 2024). Dysbiosis may also disrupt the host metabolic homeostasis contributing then to the development of metabolic disorders (Zhang C. et al., 2021) (Usman et al., 2022), including an alteration of metabolic markers such as lipid, glucose, and of phospholipid metabolism since certain bacteria were linked to phospholipids (Qiao et al., 2019b).

Moreover, since many marine species are consumed by humans, the enrichment of potential pathogens in the gut may increase the organism's susceptibility to infections or diseases, as well as have an impact on food safety and the health of marine ecosystems.

Despite all the reported alterations at cellular level reflecting crucial toxicity that can be extended to enable concluding about species populations decline and even possible risk of extinction, information on how NPs and MPs can enter living cells and thus provoke the damaging effects is very controversial. This is even more challenging when dealing with biological barriers such as brain and reproductive organs. Moreover, MPs have been reported and their effects assessed at different levels of the marine trophic chain from phytoplankton to top level organisms like cetaceans (Table 2), however and to our knowledge no experiments have been attempted to assess its trophic transfer rate throughout the entire marine food chain.

4.2 Microbial communities

Recent research has focused on the effects of different types of MPs on marine microbial communities, that are attached to MP particles or free-living, using advanced methods such as high-throughput sequencing of 16S and 18S rRNA gene, gene expression analysis, and molecular techniques. This has helped to clarify the wider ecological and health implications of MPs pollution by examining how MPs influence the composition, diversity, and function of these communities. MPs can be rapidly colonized by diverse organisms through biofouling processes (Carson et al., 2013) (Reisser et al., 2014) leading to diverse biofilm communities adapted to plastics as a colonization surface. Microplastics that act as an artificial microbial reef, have synergetic effects on the development,

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transportation, persistence, and ecology of these communities (Dey et al., 2022). Investigations into the plastisphere, the microbial communities that colonize and live on the surface of floating plastic debris in aquatic environments (Zettler et al., 2013), have revealed that it hosts a wide range of microorganisms including archaea, bacteria, eukaryotes (fungi, algae, protozoa, and even tiny invertebrates), and even viruses (Lacerda et al., 2022). Eukaryotic organisms colonizing marine MPs are particularly more benthic eukaryotes than the free-water community, notably diatoms which are early and/or dominant colonizers (Eich et al., 2015) (Zhao et al., 2021) (Davidov et al., 2024). Bacteria were seemingly the most abundant biofilm members (Dey et al., 2022), with Proteobacteria (particularly alpha- and Gamma-proteobacteria) being the most identified on marine plastics from different seas and oceans and at different depths (sea surface, seafloor, etc.). Bacteroidetes, Actinobacteria and Cyanobacteria are also common groups of plastisphere community (Bryant et al., 2016) (Dussud et al., 2018) (Harvey et al., 2020) (Chen et al., 2021), (Dey et al., 2022) (Lacerda et al., 2024). Among Proteobacteria, the two hydrocarbonoclastic orders, Oceanospirillales and Alteromonadales, are consistently the more abundant, potentially biodegrading plastisphere members (Wright et al., 2021). In addition, Photobacterium, Pseudoalteromonas, and Psychrobacter, are the dominant Proteobacteria genera that are known for their ability to biodegrade and utilize plastics as a source of carbon and nutrients (Raghul et al., 2014) (Muriel-Millán et al., 2021) (Atanasova et al., 2021). The plastic debris was also found to host taxa that play significant roles in biogeochemical cycles (e.g., cyanobacteria, Erythrobacter) and hygienically relevant bacteria (e.g., Chryseobacterium, Brevundimonas) (Koh et al., 2023). These studies emphasize the metabolic activity of plastisphere microorganisms and their potential impact on the global ecosystem functions.

Bacterial communities colonizing marine plastics are characterized by high diversity and they include pathogens for animals (Priyadharshini et al., 2024) (Radisic et al., 2020) and humans (Kirstein et al., 2016) (Rodrigues et al., 2019) (Silva et al., 2019) (Wu et al., 2019) such as *Vibrio* spp., *E. coli* (Lacerda et al., 2024) (Bowley et al., 2021), *Enterococcus faecalis* (Lear et al., 2021) and bacteria resistant to antimicrobials (Sababadichetty et al., 2024), which constitutes potential risks to health. Furthermore, studies using high throughput sequencing have demonstrated within the genomes of plastic biofilm isolates, the presence of a wide variety of antimicrobial resistance genes, including a diversity of multi-drug efflux pumps and beta-lactams (Sababadichetty et al., 2024) (Rasool et al., 2021) (Lear et al., 2022). This makes MPs a vector of pathogens as well as support for horizontal gene transfer, therefore enhancing the spread of antibiotics gene resistance which constitutes a threat for human and marine organisms' health. Furthermore, the White Spot Syndrome Virus (WSSV), a shrimp pathogen, was recently found on seawater plastics biofilm, highlighting the plastisphere's potential as a disease vector in the marine environment and its ecological and economic impacts (Lacerda et al., 2024). The plastisphere often hosts communities and or taxa that differ from that of the surrounding environments including seawater, sediment and sand beaches (Sababadichetty et al., 2024) (Amaral-Zettler et al., 2020) (Harrison et al., 2014) (Lobelle and Cunliffe, 2011), (Battulga et al., 2024) (Sun et al., 2024). Diversity, richness and taxonomic composition of plastisphere community may differ depending on several factors including environment, abiotic parameters and plastic type and morphotype. Differences in the bacterial community composition at various taxonomic levels were detected depending on the plastic type, i.e., PS *versus* PE and PP (Frère et al., 2018) and wild plastic PE *versus* PP and PS (Vaksmaa et al., 2021). An absence of changes was however found among five household plastics (LDPE), (HDPE), (PP), (PVC) and (PET) (Lear et al., 2022), as well as between seawater-collected plastics (Lacerda et al., 2024). Depending on the morphotypes of MPs (fiber, film, foam, and fragment), no significant differences in Bacterial and fungal communities' composition were found (Battulga et al., 2024). Furthermore, eukaryotic communities can be significantly influenced by plastic polymer type and time incubated. (Guo et al., 2022).

In summary, variations in MPs samples origins (geographical location, depth, sediment vs. water column, etc.), abiotic parameters, plastic types and morphotypes, and methodology all contribute to the reported variations in the taxonomic composition of plastisphere microbial communities. Future research should focus on how the native microbiota, the microorganisms already present in the environment, and the MPs features such as morphotypes, polymers, etc., could individually contribute to the structuring of the plastisphere communities.

5 Impacts on marine ecosystem services (MESs) and blue carbon ecosystems (BCEs)

Microplastics have emerged as a significant environmental contaminant with wide-reaching impacts on MESs as shown in Table 3. They present a complex threat to marine ecosystems by interfering with MESs and compromising overall marine health, which in turn affects the essential services these ecosystems offer to both humans and environment. These services include provisioning, regulating, supporting, and cultural services that humans derive from marine ecosystems. Considering the value of marine services to society, estimated at USD 49.7 trillion per year, the presence of marine plastic debris results in an annual loss of USD 0.5–2.5 trillion, with a yearly cost in terms of reduced marine natural capital ranging from USD 3,300 to USD 33,000 per ton of plastic (Beaumont et al., 2019).

5.1 Impacts on provisioning services: Fisheries and aquaculture

Provisioning services in marine environments refer to the goods that humans directly obtain from the sea, primarily food resources like fish and shellfish. Plastic pollution significantly impacts these services in various ways including contamination of seafood, health risks to humans, and economic impacts on fisheries and aquaculture leading to restricted catches due to litter in nets. For instance, 86% of Scottish fishing vessels have been impacted by plastic pollution, which damages fishing gear, reduces catch quality, and increases operating costs. This pollution costs these fleets an estimated USD 12.8–14.2 million per year, representing about 5% of the total revenue of the affected fisheries. This financial burden

TABLE 3 Impacts of MPs on marine ecosystems services (MESs).

Type of MESs	Sub-type	Impact	Description	References
Regulating services	Climate Regulation	The photosynthetic system of microalgae	Decrease of chlorophyll content and photosynthetic efficiency (ΦPSII). (*Skeletonema costatum*)	Zhang et al. (2017)
			Decrease of chlorophyll-*a* content (*Chlorella vulgaris*)	Tunali et al. (2020)
		Promotion of CH_4 and CO_2	Degradation of plastic materials induces increase in CH_4 and CO_2 emissions	Kida et al. (2023)
		Carbon input	Decrease of carbon input by inhibiting fecal deposition	Wieczorek et al. (2019)
		CO_2 emission	Inhibition of carbon ingestion by plankton	Shen et al. (2020)
			Inhibition of growth and reproduction of plankton	Liu et al. (2020)
			Inhibition of plant growth by affecting photosynthesis	Sjollema et al. (2016)
			Penetration of microalgae cell walls to inhibit CO_2 uptake	Bhattacharya et al. (2010)
	Nutrients cycle	Nitrogen input	Reduction of nitrogen input by inhibiting urease activity	Wieczorek et al. (2019)
		NH_3 input	Ammonification by inhibiting urease activity	Yu et al. (2021a)
		Phosphorus input	Reduction of soil total phosphorus content	Yu et al. (2021a)
	Water purification	Reduction in filtration capacity	Clogging and damage of filtration systems	(Li et al., 2021a) (Ding et al., 2020)
Provisioning services	Fisheries	Ecotoxicological impacts	Disruption in tissues, organs, intestinal permeability, intestinal inflammation, disorders of the intestinal microbiome, neurological functions, immune dysfunction, metabolism and brain	(Qiao et al., 2019b) (Ding et al., 2020) (Jacob et al., 2020) (Gu et al., 2020)
	Aquaculture	Reduction of efficiency and productivity	Water acidification	Gewert et al. (2015)
			Impacts on ecological balance of aquaculture environment	Zhang et al. (2022)
			Ingestion of toxins by aquaculture products	(Cai et al., 2017) (Rummel et al., 2017)
			Increase the abundance of antibiotic resistance genes in aquaculture environments and increase potential risks of losing effectiveness for antibiotics	Lu et al. (2019)
			Loading many viruses	Li et al. (2022b)
Supporting services	Habitat provision and biodiversity	Habitat alteration and loss of biodiversity	Disruption of benthic communities	Khalid et al. (2021)
			Menace nesting sea turtles	Nelms et al. (2016)
			Disruption of the wellbeing and longevity of coral reef ecosystems	Rahman et al. (2023)
			Decline of Seagrass beds	Li et al. (2023b)
			Creation of artificial habitats: plastisphere	Reisser et al. (2014)
	Invasive species spread	Ecological imbalance and displacement of native species	Promotion of colonization of various harmful algal blooms	(Zettler et al., 2013) (Masó et al., 2003) (Masó et al., 2016)
			Transport of exotic pathogenic bacterium	(Zettler et al., 2013) (Kirstein et al., 2016) (Viršek et al., 2017)
			Transport of pathogenic *vibrio* species	Rummel et al. (2017)
			Transport of hydroids, bryozoans, barnacles, mollusks, and Polychaeta worms	(Masó et al., 2016) (Yang et al., 2020)
Cultural services	Tourism/Recreation and Heritage	Physical health	Physical injuries such as cuts due to sharp debris	Beaumont et al. (2019)
		Mental health	Disruption peoples' quality of life by reducing the aesthetic appeal of the marine environment	Beaumont et al. (2019)
		Heritage of communities and individuals	Degradation of natural learning environments	Beaumont et al. (2019)

TABLE 4 Financial Impact of Microplastics on Marine Ecosystem Services (MESs) and Blue Carbon Ecosystems (BCEs). The estimated global value, the potential loss due to microplastics (MPs), and the corresponding estimation year are indicated for each service. The potential losses, expressed as a percentage of global GDP, are calculated by dividing the estimated potential loss of each ecosystem service by the global GDP for the respective estimation year indicted between brackets. The estimated global value is derived from data corresponding to the reference indicated by*.

Service type	Ecosystem service	Estimated global value USD/year	Potential loss due to MPs USD (year)	Potential loss as % of global GDP	References
Provisioning	Fisheries and aquaculture	~401* billion	~6–19 billion (2019)	0.006%–0.02%	(FAO. The State of World Fisheries and Aquaculture, 2020, 2020) (Raes et al., 2023)
Regulating	Nutrient Cycling	~13,000* billion	~650 billion (2019)	0.74%	(Costanza et al., 1997) * (Beaumont et al., 2019)
	Water Purification	~1700* billion	~100–200 billion (2019)	0.011%–0.22%	(Costanza et al., 1997) * (Beaumont et al., 2019)
	Climate Regulation	~200* billion	~2–20 billion (2019)	0.002%–0.02%	(Costanza et al., 1997) * (Beaumont et al., 2019)
Supporting	Habitat Provision	~1,000* billion	~100–300 billion (2019)	0.11%–0.34%	(Costanza et al., 1997) * (Beaumont et al., 2019)
	Biodiversity Maintenance	~3,000-5,000* billion	~300–900 billion (2019)	0.34%–1.02%	(Costanza et al., 1997) * (Beaumont et al., 2019)
	Invasive Species Transport	~120* billion	~10–50 billion (2019)	0.01%–0.05%	(Pimentel et al., 2005) * (Beaumont et al., 2019)
Cultural	Tourism, Recreation Education and Public pollution perception	~4,400 billion	~13–25 billion (2019)	0.01%–0.02%	Beaumont et al. (2019)
Blue Carbon Ecosystems	Coastal protection, carbon sequestration, water filtration and habitat provision	1,000-20000* USD/ha	>50 USD million (2011)	-	(Barbier et al., 2011) *

emphasizes how urgently sustainable waste management and pollution mitigation strategies are needed to protect marine resources and support sustainable coastal economies. Furthermore, the fishing industries faces challenges with derelict fishing gear, which can damage vessels, require the replacement of lost gear, and lead to potential catch losses, thereby reducing revenue (Arabi and Nahman, 2020). In 2002, a single trap fisher in the Scottish Clyde fishery faced losses of approximately USD 21,000 in fishing gear and USD 38,000 in lost fishing time (Macfadyen et al., 2009). MPs are found in various marine organisms including fish (Ferreira et al., 2018), bivalves (Bråte et al., 2018), cephalopods (Oliveira et al., 2020) or crustaceans (Botterell et al., 2019), entering the food chain and potentially affecting human health (Tuuri and Leterme, 2023). Fish, shellfish and other seafood that humans consume are often contaminated with MPs, leading to bioaccumulation and biomagnification of harmful chemicals (Li J. et al., 2021) (Ding et al., 2020) (Bhuyan, 2022) (Barboza et al., 2018) (Wright and Kelly, 2017). MPs impact freshwater and marine fish in several ways, including impairing their feeding ability (Wright and Kelly, 2017), causing nutritional and growth disorders (Lusher et al., 2017), and promoting behavioral changes (Liang et al., 2023). Additionally, when these fish are consumed, MPs become part of the human diet (Smith et al., 2018).

According to the International Union for Conservation of Nature (IUCN), marine plastic pollution could result in economic losses ranging from $6 billion to $19 billion annually across 87 coastal countries (Table 4). This estimate extends beyond the direct impacts on fisheries and aquaculture to include broader economic effects on coastal economies, such as degradation of ecosystem services and reduced export revenues (Raes et al., 2023). More precisely, Beaumont et al. (2019) estimated that plastic debris imposes an annual global loss of $1.3 billion USD on the fishing and aquaculture industries, specifically through reduced catch, gear damage, and clean-up costs. This estimate, which has a relatively high degree of certainty within this narrowly defined scope, is derived from an earlier assessment by UNEP (2014) based on a limited number of national case studies. However, it does not account for broader indirect impacts such as disruptions to supply chains, or wider ecosystem service degradation. While this figure highlights the sector-wide burden, framing these losses against national economies underscores their broader significance. In Norway, where fisheries and aquaculture accounted for 2.3% of mainland GDP in 2022, average vessel losses of USD 12 000 per year reflect ongoing financial strain in an industry contributing roughly six billion USD to the economy (Fish farming Expert) (Fishfarming Expert, 2025). In Fiji, 2019 losses of over 600,000 USD correspond to about 0.011% of the country's total GDP of 5.44 billion USD (World Bank, 2023a). Likewise, in Ecuador, annual losses of 8.4 million USD represent approximately 0.007% of its 118.84 billion USD GDP (World Bank, 2023) (World Bank, 2023b) and in Peru, 8.27 million USD in losses equate to around 0.003% of a 267.6 billion USD economy (World Bank, 2023c). Although these percentages may seem modest, they reveal that MPs pollution exerts a measurable drag even on national economic output, reinforcing the urgency of targeted mitigation and policy action.

The accumulation of MPs by various shellfish can lead to unique health and performance deterioration, such as toxicological implications, behavioral changes, and growth and reproductive problems (Li J. et al., 2021) (Ding et al., 2020) (Hossain et al., 2024). MPs have also been found in aquafarms, where fish and mollusks are cultured for human consumption (Zhang et al., 2020) (Rochman et al., 2015). Harmful additives and absorb pollutants released by MPs in aquaculture environments can cause toxicological effects, impact behavior, growth, and reproduction of aquaculture species, and ultimately reduce the economic benefits of aquaculture. MPs entering the human body through aquaculture products also pose potential health risks at multiple levels (Wu et al., 2023).

5.2 Impacts on regulating services: nutrient cycles, water purification and climate regulation

Microplastics have a profound and detrimental impact on marine regulating services, which are critical for maintaining the health and stability of marine ecosystems (Sridharan et al., 2021). These tiny plastic particles infiltrate marine environments and disrupt various ecological processes. Key regulating services, such as nutrient cycling, water purification, and climate regulation, are particularly affected by MPs pollution.

The economic repercussions of MPs pollution are stark. As reported in Table 4, the estimated global economic value of these services amounts to approximately USD 15 trillion per year (Costanza et al., 1997). However, MPs pollution leads to significant disruptions of these functions, resulting in estimated economic losses ranging from USD 752 to 870 billion annually. For instance, a 5% disruption in nutrient cycling could represent a loss of USD 650 billion/year, while impairments in water purification capacities could incur losses between USD 100 and 200 billion/year (Beaumont et al., 2019). Climate regulation services may also experience economic damages ranging from USD 2 to 20 billion/year under an assumed impact of 1%–10% (Beaumont et al., 2019). The potential loss as a percentage of global GDP was calculated by dividing the estimated potential loss of each ecosystem service by the global GDP in 2019 (~USD 87.55 trillion). Collectively, these losses amount to approximately 1% of global GDP, a substantial figure that reflects a significant economic impact on the global scale. (Table 4).

Nutrient cycling is significantly disrupted by MPs through the alteration of microbial dynamics and nutrient availability. MPs serve as surfaces for biofilm development, enabling microorganisms to thrive, which can shift nutrient transformations, particularly nitrogen and phosphorus cycling (Chen et al., 2020). Additionally, MPs adsorb nutrients and pollutants from surrounding waters, affecting their bioavailability, and potentially leading to imbalances in nutrient dynamics. When ingested by marine organisms, MPs disrupt food webs and influence community structures, impacting ecosystem functions related to nutrient cycling (Wang et al., 2022). Furthermore, their presence in sediments modifies the physical and chemical properties of the benthic environment, affecting nutrient release and utilization (Green et al., 2016). Collectively, these interactions highlight the complex and multifaceted role MPs in marine nutrient cycles, posing risks to marine ecosystem health.

Microplastics pose significant challenges to water purification processes in marine environments. As these small plastic particles are pervasive, they can interfere with the natural filtration mechanisms of marine ecosystems, such as wetlands and mangroves, which are crucial for improving water quality (Qian et al., 2021) (Adaro and Ronda, 2024). Filter-feeding organisms, which can filter up to 5 m^3 of water per day, become less efficient as MPs clog their systems leading to poorer water quality and decreased availability of clean habitats for other marine life (Li J. et al., 2021) (Ding et al., 2022). Microplastics also adsorb pollutants, including heavy metals and Persistent Organic Pollutants (POPs), which may be released back into the water column upon degradation, exacerbating contamination levels (Rafa et al., 2024). Due to their hydrophobicity and their relatively large surface area, MPs act as vectors of harmful pollutants, such POPs facilitating their transfer to organisms (Huang et al., 2021). These substances, many of which are endocrine disruptors, bioaccumulative and persistent, can modify the metabolic and reproductive parameters (Galloway et al., 2015) (Koelmans et al., 2016). Plastics not only have the capacity to transport toxic substances but also to increase them in the environment. Adsorption of these substances is enhanced by the presence of biofilm, which alters the hydrophobicity of the plastic particle's surfaces. Additionally, the aging of plastics in marine waters changes their surface morphology facilitating the absorption of metallic ions (Squadrone et al., 2022). Additionally, the presence of MPs hinders the growth of beneficial microorganisms that play vital roles in nutrient cycling and bioremediation, further impairing the ecosystem's ability to purify water. This disruption compromises the health of marine organisms and the integrity of coastal and oceanic ecosystems, highlighting the urgent need for effective management strategies to mitigate MPs pollution. Microplastics also affect climate regulation by altering key processes such as carbon cycling and marine ecosystem health (Li K. et al., 2024) (Galgani et al., 2023). They affect phytoplankton communities, which play a crucial role in carbon sequestration through photosynthesis, and can hinder the biological pump which represents the mechanism by which carbon dioxide is absorbed from the atmosphere and transported to deeper ocean layers (Shen et al., 2020). Additionally, MPs can contribute to the degradation of marine habitats, such as coral reefs and seagrass beds, which are vital for carbon storage. They also disrupt food webs, impacting species that contribute to carbon cycling and ecosystem resilience. As these ecosystems weaken, their ability to sequester carbon effectively diminishes, potentially exacerbating climate change effects (Li K. et al., 2024). Addressing MPs pollution is therefore essential for maintaining marine biodiversity but also for supporting climate regulation efforts.

5.3 Impacts on supporting services: habitat provision, biodiversity and invasive species transport

Microplastics profoundly affect marine supporting services, especially habitat provision, biodiversity, and the transport of invasive species. The economic losses associated with habitat degradation can be substantial, affecting tourism, fisheries, and coastal protection services. For example, the loss of coral reefs alone is estimated to cost global economies around $375 billion annually due to lost tourism, fisheries and protection against erosion and storm damage (Chatterjee and Sharma, 2019). In terms of habitat provision, 70% of MPs debris accumulates in marine environments, altering the physical and chemical characteristics of habitats such as beaches, seafloors, and coral reefs (Mouat et al., 2010). This can change the sediment's texture, reduce light penetration, and introduce harmful chemicals, making the habitat less suitable for native species. For biodiversity, MPs pose a dual threat (Beaumont et al., 2019). Physically, they can cause internal injuries or blockages in marine organisms that ingest them, while chemically, they can leach toxic substances or adsorb pollutants, leading to bioaccumulation and biomagnification within the food web (Mouat et al., 2010). These effects can reduce the survival and reproductive success of marine species, causing population declines and shifts in community structure.

Additionally, MPs facilitate the transport of invasive species (Naidoo et al., 2020). They serve as rafts for microorganisms, algae, and invertebrates, allowing these species to travel across oceans and colonize new areas. This can disrupt local ecosystems by introducing competitors, predators, or pathogens to which native species are not adapted to cope with, leading to reduced native biodiversity and altered ecosystem functions (Beaumont et al., 2019) (Carney and Eggert, 2019). Microplastics also pose risk to zooplankton, marine mammals, birds and fish, and can serve as vector for the dispersal of harmful microalgae such as *Alexandrium*, *Coolia* and *Ostreopsis* (Zettler et al., 2013) (Masó et al., 2003) (Masó et al., 2016). These harmful species produce a variety of marine biotoxins implicated in contamination events affecting filter or grazer-feeding animals and human poisoning, sometimes with lethal outcomes (Anderson et al., 2012) (Parsons et al., 2012) (Trainer et al., 2012).

Beyond these direct ecological impacts, MPs pollution also threatens the broader supporting services that underpin marine ecosystem functioning and productivity. As shown in Table 4, the global economic value of these services is estimated at USD 4.1 to 6.1 trillion per year (Costanza et al., 1997) (Pimentel et al., 2005). Microplastic pollution compromises these functions, resulting in considerable economic losses. For example, degradation of coastal habitats could result in losses of USD 100 to 300 billion/year, while erosion of biodiversity and ecosystem stability may cost between USD 300 and 900 billion/year (Beaumont et al., 2019). Additionally, increased costs related to the facilitation of invasive species transport via MPs are estimated between USD 10 and 50 billion/year (Beaumont et al., 2019). When reported as a percentage of global GDP, the losses represent between 0.5% and 1.4% of global GDP. These figures underscore the critical importance of supporting services and highlight the substantial economic impacts that persistent MPs pollution could have on the resilience and productivity of marine ecosystems. Overall, the pervasive presence of MPs in marine environments undermines habitat quality, threatens biodiversity, and enhances the spread of invasive species, posing substantial risks to marine ecosystem health, stability and resilience.

5.4 Impact on blue carbon ecosystems

Wetlands and marine ecosystems, including seagrass meadows, mangrove forests, estuaries, and salt marshes, serve as sites of MPs deposition (Yu H. et al., 2021) (Ogbuagu et al., 2022). These BCEs are increasingly impacted by MPs contamination (Garcés-Ordóñez et al., 2019) (Huang et al., 2020) (Pinheiro et al., 2022) (Yin et al., 2021). Microplastics can attach to plants and carry toxic chemical compounds (Tourinho et al., 2019) (Goss et al., 2018), which may be ingested by herbivores and subsequently enter the food web (Goss et al., 2018). Additionally, the presence of MPs can influence the physical, chemical and biological properties of sediments, providing new niches for microbial communities (Wright et al., 2020) (Su et al., 2022). Moreover, accumulated MPs can damage the delicate structures of seagrass meadows and mangrove forests and animals within these ecosystems, inhibiting their growth and reproductive capabilities (Li R. et al., 2020).

Blue Carbon Ecosystems, such as mangroves, seagrasses, and salt marshes, occupy a small portion of the global seafloor (less than 1%) but are highly efficient carbon sinks, responsible for up to 50% of the total carbon sequestration (Mcleod et al., 2011) (Duarte et al., 2013). These ecosystems play a vital role in both short-term and long-term carbon storage, making them far more efficient compared to terrestrial forests (Mcleod et al., 2011) (Pendleton et al., 2016). Mangroves, in particular, contribute to the global mean burial rate of approximately 24 Tg C per year (Breithaupt and Steinmuller, 2022). The accumulation of MPs in these ecosystems disrupts their efficiency as carbon sinks.

Microplastics reduce the carbon sequestration efficiency of BCEs through several interconnected mechanisms. They interface with the sinking of organic matter such as marine snow by increasing the buoyancy of particle aggregates. This slows their descent to the seafloor, reducing the amount of carbon that becomes permanently buried in sediment (Kiørboe, 2001). In addition, MPs negatively impact key blue carbon plants such as mangroves, seagrasses and salt marsh vegetation by causing oxidative stress, blocking light, or introducing toxic chemicals. These effects reduce primary productivity and biomass accumulation, thereby decreasing carbon capture at the ecosystem level (Microplastic Pollution in Marine Environment, 2021). Furthermore, MPs disturb the structure and function of sediment microbial communities that are essential for transforming and stabilizing organic carbon. They alter sediment porosity and oxygen levels, leading to enhanced microbial degradation of stored carbon and increased CO emissions (Seeley et al., 2020) (Microplastic Pollution in Marine Environment, 2021). Microplastics may also carry harmful pollutants or pathogens that further disrupt benthic ecosystems. Together, these effects significantly weaken the role of blue carbon ecosystems as long-term carbon sinks, threatening their contribution to global climate regulation and carbon neutrality goals.

From an economic perspective, if MPs reduce carbon sequestration by just 1%, the loss in carbon storage potential could lead to a significant financial impact. Indeed, at a conservative carbon price of $50 per ton, this reduction could result in an economic loss of approximately $12 million annually from mangroves alone (Zhou et al., 2023) (Bandh et al., 2023). Similar financial impacts are recorded in coral reef ecosystems, where coral degradation linked to plastic debris could result in losses of up to $35,000 per hectare per year in tourism and coastal protection services. When extrapolated across global reef systems, these losses may exceed several billion dollars annually (Spalding et al., 2017).

This highlights the dual threat posed by MPs pollution not only compromising marine ecosystem health but also contributing to economic losses linked to climate regulation services. In addition, BCEs provide essential services such as coastal protection, water filtration, and habitat for wildlife. The economic value of these services is estimated to be in the range of $1,000 to $20,000 per hectare per year, depending on the ecosystem type and location (Table 4). Microplastics pollution that degrades these services could result in substantial losses. For example, a 5% reduction in service value across one million hectares could lead to losses exceeding $50 million annually (Zhou et al., 2023) (Bandh et al., 2023) (Noman et al., 2024).

Therefore, the financial burden of restoring BCEs affected by MPs can also be significant. Restoration efforts can cost between $10,000 to $100,000 per hectare, depending on the ecosystem type and the degradation level. If MPs necessitate restoration across thousands of hectares, costs could escalate into the millions or even billions (Bandh et al., 2023) (Zhang et al., 2024).

5.5 Impacts on cultural services: tourism, recreation, education and public pollution perception

Many coastal communities derive cultural identity and livelihoods from their marine environment. The degradation of these ecosystems can lead to significant losses in cultural services, which might be valued at around USD 4.4 trillion per year (Table 4). Even a small degradation could represent millions in lost cultural heritage value (Chatterjee and Sharma, 2019). Marine plastic pollution has significant implications for the tourism industry, leading to decreased revenue and substantial economic costs related to cleaning and maintaining affected areas (Aminur Rahman et al., 2023). Thus, to prevent losses in tourism income and support sustainable coastal economies, some municipalities invest heavily in clean-up efforts to remove debris from beaches and public spaces. For instance, municipalities in Belgium, the Netherlands, and the UK spend between EUR 10–20 million annually on coastal debris removal to protect tourism (Rahman et al., 2023). Furthermore, the presence of MPs on beaches and in coastal waters also diminishes the aesthetic appeal and safety of these environments. Swimmers, divers, and beachgoers may encounter polluted waters and shores, reduce enjoyment and increase potential health risks from direct contact with MPs or associated toxins. In Bali, plastic pollution caused such a decline in tourist satisfaction that the local government declared a "trash emergency" in 2017. Cleanup costs surged to over $1 million per month, and tourism revenue dropped by nearly 10% during peak season due to bad publicity and reduced water quality (Jambeck et al., 2015). Similarly, in South Korea, a 1% increase in visible marine debris has been estimated to lead to a 1.29% drop in beach visits, while in the UK, polluted beaches may result in up to a 50% reduction in tourist numbers, directly affecting local revenues (Jang et al., 2014) (UNEP, 2016). This can lead to a decline in tourism, adversely affecting local economies that rely on marine-based recreational activities (Mouat et al., 2010) (Beaumont et al., 2019). In addition, as of 2021, 20 million people were engaged in subsistence fishing, and nearly 30 million worked in capture fisheries, with 90% of these individuals located in low- and lower-middle-income countries (Food and Agriculture Organization of the United Nations, 2024). Thus, recreational (sport) fishing has become an additional source of income for many communities in these regions (Zhao Q. et al., 2022) (Arlinghaus et al., 2019). While participation rates remain low in some areas, it is estimated that at least 220 million people worldwide engage in recreational fishing (Arlinghaus et al., 2019). Local governments spend considerable resources to maintain beach quality. For example, the United States spends over $500 million annually on beach cleanups, while the EU coastal states collectively invest over €630 million each year for similar purposes (Ten Brink et al., 2016). In Cape Town, the city spends approximately $2 million per year on beach cleaning to maintain its status as a tourist destination.

Public perceptions of plastic pollution and its impacts are strongly linked to pro-environmental behavior (Kumar et al., 2021). Awareness and concern about plastic pollution vary based on age and education level. Older individuals and those with lower educational attainment tend to place less emphasis on recycling as a means of tackling the plastic problem (Miguel et al., 2024). This highlights the critical role of education for sustainable development in promoting awareness and proactive engagement with environmental issues across diverse populations. By integrating sustainability principles into education systems and curricula, individuals, particularly engineers who can devise innovative solutions through their designs, can develop the knowledge and skills needed to address challenges like plastic pollution effectively (Nakad et al., 2024) (Nakad et al., 2025a) (Nakad et al., 2025b). As a result, plastic pollution has socio-cultural impacts that extend beyond direct, quantifiable effects, influencing aspect like lifestyle, mental health, and cultural heritage (Yose et al., 2023).

Environmental attitudes play a significant role in shaping consumers' intentions to reduce plastic use, particularly in the food products sector (Siddiqui et al., 2023). This highlights the close relationship between consumer behavior, environmental awareness, and efforts to address plastic pollution. For instance, the transition to sustainable packaging in the food and beverage sector has been slow and inconsistent, with many companies focusing on collection and recycling rather than adopting systemic sustainable solutions (Phelan et al., 2022).

Consumers associate plastics with more than just environmental issues, and different types of consumer awareness regarding plastics use, particularly in packaging have been identified (Rhein and Schmid, 2020) first type, awareness of environmental pollution, reflects an understanding of the global plastic waste problem, especially in oceans, and supports environmental protection efforts. The second, awareness of excessive plastic use, highlights the overuse and often unnecessary plastic packaging, particularly for items like fruits and vegetables. Third, awareness of consumers' influence recognizes the role of consumer behavior in encouraging companies towards more sustainable practices. Fourth, awareness of consumers' powerlessness expresses the feeling among some consumers that, while change is theoretically possible, practical alternatives are limited. Finally, awareness of the need for using plastic recognizes plastic's hygienic benefits in daily life, despite its environmental drawbacks. These insights suggests that consumer perceptions and attitudes toward plastic are complex and multifaceted (Rhein and Schmid, 2020).

6 Regulation

The growing environmental crisis caused by plastic pollution requires robust national and international regulations to mitigate its impact. The establishment of international regulations will ensure sustainable standards for plastic production, usage and end of life management and promote a circular economy that minimizes waste and conserves resources. By addressing this issue, plastic pollution can be reduced worldwide and protect the environment for future generations.

6.1 International and national regulations

Many international regulations have been developed like the International Convention for the Prevention of Pollution from Ships (MARPOL Convention) which aims to minimize marine pollution in seas and oceans, including pollution caused by plastics. Annex V of the MARPOL Convention addresses the prevention of pollution by garbage, such as plastics (MARPOL, 1973). Furthermore, the Basel convention aims to control the transboundary movements of hazardous waste, including plastic waste, ensuring its environmentally sound management to protect human health convention, and the environment (Rummel-Bulska, 2004). Additionally, Food and Drug Administration (FDA) regulations govern the safety of plastics used in food packaging, ensuring they do not pose health risks and encouraging recycling and proper disposal practices (FDA, 2020).

The international regulatory control system for waste transfers was created under the 1989 Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal. Starting in 2019, parties to the Basel Convention adopted an alteration targeting plastic waste to protect human and environmental health from the negative impacts of the global plastic waste trade. The new regulations can significantly change the global plastic waste trade, affecting both Basel parties and non-party trading partners like the United States (Khan, 2020).

In Africa, most countries have implemented a total ban on the production and use of plastic bags, with 25 countries having announced such regulations. However, more than half of these regulations were enforced after 2014 (Ncube et al., 2021).

In Europe, regulations governing materials and articles made of plastic are outlined in Regulation No. 10/2011 (EU 2011c). This regulation is recognized as the Plastic Implementation Measure (Steensgaard et al., 2017). According to this regulation, plastics refer to all polymers that form the primary structural component of final materials, produced through processes like polymerization, polycondensation, or polyaddition. However, ion exchange resins, rubber, and silicones are not covered under the plastics regulation. Regarding recycled plastic materials for food packaging, an appropriate quality assurance system must be employed to ensure the recycled plastic meets the specifications detailed in the authorization, as specified in Regulation (EC) No. 2023/2006 Annex. In addition, The quality of the plastic input must also comply with Article three of Regulation (EC) No. 1935/2004 (Thapliyal et al., 2024). Another regulation developed by the European Union is the Directive (EU) 2019/904 on Single-Use Plastics. This directive aims to decrease the volume and environmental impact of specific plastic products, and to mitigate the adverse effects of certain plastic items on the environment, particularly aquatic ecosystems and human health (Thapliyal et al., 2024).

In the United States, in 2021, the FDA issued the guidance document "Use of Recycled Plastics in Food Packaging: Chemistry Considerations" to help food packaging manufacturers assess the processes for integrating post-consumer recycled plastic into food packaging. The FDA states that exposure to contaminants from recycled food contact materials at levels of 1.5 μg per person per day (0.5 ppb DC) or lower is generally considered safe (FDA, 2020). On the other hand, some countries have detailed regulations and standards for recycled plastic content in food packaging, while others offer general guidance or lack specific regulations altogether. The requirements for testing and certification differ from one country to another. Different countries have established national regulations for plastic materials to reduce pollution and manage waste more effectively (Table 5).

The regulatory framework addressing MPs pollution in the marine environment focuses on mitigating their presence by restricting the use of intentionally added MPs in products and addressing unintentional releases. This includes developing standardization, certification, and regulatory measures, as well as harmonizing methods for accurately measuring MPs releases.

The regulation is mainly aimed at primary measures for controlling the original types of MPs justified by the fact that marine MPs pollution is a phenomenon caused by specific pollutants rather than a specific behavior regulated by law. The primary MPs are microscopic polymers, mostly including microbeads. Therefore, the legislation mainly focuses on prohibiting the addition of microbeads to related products and prohibiting the sales, import, and export of such products (Li, 2022). However, to our knowledge, there are no regulations related to threshold levels of MPs in the marine environment and its related resources.

Some directives emerged relative to the monitoring aspects of MPs in the marine environment such as the Marine Strategy Framework Directive MSFD (MSFD, 2008/56/EC); which provides framework for monitoring and large-scale actions to assess and mitigate the impacts of marine litter in the European region. The background of this initiative stems from the fact that, while well-established methods exist for assessing litter on beaches, in the water column and on the seafloor, there remains a need to implement monitoring schemes specifically for MPs in sediments and invertebrates. Some actions have been identified to ensure the effectiveness of monitoring efforts including the identification of accumulation areas and sources of specific types of litter; enhance monitoring of riverine and atmospheric inputs of litters including MPs, take care of accidental inputs during extreme weather events (Galgani et al., 2024).

TABLE 5 National regulations on plastic materials in different countries.

Country	Regulations	References
India	Plastic Waste Management (Amendment) Rules (2021)	Ashish (2021)
California	SB 54: Plastic Pollution Prevention and Packaging Producer Responsibility Ac	Senate Bill (2022)
Australia	The National Plastic Plan 2021	Thapliyal et al. (2024)
Japan	The Plastic Resource Circulation Act (Act No. 60 of 2021	Ministry of the Environment and Japan (2021)
United Kingdom (UK)	The Plastic Packaging Tax (2022)	GOV. UK (2022)
China	GB/T 38,082–2019 Biodegradable Plastic Bags	Thapliyal et al. (2024)
United States	Use of Recycled Plastics in Food Packaging: Chemistry Considerations	Food and Drug Administration FDA (2021)
European Union	Regulation No. 10/2011 (EU 2011c)	Thapliyal et al. (2024)
European Union	Directive (EU) 2019/904	Thapliyal et al. (2024)

6.2 Gaps and challenges in current regulatory approaches

Despite increasing global efforts to address plastic pollution, notable gaps remain in current regulations. Thus, existing policies often fail to manage the complexity of plastic waste, particularly given the rise of diverse materials and evolving global trade patterns. Furthermore, the lack of enforceable global targets and a coordinated approach has resulted in unorganized and less effective solutions. To address this challenge, it is crucial to explore current policy gaps and the need for better regulatory measures that promote sustainable practices and significantly decrease plastic pollution.

6.2.1 Binding policy instruments

As of 2019, there are no binding global policy instruments with specific and measurable targets for reducing plastic pollution. While recent legally binding amendments to the Basel Convention have made it easier to classify more types of plastic waste and recycle plastic packaging. These amendments do not include measurable targets for reducing plastic pollution, including packaging (Diana et al., 2022). A global program with specific and measurable targets could help unify fragmented national and subnational efforts to combat plastic pollution. This global response would need to be flexible enough to accommodate geographic and cultural differences. Notably, international negotiations led by the UN are currently underway to establish a global plastic treaty by the end of 2024. The ongoing negotiations underscore the recognition of environmental consequences associated with plastic pollution while emphasizing the need to enhance the knowledge base of potential human health risks (Aanesen et al., 2024). This plastic treaty is a legally binding global agreement aiming to address the full lifecycle of plastics, including the design of reusable and recyclable products. The initiative also seeks to enhance international collaboration to facilitate access to technology, capacity building, and scientific and technical cooperation.

In addition, there are many types of plastics and a gap in the plastic life cycle, and many organic pollutants resist degradation, resulting in long-term exposure risks across multiple waterways which involves the development of an approach at multiple levels (international, national) to reduce the plastic pollution of multiple plastic types (Islam et al., 2023). As regulations change, new challenges will arise. For instance, the rise in e-commerce has greatly increased the need for packaging materials, requiring innovative solutions to manage this growth sustainably (Thapliyal et al., 2024). Another growing challenge is the contamination of recycling streams, which can reduce the effectiveness of plastic regulations and recycling programs (Thapliyal et al., 2024).

6.2.2 Recycling regulations and challenges

Concerning plastic recycling regulations, several challenges arise. Firstly, the poor miscibility of polymer blends makes effective sorting of waste crucial for ensuring high-quality products from mechanical recycling. This sorting presents an economic challenge. Moreover, many plastic products are made with additives such as colors, dyes, fillers, UV protectants, fire retardants, reinforcements, and plasticizers (Seay and Ternes, 2022). These additives cause recycled plastics to differ significantly from virgin resins, limiting their suitability for many uses.

Current regulations indicate that only products and materials made from recycled plastic obtained through an authorized recycling procedure may be commercialized. Thus, a suitable quality assurance system must be in place to ensure that the recycled plastic meets the specifications outlined in the relevant regulations (Regulation (EC) No. 2023/2006 Annex) (European Union, 2822008). Therefore, the quality of the plastic input and the recycling process must demonstrate its capability to reduce any contamination of the plastic input to a concentration that does not endanger human health through a challenge test or other appropriate scientific evidence. Hence, the quality of recycled plastic must be assessed and controlled appropriately, ensuring that the finished recycled plastic material complies with established standards.

Finally, over time, plastic becomes unrecyclable and must either be discarded or used for its energy value. This issue is further increased in developing countries, where the infrastructure for collecting and sorting plastic waste is often inadequate or lacking (Seay and Ternes, 2022).

6.3 Effectiveness and enforcement of existing policies

The regulations set by different countries and international organizations play an important role in promoting sustainable practices and protecting the environment from plastic waste. However, the effectiveness of these regulations is not clear and varies between countries. For instance, in Europe, more than EUR 5.5 billion has been designated to enhance waste management capacity, with the goal of recycling an additional 5.8 million tons of plastic waste annually (Nikiema and Asiedu, 2022). An important element of the regulations mentioned above is their effectiveness and implementation. Furthermore, the EU's single-use plastics directive and circular economy action plan have set goals to cut plastic waste. These initiatives have resulted in higher recycling rates, shifting toward eco-friendly materials in the packaging (Directive, 2019). On the other hand, in India, the regulations of plastic bag management seek to reduce waste by promoting the use of compostable and biodegradable materials. These regulations and measures have decreased the demand for plastic bags while there was an increase in the demand for eco-friendly packaging materials (Thapliyal et al., 2024).

The effectiveness of regulations for plastic waste management varies. For example, the ban on plastic waste in many countries aims to enhance efforts to protect the environment and human health. In this context, gradual progress in the developing world is crucial for achieving the sustainability (Islam et al., 2023). To improve waste collection processes, the plastic waste market needs better coordination between demand and supply, along with innovation and investment. Regulations can play a vital role in securing markets for recycled plastics and encouraging producers to create demand (Islam et al., 2023). Additionally, public awareness about MPs can further enhance the effectiveness of these regulations (Nikiema and Asiedu, 2022).

Furthermore, when an economic measure such as a plastic bag tax or fee is introduced, consumers may become accustomed to the higher prices and revert to using more bags, thereby reducing the policy's effectiveness (Diana et al., 2022). The effectiveness of plastic bag policies and the potential demand for paper bags are also often tied to the availability of affordable, reusable alternatives to plastic bags (Diana et al., 2022). Governments around the world offer research grants to support efforts in reducing plastic waste and enforcing regulations. Notable examples include the UK Government's £20 million Plastics Research and Innovation Fund, £61.4 million allocated to the Commonwealth Clean Oceans Alliance, and the EU's Horizon 2020 research and innovation program (Islam et al., 2023). Many countries have developed various technologies to comply with plastic regulations. Among traditional methods, landfilling, and mechanical reprocessing are the most common and easiest to implement on a large scale. Additionally, new technologies are emerging that align with the goals of regulations (Islam et al., 2023).

7 Conclusion

Microplastics have infiltrated various ecosystems and ultimately end up in the ocean, leading to a significant threat for environmental sustainability. The quantification efforts of MPs and their implications for environmental health have been inconsistent with uneven effort across different ecosystems. Marine organisms are impacted by MPs at various structural levels, showing serval harmful effects on essential functions such as the respiratory, reproductive, neurological and immune systems. Future research should be dedicated to modeling MPs transfer across the marine food chain to gain insight into the transfer to higher trophic levels and better understand the impact of this pollution on human life. Overall, the potential loss due to MPs across all marine ecosystem services is estimated to range from 1.22% to 2.1% of global GDP, underscoring the substantial economic burden posed by marine plastic pollution. To better inform policy and management strategies, future research should incorporate comprehensive cost-benefit analyses and advanced economic modeling approaches. Integrating sector-specific losses notably from fisheries, tourism, and ecosystem restoration into these models could provide clearer insights into the long-term economic consequences of MPs pollution and support more effective investment in mitigation measures. The regulatory framework for MPs pollution in the marine environment primarily focuses on restricting intentionally added MPs in products and addressing unintentional releases through standardization and regulatory measures. However, there is currently no regulation specifying threshold levels of MPs in the marine environment and its living resources. Moreover, future regulatory frameworks should prioritize setting quantifiable and enforceable targets for plastic pollution reduction, fostering sustainable practices, and encouraging international collaboration to ensure uniformity in efforts across regions.

Author contributions

MaB: Supervision, Writing – original draft, Investigation, Conceptualization, Writing – review and editing, Resources. AB: Writing – review and editing, Writing – original draft. MA: Writing – original draft, Writing – review and editing. J-CA: Writing – original draft, Writing – review and editing. MN: Writing – review and editing, Writing – original draft. RA: Writing – original draft, Writing – review and editing. YK: Writing – original draft, Writing – review and editing. MoB: Writing – review and editing. AP: Writing – review and editing. LG: Writing – review and editing. WH: Supervision, Funding acquisition, Writing – original draft, Writing – review and editing, Visualization, Validation, Conceptualization.

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Conflict of interest

Author AP was employed by DNV Business Assurance Italy S.r.l.

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Simulation of microplastic transport and dispersion based on a three-dimensional hydrodynamic particle-tracking model in the Beibu Gulf

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The Beibu Gulf, a representative semi-enclosed bay in the South China Sea, experiences microplastic transport and dispersion governed by a complex interplay of monsoons, ocean circulation, and extreme weather events, warranting systematic investigation. We developed a numerical modeling framework by coupling a three-dimensional hydrodynamic model with a Lagrangian particle-tracking module, and validated it against in observations. The model quantitatively demonstrates high accuracy, with maximum spatial deviations below 6 km and relative standard deviations within 7%, confirming its suitability for simulating microplastic transport. The simulation results indicate that the transport of microplastics in the Beibu Gulf is primarily controlled by the oceanic hydrodynamic environment, while also being indirectly affected by the monsoon. During winter and autumn, the northeast monsoon dominates, whereas in spring and summer, the southwest monsoon prevails, with the overall circulation exhibiting a counterclockwise coastal current pattern. In spring, microplastics can disperse up to 205 km, while in summer, southwest monsoon conditions lead to the formation of nearshore high-concentration zones (~20 μg/m^3). Vertical transport significantly modulates plume structure, with summer pollution coverage expanding by over 70% compared to scenarios excluding vertical motion.Storm surge events further intensify hydrodynamic conditions. As a case study, Typhoon Yagi induced significant alterations in the hydrodynamic conditions of the Beibu Gulf: prior to the storm, tidal forces governed periodic flow variations; during and after the storm, intense circulations generated prominent counterclockwise vortices, with velocities reaching 2.8 m/s, substantially enhancing long-range microplastic transport and extending their spatial distribution. This study reveals the key characteristics of microplastic transport in the Beibu Gulf under varying seasonal and hydrodynamic conditions, providing a rigorous theoretical foundation for understanding regional microplastic dispersal patterns.

KEYWORDS

microplastics, Beibu Gulf, Lagrangian particle tracking, transport and dispersion, storm surge

1 Introduction

With the increasing exploitation and utilization of marine resources by humans, various environmental issues have emerged in the ocean, among which plastic pollution has become one of the most prominent problems affecting marine ecosystems (Wang et al., 2018). A large amount of plastic waste enters the ocean from land through various pathways. It is estimated that plastics account for approximately 60%–80% of marine debris worldwide (Barnes et al., 2009; Auta et al., 2017; Thanigaivel et al., 2025). Microplastics (MPs) are typically defined as polymer particles smaller than 5 mm in size (Andrady, 2011), exhibiting diverse morphologies, including fibers, fragments, pellets, and films. They are generated from the gradual fragmentation of larger plastic debris in the marine environment under the influence of ultraviolet radiation, mechanical forces, and biodegradation processes (Cole et al., 2011). The predominant types of microplastics include polyethylene (PE), polypropylene (PP), and polystyrene (PS) (Gao et al., 2023). At present, microplastics are widely distributed in various environmental compartments, including surface seawater, deep-sea sediments, polar ice, and atmospheric deposition (Cózar et al., 2014; Peeken et al., 2018; Cunningham et al., 2020). It is estimated that the global abundance of plastic particles currently ranges from approximately 82–358 trillion pieces, with a total mass between 1.1 and 490 million tons (Eriksen et al., 2023). Marine microplastics have the capacity to adsorb a variety of organic pollutants (Koelmans et al., 2013), heavy metals (Li and Wang, 2023), and harmful microorganisms (Arias-Andres et al., 2019), posing potential risks to the marine ecological environment. Microplastics are readily ingested by marine organisms and can be transferred and accumulated through the food chain, posing a potential threat to human health (Carbery et al., 2018; Qiao et al., 2019). Approximately 75% of microplastics in the marine environment originate from land-based anthropogenic activities (Morales-Caselles et al., 2021) and are transported over long distances from coastal zones to the open ocean under the influence of oceanic and coastal currents.

Microplastic research primarily relies on *in situ* sampling, and the accuracy of observational data is highly dependent on the sampling process. Some observed results may be incidental or influenced by sampling variability (Liu et al., 2019). Moreover, long-term, real-time monitoring of microplastics at fixed locations remains challenging under current technological constraints (Yin et al., 2022). In recent years, the development and continuous refinement of ocean numerical models have provided an effective new approach for investigating oceanic physical processes, material transport, and pollutant dispersion. The primary modeling approaches for simulating microplastic transport include Eulerian models and the Lagrangian particle tracking models (Zhang, 2017). In the context of Eulerian approaches, Mountford and Morales Maqueda (2019) developed a three-dimensional model to reveal the vertical distribution of microplastics throughout the water column. Van Sebille et al. (2020) demonstrated that the interaction between Stokes drift and the Eulerian flow field significantly influences microplastic transport, and emphasized the need for future studies to couple wave and ocean models for more comprehensive investigation. In addition, backward particle tracking methods have been employed to trace the potential sources of microplastics (Sun et al., 2022). In the context of the Lagrangian particle tracking, Lebreton et al. (2012) coupled global ocean circulation with particle tracking to identify five major accumulation zones. Mansui et al. (2015) integrated the NEMO ocean model with a Lagrangian particle tracking approach to analyze the influence of Mediterranean circulation on microplastic transport. Iwasaki et al. (2017) employed particle tracking to reveal that microplastics in the Sea of Japan are transported northeastward under the influence of the Tsushima Current. Liu et al. (2023) utilized the Regional Ocean Modeling System (ROMS) model coupled with a Lagrangian transport model to investigate the seasonal and river discharge–driven characteristics of microplastic transport in Chinese coastal waters. The horizontal transport of microplastics is primarily driven by ocean circulation, wind forcing, and tides. Previous studies have indicated that ocean currents, winds, and river discharges facilitate the long-distance transport of microplastics (Zarfl and Matthies, 2010; Cole et al., 2011). Pan et al. (2019) revealed four distinct transport pathways under the combined influence of wind and current fields. In addition to hydrodynamic control, the vertical migration of microplastics is significantly affected by processes such as settling and resuspension. The particle density, shape, and size determine their distribution characteristics within the water column (Kumar et al., 2021); the sinking velocity is constrained not only by density and fluid properties but also by particle morphology (Kowalski et al., 2016). Long-term observations have shown that biofouling can alter particle density, leading to an increase in sinking rate over time (Karkanorachaki et al., 2021). Collectively, these processes govern the vertical migration pathways of microplastics in the water column. Reisser et al. (2015) observed an exponential decrease in microplastic abundance with depth, with most particles concentrated near the surface. Eriksen et al. (2014) suggested that microplastics eventually settle on the seafloor, while Kukulka et al. (2012) pointed out that wave-enhanced turbulence also influences their vertical distribution.

In recent years, research on microplastic pollution in China's coastal waters has increased steadily (Zhao et al., 2014; Zhao et al., 2019; Sun et al., 2022; Chen et al., 2023; Gao et al., 2023; Ding et al., 2025). The Beibu Gulf, characterized by intensive coastal human activities, has experienced increasingly severe microplastic contamination (Zhu et al., 2023). Studies indicate that microplastics in the water of the Beibu Gulf are predominantly fragments, whereas fibers dominate in sediments, with higher abundances observed near urban areas. Microplastic concentrations are positively correlated with population density and economic development (Zhu et al., 2022). In seawater, microplastics are mainly white fibers, with terrestrial sources being dominant; the primary origins are household products and textiles, and nearshore waters exhibit higher concentrations than offshore regions (Zhu et al., 2023; Zhu et al., 2025). Sediments also contain widespread microplastics, whose distribution is jointly influenced by hydrodynamics, geological conditions, and anthropogenic activities, with the presence of highly toxic polymers further increasing ecological risks (Wu et al., 2025). The hydrodynamics of the Beibu Gulf are complex, with pronounced seasonal circulation patterns: cyclonic circulation dominates in spring; summer circulation exhibits cyclonic patterns in the north and anticyclonic patterns in the south; autumn and winter are characterized by a dual north-south cyclonic structure, with the cold-water mass forming in spring, peaking in summer, and dissipating in autumn (Gao et al., 2017). The primary water masses include the Coastal Current (CC), West Guangdong Coastal Current (WGCC), and South China Sea Water (SCSW), whose seasonal intrusions regulate salinity and nutrient distributions within the gulf (Lao et al., 2022). Although previous studies have investigated microplastic pollution and hydrodynamics in the Beibu Gulf, the coupling between microplastic distribution and hydrodynamic processes remains limited. The mechanisms governing microplastic dispersion, transport, and seasonal variability are still unclear. Moreover, the Beibu Gulf is frequently subjected to typhoons, with storm surges exerting a significant regulatory influence on regional material transport. Previous studies have demonstrated that typhoons disrupt thermal stratification and induce upwelling, thereby reshaping nutrient sources and distributions (Lao et al., 2023). The intense hydrodynamic disturbances triggered by storm surges are likely to further facilitate the migration and dispersion of microplastics, warranting further investigation through numerical modeling. In this study, a three-dimensional hydrodynamic model coupled with a Lagrangian particle-tracking module is developed to elucidate the spatial distribution patterns of microplastics in the Beibu Gulf and to further explore the roles of vertical transport and storm surges in governing their migration and dispersion.

2 Methods

2.1 Study area

The Beibu Gulf is located in the northwestern part of the South China Sea and is a semi-enclosed bay bordered by land on three sides. The seafloor gradually deepens from the nearshore zones toward the central region, with an average depth of approximately 40 m and a maximum depth of less than 100 m. This region is characterized by a pronounced monsoonal climate, with prevailing southwesterly winds in summer and northeasterly winds in winter (Shen et al., 2018), and features persistent northward subtidal currents maintained by tidal-induced residual currents (Liu et al., 2025). This study focuses on the northern Beibu Gulf, where the study area encompasses five representative bays and seven major rivers that discharge into the sea (Figure 1). Rivers provide significant land-based inputs to coastal bays (Kanhai et al., 2022), and serve as critical conduits linking terrestrial and marine systems. They play a key role in transporting nutrients, organic matter, and microplastics, and represent the primary pathway through which microplastics are delivered from land to the ocean (Deng et al., 2025). River runoff in the Beibu Gulf exhibits pronounced seasonal variability (Figure 2). According to river discharge monitoring data provided by the Marine Environmental Monitoring Center Station, the Nanliu River has the highest annual average discharge among the seven major rivers entering the sea (Figure 2a). The Nanliu River enters its high-flow period between August and September, with the peak discharge occurring in September at approximately 840 m^3/s. The low-flow period extends from January to March, with the lowest discharge observed in March. The other six rivers exhibit relatively lower discharge volumes and show similar intra-annual variation patterns, with their high-flow periods mainly concentrated between June and October.

2.2 Numerical model

2.2.1 Governing equation

This study employs a three-dimensional hydrodynamic model coupled with a Lagrangian particle tracking module. The hydrodynamic component is based on the solution of the three-dimensional incompressible Reynolds-averaged Navier-Stokes equations, under the assumptions of Boussinesq approximation and hydrostatic balance. The model satisfies both the continuity and momentum equations, as shown in Equations 1–4:

$$\frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z} = 0 \tag{1}$$

$$\frac{\partial u}{\partial t} + \frac{\partial u^2}{\partial x} + \frac{\partial vu}{\partial y} + \frac{\partial wu}{\partial z} = fv - g\frac{\partial \eta}{\partial x} - \frac{1}{\rho_0}\frac{\partial p_a}{\partial x} + F_u + \frac{\partial}{\partial z}\left(\nu_t \frac{\partial u}{\partial z}\right) \tag{2}$$

$$\frac{\partial v}{\partial t} + \frac{\partial v^2}{\partial y} + \frac{\partial uv}{\partial x} + \frac{\partial wv}{\partial z} = -fu - g\frac{\partial \eta}{\partial y} - \frac{1}{\rho_0}\frac{\partial p_a}{\partial y} + F_v + \frac{\partial}{\partial z}\left(\nu_t \frac{\partial v}{\partial z}\right) \tag{3}$$

$$\frac{\partial p}{\partial z} = -\rho_0 g \tag{4}$$

where t is time; x, y, z are Cartesian coordinates; η is the water surface elevation; d is the still water depth; $h = \eta + d$ is the total water depth; u, v, w are the velocity components in the x, y, z directions, respectively; f is the Coriolis parameter; g is the gravitational acceleration; ρ is the water density; ν_t is the vertical eddy viscosity coefficient; P_a is the atmospheric pressure; ρ_0 is the reference water density and F_u, F_v denote the horizontal stress terms represented using stress gradients.

The fundamental principle of the particle-tracking model is to assume that particles are transported under the influence of natural forces such as water currents or wind, with diffusion represented by the addition of a stochastic random-walk term. The differential form of the governing equation is shown in Equation 5:

$$dX_t = a(t, X_t)dt + b(t, X_t)\xi_t dt \tag{5}$$

Where X_t represents the system state variable, such as particle position or concentration; a (t, X_t) is the drift term, describing

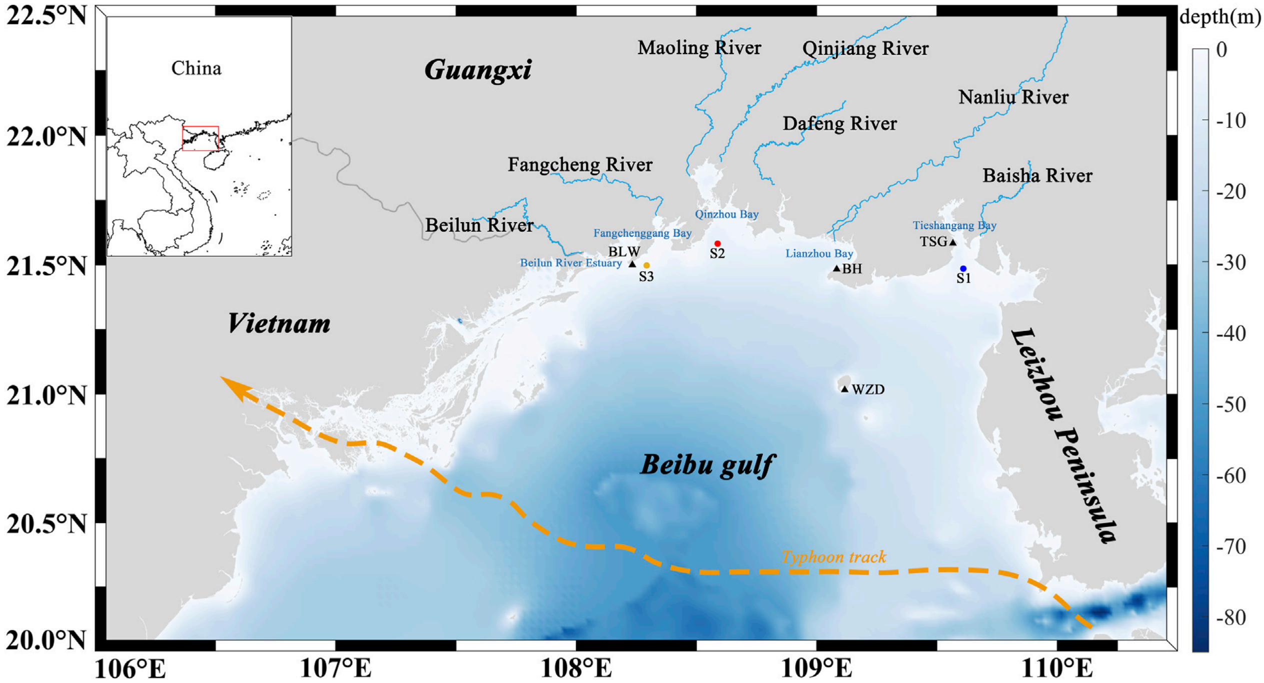

FIGURE 1
Study area and distribution of rivers and bays. The orange arrow indicates the track of Typhoon Yagi (international ID: 2411). Triangles denote tide gauge stations, and circles represent surface drifter release sites.

the mean rate of change of the system, with units of $[X_t]/[t]$; $b\,(t, X_t)$ is the diffusion coefficient, controlling the magnitude of the stochastic perturbation; and ξ_t represents a random perturbation or noise, such as the random forces generated by molecular collisions in Brownian motion. When multiplied by $b\,(t, X_t)$ and applied over dt, ξ_t has the same units as dX_t, together describing the evolution of the system under both deterministic drift and stochastic forcing.

2.2.2 Numerical solution

The spatial discretization of the computational domain was performed using the Finite Volume Method, in which the continuous domain is subdivided into non-overlapping triangular elements as shown in Equation 6.

$$\frac{\partial U}{\partial t} + \nabla \cdot F(U) = S(U) \tag{6}$$

where U is the conserved physical variable vector, F is the flux vector, and S is the source term.

Spatial discretization was performed using a second-order scheme, with spatial accuracy enhanced through a linear gradient reconstruction technique. The gradients were computed using the weighted averaging method proposed by Jawahar and Kamath (2000). To suppress numerical oscillations inherent in high-order schemes, a second-order total variation diminishing (TVD) scheme was employed for flux limiting.

The time integration of the three-dimensional model was performed using a semi-implicit scheme, in which the horizontal terms were treated implicitly, while the vertical terms were handled using fully implicit, partially explicit, and partially implicit approaches. In general, a semi-implicit equation can be expressed as Equation 7:

$$\frac{\partial U}{\partial t} = G_h(U) + G_v(BU) = G_h(U) + G_v^I(U) + G_v^V(U) \tag{7}$$

In the equation, the subscripts h and v refer to the horizontal and vertical terms, respectively, while the superscripts I and V denote inviscid and viscous components.

We employed a high-order scheme for the three-dimensional transport equation, as shown in Equations 8 and 9:

$$\begin{aligned} U_{n+1/2} - \frac{1}{4}\Delta t\left(G_V^V(U_{n+1/2}) + G_V(U_n)\right) \\ = U_n + \frac{1}{2}\Delta t G_h(U_n) + \frac{1}{2}\Delta t G_V^I(U_n) \end{aligned} \tag{8}$$

$$\begin{aligned} U_{n+1/2} - \frac{1}{2}\Delta t\left(G_V^V(U_{n+1/2}) + G_V^V(U_n)\right) \\ = U_n + \Delta t G_h(U_{n+1/2}) + \Delta t G_V^I(U_{n+1/2}) \end{aligned} \tag{9}$$

The horizontal and vertical advection terms were integrated using a second-order Runge-Kutta scheme, while the vertical term was integrated using a second-order implicit trapezoidal rule.

The boundary conditions include the following: at land or solid boundaries, the normal velocity is set to zero to prevent fluid penetration; at open boundaries, time-varying water levels or discharge are imposed to represent tidal or runoff forcing; in areas with alternating wet and dry conditions, a dynamic wetting–drying algorithm proposed by Zhao et al. (1994) and Sleigh et al. (1998) is applied. In this approach, cells are classified

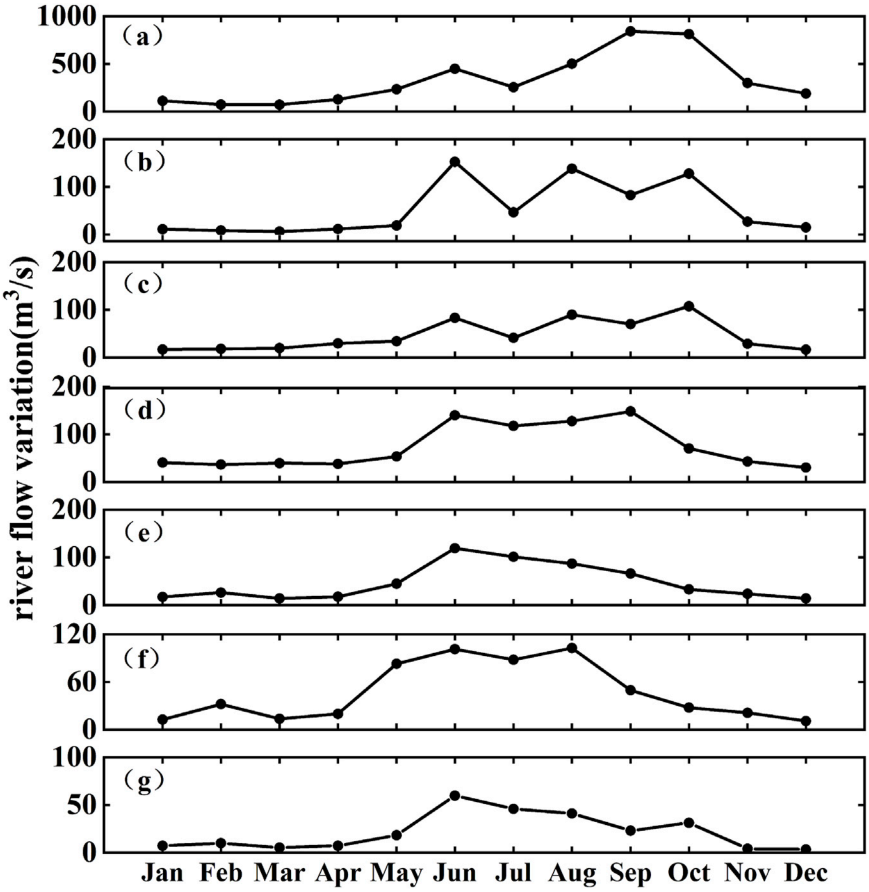

FIGURE 2
Monthly average river discharge time series in the Beibu gulf region in 2023. **(a)** Nanilu river. **(b)** Dafeng river. **(c)** Qinjiang river. **(d)** Maoling river. **(e)** Fangcheng river. **(f)** Beilun river. **(g)** Baisha river.

as dry, partially wet, or wet based on local water depth h, with corresponding flux calculation rules applied, ensuring that $h_{dry} < h_{flood} < h_{wet}$. Cells with water depth below the dry threshold are excluded from computations, while in shallow or transitional areas, only the continuity equation is solved.

2.2.3 Model configuration

The model was configured with a time step of 180 s. The horizontal computational domain was discretized using an unstructured triangular grid, and the vertical direction was divided into three layers after applying the σ-coordinate transformation (Haidvogel et al., 1991). A total of 197,859 grid cells were generated within the computational domain, with spatial resolution gradually transitioning from 60 m near the coastline to 5,000 m at the open boundary, balancing the need for nearshore detail and large-scale dynamic representation. To systematically investigate the seasonal dispersion characteristics of microplastics, the model was independently simulated for each of the four seasons. The initial condition was set to zero, and each simulation was run for 90 days, including a 24-h spin-up time to ensure the stability of the model dynamics. The spatial distribution of microplastics for each season was extracted at the end of the simulation period. In the model, rivers were represented as point sources, while no microplastic flux was prescribed at the open ocean boundaries. The model was driven by tidal forcing conditions. The tidal elevations at each grid node along the open boundary were obtained from the MIKE Global Tide Model, which determines the harmonic constants based on 8 major tidal constituents (M2, S2, N2, K2, K1, O1, Q1 and P1). The tidal elevation is calculated as Equation 10:

$$\eta = \sum_{i=1}^{8} f_i h_i \cos\left(\omega_i t + v_{0i} + u_i - g_i\right) \tag{10}$$

Where η represents the predicted tidal elevation at the open boundary; f_i is the nodal factor of the ith tidal constituent; h_i and g_i denote the amplitude and phase lag of the ith constituent,

TABLE 1 Model parameter setting.

Process	Unit	Parameter	References
Decay	% per day	0.01	Andrady (2011)
Settling	m/s	0.004	Karkanorachaki et al. (2021)
Erosion	N/m^2	0.01	Zhao et al. (2024)

TABLE 2 The riverine input flux of microplastics (unit: mg/s).

River	Spring	Summer	Autumn	Winter
Beilun River	11.96	29.96	10.11	5.73
Fangcheng River	7.54	30.01	12.02	5.66
Maoling River	0.94	3.01	1.59	0.89
Qinjiang River	1.24	3.13	3.02	0.77
Dafeng River	0.60	5.38	3.80	0.57
Nanliu River	5.22	14.42	23.33	4.54
Baisha River	0.57	2.68	1.07	0.38

respectively; ω_i is the angular frequency of the ith constituent; and $v_{0i} + u_i$ represents its astronomical argument.

Bathymetric data were obtained from NOAA (https://www.ngdc.noaa.gov/) and supplemented with local nautical charts. Atmospheric forcing was derived from ERA5 reanalysis data provided by ECMWF (https://www.ecmwf.int/), with hourly 10 m wind speed and mean sea level pressure from a single vertical level extracted as inputs. In the particle tracking module, this study parameterized the degradation, settling, and resuspension processes of microplastics (Table 1). Observational data were obtained during two research cruises in the Beibu Gulf in 2020, namely, "Yueke 1" and "Yuexiayuzhi 20028," yielding a total of 75 seawater samples and 66 surface sediment samples. Seawater samples were collected using a Manta trawl equipped with a 330 μm nylon mesh, while surface sediment samples (sampling depth >5 cm) were obtained with a Van Veen grab sampler. In the model, the microplastic input concentrations from riverine point sources were set based on measurements from sampling sites near the estuaries. Based on field measurements, this study selected the dominant type of microplastic in the Beibu Gulf—polystyrene (PS) fragments for analysis. The fluxes of microplastic inputs were estimated using observed microplastic abundances, particle volume and density, and river discharge (Table 2). The model outputs are presented in terms of concentration (μg/m^3). For the storm surge simulations, the 2024 super typhoon "Yagi" (formation: 1 September 2024; dissipation: 8 September 2024) was selected as the study case, with relevant data obtained from the China Meteorological Administration Typhoon Network (https://typhoon.nmc.cn/web.html). Based on this typhoon event, the numerical model was employed to investigate the mechanisms by which storm-induced hydrodynamic processes influence microplastic dispersion and transport.

2.3 Model validation data

Tidal level data were validated using observations from four tide gauge stations located within the Beibu Gulf (Figure 1). In addition, to verify particle transport trajectories, three surface drifters were deployed in the Beibu Gulf in June 2025 to collect in surface current and transport path data for comparison with the model results (Table 3). Each drifter consisted of a float, a positioning system, and a control mechanism, and was equipped with a real-time positioning module supporting both GPS and BeiDou satellite systems. The floating depth was adjusted by regulating the internal water volume to ensure that the drifters remained in the surface layer of the water column. Drifter trajectories were recorded at 3-min intervals, providing validation data for the simulation of microplastic transport.

2.4 Assessment of Lagrangian particle tracking performance

In this study, the performance of the Lagrangian particle tracking model was evaluated using the correlation coefficient (R), mean absolute error (MAE), root mean square error (RMSE), and relative separation distance (RSD) (Hu et al., 2023; Kang and Xia, 2025). The correlation coefficient (R) is used to measure the linear relationship between the simulated and observed values, reflecting the model's ability to capture temporal trends. The mean absolute error (MAE) characterizes the average magnitude of absolute deviations between simulated and observed values, providing an intuitive assessment of overall error. The root mean square error (RMSE), which places greater emphasis on larger deviations, quantifies the overall level of discrepancy. The relative separation distance (RSD) evaluates the spatial agreement between simulated and observed trajectories, serving as an indicator of the model's capability to reproduce particle spatial distribution as shown in Equations 11–14.

$$R = \frac{\sum_{i=1}^{n} (o_i - \bar{o})(m_i - \bar{m})}{\sqrt{\sum_{i=1}^{n} (o_i - \bar{o})^2} \cdot \sqrt{\sum_{i=1}^{n} (m_i - \bar{m})^2}} \tag{11}$$

$$MAE = \frac{1}{n}\sum_{i=1}^{n} (o_i - m_i) \tag{12}$$

$$RMSE = \sqrt{\sum_{i=1}^{n} (o_i - \bar{o})^2 \Big/ n} \tag{13}$$

$$RSD = \frac{d_n}{\sum_{i=1}^{n} l_{oi}} \tag{14}$$

TABLE 3 Surface drifter deployment information.

Label	Release location			Release time (UTC)	
	Bay	Lon	Lat	Start time	End time
S1	Tieshangang Bay	109.61°	21.49°	Jun.6 12:06,2025	Jun.6 23:27,2025
S2	Qinzhou Bay	108.59°	21.58°	Jun.11 10:32,2025	Jun.12 08:20,2025
S3	Fangchenggang Bay	108.29°	21.50°	Jun.11 08:53,2025	Jun.12 11:26,2025

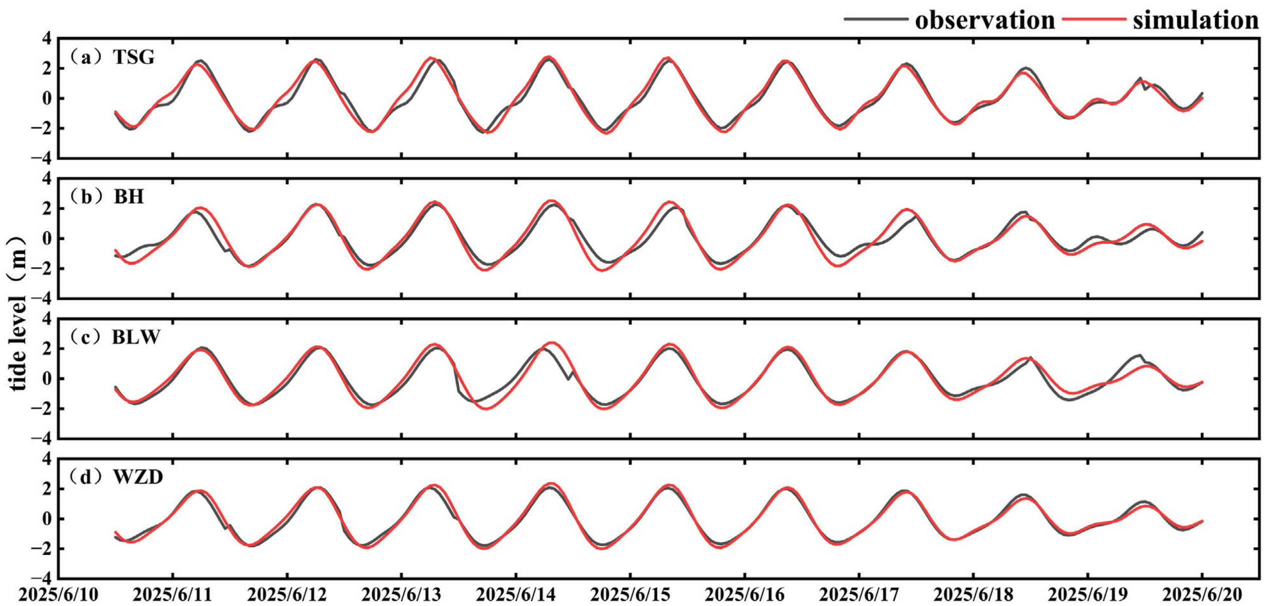

FIGURE 3
Tidal data validation. **(a)** Located at Tieshangang; **(b)** Located at Beihai; **(c)** Located at Fangchenggang; **(d)** Located at Weizhou Island.

Where o represents the observed data, m denotes the model output, d_n is the positional distance between the modeled particle and the drifter, and l_{oi} is the total displacement of the drifter.

3 Results and discussion

3.1 Comparison between drifter trajectories and simulated particle tracking paths

In June 2025, three surface drifters were deployed in the Beibu Gulf to obtain nearshore drift characteristics. The first drifter (ID: S1, blue in Figure 1) was released in Tieshan Bay at 12:06 UTC on June 6, and was tracked for approximately 13 h. Due to grounding in shallow waters, the usable data were limited. The total drift distance was 5.43 km, with a general northwestward trajectory. The drifter exhibited oscillatory movement under tidal forcing, with a maximum and mean drift velocity of 0.49 m/s and 0.14 m/s, respectively. The second drifter (ID: S2, red in Figure 1) was deployed in the outer Qinzhou Bay at 10:32 UTC on June 11, and was tracked for approximately 1 day. It drifted predominantly from northeast to southwest, covering a total distance of 21.82 km, with maximum and average velocities of 0.67 m/s and 0.28 m/s, respectively. The third drifter (ID: S3, orange in Figure 1) was deployed in Fangcheng Bay at 08:53 UTC on June 11, with a tracking duration of about 1 day. Its drift path also followed a northeast-to-southwest direction, with a total displacement of 19.12 km. The maximum and mean drift speeds were 0.44 m/s and 0.26 m/s, respectively.

To evaluate the reliability of the particle tracking model, we first validated the simulated tidal levels against observations from four tide gauge stations (Figure 3). Subsequently, the modeled particle drift trajectories were compared with the observed paths of the three surface drifters (Figure 4). For drifter S1, the simulated drift speed showed good correlation with the observed values, with an R value of 0.6900, MAE of 0.0176 m/s, and RMSE of 0.0624 m/s (Figure 4a). The correlation coefficients for the u and v-components were 0.5090 and 0.8897, with corresponding MAE values of 0.0322 m/s and 0.0117 m/s, and RMSE values of 0.0859 m/s and 0.0631 m/s (Figures 4a3, 4a4). The maximum spatial deviation was 1.43 km, and the relative separation distance (RSD) was 6.6% (Figure 4a2). For drifter S2, the modeled drift speed was well reproduced, with R = 0.8675, MAE = 0.0096 m/s, and RMSE = 0.0817 m/s (Figure 4b). The u and v-component R values were 0.7678 and 0.9016, with MAE values of 0.0197 m/s and 0.0661 m/s, and RMSE values of 0.09387 m/s and 0.1135 m/s (Figures 4b3, 4b4). The simulated trajectory closely matched the observed path during the initial phase, but the deviation increased during the flood tide due to localized hydrodynamic influences. The maximum spatial deviation reached 5.70 km, with an RSD of 6.1% (Figure 4b2). For drifter S3, the simulation also demonstrated strong agreement with the observations. The correlation coefficient for drift speed was 0.6803, with MAE and RMSE values of 0.0370 m/s and 0.0875 m/s, respectively (Figure 4c). The R values for the u and v-components were 0.4757 and 0.6731, with MAE values of 0.0427 m/s and 0.0250 m/s, and RMSE values of 0.1126 m/s and 0.1249 m/s (Figures 4c3, 4c4). In the Fangchenggang Bay region, the simulated trajectory exhibited overall good agreement with the observed path, with a maximum spatial deviation of 4.31 km and an RSD of 5.2% (Figure 4c2).

As Lagrangian particle tracking is highly sensitive to initial conditions and boundary forcing, and particle trajectories respond strongly to environmental perturbations, many studies have highlighted the challenges associated with its validation (Havens et al., 2010; Wei et al., 2016; Khatmullina and Chubarenko, 2019). Given the highly dynamic and complex

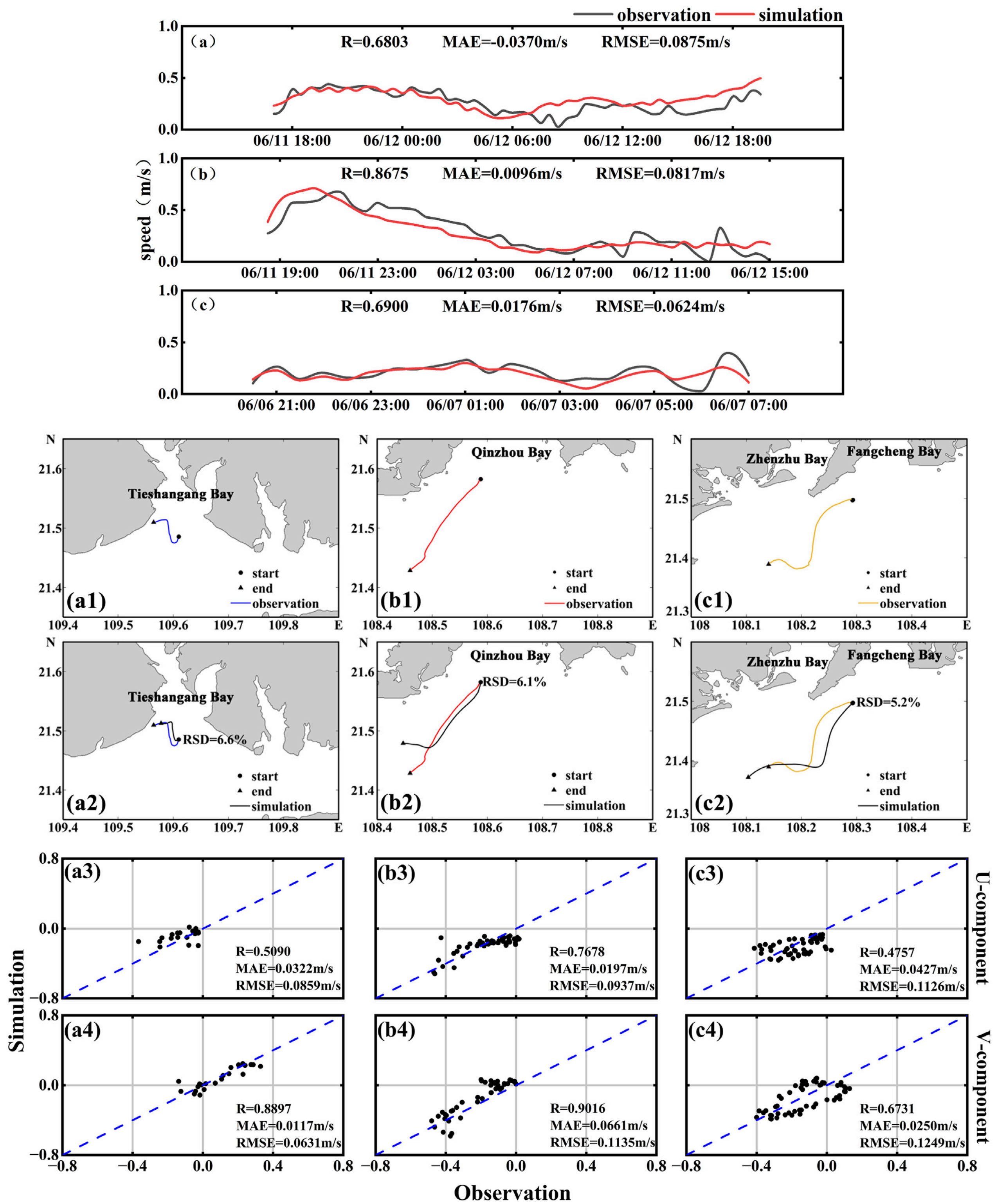

FIGURE 4
(a–c) Comparisons between observed and simulated velocities for surface drifters S1-S3. For each drifter: (a1-a4) correspond to S1; (b1-b4) to S2; and (c1-c4) to S3. The subplots show the observed trajectory, trajectory comparison, and scatter plots of velocity components.

nature of the marine environment, accurately reproducing the trajectory of an individual drifter remains challenging. The particle-tracking model employed in this study, which explicitly accounts for key hydrodynamic mechanisms such as tidal variability and wind-induced forcing, demonstrates reasonable skill in replicating drift velocities, directions, and trajectories, thereby exhibiting a credible level of reliability. However, the surface drifters used in this study were constrained by short tracking

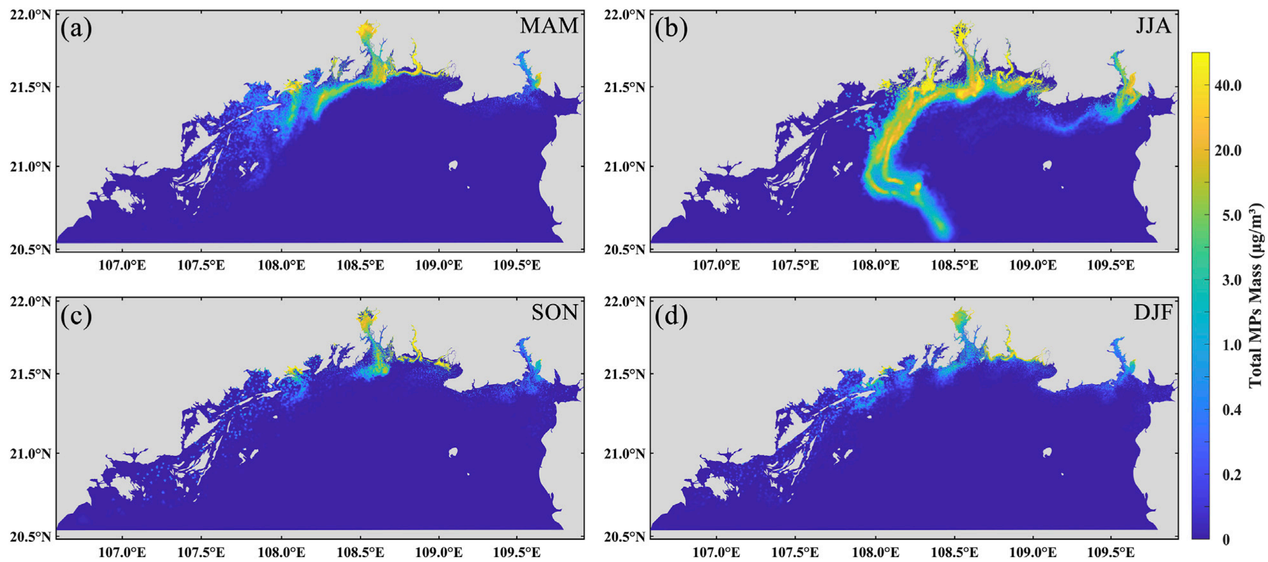

FIGURE 5
Spatial distribution of microplastics by season. **(a)** Spring MAM (March-May); **(b)** Summer JJA (June-August); **(c)** Autumn SON (September-November); **(d)** Winter DJF (December-February).

durations, preventing full coverage of all stages of microplastic transport. This limitation restricts the ability to comprehensively evaluate long-term transport pathways and the cumulative effects of external disturbances, potentially introducing uncertainty when characterizing the prolonged migration and accumulation of microplastics. In addition, temporal mismatches between the simulation period and the corresponding observational year may lead to discrepancies between model results and measurements. To enhance the applicability and reliability of the model for large-scale and long-term processes, future studies should consider incorporating longer time series of drifter observations or integrating other long-term monitoring data to achieve greater consistency in external forcing conditions, thereby improving the robustness and reliability of the simulation results. Despite these limitations, comparative analyses between model simulations and observed drifter trajectories demonstrate that the model achieves a high degree of accuracy in reproducing regional particle transport patterns. These results provide both robust technical support and a theoretical foundation for advancing the understanding and prediction of drift dynamics in the Beibu Gulf.

3.2 Seasonal characteristics of microplastic distribution

The maximum dispersion distance of microplastic pollution plumes increases over time in all seasons, but exhibits significant seasonal variation (Figure 5). In spring, autumn, and winter, the transport patterns are relatively similar, with microplastics predominantly dispersing southwestward along the coast. In spring, microplastics enter the gulf from river estuaries, with part of the particles continuing westward along the western shoreline, while others spread toward the central and northeastern parts of the gulf (Figure 5a). In autumn, the transport pathway shifts back to a primarily southwestward direction (Figure 5c). During winter, under the influence of prevailing northeasterly monsoons, with microplastics transported westward along the Chinese coast and then southward along the Vietnamese coast (Figure 5d). In contrast, summer is characterized by the influence of southwesterly monsoons, which drive microplastics offshore and northward, resulting in an offshore transport path toward the central Beibu Gulf (Figure 5b). The maximum transport distance in winter is approximately 180 km—the shortest among all four seasons. Autumn shows a similar extent of transport, with a maximum dispersion distance of around 170 km. In comparison, spring exhibits faster and broader microplastic dispersion, with particles reaching up to approximately 205 km from the source. In summer, although the maximum distance (~185 km) is shorter than that in spring, high-concentration zones are rapidly carried offshore over a short time span. Both winter and autumn are dominated by northeasterly monsoons. Microplastics are transported southwestward along the Guangxi coastline and gradually accumulate along the Vietnamese coast. High-concentration zones are primarily located from the estuaries toward the southwestern part of the gulf and along the Vietnam shoreline, while low-concentration areas extend into the southwestern open sea. In contrast, spring and summer are influenced by southwesterly monsoons. Unlike the nearshore-dominated transport observed in winter and autumn, microplastic movement in these seasons is primarily directed offshore and toward the central gulf from the river mouths. The areal extent of high-concentration microplastic pollution zones varies significantly by season. The largest extent occurs in summer, covering approximately 4,968 km^2 followed by spring at around 2,666 km^2 (Table 4). Autumn and winter exhibit smaller affected areas, approximately 1,278 km^2 and 2,006 km^2 (Table 4), respectively. Distinct seasonal differences in microplastic concentrations are also observed between nearshore (0–20 km) and offshore (>20 km) zones. Concentrations are highest

TABLE 4 Temporal evolution of microplastic dispersion area in each season (unit: km^2).

Season	15 days	30 days	45 days	60 days	75 days	90 days
Spring (MAM)	105	246	411	927	1722	2666
Summer (JJA)	412	827	2431	3,149	4,353	4,968
Autumn (SON)	234	307	455	621	854	1,278
Winter (DJF)	415	743	992	1,432	1767	2006

near riverine sources and lowest in open waters. In winter, spring, and autumn, nearshore concentrations exceed 40 μg/m^3, whereas offshore values are mostly below 1 μg/m^3, with some regions approaching background levels (<0.1 μg/m^3), indicating that pollution plumes are largely confined to coastal zones. In summer, however, in addition to elevated nearshore concentrations, a prominent high-concentration region (~20 μg/m^3) appears between 20 km and 80 km offshore—significantly higher than in other seasons. This offshore enrichment is unique to summer and suggests that microplastics are effectively transported into the open sea. In contrast, offshore concentrations in winter, spring, and autumn remain consistently low, with no evidence of similar accumulation.

The results indicate pronounced seasonal variability in microplastic dispersion within the Beibu Gulf. During winter and autumn, under the influence of the northeast monsoon, microplastics predominantly migrate southwestward along the coast, forming distinct accumulation zones along the Vietnamese shoreline, with a relatively limited dispersal range. In contrast, during spring and summer, when the southwest monsoon prevails, microplastics more readily disperse toward offshore areas and the central gulf, with summer exhibiting mesoscale high-concentration enrichment extending from nearshore to mid-shelf regions. These results indicate that the transport of microplastics in the Beibu Gulf is primarily regulated by ocean circulation, while the monsoon indirectly influences microplastic transport by altering the dynamics of the upper-layer seawater. Analysis of the dispersion maps shows that the overall hydrodynamic characteristics are consistent across the four seasons, with coastal currents exhibiting a predominantly counterclockwise pattern, aligning well with previous hydrodynamic studies in the Beibu Gulf (Gao et al., 2017). Similarly, Wang et al. (2018) reported a summer transport tendency of coastal particles toward the central gulf. Prior studies further indicate that microplastic migration and dispersion in marine environments are jointly controlled by hydrodynamics, monsoonal forcing, and estuarine inputs, exhibiting clear seasonal patterns. For instance, simulations in Chinese coastal waters reveal that floating microplastics achieve the longest transport distances in summer, while remaining predominantly nearshore during autumn and winter (Liu et al., 2023). In the Adriatic Sea, the coastline serves as the primary microplastic deposition sink, with surface concentrations exhibiting significant seasonal variability (Liubartseva et al., 2016). Additional research highlights that microplastics discharged from coastal cities primarily accumulate along shorelines and adjacent islands, with estuarine regions experiencing particularly severe contamination and elevated abundances during the rainy season (Jiang et al., 2022; Li et al., 2023). These findings are consistent with the simulated nearshore high-concentration zones and source-region input patterns in the Beibu Gulf, providing strong support for the proposed seasonal dispersion mechanisms of microplastics in the region.

3.3 Effect of vertical velocity on microplastic transport

To investigate the influence of vertical velocity on the transport and distribution of microplastics, this study selected two representative seasons for analysis: winter, characterized by the minimum dispersion range, and summer, characterized by the maximum dispersion range. To further examine the role of vertical velocity, we used a two-dimensional model to simulate microplastic transport under conditions without vertical flow. The parameter settings of the two-dimensional model are consistent with those of the three-dimensional model to ensure comparability, thereby allowing a clear assessment of the regulatory effect of vertical velocity on microplastic transport. Calculations indicate that the vertical velocity in the study area ranges from −0.0434 m/s to 0.0236 m/s in summer, with a maximum absolute value of 0.0434 m/s, and from −0.0336 m/s to 0.0275 m/s in winter, with a maximum absolute value of 0.0336 m/s. In terms of mean vertical velocity, the range is −0.00036 m/s to 0.00022 m/s in summer and −0.00022 m/s to 0.00029 m/s in winter. Overall, vertical motions are more intense in summer, indicating more active vertical water exchange, which facilitates the vertical resuspension and mixing of microplastics. The simulated distribution patterns (Figure 6) reveal distinct spatial variations and concentration differences in microplastics under different conditions. In winter (Figures 6a,c,e), microplastics are mainly concentrated in Fangchenggang Bay, Qinzhou Bay, Lianzhou Bay, Tieshangang Bay, and adjacent nearshore areas. In the absence of vertical velocity, the overall dispersal area is reduced, with the plume-covered area and maximum offshore transport distance decreasing to 1,359 km^2 and 170 km, respectively—representing reductions of approximately 32.2% and 6.3%. In summer, the spatial distribution of microplastics has shifted significantly westward, with high concentrations no longer concentrated in the central part of the sea (Figures 6b,d,f). When vertical velocity is excluded, the plume-covered area and maximum transport distance decrease to 1,407 km^2 and 177 km, respectively—reductions of approximately 71.7% and 4.2%.

These results suggest that the vertical velocity has a relatively minor influence on the transport direction and distance of microplastics. Although it does not play a dominant role in controlling the transport direction, the vertical velocity

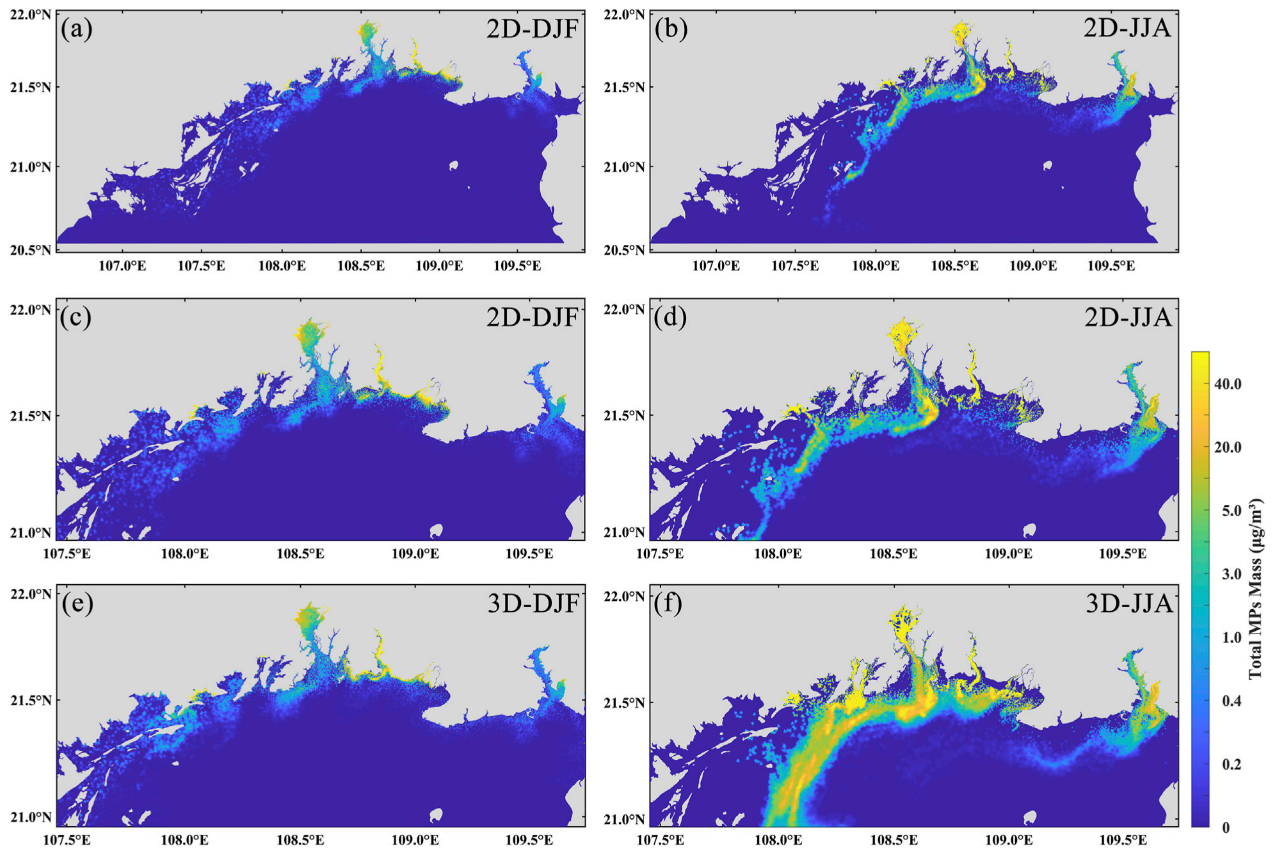

FIGURE 6
Spatial distribution of microplastics in the case of vertical flow velocity (3D) versus no vertical flow velocity (2D). **(a,b)** shows the two-season distribution of the 2D model; **(c,d)** shows the two-season localized distribution of the 2D model; **(e,f)** shows the two-season localized distribution of the 3D model.

significantly affects the dispersion and concentration distribution of microplastics. Moreover, a consistent seasonal-independent pattern is observed: vertical velocity tends to promote the dispersion and spreading of microplastics. During summer, enhanced southwesterly monsoon winds and riverine discharge intensify vertical circulation, bringing deeper microplastics to the surface layer and facilitating their wider dispersal. In addition, the prevailing circulation patterns and coastal upwelling processes in summer further contribute to offshore microplastic transport (Gao et al., 2015; Pang et al., 2025). In winter, strong northeasterly winds can induce intense vertical mixing (Gao et al., 2017), which drives surface microplastics in nearshore areas downward and offshore, resulting in lower surface concentrations compared to conditions without vertical flow.

3.4 Impact of storm surges on microplastic transport

We further investigated another key factor influencing microplastic dispersion in the Beibu Gulf—storm surges. Based on the simulation results of Typhoon Yagi, we systematically analyzed the spatial distribution characteristics and evolution mechanisms of microplastics before, during, and after the typhoon, as well as under non-typhoon conditions (Figure 7). Prior to the typhoon event, microplastics were mainly concentrated in nearshore areas such as river estuaries and harbors, which are characterized by strong land-based inputs. A distinct nearshore high-concentration zone and a clear land–sea concentration gradient were observed (Figure 7a). During the typhoon event, strong wind stress, wave-induced turbulence, and alterations in large-scale circulation markedly modified microplastic transport and dispersion, leading to accumulation along the coastline and the formation of filamentous or patchy high-concentration bands (Figure 7b). Following the typhoon, the transport distance of microplastics increased substantially, with elevated concentrations observed west of the Beilun River and along the Vietnamese coast, and an overall transport pattern indicating southwestward dispersal along the coastline (Figure 7c). Compared to the distribution under normal hydrodynamic conditions (Figure 7d), the typhoon significantly enhanced the spatial redistribution of microplastics, highlighting its strong facilitative effect on offshore transport. Under typical hydrodynamic conditions, the limited circulation in the Beibu Gulf constrains offshore microplastic transport. The typhoon-induced storm surge disrupted this limitation, substantially intensifying hydrodynamic conditions: prior to the typhoon, tidal forces dominated circulation with maximum velocities of

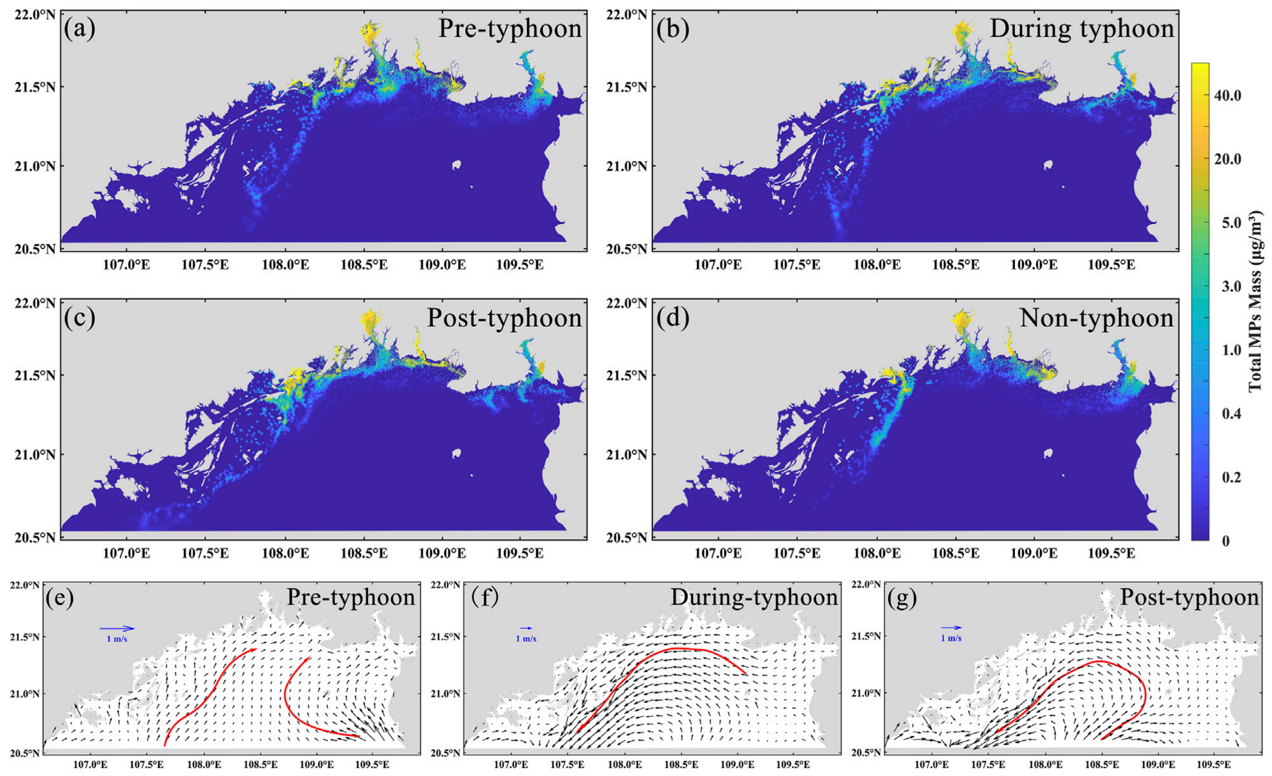

FIGURE 7
Simulated spatial distribution of microplastics in Beibu Gulf during Typhoon Yagi in 2024. **(a-d)** are the distributions of microplastics before, during, after, and without the typhoon, respectively; **(e–g)** are the flow fields in the Beibu Gulf before and after the typhoon; the red arrows show the main direction of the flow field at that moment.

approximately 0.56 m/s, whereas during the typhoon, storm-driven flow velocities reached up to 2.8 m/s. This pronounced increase in flow speed accelerated the dispersal of microplastics toward offshore waters, extending their transport distance by approximately 68 km compared to non-storm conditions, thereby significantly altering the spatial distribution patterns of microplastics in the Beibu Gulf.

Further analysis reveals that the typhoon event exerted a significant influence on the hydrodynamic characteristics of the Beibu Gulf. Prior to the typhoon, circulation was primarily tide-driven, with flow directions varying periodically in response to the tidal cycle. During the mid and late stages of the typhoon, however, storm-induced intense circulation generated a pronounced counterclockwise vortex within the gulf (Figures 7f,g). This vortex not only altered transport pathways but also had a substantial impact on microplastic distribution: a portion of the microplastics accumulated along the coastline under the vortex's influence, while another portion was advected offshore, exhibiting clear spatial redistribution. The dominant mechanisms driving microplastic transport during the storm surge comprised variations in surface flow velocity, wind-stress–induced convection and vertical mixing (Nel et al., 2018; Prevenios et al., 2018), vortex circulation structures, and the cumulative effects of storm surge dynamics. The formation of offshore high-concentration zones after the typhoon is largely attributed to the counterclockwise mesoscale eddies generated during the storm event. Following the typhoon, nearshore circulation exhibited a reversal in flow direction. On one hand, this reversal transported part of the microplastics back toward the coast; on the other, it facilitated the formation of secondary accumulation zones in semi-enclosed or leeward areas. In addition, intense storm surges can induce the resuspension of microplastics embedded in marine sediments, introducing them back into the surface water layer. This process expands the spatial distribution of microplastics while enhancing their suspension and persistence in the water column (Lo et al., 2020), thereby exacerbating the overall environmental load. Extreme weather events such as typhoons not only alter the short-term distribution patterns of microplastics, but also have the potential to reshape their migration pathways among the water column, seabed sediments, and coastal zones, thereby significantly affecting the regional fate of microplastics.

4 Conclusion

In this study, a three-dimensional numerical model coupled with a Lagrangian particle tracking module was employed to simulate the migration and dispersion of microplastics in the Beibu Gulf. The model's reliability was assessed through comparison with observed drifter trajectories. The simulation results revealed the seasonal characteristics of riverine microplastic transport, as well as the effects of vertical velocity and storm surges. The main conclusions are as follows:

1. Microplastic transport exhibits pronounced seasonal variability. Driven by the counterclockwise coastal current, microplastics predominantly disperse southwestward throughout all four seasons. In spring, autumn, and winter, pollution plumes are concentrated near estuarine zones and along the Vietnamese coast. In summer, under the influence of the southwest monsoon, transport pathways shift toward the central gulf, substantially expanding the affected region and generating high-concentration zones.
2. Vertical velocity has a significant impact on the spatial distribution of microplastics. Although its influence on transport distance is limited, vertical flow markedly alters the coverage area of the pollution plume and the location of high-concentration zones.
3. Using Typhoon Yagi as an example, the storm surge rapidly altered the spatial distribution of microplastics in the Beibu Gulf. Prior to the typhoon, microplastics were mainly concentrated in nearshore waters; during the event, strong wind stress and circulation forced their accumulation along the coastline while driving dispersion offshore. After the typhoon, transport distances increased, with an overall southwestward dispersal pattern. Compared to non-typhoon conditions, the storm surge enhanced the redistribution capacity of microplastics, promoted offshore dispersion, and reshaped their transport pathways within the gulf through vortex-driven circulation.

However, this study still has certain limitations and does not fully capture the distribution characteristics of microplastics in real marine environments. The source input settings considered only riverine inputs, and the use of single-point microplastic observation data cannot accurately represent the actual transport flux from rivers, thus affecting the accuracy of the input data. In addition, the parameterization of processes such as microplastic settling, degradation, and resuspension in the Lagrangian particle tracking model remains simplified, limiting the representation of the complex biogeochemical behavior of microplastics. These limitations suggest that future research should incorporate multi-source observational data to improve the estimation of input fluxes and further introduce dynamic processes involving interactions between microplastics and environmental factors, in order to enhance the accuracy and applicability of simulation results.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary Material, further inquiries can be directed to the corresponding authors.

Author contributions

CS: Data curation, Formal Analysis, Methodology, Writing – original draft. YG: Data curation, Writing – review and editing, Funding acquisition, Writing – original draft. WH: Writing – review and editing, Data curation, Methodology, Software. HW: Investigation, Writing – review and editing, Data curation. XH: Writing – review and editing, Investigation. HJ: Writing – review and editing, Supervision. QS: Writing – review and editing, Supervision. XZ: Visualization, Writing – review and editing. JC: Visualization, Writing – review and editing. ZZ: Data curation, Funding acquisition, Project administration, Writing – review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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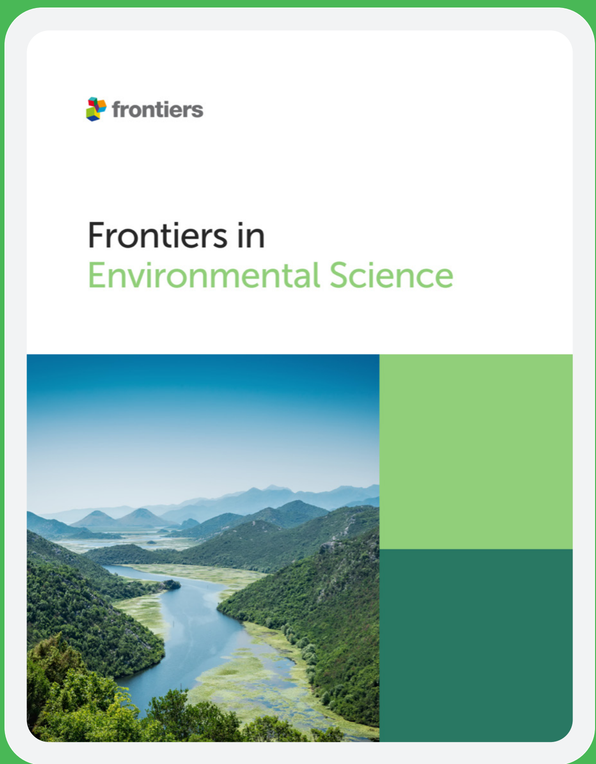

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