Journal of Hazardous Materials 492 (2025) 138078

Contents lists available at ScienceDirect

Journal of Hazardous Materials

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

Colonization time of plastisphere drives the dynamics of organic carbon stability and microbial communities in seagrass bed sediments

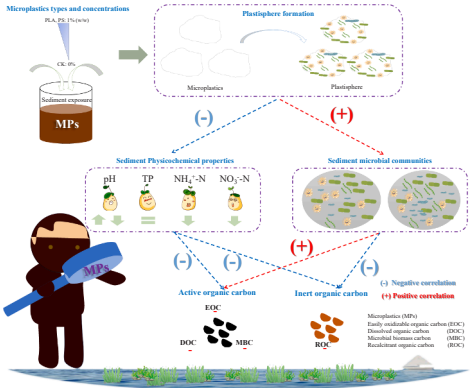
Shuailing Hou [a,b,c], Jun Wang [a,b,c], Dan Feng [a,b,c,*,1,2,3], Dongming Wu [d], Shiquan Chen [e], Jinluo Fan [a,b,c], Mingguang Luo [a,b,c], Hui Deng [a,b,c], Xing Yang [a,b,c], Jiwei Luo [a,b,c], Yuanyuan Zhao [a,b,c], Chengjun Ge [a,b,c]

[a] Key Laboratory of Agro-Forestry Environmental Processes and Ecological Regulation of Hainan Province, Hainan University, Haikou 570228, China
[b] Haikou Key Laboratory of Environmental Toxicology, Hainan University, Haikou 570228, China
[c] School of Environmental Science and Engineering, Hainan University, Haikou 570228, China
[d] Institute of Environment and Plant Protection, Chinese Academy of Tropical Agricultural Sciences, Haikou 571101, China
[e] Hainan Academy of Ocean and Fisheries Sciences, Haikou, Hainan Province 571126, China

HIGHLIGHTS

- The formation and evolution of the plastisphere is governed by polymer type and colonization time.
- Colonization time drives microbial community composition and assembly in the plastisphere and seagrass bed sediments.
- Plastisphere formation significantly influence MBC and organic carbon pool stability of sediments.
- MPs modified soil carbon pools by altering physicochemical properties and microbial communities.

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
Plastisphere
Seagrass bed sediments
Organic carbon stability

ABSTRACT

Microplastic (MP) pollution in seagrass bed ecosystems has emerged as a significant global concern. However, the effects of plastisphere formation on organic carbon pools and microbial communities in these ecosystems remain unknown. We conducted a 56-day microcosm incubation experiment to study the dynamic changes in physicochemical characteristics, organic carbon fractions and stability, and bacterial community structure in

* Corresponding author at: Key Laboratory of Agro-Forestry Environmental Processes and Ecological Regulation of Hainan Province, Hainan University, Haikou 570228, China.
E-mail addresses: 22220951320112@hainanu.edu.cn (S. Hou), wj13424050389@163.com (J. Wang), 995096@hainanu.edu.cn (D. Feng), wudomy@126.com (D. Wu), chensq@hnhky.cn (S. Chen), fallsong@hainanu.edu.cn (J. Fan), 23210830000021@hainanu.edu.cn (M. Luo), 183237@hainanu.edu.cn (H. Deng), xing.yang@hainanu.edu.cn (X. Yang), luojiwei@hainanu.edu.cn (J. Luo), zhyk0610@163.com (Y. Zhao), 990963@hainanu.edu.cn (C. Ge).
[1] Key Laboratory of Agro-Forestry Environmental Processes and Ecological Regulation of Hainan Province, Hainan University, Haikou 570228, China
[2] Key Laboratory of Environmental Toxicology, Hainan University, Ministry of Education, Haikou 570228, China.
[3] School of Environmental Science and Engineering, Hainan University, Haikou 570228, China

https://doi.org/10.1016/j.jhazmat.2025.138078
Received 17 October 2024; Received in revised form 27 January 2025; Accepted 24 March 2025
Available online 28 March 2025
0304-3894/© 2025 Elsevier B.V. All rights are reserved, including those for text and data mining, AI training, and similar technologies.

Colonization time
Bacterial community

seagrass bed sediments during the plastisphere formation process for polystyrene (PS) and polylactic acid (PLA). The results revealed significant weathering and biofilm formation on both PS and PLA. MPs altered the microbial community structure in seagrass bed sediments, leading to species turnover. Colonization time emerged as the key factor driving microbial community assembly, with ecological processes shifting from dispersal limitation to ecological drift in the plastisphere, while sediments maintained dispersal limitation as the dominant process. The formation of the plastisphere significantly influenced seagrass bed sediment microbial carbon (MBC) and organic carbon pool stability. MPs weathering negatively correlated with sediment properties but positively correlated with microbial communities, jointly modulating carbon pool stability. This study provided a new insight into the potential risks posed by MPs to carbon cycling and the ecological functioning of seagrass bed ecosystems.

1. Introduction

Plastics are extensively used in daily life due to their durability and lightweight, including electronics, automobiles, construction, clothing, cosmetics, toys, packaging, utensils, and disposable plastic products [1]. However, plastics in the environment are challenging to recycle effectively and often degrade into smaller particles or fragments, known as microplastics (MPs) [2,3], which are defined as particles smaller than 5 mm [4–6]. MPs are ubiquitously detected across various environments, including marine environments [7], beaches [8], polar regions [9], and surface waters [10]. Upon entering the environment, MPs provide new ecological niches for microbial colonization, leading to the formation of biofilms known as the “Plastisphere” [2,11]. This phenomenon has been documented in marine ecosystems, potentially impacting marine ecosystem functioning, including carbon cycling, xenobiotic compound degradation [12], and horizontal gene transfer [13]. While significant progress has been made in understanding the composition and diversity of plastisphere communities in marine ecosystems, there remains a gap in our knowledge regarding the microbiota of the plastisphere and their ecological roles in seagrass bed ecosystems.

Seagrass beds, one of the important “blue carbon” ecosystems, can prevent coastal erosion, improve water quality, and provide essential habitats for various marine organisms [14], thus offering significant ecological and economic benefits. Simultaneously, seagrass bed ecosystems act as highly efficient natural carbon sinks that can fix up to several hundred grams of carbon per square meter per year, accounting for about 10 % of the organic carbon annually fixed in the ocean. However, recent studies have shown that MPs pose a potential threat to seagrass bed ecosystems, which have been identified as a long-term sink for MPs. The sources of MP pollution in seagrass beds are predominantly attributed to anthropogenic activities, including marine tourism, fishing operations, maritime transport, and oceanic exploration, which contribute substantial amounts of plastic waste and derelict fishing gear [15,16]. Recent investigations have reported that MP particles were detected on both sediment surfaces and seagrass blades [17,18], with emerging evidence suggesting that seagrass meadows can act as natural filters for microplastics, capturing and retaining diverse MP types [11–13]. The interaction between MPs and seagrass systems includes multiple mechanisms. Upon contact, MPs adhering to seagrass blades may be physically similar to the epiphytes, creating a shading effect by clogging seagrass cells, thereby reducing light attenuation and nutrient transfer [19,20]. Prolonged MP exposure may alter seagrass structure in seagrass, prevent rhizome vertical growth, and increase vulnerability to colonization and sedimentation [21]. Furthermore, epiphytic communities on seagrass leaves have been demonstrated to facilitate MP accumulation [17,22]. Once in seagrass bed sediments, MPs may affect microbial communities, leading to altered nutrient cycling [4], and can also serve as carriers of heavy metals and residual monomers, potentially elevating local sediment contamination levels [23]. Particularly, MPs can be absorbed by seagrass blades and subsequently enter the food web, accumulating at higher trophic levels, thereby jeopardizing food security and human health [17].

Sediment organic carbon (SOC) pools play a crucial role in global carbon cycling and climate regulation, serving as significant carbon sinks in various ecosystems [24]. Coastal ecosystems, particularly seagrass beds, are recognized as major CO_2 sinks, capable of storing large carbon reserves in sediments and sequestering them over millennial timescales [25]. Recent studies suggest that multiple factors, including sediment grain size, microbial activity, and vegetation dynamics, can influence SOC accumulation and stability in coastal sediments [26,27]. MP-derived carbon was initially captured by seagrass leaves and subsequently sequestered in sediments. This MP carbon shows reduced mobility within marine carbon cycles and enhanced resistance to microbial degradation[28]. This characteristic may potentially enhance long-term carbon storage in seagrass bed sediments and alleviate MP pressure on adjacent ecosystems, which is evidenced by the documented positive correlation between particulate organic carbon (POC) and MP accumulation in seagrass sediments [29]. The chronic accumulation of MPs may change the structure of the sediment, affecting their density and stability [30]. These changes could have profound impacts on ecosystem-level carbon cycling. Moreover, the formation of plastisphere communities in seagrass bed environments may introduce additional complexity to these dynamics. This biological-MP interface may modify sediment physicochemical properties and reshape microbial community structures, subsequently influencing organic matter transformation processes. However, the mechanistic understanding of these interactions and their implications for carbon pool stability remains inadequately explored.

The present study aims to investigate the effects of plastisphere formation on the dynamics of organic carbon fractions and bacterial communities in seagrass bed sediments. We hypothesize that (1) the characteristics of the plastisphere will vary depending on the type of polymer and colonization time; (2) the plastisphere will affect sediment organic carbon fractions through both direct and indirect mechanisms; (3) colonization time will be a key driver in the stability of organic carbon pools and the structure of microbial communities in seagrass bed sediments. This study has theoretical and practical implications for assessing the long-term environmental and ecological of MPs in seagrass bed ecosystems.

2. Materials and methods

2.1. Microcosm incubation experiment

A microcosm incubation experiment was conducted in a climate-controlled chamber based on protocols from previous studies [31]. The experiment focused exclusively on sediments from seagrass habitats, excluding the seagrass itself, with a 56-day exposure period to simulate MP contamination in sediments (details in Text S1, Supplementary Materials). Previous research has demonstrated that microbial colonization on microplastic surfaces typically undergoes distinct succession stages, with initial rapid colonization followed by community stabilization, which generally occurs within 6–8 weeks [32]. This duration of 56 in our study is sufficient for the development and stabilization of plastisphere microbial communities, thus providing a comprehensive view of the temporal changes in community assembly processes and organic carbon dynamics. A single medium concentration (1 %, w/w) of polystyrene (PS) and polylactic acid (PLA) microplastics

was selected to represent non-biodegradable and biodegradable MPs, respectively. The background MP levels were not measured in this study, previous investigations in the same region have reported concentrations ranging from 93.3 to 780.2 particles/kg in seagrass bed sediments [33]. All experimental sediments were collected from the same location to ensure consistent background conditions across treatments. The selection of 1 % (w/w) MP addition to sediment was informed by previous studies [21–23]. Although this concentration exceeds current environmental levels, it represents a potential future scenario due to the increasing trend of plastic pollution in marine ecosystems. This higher concentration allows for a clearer observation of potential impacts and mechanisms.

Commercial PS and PLA microplastic pellets (size: 0.55–0.83 mm) were purchased from Beijing Anterail Technology Co., Ltd, and their main characteristics and pretreatments are shown in Text S1 and Table S1. Before starting the incubation experiments, microplastic pellets were surface-sterilized under UV light for 30 min. Sediment samples (1110 g) were then mixed with microplastic particles (11.1 g). The mixture was thoroughly homogenized using a rolling mixer for 30 min to ensure uniform distribution. The experiment comprised a control (without MPs) and two MP treatments, and each treatment was conducted in triplicate. The incubation temperature was maintained at 25℃ in the dark, representing typical summer sediment temperatures in seagrass habitats.

To reactivate the microbial communities in the sediments, 400 mL of artificial seawater was added for one week of domestication. Artificial seawater was prepared by dissolving 33.33 g of sea salt (Guangzhou Yier Bioengineering Co., Ltd) in 1 L of deionized water, as detailed in Text S1. Following the one-week reactivation period, the formal incubation began, with daily additions of artificial seawater to maintain a 2 cm water column above the sediment surface. Sampling was conducted at intervals of 0, 7, 14, 28, and 56 d. To avoid disruption of the incubation system dynamics during sampling, a destructive sampling approach was implemented. At each sampling point, 250 g of the sediment-MP mixture was collected for the analysis of both MPs and sediments, as well as for microbiological analyses.

2.2. Characterization of physicochemical properties

The pristine and incubated MPs were characterized using several analytical techniques, including scanning electron microscopy (SEM, TESCAN MIRA LMS), Attenuated Total internal Reflectance Fourier Transform Infrared spectroscopy (ATR-FTIR, PerkinElmer Spectrum 3TM), X-Ray Diffraction (XRD, MiniFlex600), and X-Ray Photoelectron Spectroscopy (XPS, Thermo Fisher Scientific ESCALAB Xi), to assess surface morphology, functional groups, crystallinity, and chemical composition. Biofilm biomass on pristine and incubated MPs was quantified using the crystal violet staining method [34]. The carbonyl index (CI), an important indicator for evaluating the degree of surface weathering and oxidation of microplastic particles [35], was calculated from the obtained ATR-FTIR spectra [36]. Further details are provided in Text S2. In addition, sediment properties were also analyzed, including pH, total phosphorus (TP), NH_4^+-N, and NO_3^--N and detailed information is available in Text S3.

2.3. Determination of sediment organic carbon fractions and calculation of carbon pool management index

Sediment organic carbon (SOC) was quantified using commercial assay kits (Suzhou Keming Biotechnology Co., Ltd). Dissolved organic carbon (DOC) in the sediments was extracted by shaking the samples (30 min, 250 rpm) with deionized water at a water-sediment ratio of 5:1 followed by filtration through a 0.45 μm membrane. The filtrate was then determined using a TOC analyzer (TOC-L CPH CN200). Sediment microbial biomass carbon (MBC) was determined by the chloroform fumigation-extraction method [37]. Easily oxidizable organic carbon (EOC) content was measured by the potassium permanganate ($KMnO_4$) oxidation colorimetric method [38]. The concentration of sediment recalcitrant organic carbon (ROC) was calculated using the formula ROC =SOC-EOC [39].

The Carbon management index (CMI) was used to evaluate the effects of plastisphere formation on the stability of the sediment carbon pool. The CMI was calculated following established methods from previous studies[38], with further details provided in Text S4.

2.4. DNA extraction and high-throughput sequencing

Sediment and incubated MP samples were collected at 14 and 56 days for bacterial community analysis. Detailed procedures for bacterial community analysis are available in Text S5. Sequences with 97 % similarity were clustered into operational taxonomic units (OTUs) using USEARCH software, and the taxonomy of the OTUs was assigned according to the RDP classifier. Detailed information for alpha diversity indices (ACE, Chao1, Shannon, Simpson, and observed OTUs), beta diversity, potential metabolic functions, and co-occurrence network analysis is shown in Text S5. To examine bacterial community assembly and ecological processes, β-nearest taxon index (β-NTI) and Bray-Curtis-based-Crick Index (RC_{Bray}) values were calculated, with further details shown in Text S6. Relationships between MP characteristics, sediment physicochemical properties, microbial community compositions, and sediment organic carbon fractions were estimated using Partial Least Squares Structural Equation Modeling (PLS-SEM) in Smart-PLS software [40].

2.5. Statistical analysis

All experimental data are presented as mean ± standard deviation (SD) based on three replicates. Data analysis was conducted using Excel 2019 and SPSS 26.0 software, while figures were generated using R version 4.3.3 and OriginPro 2024. We mainly conducted statistical analysis on response variables such as the physical properties of microplastics, sediment organic carbon pool, sediment physicochemical parameters, and microbial community composition indicators. Prior to analysis, the Kolmogorov-Smirnov test was used to assess the normality of the data, and then Levene's test was used to further verify whether the data satisfied the homogeneity of variance to ensure the appropriateness of the application of analysis of variance (ANOVA). A one-way ANOVA and Duncan analysis multiple comparisons were used to assess the significance of differences among the experimental treatments, and the difference with p values ≤ 0.05 was statistically significant. Moreover, to elucidate the temporal dynamics and polymer-specific effects on plastisphere formation, a two-way ANOVA analysis was applied with colonization time and MP polymer type as independent variables. Principal coordinate analysis (PCoA) based on Bray-Curtis dissimilarity matrices was performed to evaluate beta diversity and visualize the differences in microbial community composition across treatments using majorbio online platform. Neutral community models were implemented to investigate the ecological processes governing bacterial community assembly in both plastisphere and MP-treated seagrass bed sediments using R statistical software. Co-occurrence network analysis was performed using Spearman rank correlations[41] (thresholds: r value $>$ 0.65, p value $<$ 0.05) with R software, and visualized by the Fruchterman-Reingold layout in the Gephi v.0.10.1 software. The putative functional profiles of the bacterial communities in sediments and plastisphere were predicted by the Functional Annotation of Prokaryotic Taxa (FAPROTAX) database [42].

3. Results

3.1. Characteristics of MPs and their biofilms during the incubation process

3.1.1. Surface functional groups and chemical composition

ATR-FTIR spectra (Fig. 1a-b) show the changes in functional groups of the pristine and aged MPs in seagrass bed sediments during the incubation process. After 56 d of exposure, significant changes in the characteristic peaks of the studied PLA (-C═O- at 1750 cm^{-1}, -C-O- at 1185, 1080 cm^{-1}, and Si-O at 534 cm^{-1}) and PS (C-O at 1000 cm^{-1} and $-CH_2$- at 536 cm^{-1}) were observed. Biodegradable PLA exhibited more dynamic changes compared with the gradual oxidation of PS, as evidenced by stronger fluctuations in CI index values (Fig. 1e). The initial

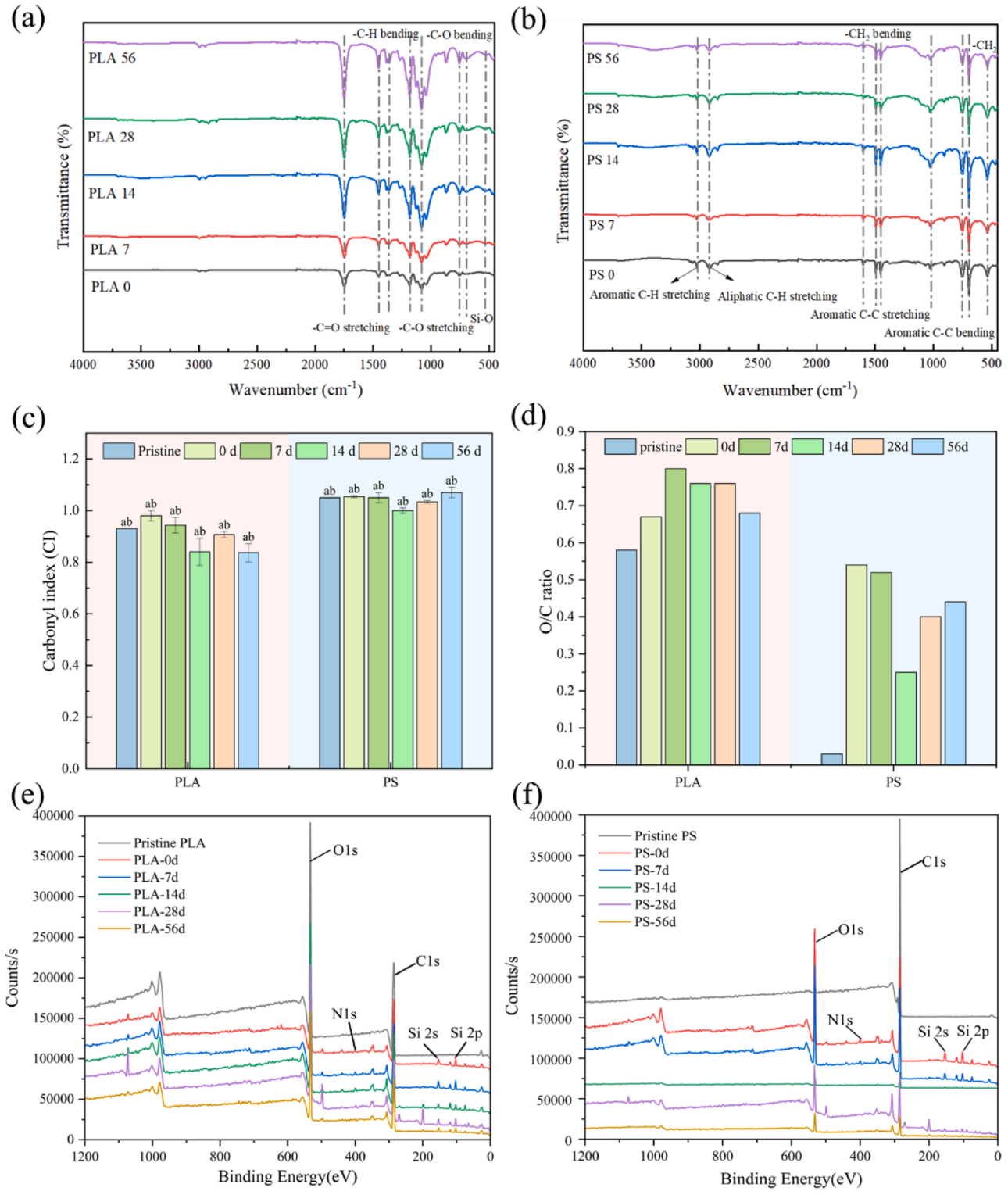


Fig. 1. FTIR spectra (a-b), XPS wide scan (c-d), and changes of carbonyl index (CI) (e) and O/C ratio (f) of the different pristine and incubated MPs in seagrass bed sediments over colonization time.

CI reduction in PS might be attributed to the degradation or detachment of surface oxidized layers, while the later increase suggests ongoing slow oxidation.

XPS spectra and O/C ratio of pristine in seagrass bed sediments are presented in Fig. 1 and Fig. S1. XPS (Fig. 1c-d) analysis revealed significant alterations in the surface elemental composition of MPs during the incubation process in seagrass bed sediments. A considerable proportion increase in oxygen (O, 530.90 eV) was observed, indicating extensive surface oxidation. Moreover, a small amount of nitrogen (N, 400.04 eV) was detected, revealing the accumulation of N-containing organic substances after incubation. Meanwhile, silicon (Si, 103.4 eV) appeared on the MP surface, indicating the adherence of diatoms or silicon-containing substances to the surface of incubated MPs, consistent with the FTIR results. The O/C ratio (Fig. 1f) exhibited distinct temporal patterns for both polymers. PLA showed an initial increase followed by a decrease, while PS demonstrated a more complex oscillatory pattern with multiple fluctuations. Both polymers experienced a notable decline in the O/C ratio at day 14, with PS showing a more pronounced reduction.

High-resolution XPS analysis of C1s and O1s spectra (Fig. S1) revealed distinct surface chemical alterations in two types of MPs following incubation. PLA exhibited no formation of new oxygen-containing functionalities, but the relative proportions of pre-existing oxygen-containing functional groups changed, particularly evidenced by a marked decrease in ether bond content. For PS, the O1s spectrum showed the emergence of C═O and O═C-O functionalities post-incubation. These findings suggest the significant impact of weathering on the surface chemistry of MPs in seagrass bed sediments, specifically the generation of oxygenated functional groups.

3.1.2. *Surface crystallinity structure*

Fig. S2a-b shows the XRD scans of pristine and incubated MPs in

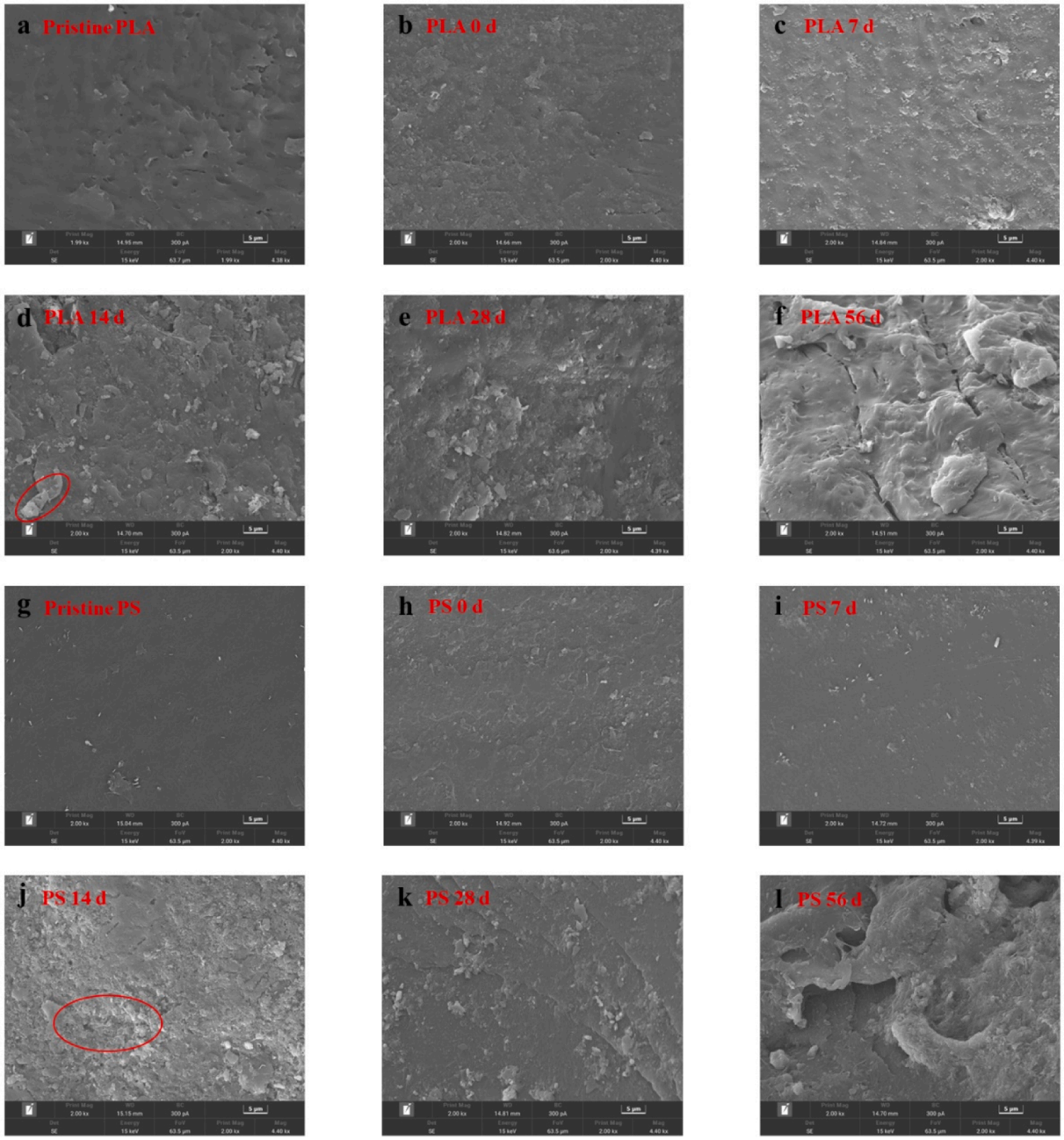


Fig. 2. SEM images of the studied MPs before and after exposure to seagrass bed sediments.

seagrass bed sediments. PLA showed progressive increases in crystallinity degree, reaching 13.56, 10.51, 11.91, 10.18, and 4.30 times higher than the pristine samples at successive time points. In contrast, PS samples maintained relatively stable diffraction patterns in the early stages of incubation, followed by a sharp increase in peak intensities at 56 d, accompanied by a substantial increase in crystallinity degree. These observations suggest significant structural reorganization of the polymer matrices during the incubation process.

3.1.3. Surface morphology and biofilm biomass

Fig. 2 presents the surface morphology variations of pristine and incubated MPs in the seagrass bed sediments during the cultural process. As shown in Fig. 2a and g, pristine MPs exhibited relatively smooth and homogeneous surfaces. With the extension of incubation time, the surfaces of PLA and PS showed different degrees of erosion, with cracks and pores increasing. PLA demonstrated more pronounced erosion characteristics than PS, with both polymers showing intensified surface erosion at 14 d, characterized by irregular pits and pores with different sizes. This critical degradation phase coincided with significant microbial colonization, as verified by biofilm biomass measurements (OD_{595}, Fig. S3). For both MPs, the biofilm biomass showed a trend of increasing and then decreasing, which may be related to the development and succession process of the biofilm (Fig. S3). Compared to PS, PLA demonstrated more rapid initial colonization (0–7 d) and faster microbial succession rates.

3.2. Sediment physicochemical and organic carbon pools characterization

3.2.1. Changes in sediment physicochemical characteristics

MP exposure presented significant and time-dependent alterations in sediment nitrogen dynamics (Fig. S4). Compared to control, PLA significantly elevated NH_4^+-N content at both 14 and 56 d, while PS showed a 0.6-fold increase at 56 d (Fig. S4a, F=22.19, $p < 0.01$). Both PLA and PS treatments significantly reduced sediment NO_3-N levels at the initiation of the experiment (Fig. S4b, F-value=14.66, $p < 0.05$). As colonization time extended, the impact of MPs on NO_3-N contents diminished, with all treatments showing a significant reduction followed by stabilization.

The presence of MPs slightly affected the pH of the seagrass bed sediments (Fig. S4c). PLA initially decreased sediment pH, while PS caused a slight increase, with both treatments showing reduced pH levels by day 56 compared to control. Additionally, the TP content in sediments exhibited complex and polymer-specific trends over the colonization time (Fig. S4d). PLA and PS treatments showed opposing effects on sediment TP content, with their effects reversing by day 56, thereby emphasizing the dynamic nature of MPs-sediment interactions.

3.2.2. Changes in sediment organic carbon fractions

The effects of MPs on the organic carbon fractions of seagrass bed sediments were assessed over various incubation periods (Fig. 3). Our analysis revealed distinct temporal patterns in the response of various organic carbon fractions to MP exposure in seagrass bed sediments. The labile carbon pools (DOC, EOC, and MBC) presented similar trends over time: In the early stages (0–14 d), sediment DOC, EOC, and MBC significantly decreased (Fig. 3a, b, and c). Subsequently, they increased from day 14–28 and became stable from day 28–56. In contrast, sediment SOC and ROC fractions remained relatively stable across all treatments and time points, suggesting limited MP impact on these persistent carbon fractions over the experimental duration.

3.2.3. Changes in carbon pool stability index

The effects of the addition of both PLA and PS microplastics on sediment CA, LI, and CMI were significant with the extension of incubation time in Table S2. The CA of PLA generally exceeded that of PS (except 7, 56 d), suggesting enhanced organic carbon transformation and microbial utilization efficiency in PLA-treated sediments. The CMI values for sediments treated with PLA were consistently lower than those treated with PS, indicating that the addition of PLA increased the

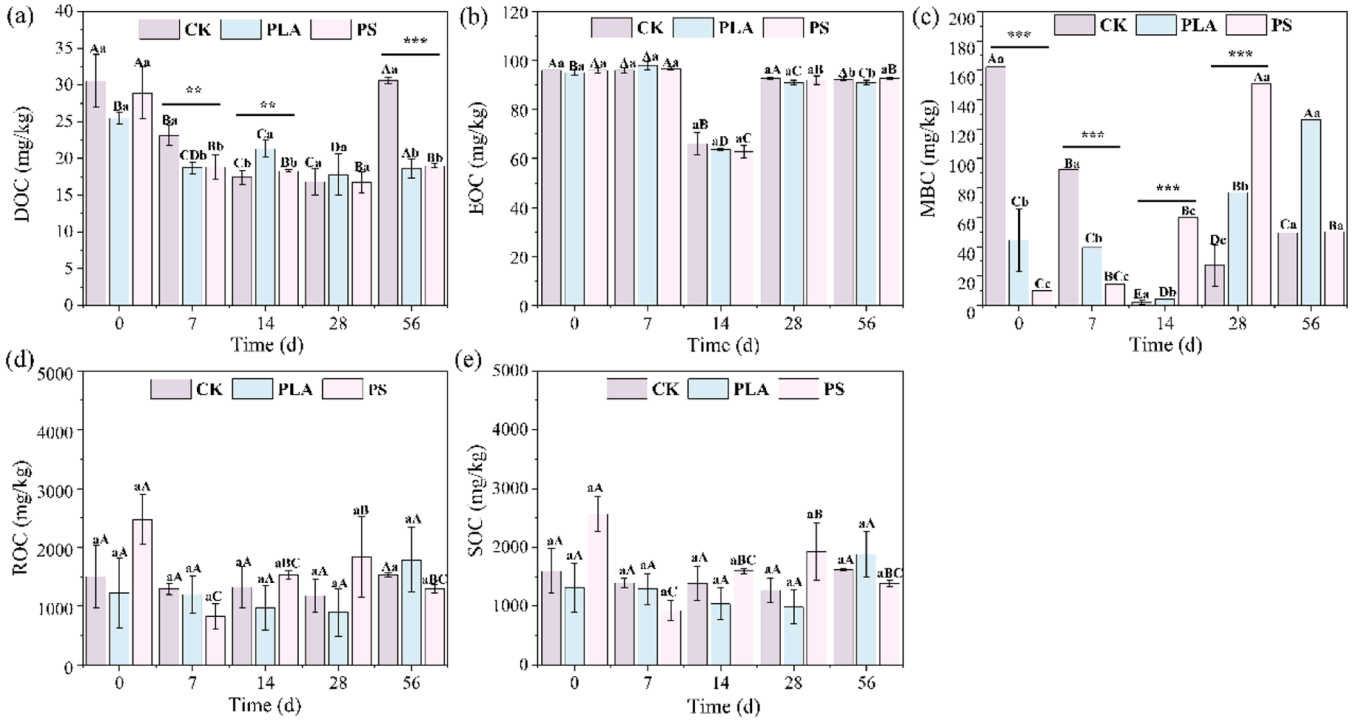


Fig. 3. Temporal dynamics of organic carbon fractions in sediments exposure to different MP types: DOC (a), EOC (b), MBC (c), ROC (d), and SOC (e) contents under different incubation times. Values are means ± SD with three replicates. Capital letters (A-E) indicate significant differences across incubation times within the same MP treatment ($p < 0.05$). Lowercase letters (a-c) indicate a significant difference between PLA and PS treatment at each time point ($p < 0.05$). Unmarked letters and * indicate non-significant differences. Asterisks denote levels of significance: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$.

organic carbon fraction dynamics but decreased the stability of the sediment carbon pool. It is worth noting that LI and CMI of sediments peaked at 14 days followed by a decline, highlighting the temporal evolution of MP-sediment interactions and their influence on carbon pool dynamics.

3.3. Changes in plastisphere and sediment bacterial compositions and functions

3.3.1. Diversity and compositions of bacterial communities

We explored the differences in diversity and composition of bacterial communities between the plastisphere and the MP-treated sediments. At 14 d of incubation, the bacterial diversity on PLA surfaces was observed to be higher than that of PS with a higher OTU number, ACE, Chao1, and Shannon index (Fig. S5A). However, as the incubation period extended (56 d), the differences in bacterial community diversity between PLA and PS diminished. The changing trends of bacterial community diversity in MPs-treated sediments over colonization time were consistent with those of plastisphere (Fig. 4B). Throughout the incubation period, the total amount of observed OTUs, ACE, Chao, and Shannon indices at 56 d in plastisphere and MP-treated sediments were significantly higher than those at 14 d. Conversely, the Simpson index decreased over colonization time. The results also demonstrated substantial differences in community diversity between the plastisphere and sediments, as well as temporal changes in bacterial community structure.

The beta diversity of bacterial communities for plastisphere and sediments was analyzed by PCoA based on Bray-Curtis dissimilarity (Fig. S6). Initially, at 14 d, plastisphere bacterial communities showed a clear separation between PLA and PS, with MP type strongly influencing community composition (70.98 % variation explained by PC1). Concurrently, sediment bacterial communities exhibited distinct clustering among CK, PLA-treated, and PS-treated groups, suggesting that the presence of MPs significantly affected sediment bacterial composition. By 56 d, both plastisphere and sediment bacterial communities demonstrated convergence trends, with reduced separation between treatments, suggesting adaptive processes over time. Notably, plastisphere bacterial communities displayed more pronounced initial differences and subsequent convergence compared to MP-treated sediment communities, indicating the greater heterogeneity of the plastisphere as a microbial habitat.

It was also found that the formation of the plastisphere induced changes in the composition of the sediment bacterial community. At day 14, the plastisphere was dominated by *Proteobacteria* (relative abundances of 60.98 % and 72.63 % for PLA and PS, respectively) and *Bacteroidota* (relative abundances of 13.12 % and 11.79 % for PLA and PS, respectively) (Fig. 4A). As the incubation duration increased, *Proteobacteria* abundance increased on PLA biofilms but decreased on PS biofilms, while *Bacteroidota* abundance increased on both MPs. Control sediments without MPs were primarily dominated by *Proteobacteria* (48.54 %) and *Firmicutes* (14.25 %). The addition of MPs significantly enhanced the relative abundances of *Proteobacteria*, *Bacteroidetes*, and *Desulfobacterota* (Fig. 4B). Moreover, the relative abundance of *Proteobacteria* in sediments at 56 d was significantly lower than that at 14 d across all treatments. The plastisphere exhibited distinct bacterial community patterns compared to sediments, characterized by higher *Proteobacteria* abundance and lower *Firmicutes* representation. Interestingly, *Planctomycetota* and *Patescibacteria* appeared in the plastisphere in the late stage of incubation, and the relative abundance of these two bacteria on the PS surface was significantly higher than that on the PLA surface, indicating that biodegradable and non-biodegradable microplastics showed different preferences for bacterial communities. In contrast, *Campilobacterota* and *Bdellovibrionota* disappeared from both plastispheres. This phenomenon was also similarly found in the sediments, i.e., *Gemmatimonadota* and *Planctomycetota* appeared in the late incubation stage, with the disappearance of *Bdellovibrionota* and *Myxococcota*.

In order to further explore the complexity and stability of the bacterial community structures in plastisphere and sediments, the co-occurrence networks analysis was carried out (Fig. 4C). At plastisphere 14 d, the network associated to PS and PLA plastisphere exhibited 3023 edges and 188 nodes, whereas at 56 d, it increased to 3422 edges and 216 nodes. In comparison, the sediment networks demonstrated

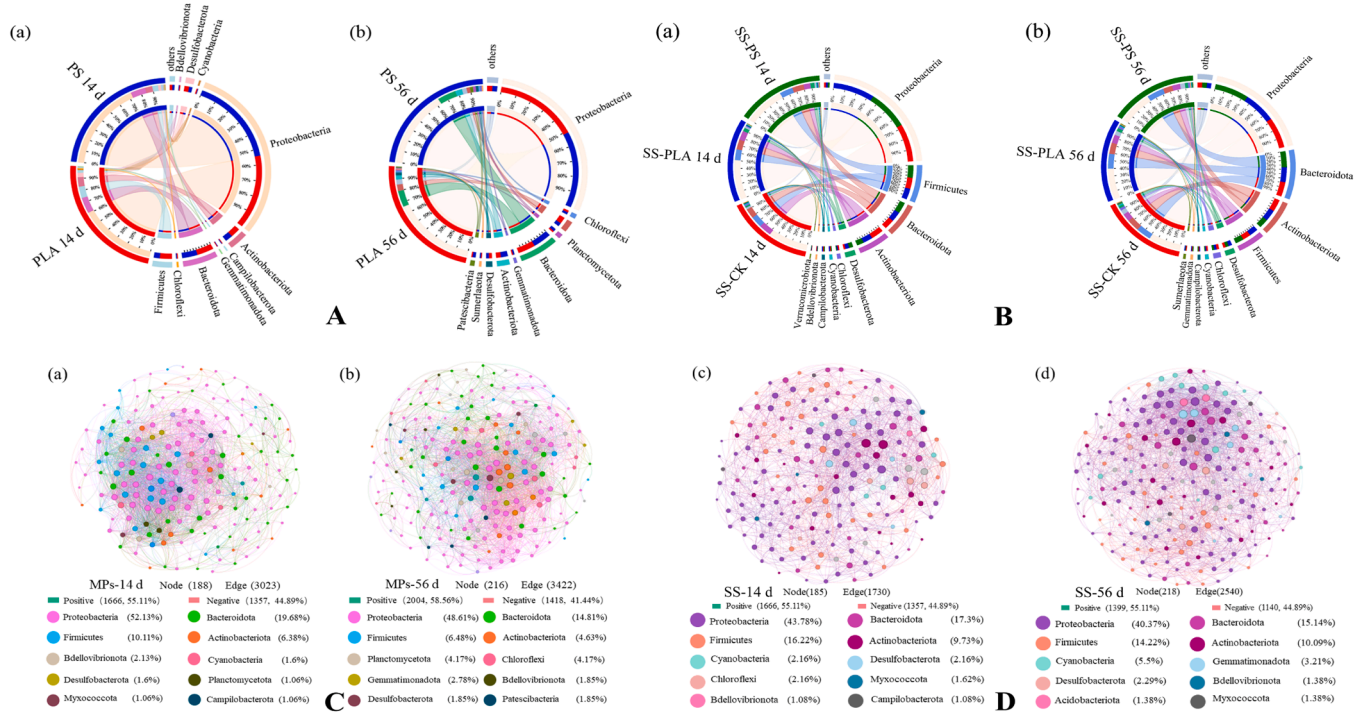


Fig. 4. Relative abundance of microbial communities in the incubated MPs (A) and MPs-treated seagrass bed sediments (B) at the phylum level at 14 d and 56 d Co-occurrence network of bacterial communities in the incubated MPs (C) and MPs-treated seagrass bed sediments (D).

1730 edges and 188 nodes at 14 d, and 2540 edges and 218 nodes at 56 d (Fig. 4D). With the increase of colonization time, the network structure of plastisphere and sediment bacterial communities tended to be complex and stable. Moreover, the microbial network in the plastisphere was more intricate and stable compared to those in sediments. This trend is consistent with the results of the alpha and beta diversity analysis described above.

3.3.2. Functional bacterial communities in plastisphere and sediment

FAPROTAX analysis identified 54 significantly altered functional groups out of 70 total groups (Fig. S7). Fig. 5A showed that the carbon (C), nitrogen (N), and sulfur (S) cycles in the plastisphere changed significantly with the increase of colonization time. For C cycles, both PLA and PS plastisphere initially suppressed aerobic chemoheterotrophy and hydrocarbon degradation, while photoautotrophy and fermentation slightly increased, with more pronounced effects in PLA. In sediments, the relative abundance of most functional groups increased over time with the presence of MPs. Notably, the dominant chemoheterotrophy and aerobic_chemoheterotrophy communities decreased significantly over time (Fig. 5B).

Nitrogen cycling functions demonstrated contrasting trends between polymer types: increasing in PLA plastisphere while decreasing in PS plastisphere. It is worth noting that despite an overall decrease, nitrification and aerobic nitrite oxidation functional groups increased significantly. A similar increase of nitrification and aerobic nitrite oxidation over time was observed in sediments.

In terms of S cycling, hypoxic photoautotrophic phenomena increased over time, especially on PLA plastisphere (Fig. 5A). Conversely, dark oxidation, dark sulfur oxidation, and dark sulfite oxidation processes decreased with colonization time. In sediments, PLA initially enhanced dark sulfur oxidation and dark sulfite oxidation, while both MPs subsequently inhibited sulfur respiration and thiosulfate respiration compared to controls. These findings demonstrate the temporal evolution of MP-induced alterations in bacterial functional ecology within seagrass bed sediments.

3.3.3. Key ecological processes assembling bacterial community in plastisphere and sediments

The balance between ecological niche (determinism) and neutrality (randomness) plays a key role in shaping microbial communities, thus it is essential to evaluate the relative contributions of these ecological processes in the assembly of bacterial communities on the plastisphere and sediments. Neutral Community Model (NCM) indicated that the construction of plastisphere and sediment bacterial communities was predominantly influenced by stochastic processes (Fig. S7). As the colonization time increased from 14 to 56 d, the r^2 value decreased from 0.859 to 0.8466, suggesting an increased influence of deterministic processes on bacterial community construction. This shift was further supported by decreasing migration rates (m), indicating enhanced local stochastic drift in community assembly over time.

β-NTI values were further calculated to assess the relative importance of deterministic and stochastic processes in the assembly of bacterial communities in plastisphere and sediments (Fig. S8). The results revealed predominant stochastic processes in both environments, with values ranging from −1.75–1.37 in plastisphere, and −2.388–1.55 in sediments. This finding is consistent with the conclusions of the NCM analysis. The wider β-NTI range in sediments suggests stronger deterministic influences compared to the plastisphere. Combined β-NTI and

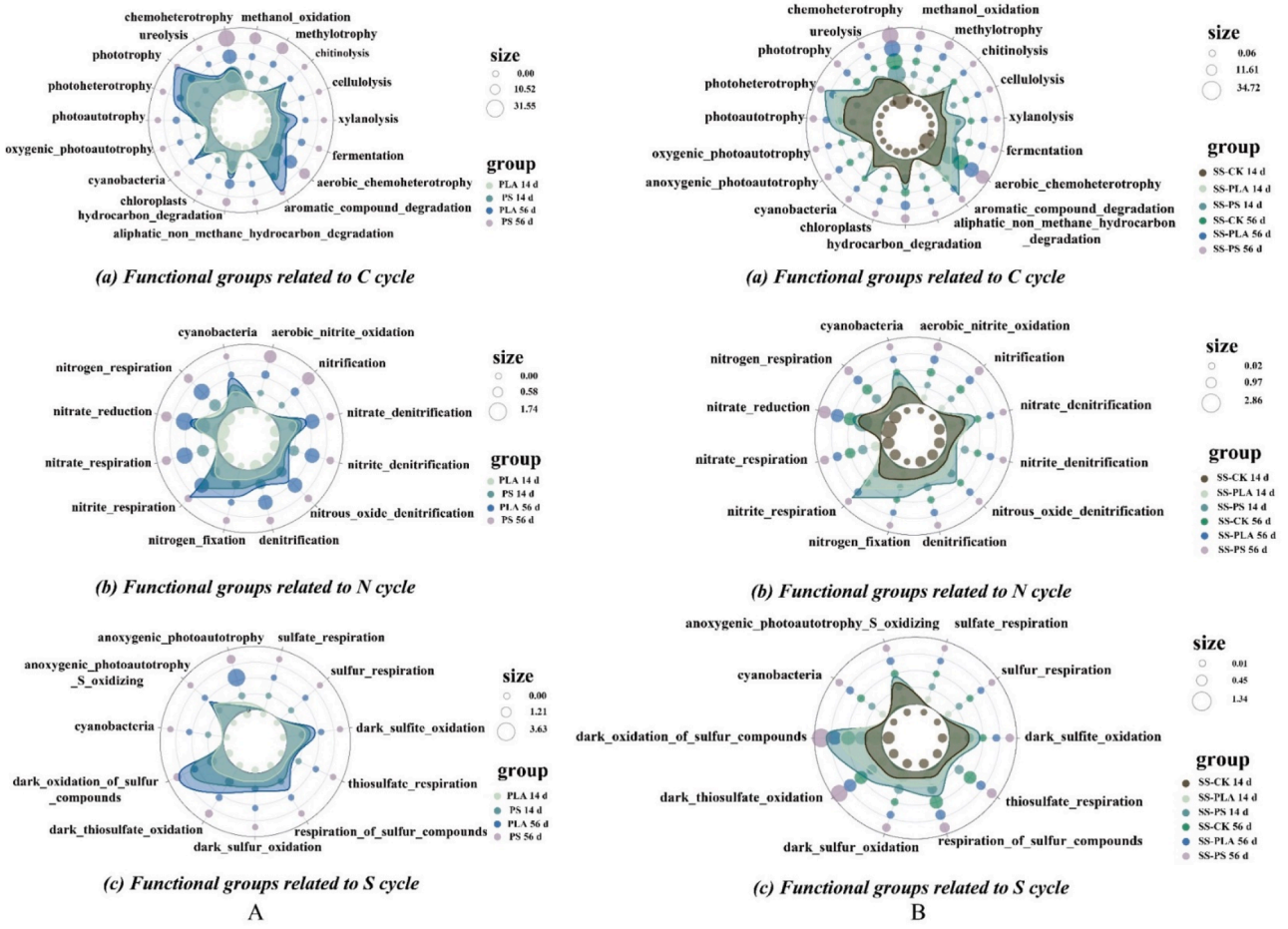


Fig. 5. Relative abundance of ecological groups involved in C (a), N (b), and S (c) metabolism in plastisphere (A) and sediments (B).

Bray-Curtis-based Raup-Crick (RCbray) analyses were utilized to approximate the relative contributions of different assembly processes. In the plastisphere, RCbray values were predominantly above 0.95 at day 14, indicating that dispersal limitation was the primary factor influencing bacterial community assembly. However, as the incubation period extended to 56 d, we observed a shift in RCbray values below 0.95, suggesting a transition towards ecological drift or other stochastic processes. These findings illuminate a dynamic progression in the bacterial community assembly mechanisms on plastisphere over time, highlighting the importance of colonization time as a key factor in the development and succession of plastisphere communities. However, sediment communities maintained dispersal limitation (RCbray > 0.95) throughout the incubation period, demonstrating temporal stability in their assembly processes.

3.4. Mechanisms governing sediment organic carbon dynamics during plastisphere formation

PLS-SEM analysis was used to reveal the possible mechanisms affecting the organic carbon dynamics of seagrass bed sediments during the plastisphere formation. Fig. 6 shows the complex interplay between MP weathering characteristics, sediment physicochemical properties, and bacterial community dynamics in modulating the interactions among sediment organic carbon fractions and their collective impact on the stability of the overall organic carbon pool. The goodness-of-fit index is 0.314, which indicates that the model can explain this relationship well. MPs weathering characteristics demonstrated a significant negatively correlation on sediment physicochemical properties (path coefficient of −0.759) and reactive organic carbon (path coefficient of −0.604), and presented a weak positively correlation on sediment microbes (path coefficient of 0.233). Additionally, significant positively correlations between sediment microbes and the organic carbon fraction were observed, suggesting the crucial role of microbial communities in C cycling process. These observations collectively indicate that MPs weathering can directly or indirectly affect sediment organic carbon fractions by regulating sediment physicochemical properties and microbial community structures or functions, ultimately shaping the stability of seagrass bed sediment organic carbon pools.

4. Discussion

4.1. Key factors affecting plastisphere formation

After 56 d of exposure, significant changes in the surface properties of the studied MPs were observed. ATR-FTIR and XPS analyses revealed an increase in oxygen-containing functional groups and the presence of silicon on the MP surfaces, indicating radical-based oxidation processes and the adsorption of silica-containing particles. These findings are similar to previously reported results in mangrove sediments and other aquatic environments [43,44]. XRD analyses revealed that the MP diffraction peak intensities and crystallinity exhibited significant differences in all treatment groups. In addition, observation of the surfaces of aged MPs using SEM revealed the formation of irregular pits and holes, creating diverse microhabitats conducive to plastisphere development. Similar morphological changes have been reported in other oceanic environments, including estuary [45], seawater [34], beach [46], and mangrove sediments [40,44].

The weathering characteristics of MPs and plastisphere formation in seagrass bed sediments are governed by a complex interplay of polymer type and duration of colonization. Compared with the gradual oxidation of PS, PLA exhibited more dynamic changes with stronger fluctuations in CI index values, more pronounced erosion features, and reduced crystallinity degree of PLA, which might be attributed to the occurrence of chain scission of MP polymer during weathering [40]. Compared to PS, PLA surfaces experienced faster and more extensive microbial colonization during the early incubation period and more rapid succession rates later in the experiment. This could be attributed to the higher surface reactivity and biodegradable nature of PLA, potentially offering more favorable attachment sites for early colonizing pioneering microorganisms or releasing bioavailable oligomers that serve as nutrient sources, stimulating microbial growth during the initial period. In contrast, the relatively inert nature of PS led to a less dynamic microbial community. These findings aligned with previous studies, which have shown that biodegradable MPs are more susceptible to environmental weathering than non-biodegradable ones [47], likely due to differences in polymer properties such as molecular weight, crystallinity, and co-polymer composition [48].

Two-way ANOVA analysis (Table S3) revealed that colonization time, rather than MP types, is the dominant factor driving the formation and evolution of the plastisphere. ATR-FTIR and XPS analyses revealed a gradual increase in oxygen-containing functional groups over time. This finding aligned with previous observations in mariculture settings [34]. Both MPs exhibited their lowest CI values, increased surface erosion, and a decreased O/C ratio at 14 d, suggesting a potential inflection point in the weathering process within seagrass bed sediments. PLA showed greater sensitivity to colonization time in the early incubation stages, possibly attributed to the degradation of crystalline regions or polymer chain hydrolysis. In contrast, PS exhibited more significant changes in the later stages, possibly due to stress-induced crystallization or formation of orderly structures through chain scission and reorientation

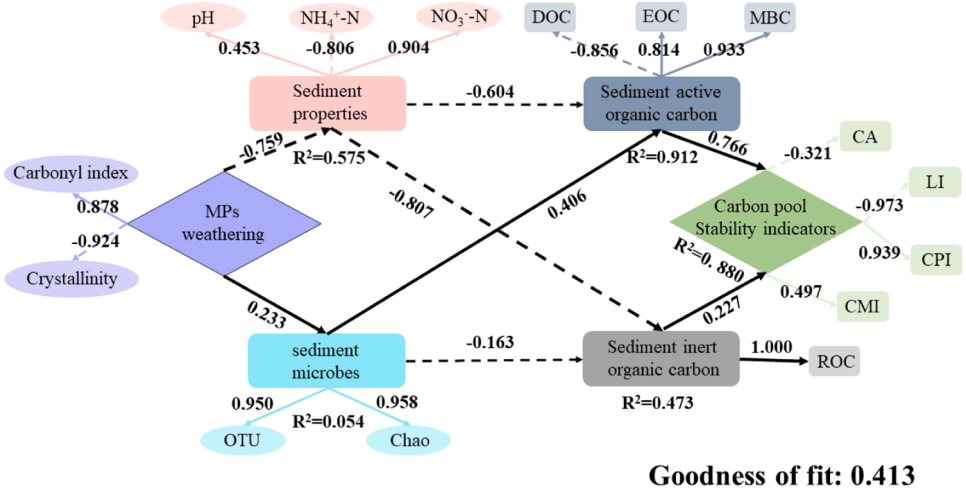


Fig. 6. Interaction of MPs aging characteristics, sediment properties, microbial communities and organic carbon pools based on PLS-SEM analysis.

after prolonged environmental exposure. Moreover, biofilm biomass on MP surfaces exhibited a non-linear trend over colonization time, reflecting distinct phases of plastisphere evolution. During the initial stage (0–7 d), MPs rapidly adsorbed nutrients, providing a favorable substrate for microbial attachment. Between 7 and 14 d, biofilm growth slowed as communities matured. After 14 d, biofilm biomass decreased, possibly due to nutrient depletion and waste accumulation, leading to microenvironment deterioration detachment of microbial colonies. This pattern diverged from previous studies reporting continuous biomass increases [49], suggesting a more complex relationship between colonization time and biofilm development. The observed decline in later stages warranted further investigation into potential mechanisms, such as resource limitations, interspecies competition, or changes in MP surface properties.

4.2. Effects of time-course plastisphere formation on sediment physiochemical properties

PLA and PS showed distinct impacts on nitrogen cycling in sediments. NH_4^+-N levels were consistently lower in PLA-treated sediments compared to the control, suggesting enhanced ammonium uptake or increased nitrification [50]. This effect could be attributed to the biodegradable nature of PLA, which may support ammonium-utilizing microbial communities or create microenvironments favorable to nitrifying bacteria. In contrast, the low and gradual increase in NH_4^+-N in PS-treated sediments may result from initial colonization by ammonia-oxidizing microorganisms, followed by changes in microbial community function or sediment conditions that promote ammonium accumulation over time. Compared to the control, PLA and PS treatments initially reduced sediment NO_3^--N contents, suggesting that MPs may favor denitrification or inhibit nitrification in the short term [37]. As colonization time extended, however, the impact of MPs on NO_3^--N contents diminished, with all treatments showing a significant reduction followed by stabilization. The convergence of NO_3^--N levels in the later stages suggested that a new equilibrium in N cycling may have been established as the sediment microbial community composition adjusted with the maturation of the plastisphere. In addition, MPs addition increased the release of N_2O, suggesting that MPs may impact denitrification processes [51]. The observed changes in nitrogen dynamics coincided with the development and maturation of the plastisphere. The peak changes in NH_4^+-N and NO_3^--N levels at 14 d aligned with the period of maximum biofilm formation, suggesting a strong link between microbial colonization of MPs and alterations in sediment N cycling.

4.3. Colonization time driving composition and assembly of microbial communities in the plastisphere and seagrass bed sediments

As colonization time increased, we observed higher OTU, ACE, Chao1, and Shannon indices in both the plastisphere and MP-treated sediments, indicating an overall increase in species richness and diversity over colonization time. Similar results have been reported in freshwater lakes and rivers[47]. At 14 d, PLA surfaces and PLA-treated sediments exhibited greater bacterial diversity, with higher OTU numbers, ACE, Chao1, and Shannon indices than PS. However, as the incubation extended to 56 days, the differences in bacterial community diversity between PLA and PS diminished, indicating the convergence of microbial communities in the plastisphere and sediments, consistent with previous reports [52].

Over time, MP surfaces degrade and develop biofilms, creating microenvironments similar to surrounding sediments [41,53]. Microbial communities on MPs and in sediments adapt, develop shared metabolic capabilities [2,13], and experience continuous dispersal [54], leading to the convergence of bacterial communities between MPs and sediments. This convergence is further driven by the equalization of selection pressures and resource availability over time [55], along with the development of functionally similar microbial communities [56]. This trend was further verified by the decreasing distance for both the plastisphere and sediments in the PCoA analysis. Moreover, more pronounced initial differences in bacterial communities were observed in the plastisphere compared to MP-treated sediments, followed by a stronger convergence over time. This can be attributed to the higher heterogeneity of the plastisphere as a microbial habitat compared to the relatively homogeneous sediment environment.

Bacterial community compositions in the plastisphere and MP-treated sediments also varied significantly with colonization time, confirming temporal heterogeneity in bacterial succession. Previous studies also found similar results that colonization time drives community structure on different MP surfaces and in surrounding marine environments [57]. In our study, *Proteobacteria* and *Bacteroidota* dominated the plastisphere, likely due to their rapid colonization and biofilm-forming abilities, while *Proteobacteria and Firmicutes* dominated the sediments. As colonization time increased, the relative abundance of *Proteobacteria* increased on PLA surfaces, possibly attributed to their metabolic versatility and ability to degrade PLA [56]. In contrast, their abundance decreased on PS surfaces and in sediments, likely due to the recalcitrant nature of PS and the depletion of labile organic matter in sediments, leading to more diverse and stable microbial communities. Notably, species turnover was observed in the later stages of incubation, with new species appearing and old ones disappearing, suggesting succession within the communities. *Planctomycetota* emerged in both the plastisphere and sediments, likely due to their ability to degrade complex organic matter and their particle-associated lifestyles [58]. *Patescibacteria* appeared in the plastisphere, possibly related to their symbiotic role in mature biofilms [59]. *Gemmatimonadota* emerged in sediments, potentially attributed to their adaptation to low moisture and nutrient conditions [60]. *Campilobacterota* and *Bdellovibrionota* disappeared from MP surfaces, while *Bdellovibrionota* and *Myxococcota* were no longer detected in sediments, suggesting shifts in community structure and nutrient availability. This succession pattern indicates a transition from early colonizers to a more specialized community adapted to the evolving microenvironments [2,61].

The community succession was further verified by co-occurrence network analysis. As colonization progressed, bacterial communities in both plastisphere and sediments diversified, leading to more complex networks with increased interconnections, indicating the establishment of syntrophic relationships, metabolic dependencies, and niche partitioning [62]. Moreover, the increased stability of these networks over time indicates the formation of a more resilient community structure, better adapted to the specific microenvironments of the plastisphere and MP-treated sediments [54].

FAPROTAX results indicated that C metabolic processes accounted for 25.7 % and 25.6 % of the predicted functional groups in the plastisphere and sediments, respectively. With the increase of colonization time, methanol-related metabolisms (methanol decomposition and oxidation) became more prominent in the plastisphere, especially on PS. This may be due to the degradation of plastic additives or the breakdown of adsorbed organic matter [63]. Additionally, slight increases in photoautotrophy and fermentation in the plastisphere suggested the development of oxygen gradients in mature biofilms, potentially influencing C cycling at the micro-scale [64]. In sediments, the abundance of most functional groups increased over time with MP exposure, except for dominant chemoheterotrophy and aerobic chemoheterotrophy, indicating a shift in carbon utilization strategies. The decline in chemoheterotrophic abundance could be linked to the depletion of labile carbon sources [13]. For N cycling, notable increases in nitrification and aerobic nitrite oxidation were observed in both the plastisphere and sediments, consistent with the changes in sediment characteristics. In terms of S cycling, hypoxic photoautotrophy increased over time, especially on the PLA plastisphere, while dark oxidation processes decreased. In sediments, dark sulfur and sulfite oxidation initially increased with PLA exposure, but both PLA and PS inhibited sulfur and thiosulfate respiration in the later stages of the experiment.

The NCM model revealed that stochastic processes predominantly governed the assembly of bacterial communities in both the plastisphere and sediments. *Proteobacteria*, which dominated all treatments, exhibited a wide ecological niche, and their community assembly was primarily driven by random collisions and colonization, confirming the dominance of stochastic processes in bacterial community assembly [65]. The prevalence of stochastic processes in plastisphere community construction has been observed in both aquatic and soil environments [66]. In addition, the influence of deterministic processes on bacterial communities in the plastisphere increased slightly over time, suggesting the gradual maturation and specialization of these bacterial communities. It was also reported that the prokaryotic communities in different lakes, where assembly shifted from stochastic to deterministic processes with increasing colonization time [67]. This shift towards deterministic assembly may be attributed to several factors: (1) stronger environmental filtering in the microenvironments on MP surfaces and in MP-treated sediments [55]; (2) complex biofilms development on MPs, creating distinct niches for specifically adapted species [68]; and (3) MP-induced changes in physicochemical properties, potentially enhancing deterministic selection [69].

Furthermore, we carried out a comprehensive analysis of β-NTI and RCbray. The results revealed that ecological processes in the plastisphere shifted from diffusion limitation in the early stage (14 d) to ecological drift and other processes in the late stage (56 d). Initially, diffusion limitation dominated, as species were dispersing to colonize a new environment that was not yet fully occupied [53,55]. As the colonization period extended, dispersal limitation diminished as more species have successfully colonized on the MP surfaces, leading to a more saturated and competitive environment [70]. Thus, ecological drift became the dominant stochastic process, reflecting random fluctuations in species abundance within the established community [71]. In contrast, in the sediments, dispersal limitation dominated the entire incubation process, suggesting a relatively stable community assembly process over time. This stability may be attributed to the fact that sediment environments generally provide more stable and diverse microhabitats compared to the MP surfaces, reducing the need for rapid community turnover [72]. The contrast between the stable, dispersal-limited processes in sediments and the dynamic changes in the plastisphere highlights the unique nature of MPs as a novel habitat. The plastisphere undergoes rapid ecological succession with a shift from dispersal limitation to ecological drift as the community matures. In summary, colonization time played a key role in driving the composition and assembly of microbial communities in both the plastisphere and sediments, with distinct ecological processes governing each habitat over time.

4.4. *Effects of time-course plastisphere formation on sediment organic carbon fractions and carbon pool stability*

In this study, the formation of the plastisphere significantly influences organic carbon fractions and overall carbon pool stability in seagrass bed sediments. In the early stages of the plastisphere (0–14 d), DOC, EOC, and MBC significantly decreased, possibly due to temporary perturbations caused by experiment condition changes. Liu et al. [37] also concluded that DOM changes observed at the beginning of the incubation may be related to sample interference during mixing. The subsequent increases from day 14–28 indicate a recovery phase as the sediment system adapts to new conditions, with stabilization from day 28–56, suggesting the establishment of a new equilibrium. MP-treated sediments showed overall lower DOC levels compared to the control, possibly due to enhanced microbial metabolism or DOC adsorption onto MP surfaces [73]. During most of the colonization time (except for 14 d), MPs had little effect on EOC content, indicating that MPs may not substantially alter readily available organic substrates in sediments. MPs significantly affected sediment MBC, with PS having a more pronounced impact. The biphasic response in MBC highlights the complex and time-dependent nature of microbial community responses to plastisphere formation. Initially, lower MBC in MP-treated sediments (first 14 days) compared to the control indicates an acute stress response, possibly due to MP additive release [27]. The subsequent MBC increase above control levels suggests a remarkable adaptive capacity of sediment microbial communities. This adaptation may involve: 1) biofilm development on MPs creating new microbial niches; 2) microbial adaptation to use MPs or their degradation products as carbon sources; 3) community composition shifts favoring taxa suited to the altered sediment environment. Overall, PLA had a greater effect on EOC and DOC, and PS had a more significant effect on MBC and ROC.

The trend of ROC was consistent with that of SOC. It also concluded that the change of soil organic carbon was mainly related to the active components [74]. The relative stability of SOC and ROC suggests that short-term MP exposure may have limited impacts on long-term carbon sequestration in sediments. The more persistent carbon fractions in seagrass bed sediments may be relatively resistant to short-term perturbations caused by MP pollution. The stability of SOC and ROC may be due to physical protection within soil aggregates or chemical recalcitrance, rendering them less susceptible to MP-induced changes.

PLA-treated sediments presented a higher carbon pool activity index, suggesting that biodegradable PLA enhanced the turnover and recycling of organic carbon in sediments. This may be due to: 1) the biodegradable nature of PLA providing an additional carbon source, stimulating metabolic activity; 2) PLA creating a favorable surface for microbial colonization, leading to the development of a more active plastisphere influencing surrounding sediment processes; 3) PLA potential inducing a priming effect, where labile carbon from its degradation stimulates breakdown of native sediment organic matter. PS treatment showed lower carbon pool activity, suggesting less impact on carbon turnover compared to PLA. However, PLA-treated sediments consistently had lower CMI values than PS-treated sediments, indicating that although PLA increases organic carbon fraction dynamics, it simultaneously decreases sediment carbon pool stability. Furthermore, the observed peak of LI and CMI at 14 d may be attributed to new organic input stimulating microbial activity, increasing carbon pool turnover and temporary carbon storage. The subsequent decline may be due to: 1) substrate depletion: consumption of easily accessible carbon, leaving recalcitrant organic matter; 2) community adaptation: microbial compositional shifts, or functional adaptations to MPs, potentially altering carbon cycling dynamics. In conclusion, our results demonstrate that MPs significantly influence seagrass bed sediment carbon dynamics, with effects that are both polymer-specific and time-dependent.

PLS-SEM analysis indicates that MPs weathering can affect sediment organic carbon fractions by affecting sediment physicochemical properties and microbial community. MPs weathering presented a significantly negative correlation with sediment properties (-0.759, especially for NO_3^--N and NH_4^+-N), in turn, negatively correlating with both active and inert organic carbon fractions. Concurrently, MPs weathering slightly positively correlated with sediment microbes (0.233), in turn positively (0.406) and negatively (-0.163) correlating with active and inert organic carbon fractions, respectively. Ultimately, these changes cascade to affect carbon pool stability indicators, with active organic carbon exerting a stronger positive correlation compared to inert organic carbon. The negative correlation of MP weathering on sediment physicochemical characteristics and reactive carbon may potentially destroy the stability of organic carbon pools, while the positive correlation on microbial communities and their subsequent influence on organic carbon fractions may partially counteract this destabilization. This process underscores the intricate impact of microplastic pollution on carbon cycling in aquatic ecosystems, emphasizing the need for further research into these complex interactions.

4.5. *Limitation and future research prospects*

This study provides novel insights into plastisphere-mediated carbon

dynamics in seagrass ecosystems. However, it must be acknowledged that this study still has some limitations and further research could be investigated. Firstly, the challenges in quantifying and controlling background micro- and nanoplastic levels persist in sediments. The development of non-destructive methods to remove existing MPs from experimental substrates remains crucial to precisely assess MP impacts. Moreover, the comprehensive initial microbial community characterization needs further improvement to establish reliable baselines for community succession studies. Secondly, although our 56-day experimental period provided valuable insights, long-term field studies would better elucidate the succession patterns of plastisphere communities and their implications for sediment biogeochemistry under various environmental conditions, facilitating a more accurate assessment of MP-induced changes in seagrass ecosystem functions and potential ecological risks. It is also necessary to investigate the interactive effects between seagrass and plastisphere formation, the long-term consequences for seagrass ecosystem functioning, and potential feedback mechanisms between seagrass, sediment microbes, and plastic degradation. Furthermore, investigating the temperature-dependent dynamics of plastisphere development and associated ecosystem processes emerges as a critical research priority. Despite these limitations, our study provides valuable insights into MP effects on the sediment carbon cycle in seagrass ecosystems. The findings in this study establish a foundation for future research addressing the complex challenges of MP pollution in seagrass bed ecosystems.

5. Conclusions

This study revealed the dynamic nature of plastisphere formation and its significant impacts on sediment organic carbon pools and microbial communities in the seagrass bed ecosystem. Both PLA and PS exhibited obvious weathering characteristics and biofilm formation. MPs affected sediment physicochemical properties, organic carbon fractions, and carbon pool stability, especially during the first 14 days. MPs also altered the structure of microbial communities in seagrass bed sediments, leading to shifts in dominant species. Colonization time significantly influenced bacterial community composition and assembly in both plastisphere and sediments. Over time, the sediment bacterial communities became more stable, while plastisphere bacterial communities shifted from dispersal limitation to ecological drift. In contrast, sediment bacterial communities continued to exhibit dispersal limitation, highlighting the distinct ecological characteristics of MPs as novel microbial habitats. MPs negatively affected sediment organic carbon pools by influencing sediment physicochemical properties and positively influencing microbial communities. Long-term MP exposure may alter the carbon sequestration capacity of seagrass bed sediments. Future research should focus on long-term field studies to further explore these effects and investigate potential synergistic interactions between MP pollution and other environmental stressors (e.g., climate change, eutrophication) on the functioning of the seagrass ecosystem.

Environmental implication

This study reveals the significant impact of plastisphere colonization on seagrass bed sediments, affecting both organic carbon stability and microbial communities. These findings emphasize the broader ecological consequences of plastic pollution in marine ecosystems, potentially altering carbon sequestration capacity and biogeochemical cycles. The research underscores the pressing need for enhanced plastic waste management strategies and targeted conservation efforts to safeguard coastal blue carbon ecosystems, which play a critical role in mitigating climate change and supporting marine biodiversity.

CRediT authorship contribution statement

Fan Jinluo: Software, Methodology. **Luo Mingguang:** Resources, Methodology. **Deng Hui:** Writing – review & editing, Software, Methodology. **Yang Xing:** Writing – review & editing. **Wang Jun:** Methodology, Formal analysis. **Feng Dan:** Writing – review & editing, Supervision, Resources. **Wu Dongming:** Writing – review & editing, Methodology, Formal analysis. **Chen Shiquan:** Writing – review & editing, Methodology. **Hou Shuailing:** Writing – original draft, Software. **Luo Jiwei:** Writing – review & editing, Software, Resources. **Zhao Yuanyuan:** Software, Methodology, Investigation. **Ge Chengjun:** Writing – review & editing, Supervision, Funding acquisition.

Declaration of Competing Interest

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

Acknowledgments

This work was financially supported by the Natural Science Foundation of Hainan Province (422QN262), the National Science Foundation of China (No. 42277315, 22066009, 22266014), Innovational Fund for Scientific and Technological Personnel of Hainan Province (KJRC2023C12), Key research and development projects of Hainan Province (ZDYF2022SHFZ025) and Hainan University Scientific Research Start-up Fund (KYQD(ZR)-21017).

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2025.138078.

Data availability

Data will be made available on request.

References

[1] Filho, W.L., Havea, P.H., Balogun, A.-L., Boenecke, J., Maharaj, A.A., Ha'apio, M., Hemstock, S.L., 2019. Plastic debris on Pacific Islands: ecological and health implications. Sci Total Environ 670, 181–187. https://doi.org/10.1016/j.scitotenv.2019.03.181.
[2] Amaral-Zettler, L.A., Zettler, E.R., Mincer, T.J., 2020. Ecology of the plastisphere. Nat Rev Microbiol 18, 139–151. https://doi.org/10.1038/s41579-019-0308-0.
[3] Zettler, E.R., Mincer, T.J., Amaral-Zettler, L.A., 2013. Life in the "plastisphere": microbial communities on plastic marine debris. Environ Sci Technol 47, 7137–7146. https://doi.org/10.1021/es401288x.
[4] Seeley, M.E., Song, B., Passie, R., Hale, R.C., 2020. Microplastics affect sedimentary microbial communities and nitrogen cycling. Nat Commun 11, 2372. https://doi.org/10.1038/s41467-020-16235-3.
[5] Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., Russell, A.E., 2004. Lost at sea: where is all the plastic?, 838-838 Science 304. https://doi.org/10.1126/science.1094559.
[6] Thompson, R.C., Courtene-Jones, W., Boucher, J., Pahl, S., Raubenheimer, K., Koelmans, A.A., 2024. Twenty years of microplastic pollution research—what have we learned? Science 386, eadl2746. https://doi.org/10.1126/science.adl2746.
[7] Ivar do Sul, J.A., Costa, M.F., 2014. The present and future of microplastic pollution in the marine environment. Environ Pollut 185, 352–364. https://doi.org/10.1016/j.envpol.2013.10.036.
[8] Karthik, R., Robin, R.S., Purvaja, R., Ganguly, D., Anandavelu, I., Raghuraman, R., Hariharan, G., Ramakrishna, A., Ramesh, R., 2018. Microplastics along the beaches of southeast coast of India. Sci Total Environ 645, 1388–1399. https://doi.org/10.1016/j.scitotenv.2018.07.242.
[9] Obbard, R.W., Sadri, S., Wong, Y.Q., Khitun, A.A., Baker, I., Thompson, R.C., 2014. Global warming releases microplastic legacy frozen in Arctic Sea ice. Earth'S Future 2, 315–320. https://doi.org/10.1002/2014EF000240.
[10] M. Bergmann, L. Gutow, M. Klages, Alfred-Wegener-Institut, Göteborgs universitet, eds., Marine anthropogenic litter, Springer, Cham Heidelberg New York Dordrecht London, 2015.
[11] Li, C., Gillings, M.R., Zhang, C., Chen, Q., Zhu, D., Wang, J., Zhao, K., Xu, Q., Leung, P.H., Li, X., Liu, J., Jin, L., 2024. Ecology and risks of the global plastisphere as a newly expanding microbial habitat. Innovation 5, 100543. https://doi.org/10.1016/j.xinn.2023.100543.
[12] Bhagwat, G., Zhu, Q., O'Connor, W., Subashchandrabose, S., Grainge, I., Knight, R., Palanisami, T., 2021. Exploring the composition and functions of plastic

microbiome using whole-genome sequencing. Environ Sci Technol 55, 4899–4913. https://doi.org/10.1021/acs.est.0c07952.
[13] Arias-Andres, M., Kettner, M.T., Miki, T., Grossart, H.-P., 2018. Microplastics: new substrates for heterotrophic activity contribute to altering organic matter cycles in aquatic ecosystems. Sci Total Environ 635, 1152–1159. https://doi.org/10.1016/j.scitotenv.2018.04.199.
[14] Nordlund, L.M., Jackson, E.L., Nakaoka, M., Samper-Villarreal, J., Beca-Carretero, P., Creed, J.C., 2018. Seagrass ecosystem services – what's next? Mar Pollut Bull 134, 145–151. https://doi.org/10.1016/j.marpolbul.2017.09.014.
[15] Ajith, N., Arumugam, S., Parthasarathy, S., Manupoori, S., Janakiraman, S., 2020. Global distribution of microplastics and its impact on marine environment—a review. Environ Sci Pollut Res 27, 25970–25986. https://doi.org/10.1007/s11356-020-09015-5.
[16] Andrady, A.L., 2011. Microplastics in the marine environment. Mar Pollut Bull 62, 1596–1605. https://doi.org/10.1016/j.marpolbul.2011.05.030.
[17] Goss, H., Jaskiel, J., Rotjan, R., 2018. *Thalassia testudinum* as a potential vector for incorporating microplastics into benthic marine food webs. Mar Pollut Bull 135, 1085–1089. https://doi.org/10.1016/j.marpolbul.2018.08.024.
[18] Jones, K.L., Hartl, M.G.J., Bell, M.C., Capper, A., 2020. Microplastic accumulation in a *Zostera marina* L. bed at Deerness Sound, Orkney, Scotland. Mar Pollut Bull 152, 110883. https://doi.org/10.1016/j.marpolbul.2020.110883.
[19] Brodersen, K.E., Koren, K., Revsbech, N.P., Kühl, M., 2020. Strong leaf surface basification and CO_2 limitation of seagrass induced by epiphytic biofilm microenvironments. Plant Cell Environ 43, 174–187. https://doi.org/10.1111/pce.13645.
[20] Silberstein, K., Chiffings, A.W., McComb, A.J., 1986. The loss of seagrass in cockburn sound, Western Australia. III. The effect of epiphytes on productivity of *Posidonia australis* Hook. F. Aquat Bot 24, 355–371. https://doi.org/10.1016/0304-3770(86)90102-6.
[21] Menicagli, V., Balestri, E., Vallerini, F., De Battisti, D., Lardicci, C., 2021. Plastics and sedimentation foster the spread of a non-native macroalga in seagrass meadows. Sci Total Environ 757, 143812. https://doi.org/10.1016/j.scitotenv.2020.143812.
[22] Gerstenbacher, C.M., Finzi, A.C., Rotjan, R.D., Novak, A.B., 2022. A review of microplastic impacts on seagrasses, epiphytes, and associated sediment communities. Environ Pollut 303, 119108. https://doi.org/10.1016/j.envpol.2022.119108.
[23] Van Cauwenberghe, L., Devriese, L., Galgani, F., Robbens, J., Janssen, C.R., 2015. Microplastics in sediments: A review of techniques, occurrence and effects. Mar Environ Res 111, 5–17. https://doi.org/10.1016/j.marenvres.2015.06.007.
[24] Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627. https://doi.org/10.1126/science.1097396.
[25] Ganguly, D., Singh, G., Ramachandran, P., Selvam, A.P., Banerjee, K., Ramachandran, R., 2017. Seagrass metabolism and carbon dynamics in a tropical coastal embayment. Ambio 46, 667–679. https://doi.org/10.1007/s13280-017-0916-8.
[26] Serrano, O., Lavery, P.S., Duarte, C.M., Kendrick, G.A., Calafat, A., York, P.H., Steven, A., Macreadie, P.I., 2016. Can mud (silt and clay) concentration be used to predict soil organic carbon content within seagrass ecosystems. Biogeosciences 13, 4915–4926. https://doi.org/10.5194/bg-13-4915-2016.
[27] Liang, C., Schimel, J.P., Jastrow, J.D., 2017. The importance of anabolism in microbial control over soil carbon storage. Nat Microbiol 2, 1–6. https://doi.org/10.1038/nmicrobiol.2017.105.
[28] Hou, X., Li, C., Zhao, Y., He, Y., Li, W., Wang, X., Liu, X., 2024. Distinct impacts of microplastics on the carbon sequestration capacity of coastal blue carbon ecosystems: a case of seagrass beds. Mar Environ Res 202, 106793. https://doi.org/10.1016/j.marenvres.2024.106793.
[29] Huang, Y., Xiao, X., Effiong, K., Xu, C., Su, Z., Hu, J., Jiao, S., Holmer, M., 2021. New insights into the microplastic enrichment in the blue carbon ecosystem: evidence from seagrass meadows and mangrove forests in coastal South China sea. Environ Sci Technol. https://doi.org/10.1021/acs.est.0c07289.
[30] Rillig, M.C., Leifheit, E., Lehmann, J., 2021. Microplastic effects on carbon cycling processes in soils. PLoS Biol 19, e3001130. https://doi.org/10.1371/journal.pbio.3001130.
[31] Fuller, S., Gautam, A., 2016. A procedure for measuring microplastics using pressurized fluid extraction. Environ Sci Technol 50, 5774–5780. https://doi.org/10.1021/acs.est.6b00816.
[32] Zhang, S.-J., Zeng, Y.-H., Zhu, J.-M., Cai, Z.-H., Zhou, J., 2022. The structure and assembly mechanisms of plastisphere microbial community in natural marine environment. J Hazard Mater 421, 126780. https://doi.org/10.1016/j.jhazmat.2021.126780.
[33] Huang, Y., Xiao, X., Xu, C., Perianen, Y.D., Hu, J., Holmer, M., 2020. Seagrass beds acting as a trap of microplastics - emerging hotspot in the coastal region. Environ Pollut 257, 113450. https://doi.org/10.1016/j.envpol.2019.113450.
[34] Tu, C., Chen, T., Zhou, Q., Liu, Y., Wei, J., Waniek, J.J., Luo, Y., 2020. Biofilm formation and its influences on the properties of microplastics as affected by exposure time and depth in the seawater. Sci Total Environ 734, 139237. https://doi.org/10.1016/j.scitotenv.2020.139237.
[35] Almond, J., Sugumaar, P., Wenzel, M.N., Hill, G., Wallis, C., 2020. Determination of the carbonyl index of polyethylene and polypropylene using specified area under band methodology with ATR-FTIR spectroscopy. E-Polym 20, 369–381. https://doi.org/10.1515/epoly-2020-0041.
[36] Chen, C.-F., Ju, Y.-R., Lim, Y.C., Chen, C.-W., Dong, C.-D., 2021. Seasonal variation of diversity, weathering, and inventory of microplastics in coast and harbor sediments. Sci Total Environ 781, 146610. https://doi.org/10.1016/j.scitotenv.2021.146610.
[37] Liu, H., Yang, X., Liu, G., Liang, C., Xue, S., Chen, H., Ritsema, C.J., Geissen, V., 2017. Response of soil dissolved organic matter to microplastic addition in Chinese loess soil. Chemosphere 185, 907–917. https://doi.org/10.1016/j.chemosphere.2017.07.064.
[38] Blair, G.J., Lefroy, R.D.B., Lisle, L., 1995. Soil carbon fractions based on their degree of oxidation, and the development of a carbon management index for agricultural systems. Aust J Agric Res 46, 1459–1466. https://doi.org/10.1071/ar9951459.
[39] Modak, K., Ghosh, A., Bhattacharyya, R., Biswas, D.R., Das, T.K., Das, S., Singh, G., 2019. Response of oxidative stability of aggregate-associated soil organic carbon and deep soil carbon sequestration to zero-tillage in subtropical India. Soil Tillage Res 195, 104370. https://doi.org/10.1016/j.still.2019.104370.
[40] Deng, H., Zhang, Y., Li, D., Fu, Q., He, J., Zhao, Y., Feng, D., Yu, H., Ge, C., 2023. Mangrove degradation retarded microplastics weathering and affected metabolic activities of microplastics-associated microbes. J Hazard Mater 445, 130535. https://doi.org/10.1016/j.jhazmat.2022.130535.
[41] Li, C., Wang, L., Ji, S., Chang, M., Wang, L., Gan, Y., Liu, J., 2021. The ecology of the plastisphere: microbial composition, function, assembly, and network in the freshwater and seawater ecosystems. Water Res 202, 117428. https://doi.org/10.1016/j.watres.2021.117428.
[42] Wu, Z., Lyu, H., Liang, W., Jing, X., Wang, Y., Ma, X., 2021. Microbial community in indoor dusts from university dormitories: characteristics, potential pathogens and influence factors. Atmos Pollut Res 12, 321–333. https://doi.org/10.1016/j.apr.2020.12.018.
[43] Duan, J., Bolan, N., Li, Y., Ding, S., Atugoda, T., Vithanage, M., Sarkar, B., Tsang, D.C.W., Kirkham, M.B., 2021. Weathering of microplastics and interaction with other coexisting constituents in terrestrial and aquatic environments. Water Res 196, 117011. https://doi.org/10.1016/j.watres.2021.117011.
[44] Zhou, Q., Tu, C., Fu, C., Li, Y., Zhang, H., Xiong, K., Zhao, X., Li, L., Waniek, J.J., Luo, Y., 2020. Characteristics and distribution of microplastics in the coastal mangrove sediments of China. Sci Total Environ 703, 134807. https://doi.org/10.1016/j.scitotenv.2019.134807.
[45] Gupta, P., Saha, M., Rathore, C., Suneel, V., Ray, D., Naik, A., U. K, D. M, Daga, K., 2021. Spatial and seasonal variation of microplastics and possible sources in the estuarine system from central west coast of India. Environ Pollut 288, 117665. https://doi.org/10.1016/j.envpol.2021.117665.
[46] Tiwari, M., Rathod, T.D., Ajmal, P.Y., Bhangare, R.C., Sahu, S.K., 2019. Distribution and characterization of microplastics in beach sand from three different Indian coastal environments. Mar Pollut Bull 140, 262–273. https://doi.org/10.1016/j.marpolbul.2019.01.055.
[47] Zhang, W., Bhagwat, G., Palanisami, T., Liang, S., Wan, W., Yang, Y., 2024. Lacustrine plastisphere: Distinct succession and assembly processes of prokaryotic and eukaryotic communities and role of site, time, and polymer types. Water Res 248, 120875. https://doi.org/10.1016/j.watres.2023.120875.
[48] Shruti, V.C., Kutralam-Muniasamy, G., 2019. Bioplastics: missing link in the era of Microplastics. Sci Total Environ 697, 134139. https://doi.org/10.1016/j.scitotenv.2019.134139.
[49] Deng, H., Fu, Q., Zhang, Y., Li, D., He, J., Feng, D., Zhao, Y., Yu, H., Ge, C., 2022. Bacterial communities on polyethylene microplastics in mangrove ecosystems as a function of exposure sites: compositions and ecological functions. J Environ Chem Eng 10, 107924. https://doi.org/10.1016/j.jece.2022.107924.
[50] Li, C., Cui, Q., Li, Y., Zhang, K., Lu, X., Zhang, Y., 2022. Effect of LDPE and biodegradable PBAT primary microplastics on bacterial community after four months of soil incubation. J Hazard Mater 429, 128353. https://doi.org/10.1016/j.jhazmat.2022.128353.
[51] Ren, X., Tang, J., Liu, X., Liu, Q., 2020. Effects of microplastics on greenhouse gas emissions and the microbial community in fertilized soil. Environ Pollut 256, 113347. https://doi.org/10.1016/j.envpol.2019.113347.
[52] Martínez-Campos, S., González-Pleiter, M., Rico, A., Schell, T., Vighi, M., Fernández-Piñas, F., Rosal, R., Leganés, F., 2023. Time-course biofilm formation and presence of antibiotic resistance genes on everyday plastic items deployed in river waters. J Hazard Mater 443, 130271. https://doi.org/10.1016/j.jhazmat.2022.130271.
[53] Datta, M.S., Sliwerska, E., Gore, J., Polz, M.F., Cordero, O.X., 2016. Microbial interactions lead to rapid micro-scale successions on model marine particles. Nat Commun 7, 11965. https://doi.org/10.1038/ncomms11965.
[54] Wright, R.J., Erni-Cassola, G., Zadjelovic, V., Latva, M., Christie-Oleza, J.A., 2020. Marine plastic debris: a new surface for microbial colonization. Environ Sci Technol 54, 11657–11672. https://doi.org/10.1021/acs.est.0c02305.
[55] Oberbeckmann, S., Kreikemeyer, B., Labrenz, M., 2018. Environmental factors support the formation of specific bacterial assemblages on microplastics. Front Microbiol 8. https://doi.org/10.3389/fmicb.2017.02709.
[56] Jacquin, J., Cheng, J., Odobel, C., Pandin, C., Conan, P., Pujo-Pay, M., Barbe, V., Meistertzheim, A.-L., Ghiglione, J.-F., 2019. Microbial ecotoxicology of marine plastic debris: a review on colonization and biodegradation by the "plastisphere. Front Microbiol 10. https://doi.org/10.3389/fmicb.2019.00865.
[57] Xu, X., Wang, S., Gao, F., Li, J., Zheng, L., Sun, C., He, C., Wang, Z., Qu, L., 2019. Marine microplastic-associated bacterial community succession in response to geography, exposure time, and plastic type in China's coastal seawaters. Mar Pollut Bull 145, 278–286. https://doi.org/10.1016/j.marpolbul.2019.05.036.
[58] Delmont, T.O., Quince, C., Shaiber, A., Esen, Ö.C., Lee, S.T., Rappé, M.S., McLellan, S.L., Lücker, S., Eren, A.M., 2018. Nitrogen-fixing populations of planctomycetes and Proteobacteria are abundant in surface ocean metagenomes. Nat Microbiol 3, 804–813. https://doi.org/10.1038/s41564-018-0176-9.
[59] Tian, R., Ning, D., He, Z., Zhang, P., Spencer, S.J., Gao, S., Shi, W., Wu, L., Zhang, Y., Yang, Y., Adams, B.G., Rocha, A.M., Detienne, B.L., Lowe, K.A.,

Joyner, D.C., Klingeman, D.M., Arkin, A.P., Fields, M.W., Hazen, T.C., Stahl, D.A., Alm, E.J., Zhou, J., 2020. Small and mighty: adaptation of superphylum Patescibacteria to groundwater environment drives their genome simplicity. Microbiome 8, 51. https://doi.org/10.1186/s40168-020-00825-w.

[60] DeBruyn, J.M., Nixon, L.T., Fawaz, M.N., Johnson, A.M., Radosevich, M., 2011. Global biogeography and quantitative seasonal dynamics of gemmatimonadetes in soil. Appl Environ Microbiol 77, 6295–6300. https://doi.org/10.1128/AEM.05005-11.

[61] Zhao, Z., Wang, Y., Wei, Y., Peng, G., Wei, T., He, J., Li, R., Wang, Y., 2024. Distinctive patterns of bacterial community succession in the riverine microplastisphere in view of biofilm development and ecological niches. J Hazard Mater 480, 135974. https://doi.org/10.1016/j.jhazmat.2024.135974.

[62] Zengler, K., Zaramela, L.S., 2018. The social network of microorganisms — how auxotrophies shape complex communities. Nat Rev Microbiol 16, 383–390. https://doi.org/10.1038/s41579-018-0004-5.

[63] Zhu, D., Chen, Q.-L., An, X.-L., Yang, X.-R., Christie, P., Ke, X., Wu, L.-H., Zhu, Y.-G., 2018. Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. Soil Biol Biochem 116, 302–310. https://doi.org/10.1016/j.soilbio.2017.10.027.

[64] Bryant, J.A., Clemente, T.M., Viviani, D.A., Fong, A.A., Thomas, K.A., Kemp, P., Karl, D.M., White, A.E., DeLong, E.F., 2016. Diversity and activity of communities inhabiting plastic debris in the North Pacific Gyre. mSystems 1, e00024-16. https://doi.org/10.1128/mSystems.00024-16.

[65] Wang, K., Hu, H., Yan, H., Hou, D., Wang, Y., Dong, P., Zhang, D., 2019. Archaeal biogeography and interactions with microbial community across complex subtropical coastal waters. Mol Ecol 28, 3101–3118. https://doi.org/10.1111/mec.15105.

[66] Zhang, W., Chen, L., Chen, H., Liu, W., Yang, Y., 2022. Geographic dispersal limitation dominated assembly processes of bacterial communities on microplastics compared to water and sediment. Appl Environ Microbiol 88, e00482-22. https://doi.org/10.1128/aem.00482-22.

[67] Zhang, W., Liang, S., Grossart, H.-P., Christie-Oleza, J.A., Gadd, G.M., Yang, Y., 2024. Convergence effect during spatiotemporal succession of lacustrine plastisphere: loss of priority effects and turnover of microbial species. ISME Commun 4 (1), ycae056.

[68] Kirstein, I.V., Wichels, A., Krohne, G., Gerdts, G., 2018. Mature biofilm communities on synthetic polymers in seawater - specific or general? Mar Environ Res 142, 147–154. https://doi.org/10.1016/j.marenvres.2018.09.028.

[69] Rillig, M.C., Ingraffia, R., de Souza Machado, A.A., 2017. Microplastic incorporation into soil in agroecosystems. Front Plant Sci 8. https://doi.org/10.3389/fpls.2017.01805.

[70] Fukami, T., 2015. Historical contingency in community assembly: integrating niches, species pools, and priority effects. Annu Rev Ecol, Evol, Syst 46, 1–23. https://doi.org/10.1146/annurev-ecolsys-110411-160340.

[71] Zhou, J., Ning, D., 2017. Stochastic community assembly: does it matter in microbial ecology. Microbiol Mol Biol Rev 81, e00002-17. https://doi.org/10.1128/MMBR.00002-17.

[72] Nemergut, D.R., Schmidt, S.K., Fukami, T., O'Neill, S.P., Bilinski, T.M., Stanish, L. F., Knelman, J.E., Darcy, J.L., Lynch, R.C., Wickey, P., Ferrenberg, S., 2013. Patterns and processes of microbial community assembly. Microbiol Mol Biol Rev 77, 342–356. https://doi.org/10.1128/MMBR.00051-12.

[73] Yu, H., Zhang, Z., Zhang, Y., Fan, P., Xi, B., Tan, W., 2021. Metal type and aggregate microenvironment govern the response sequence of speciation transformation of different heavy metals to microplastics in soil. Sci Total Environ 752, 141956. https://doi.org/10.1016/j.scitotenv.2020.141956.

[74] Lal, R., 2006. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. Land Degrad Dev 17, 197–209. https://doi.org/10.1002/ldr.696.