Title

Micro- and macroplastic accumulation in a newly formed Spartina alterniflora colonized estuarine saltmarsh in southeast China

Authors

Yao, Weimin
Di, Di
Wang, Zhenfeng
et al.

Publication Date

2019-12-01

DOI

10.1016/j.marpolbul.2019.110636

Marine Pollution Bulletin 149 (2019) 110636

Contents lists available at ScienceDirect

Marine Pollution Bulletin

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

Micro- and macroplastic accumulation in a newly formed *Spartina alterniflora* colonized estuarine saltmarsh in southeast China

Weimin Yao[a,b], Di Di[a], Zhenfeng Wang[a], Zhonglu Liao[a], Hong Huang[a], Kun Mei[a], Randy A. Dahlgren[a,c,**], Minghua Zhang[a,c], Xu Shang[a,*]

[a] *Key Laboratory of Watershed Sciences and Health of Zhejiang Province, School of Public Health and Management, Wenzhou Medical University, Wenzhou, 325035, China*

[b] *Wenzhou Marine Environmental Monitoring Center Station, SOA, Wenzhou, 325027, China*

[c] *Department of Land, Air and Water Resources, University of California Davis, CA, 95616, USA*

ARTICLE INFO

Keywords:
Microplastic
Macroplastic
Saltmarsh
Spartina alterniflora
Degradation

ABSTRACT

In recent years, there is great concern about plastic pollution due to the identification of several environmental risks associated with microplastics (< 5 mm). This study investigated microplastic and macroplastic accumulation patterns in a newly formed *Spartina alterniflora* colonized saltmarsh of an estuary in southeastern China. Abundance of microplastic and macroplastic particles was in the range of 9600–130725 and 200-4350 n/m^2, respectively. Abundances of microplastics and macroplastics were highest at the saltmarsh edge, but the mass of macroplastics was highest in the saltmarsh interior. Mass of microplastics and macroplastics in bareflats was significantly lower than vegetated areas. Although microplastics accounted for 96.3% of total plastic abundance, macroplastics accounted for 90% of total plastic mass. Results showed that *S. alterniflora* dominated saltmarshes have a strong ability to trap plastic debris, especially macroplastics. Thus, coastal saltmarshes may serve as a transformer of macroplastics to microplastics and consequently as a source of microplastics to the ocean.

1. Introduction

Worldwide production of plastics has grown exponentially since the middle of the 20th century (Andrady, 2011). Due to poor waste management (e.g., low recycling rates), plastics are released into the environment and become a severe hazard (Rochman et al., 2013). An estimated 275 million tons of plastic waste were generated in 2010, nearly equivalent to the entire global plastic production (Jambeck et al., 2015). Up to 10% of plastic waste is eventually transported to oceans where it accumulates due to its low biodegradability (Jambeck et al., 2015). The cumulative quantity of plastic waste projected to enter the marine environment from land is predicted to increase an order of magnitude by 2025 compared to 2010, thus exacerbating ecological impacts (Thompson et al., 2009).

Consumption of plastics by marine organisms, such as birds, fishes, invertebrates and even mammals, is well documented (Browne et al., 2008; Oehlmann et al., 2009). Although it is unknown exactly how many animals are killed by plastic pollution, as many as 100,000 marine mammals are estimated to die from entanglement every year (Gregory, 2009). Further, plastic debris can cause negative effects on the physiology, reproduction, and diversity of marine biota (Foley et al., 2018). Plastics also serve as a vector for exposure to persistent organic pollutants (POPs), such as flame retardants, bisphenol A, and antimicrobials, which are adsorbed to the surface of plastics (Browne et al., 2013). Toxic chemicals leached from plastic debris also create potential health risks to wildlife and humans (Bouwmeester et al., 2015; Rist et al., 2018). Plastic debris is broken down into smaller pieces by physical, chemical and biological processes, such as photodegradation and mechanical abrasion (Andrady, 2011). Microplastics (< 5 mm) are of increasing concern as a global environmental threat (Cole et al., 2011; Eerkes-Medrano et al., 2015; Wright and Kelly, 2017). Controlled laboratory experiments showed microplastics present ecotoxicological risk to many aquatic animals like zooplankton, lugworms, amphipods, mussels and fishes (Von Moos et al., 2012; Oliveira et al., 2013; Cole et al., 2015; Chua et al., 2014). As microplastics are highly persistent and their concentrations are projected to rapidly increase in the future, microplastic risks to humans is of increasing concern via exposure through the food web (Browne et al., 2008; Wright and Kelly, 2017).

* Corresponding author.
** Corresponding author. Key Laboratory of Watershed Sciences and Health of Zhejiang Province, School of Public Health and Management, Wenzhou Medical University, Wenzhou, 325035, China.
E-mail addresses: radahlgren@ucdavis.edu (R.A. Dahlgren), xshang@wmu.edu.cn (X. Shang).

https://doi.org/10.1016/j.marpolbul.2019.110636
Received 17 May 2019; Received in revised form 25 September 2019; Accepted 25 September 2019

Plastic debris is found throughout the world's ocean, from surface waters to seabed sediments, shoreline to open ocean, and tropics to polar regions (Cole et al., 2011). As rivers are considered as primary vectors for plastic debris to enter the ocean (Klein et al., 2015), estuaries likely play an important role in its fate and transport processes. Estuaries are the key pathway for transport of materials across the land-ocean interface and act as effective filters for nutrients and contaminants (Van Beusekom and de Jonge, 1998; Thévenot et al., 2007). Depositional processes predominant in estuarine areas create extensive tidal flats that may act as an important sink for anthropogenic pollutants. Terrestrial pollutants, like heavy metals and persistent organic pollutants, tend to accumulate in tidal flats and biomagnify through the food web (Gedan et al., 2017). As estuarine tidal flats provide a vital feeding ground for migratory birds, as well as nursery areas for fish and invertebrate (Boesch and Turner, 1984; Kneib, 1997), the estuarine accumulation of pollutants poses a threat to estuarine biota, as well as offshore marine organisms. Many studies report ubiquitous distribution of plastic debris in coastal regions, but they are generally restricted to high energy beach environments (Van Cauwenberghe et al., 2015). Lower hydrodynamic power in tidal flats is considered favorable for retention of plastic debris (Browne et al., 2011). For example, tidal flats with low water movement presented higher microplastic abundance compare to a nearby beach (Liebezeit and Dubaish, 2012; Vianello et al., 2013; Lo et al., 2018). Additionally, shoreline vegetation can significantly promote sediment retention on tidal flats, such as high plastic deposition found in mangroves (Nor and Obbard, 2014). Notably, one study showed that plastic degradation proceeded relatively quickly in the saltmarsh environment (Weinstein et al., 2016). Given the importance of coastal and estuarine ecosystems in regulating the fate and transport of pollutants, the paucity of information for microplastic dynamics in these systems emphasizes the critical need for additional research.

China is the largest producer of plastic waste, and its output of plastic to the ocean is assumed to exceed 25% of the total global loading (Jambeck et al., 2015). Investigations show that microplastic pollution is especially serious in coastal and estuarine regions of China from north to south (Zhao et al., 2014; Qiu et al., 2015; Peng et al., 2017; Zhu et al., 2018), including the Ou River estuary at Southeast China where this study is located (Zhao et al., 2015). Similar to most coastal zones in China, the sediment settling rate and hence expansion of tidal mudflats surrounding the Ou River estuary are greatly accelerated by invasion of the saltmarsh cordgrass *Spartina alterniflora* (Wang et al., 2015). Following a recent large-scale land reclamation project, a tidal flat saltmarsh quickly formed outside of the reclamation area in the Ou River estuary largely due to *S. alterniflora* colonization. *S. alterniflora* is considered to greatly enhance the accretion rate of saltmarshes by reducing water flow velocity and trapping suspended particles (Stumpf, 1983). Therefore, the present study tested the hypothesis that saltmarsh environments have a higher trapping efficiency for plastic debris than adjacent unvegetated mudflats, and these coastal wetlands may act as an important sink for trapping plastics prior to entering the ocean.

2. Materials and methods

2.1. Study area

The study was carried out in a newly formed saltmarsh on Linkun Island located in the mouth of the Ou River, southeast Zhejiang, China (27º56′40″-27º57′39″N, 120º56′25″-120º57′54″E) (Fig. S1). The Ou River is the second largest river in Zhejiang Province and has a drainage area of more than 18000 km^2 with mean annual runoff greater than 20 billion m^3. The Ou River watershed is among the fastest developing areas of China, with a population of more than 10 million and GDP of ~400 billion RMB. Linkun Island is an estuarine deposited island that expanded in area from ~25 km^2 in 2007 to ~50 km^2 in 2016 due to extensive reclamation activities since 2010 (Fig. S1).

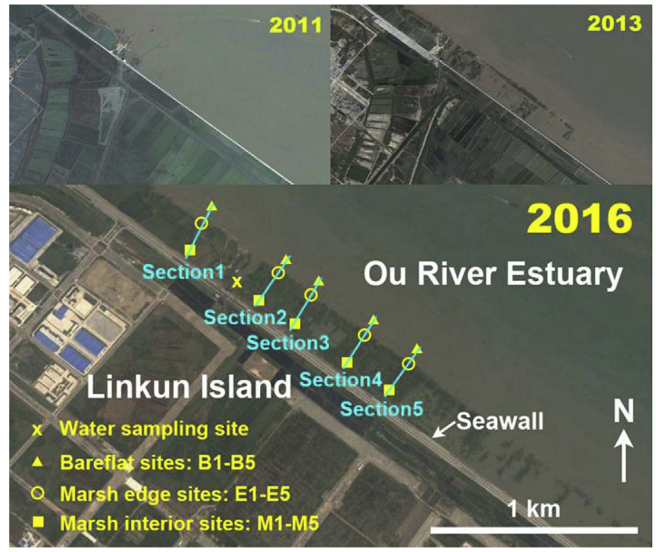

Fig. 1. Sampling sites in the saltmarsh outside the constructed seawall on Linkun Island. Satellite photos show the formation of the *S. alterniflora* dominated saltmarsh from 2011 to 2016.

The saltmarsh naturally formed outside of the seawall following its construction in 2010 (Fig. 1). *Spartina alterniflora*, which is the dominant invasive species along the China coast, contributed to rapid saltmarsh formation due to its ability to stabilize the seashore following tidal land reclamation (Fig. 1). *S. alterniflora* height was approximately 100–150 cm and density ranged from 50-200 plants/m^2.

2.2. Sampling

Samples were collected during low tide along 5 transects perpendicular to the coastline in the dry season (December 2016) (Fig. 1). Along each transect (250–300 m), three sites were established consisting of bare mudflat (intertidal), saltmarsh edge (covered by 2–3 cm of litter), and saltmarsh interior (120–150 cm elevation, covered by 10–15 cm of litter). At each of the three transect sites, 3 replicate samples located 5–10 m apart were randomly collected for a total of 45 samples (5 transects × 3 sites × 3 replicates). Areas covered by highly localized, excessive plastic debris (i.e., non-representative hotspots) were avoided. Litter on the surface and the sediment of 0–2 cm and 2–4 cm depths were collected from a 0.04 m^2 area using a stainless steel frame (20 × 20 cm). Sampling equipment was rinsed free of sediments with distilled water between samples to avoid cross contamination. Samples were stored in 1 L glass jars for transport to laboratory. Additionally, triplicate 100 L samples of surface water from the tidal creek between transects 1 and 2 were collected during high tide. Water samples were passed through a 25-μm stainless-steel sieve and residue on the sieve was transferred with distilled water to 250 ml glass jars for further processing.

2.3. Sample preparation

In the laboratory, litter samples were rinsed 5 × with distilled water and sieved with a 5-mm stainless-steel screen to separate macroplastic particles (> 5 mm). All > 5 mm particles were visually inspected and the suspected plastic items retained for further identification. Microplastics (< 5 mm) passing the 5-mm sieve were passed through a 25-μm stainless-steel sieve and the residue on the sieve transferred to glass petri dishes for further identification.

For sediment samples, 500 g wet sediment from each depth (0-2 & 2–4 cm) was dried at 60 °C for 48 h. Triplicate sediment samples from each site were then pooled to give a total of 15 surface (0–2 cm) and 15 sub-surface (2–4 cm) composite samples. A 100 g subsample from each well-mixed composite sample was sieved through a 5-mm stainless-steel sieve; items retained on the 5-mm sieve were visually inspected and the suspected macroplastics were retained for further identification.

The < 5-mm sediment fraction was sieved through a 25-μm stainless-steel sieve to retain plastic particles > 25 μm, thereby defining microplastics as the 25 μm to 5 mm fraction in this study. To recover microplastics from sediment, the 25 μm ~ 5 mm fraction was treated with 30% hydrogen peroxide oxidation at 70 °C for 2 h to digest natural organic matter present in sediments (Masura et al., 2015). Then, density separation using a $ZnCl_2$ solution (density = 1.7 g ml^{-1}) was utilized to separate microplastics through floatation. The density separation step was repeated and supernatants combined. The supernatant with isolated microplastics was filtered through a 25-μm stainless-steel sieve and the residue transferred to glass petri dishes for further identification.

2.4. Analysis

A combined method of visual identification and Micro Fourier Transform Infrared Spectrometer analysis (μ-FT-IR) was applied to minimize false-positive misidentifications (i.e., non-plastics recorded as microplastics). Based on their characteristic shape, cleavage and color, most plastic items were easy to visually distinguish from non-plastic items (Crawford and Quinn, 2017). All remaining > 5-mm items that could not be identified as plastic versus non-plastic during initial sorting, and 20 items in each 25 μm ~ 5 mm fraction (about 10–30% of total suspected microplastics) were randomly selected to verify polymer composition using μ-FT-IR (VERTEX 70 plus HYPERION 2000; Bruker, USA). Thus, μ-FT-IR results were used to establish a general rule for excluding non-plastic items in the visual examination using a microscope. Resulting spectra were compared to a known polymer spectra library to identify the chemical composition using a criterion of at least 60% similarity for confirmation.

After isolation from non-plastic items, macroplastics were counted, measured under a stereo microscope (XTZ-D, Sgaaa) at up to 40 × magnification, and weighed on pre-weighed filters to 0.1 mg using a digital balance (BSA224S, Sartorius, Germany). Microplastics were counted and measured using a stereo microscope (M165FC, Leica) at up to 120 × magnification allowing identification of particles > 25 μm. According to morphology, microplastics were classified as fragment (fragment of large plastic waste), fiber (fibrous plastic), pellet (industrial plastic pellet), or foam (lightweight, sponge-like plastic). Based on their size, microplastics were divided into four size classes (based on longest particle dimension): 25-100, 100-300, 300-1000, and 1000–5000 μm. After counting and measurement, microplastics > 300 μm (Large microplastics, L-MPs) were hand-sorted using a forceps and needle for weighing as size groups. Since microplastics < 300 μm (small microplastics, S-MPs) were difficult to isolate and weigh, the weight of these particles was determined by assuming a density and multiplying by the volume; the volume determined by using the biovolume calculation for microalgae of varied shapes by geometric shape-based mathematical equations (See supporting information). Microplastic content was converted to a whole sample dry-weight equivalent, and then converted to an area basis using sediment bulk weight and sampling area.

To avoid contamination from airborne microplastics, extraction processes were performed in a laminar-flow hood, and all glassware was thoroughly cleaned before use. All samples and equipment were covered with glass petri dishes or aluminum foil after cleaning. Laboratory blanks of distilled water were used as a "negative control" during extraction and identification processes, which confirmed that background contamination was negligible (See supporting information).

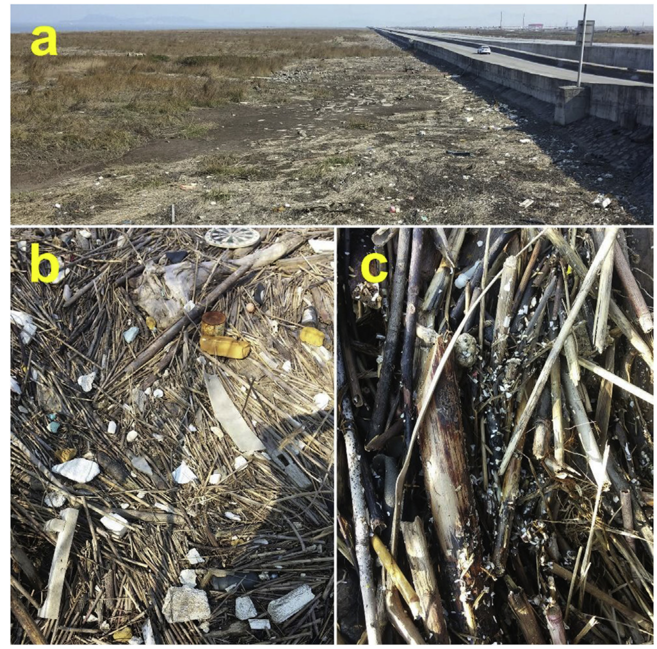

Fig. 2. Plastic debris in the *Spartina alterniflora* dominated saltmarsh outside the seawall on Linkun Island. (a) Riverine litter accumulated in the saltmarsh; (b) Macroplastics in plant litter on marsh interior site; (c) Microplastics in plant litter on marsh edge site.

2.5. Statistical analysis

One-way ANOVA followed by the Holm-Sidak all-pairwise multiple comparison test was used to analyze differences in spatial variation of mass per item of plastic debris, spatial size group variation of plastic debris on the beach surface, and vertical size distribution of plastic debris in the litter layer and beach sediments. All statistical differences were evaluated at a $p \leq 0.05$ level of significance.

3. Results

3.1. Abundance of plastic debris

Micro- and macroplastics were ubiquitous at all sampling sites. Most of the surface area for the salt marsh and edge sites was covered by plant litter that incorporated large quantities of micro- and macroplastics (Fig. 2). Abundances of micro- and macroplastics on the beach ranged between 9600-130725 and 200-4350 n/m^2, respectively (Table 1). Although the abundance of microplastics was significantly higher than macroplastics, the mass of macroplastics (126.9 ± 95.4 g/m^2) was an order of magnitude higher than microplastics (14.3 ± 18.3 g/m^2) (Table 1). In the adjacent tidal creek water, micro- and macroplastic abundances were 3470 n/m^3 and 70 n/m^3, corresponding to masses of 0.28 g/m^3 and 0.85 g/m^3, respectively. The average mass of individual microplastic and macroplastic item on the beach vs water column was 4.1 and 8.4 times, respectively.

3.2. Size distribution of plastic debris

Microplastics constituted the majority of plastic debris (96.3%) in terms of particle abundance with the mean abundance of L-MPs (300 ~ 5000 μm) slightly higher than that of S-MPs (< 300 μm) (Fig. 3a). However, nearly 90% of plastic debris mass was contributed by macroplastics; L-MPs contributed about 10% and S-MPs were negligible (Fig. 3b). Considering microplastic size distribution, the most prominent size range was 50–100 μm with a decreasing abundance with increasing size on the beach, while microplastics smaller than 50 μm

Table 1
Abundance and mass of micro- and macroplastics on the beach and tidal creek water column.

		Abundance (n/m^2)		Mass (g/m^2)		Mass per item (mg/item)	
		Micro-	Macro-	Micro-	Macro-	Micro-	Macro-
Beach	Min-Max	9600-130725	200-4350	1.1-55.1	0.01-298.8	ND	ND
	Mean ± SD	37375 ± 40325	1417 ± 1194	14.3 ± 18.3	126.9 ± 95.4	0.33 ± 0.22	101.9 ± 102.9
Water		3470*	70*	0.28**	0.85**	0.08	12.1

ND = not determined; *n/m^3; **g/m^3.

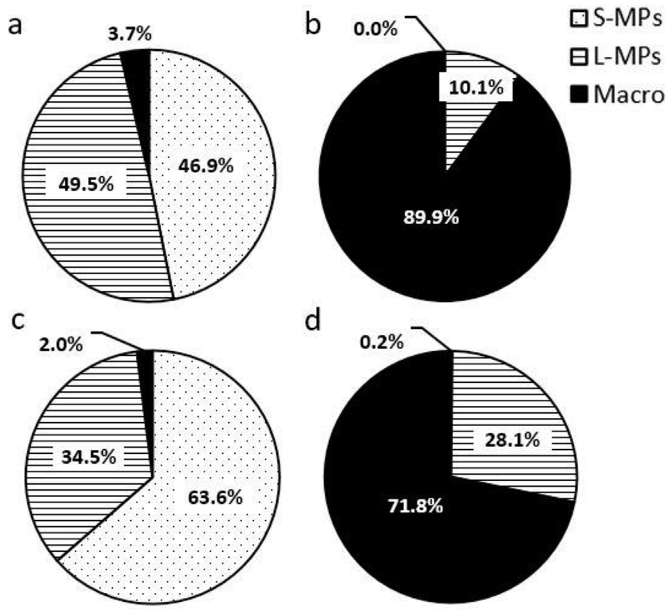

Fig. 3. Size distribution of small (S-MPs, 25–300 μm), large (L-MPs, 300–5000 μm) microplastics and macroplastics (Macro, > 5000 μm) on the beach (a, b) and in the tidal creek water (c, d). a, c: based on abundance (n/m^2); b, d: based on mass (g/m^2).

were predominant in tidal creek water (Fig. S4).

In tidal creek water, macroplastic abundance (2.0%) was less than that on the beach (3.7%) (Fig. 3c), but macroplastics still dominated in mass (Fig. 3d). In contrast to beach plastics, abundance of S-MPs (63.6%) was higher than L-MPs (34.5%) in tidal creek water (Fig. 3c). Although S-MPs accounted for nearly two-thirds of total plastic abundance, they only contributed 0.2% of total mass (Fig. 3d).

While the amount of plastic debris on Linkun Island's saltmarsh beach was dominated by microplastics, the contribution of microplastics to total plastic mass was negligible (Fig. 3). Average mass per item did not differ among S-MPs in the tidal creek water and beach samples of different elevations, but L-MPs mass on the beach was greater than those in tidal creek water (Fig. S5). With increasing elevation, the mass per item of macroplastics increased significantly from tidal creek water to marsh interior sites (Fig. S5). The mass of macroplastics made up only 3.7% of the total plastic abundance, but made up 90% of total plastic mass (Fig. 3).

3.3. Shape distribution of plastic debris

Fragment shaped plastics dominated the micro- and macroplastic fractions on the beach followed by foams. The mass of fragments exceeded 90% of the total microplastic mass and about 50% of macroplastics (Fig. 4). Fibers contributed 14.9% of microplastics and 14.1% of macroplastics in number, but were negligible in mass for both the micro- and macroplastic fractions. Pellets contributed 13.3% of microplastics and 2.1% of macroplastics by number, but contributed less than 3% of microplastic mass and a negligible fraction of macroplastic mass. Foams accounted for only 6.4% of microplastic mass, but were much more important for macroplastic mass (49.8%).

The shape of microplastics in the tidal creek water was different from the beach, as fibers dominated in number (36.6%), but were negligible in mass (Fig. 4). Although fragments accounted for less than 1/3 of total microplastics in number, they contributed more than 80% of microplastic mass. Compared to beach samples, the proportion of foams in the tidal water sample was similar (24.5–20.5%) in number but much higher (18.3–6.4%) in mass.

3.4. Spatial variation of plastic debris

Spatial distribution of plastic debris was highly variable among different sites along the beach transect (e.g., bareflat vs edge vs marsh). The abundance of micro- and macroplastics in edge sites was 85830 ± 35697 and 2835 ± 980 n/m^2, respectively, both significantly higher than those in bareflat and marsh sites (Fig. S6a). Although the mass of macroplastics in marsh sites (192.4 ± 64.4 g/m^2) was similar to edge sites (174.7 ± 56.8 g/m^2), the mass of microplastics in edge sites (35.5 ± 17.8 g/m^2) was much higher than marsh sites (5.3 ± 3.3 g/m^2), thereby resulting in the highest mass of total plastic debris at edge sites (Fig. S6b). The abundance of microplastics in bareflat sites was higher than marsh sites, but both the mass of micro- and macroplastics in bareflat sites were far below that in edge and marsh sites, resulting in a significantly lower mass of plastic debris in bareflat sites.

The contribution of different plastic size fractions varied among sampling sites. The highest contribution of microplastics was found in tidal creek water, while the highest abundance of macroplastics occurred in marsh sites (Fig. S7a). With increasing elevation from tidal creek to marsh sites, the fraction of macroplastics to total plastic mass increased from 71.8 to 95.9% (Fig. S7b). While S-MPs dominated total plastic debris in tidal creek water and bareflat sites, the contribution of L-MPs was higher in edge and marsh sites.

Fragments were the largest proportion of microplastics in the bareflat and edge sites, accounting for 32.3% and 57.1% of the microplastic fraction, respectively (Fig. S8a). Foam dominated microplastics in marsh sites, but fragments contributed most of the microplastic mass at all transect positions, such as 87.3% in the bareflat, 93.0% in the marsh edge and 77.8% in the saltmarsh interior (Fig. S8c). In contrast, the shape of macroplastics varied widely among transect positions. With rising elevation, the proportion of fragments in the macroplastic category decreased from 72.7% in the bareflat to 26.2% in the marsh center, while the proportion of foams increased from zero to 56.9% (Fig. S8b). Accordingly, the proportion of fragments in the total macroplastic mass decreased gradually from nearly 100% in the bareflat to only 14.3% in the marsh interior, where the remaining 85.7% of the mass was contributed by large foams (Fig. S8d).

Average abundance of plastic debris (13348 ± 15563 n/m^2) in beach litter was less than that in the sediment (surface plus subsurface) (29893 ± 31040 n/m^2) (Fig. 5a). In contrast, the average mass of plastic debris in the litter layer (107.8 ± 85.7 g/m^2) was higher than in the sediment (69.3 ± 78.1 g/m^2) (Fig. 5b). In spite of the large differences in mean values, they were not significant due to the high

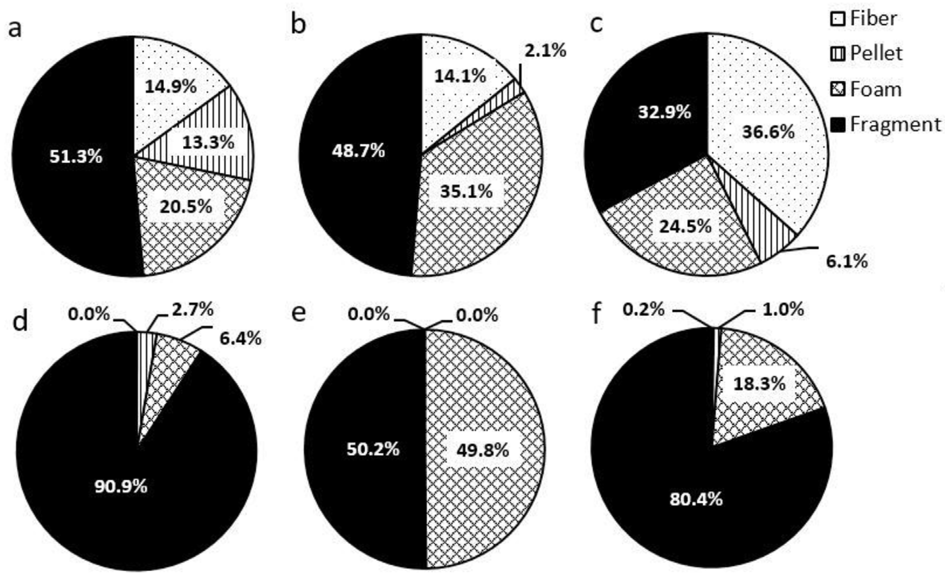

Fig. 4. Shape characteristics of microplastics (a, d) and macroplastics (b, e) on the beach and microplastics (c, f) in the tidal creek water. a, b, c: composition based on abundance (n/m^2); d, e, f: composition based on mass (g/m^2).

variability within a given category. While the abundance and mass of microplastics in the surface sediment were all higher than in the overlying litter layer, the higher abundance and mass of macroplastics resulted in the litter layer accumulating more total plastic mass. The abundance and mass of plastic debris in the sub-surface sediment were lower than the overlying layers. Larger plastic debris contributed more in the litter layer, while macroplastics in the sub-surface sediment contributed more abundance and mass than that in surface sediment (Fig. 5c and d).

3.5. Polymer composition of plastic debris

FT-IR identified more than 20 polymers types among randomly selected microplastics (n = 823) and all macroplastics (n = 172, besides visually distinguished expanding polystyrene (EPS)). Polyethylene (PE),

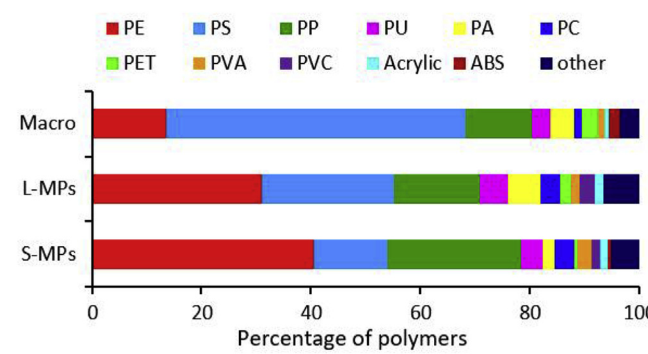

Fig. 6. Mean polymer composition of plastic debris on beach.

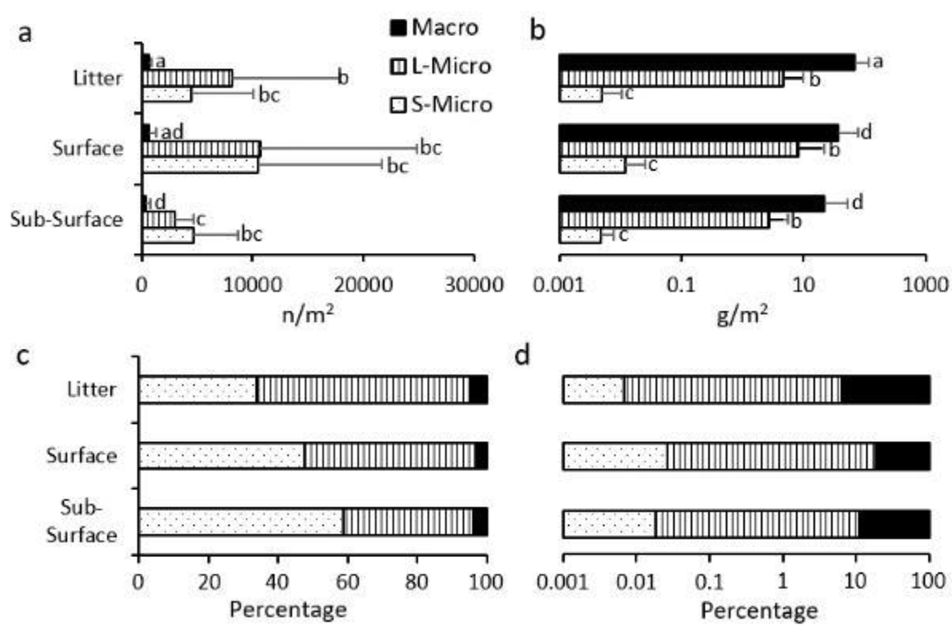

Fig. 5. Vertical size distribution of plastic debris in the litter layer and beach sediments. a: abundance of size groups; b: mass of size groups; c: contribution of size groups in amount; d: contribution of size groups in mass.

polystyrene (PS, including EPS), and polypropylene (PP) dominated the plastic debris (Fig. 6). PE was the most abundant polymer type in S-MPs (40.6%) and L-MPs (31.0%), followed by PP in S-MPs (24.4%) and PS in L-MPs (24.2%). Among macroplastics, PS was the most abundant polymer type (48.6%) followed by PE (19.8%) and PP (12.1%) (Fig. 6).

4. Discussion

In this study, large amounts of micro- and macroplastic debris on the saltmarsh beach accumulated rapidly (8 years) in conjunction with invasion of *S. alterniflora*. Compared with surveys on beaches in other regions worldwide, the amount of microplastics on the beaches of Linkun Island's saltmarsh was among the highest reported (Table S2). However, the amount of microplastics in saltmarsh sediments was much less than in sediments from a nearby inland urban river (Wang et al., 2018). This discrepancy was mainly due to differences in the small microplastic fraction < 300 μm, which accounted for no more than 50% of total microplastics on the beach compared to about 85% of total microplastics in the urban river sediments. Although the quantity of large plastics on the saltmarsh tidal flat was far less than that of microplastics, it contributed about 90% of total plastic mass. The amount of microplastics in the tidal creek water column was similar to that in the Yangtze Estuary, China (Zhao et al., 2014), but it was one order of magnitude higher than previously found in the Ou River estuary (Zhao et al., 2015). This maybe because we used a sieve diameter of 25 μm rather than the 333 μm net used in the previous study.

Different parts of the tidal flats varied in sedimentation/trapping efficiency due to differences in elevation and vegetation canopy. Accordingly, the abundance, size and shape of plastic debris deposited on tidal flats showed significant spatial variation. The abundance and mass of plastic debris at the edge of the *S. alterniflora* saltmarsh were higher than those inside the saltmarsh and on the bareflat. This indicates that the edge area where the vegetation meets the seawater has a strong ability to retain plastic debris, especially the smaller size fractions. The abundance and mass of microplastics inside the saltmarsh were significantly lower than at the marsh edge, which implies microplastics are selectively retained at the marsh edge hindering their transport into the saltmarsh interior (Fig. S6). The abundance of macroplastics in the marsh interior was not the highest among the different tidal flat positions, but the mass was the highest. The mass per macroplastic item within the marsh interior was significantly higher than the marsh edge and bareflat indicating that the interior saltmarsh position was especially effective in trapping large plastic debris (Fig. S5). Some previous studies demonstrated a strong positive correlation (r^2 from 0.59 to 0.93) between microplastics and larger plastic debris on beaches (Lee et al., 2013). However, this trend was only apparent for the bareflat position of the Linkun Island saltmarsh, while a negative correlation was found within the vegetation in the saltmarsh interior. A previous study examining differences in plastic debris retention efficiency as a function of beach position found a trend of strand line > intertidal zone > subtidal zone (Claessens et al., 2011). In contrast, our study showed that the dense vegetation changed sedimentation dynamics at the water-land interface resulting in differential sedimentation for plastic debris of different sizes, densities and shapes on the tidal flat. As a result, the saltmarsh edge below the strand line accumulated the most plastic debris in this study.

Owing to the presence of a large amount of dead plant debris, a thick litter layer (2–15 cm) formed on the surface of the edge and interior positions of the Linkun Island saltmarsh. The abundance and mass of microplastics in the litter layer were slightly less than those in the underlying surface sediments, but the number of macroplastics was similar to that in surface sediments, while the mass of macroplastics was higher than in the underlying surface sediments (Fig. 5). Both the abundance and mass of micro- and macroplastics in the subsurface sediment layer were lower than those in the upper sediment layer. Other studies have also pointed out preferential accumulation of plastic particles in the upper 5 cm of beach sediments with an overall decrease with increasing depth (Carson et al., 2011; Hengstmann et al., 2018). The litter layer in our study can also be considered as the part of the surface sediment layer as defined by previous studies and was found to preferentially accumulate large plastic debris dominated by foam.

Morphologically, fragments were the main plastic shape found on the Linkun Island's saltmarsh beach. Plastic fragments with various irregular shapes are physical and chemical degradation products of many kinds of plastic products, which make their sources much more diverse than microfibers, pellets and foams. Foams were another important type of plastic debris on the tidal flats, especially inside the saltmarsh. Even though foam density was low, their large abundance and size resulted in foams contributing half of the macroplastic mass (Fig. 4). Low density of foams also makes their deposition behavior different from other plastic types. In fact, foams will re-suspend on the water surface during each spring tide. Thus, they are more likely to be "trapped" by the tidal vegetation rather than depositing on the bareflats.

Several studies showed that plastic fibers were the dominant form of microplastics in coastal environments (Cesa et al., 2017). These microplastics mainly come from washing of plastic fiber fabrics resulting in their ubiquitous occurrence in beach sediments (Stolte et al., 2015; Cesa et al., 2017). However, other studies along the western Pacific Coast showed fragments as the dominant form in beach sediments (Eo et al., 2018). Fibers were the most abundant microplastic shape in tidal creek waters at Linkun Island. In contrast, the proportion of fibers deposited on the beach was much lower, and the contribution of fibers to the total mass of plastic debris was negligible because the mass per fiber was very low (Fig. 4). Plastic pellets are usually used as an abrasion material in industrial production or in cleaning and cosmetic products (Auta et al., 2017). Because their size range is dominantly less than 1 mm, their contribution to the total mass of plastic debris was very low.

The dominance of PE, PP and PS (Fig. 6) is consistent with their worldwide production and use statistics (Geyer et al., 2017). In addition, polyether urethane (PU), polyethylene terephthalate (PET), polyvinyl acetate (PVC) and other common plastics found in this study are also among the most prevalent plastics in use (Geyer et al., 2017). PS, which accounted for half of the macroplastics, was dominantly in the EPS form. EPS debris was mainly found in the surface litter layer on the marsh interior and was significantly larger in size than the other plastic types, thus contributing about 50% of the total macroplastic mass in spite of its much lower density.

While the amount of plastic debris on Linkun Island's saltmarsh beach was dominated by microplastics, the contribution of microplastics to total plastic mass was negligible (Fig. 3). Average mass per item did not differ among S-MPs in the tidal creek water and beach samples of different elevations, but L-MPs mass on the beach was greater than those in tidal creek water (Fig. S5). With increasing elevation, the mass per item of macroplastics increased significantly from tidal creek water to marsh interior sites (Fig. S5). The mass of macroplastics made up only 3.7% of the total plastic abundance, but made up 90% of total plastic mass (Fig. 3). As with other studies, the proportion of microplastics with particle size < 300 μm can be neglected in the total plastic mass because of its extremely low unit mass (Martins and Sobral, 2011; Klein et al., 2015). However, small microplastics are considered to pose the greatest threat to aquatic ecosystems thereby attracting much more environmental concern (da Costa et al., 2016). The dominant microplastic types in the Linkun Island saltmarsh were fragments and foams that originate from physical and chemical degradation of large plastics. Thus, macroplastics should not be overlooked as an importance source of microplastics, as well as posing a threat to larger marine organisms (e.g., whales, sharks, sea birds, etc.).

The large range in size of plastics (e.g., micron to meter) complicates the estimate of environmental impacts based only on particle numbers, especially when considering the degradability of plastics in the environment. A single, large plastic particle may continuously release thousands to millions of tiny plastic particles during degradation processes, which makes large plastic debris an important source of microplastic pollution in the environment over the long term. Therefore, to more objectively reflect the degree of plastic pollution in a given environment, not only the number of individual plastic particles is important, but also their mass should be reported as an important quantitative indicator.

The vegetation canopy in saltmarshes can significantly reduce near-bed current velocity, making saltmarshes more effective than bareflats in trapping suspended materials from estuarine waters (Leonard and Luther, 1995; Shi et al., 2000). This trapping ability increases with the extension of saltmarsh development time and increasing vegetation coverage (Rooth et al., 2003; Mudd et al., 2010). *S. alterniflora* is known for its ability to trap suspended matter from water, thereby facilitating enhanced sedimentation and rapid formation of saltmarshes (Stumpf, 1983). Historical images showed the tidal flats rapidly formed within 8 years after completion of the seawall on Linkun Island (Fig. 1), similar to other areas invaded by *S. alterniflora* in the same estuary (Wang et al., 2015). The invasion of *S. alterniflora* also appears to enhance the ability of saltmarshes to trap plastic debris.

Our results showed a large amount of plastic debris, especially macroplastics, was deposited in the saltmarsh interior and edge positions where *S. alterniflora* dominated. In contrast, microplastic abundance in the surface sediment layer beneath the vegetation canopy in the saltmarsh interior was lower than at the saltmarsh edge. This indicates that the saltmarsh interior was more efficient at trapping large-sized plastic debris than microplastics. A similar phenomenon was found for intertidal mangrove wetland, which is considered an important coastal ecosystem for trapping plastic debris (Nor and Obbard, 2014; Li et al., 2018). The marine plastic debris trapped in the mangroves was considered to degrade into microplastics (Nor and Obbard, 2014). However, the amount of microplastics in sediments under mangroves was significantly lower than the adjacent area immediately outside of the mangroves (Li et al., 2018), implying a removal mechanism for microplastics in mangroves.

Macroplastics were found to preferentially deposit on tidal flats along the western Pacific Ocean in Japan (Isobe et al., 2014; Kataoka et al., 2015). Under the influence of swash waves and wave-induced nearshore currents, meso- and macroplastics were washed ashore while microplastics were backwashed offshore. A numerical simulation model of wave-beach interactions supported the near-shore trapping of larger plastic debris (Isobe et al., 2014). The model results showed that meso- and macroplastics were selectively conveyed onshore by a combination of Stokes drift and terminal velocity factors that were dependent on fragment size (Isobe et al., 2014). Field trials also demonstrated that small suspended particles were likely to be trapped by eddies due to their smaller UTV (upward terminal velocity) and be backwashed to the surf zone by the backwash waves (Hinata et al., 2017). In contrast, the large suspended materials drifted on the surface due to larger UTV and were pushed to the upper backshore by swash waves (Hinata et al., 2017). The ability of *S. alterniflora* to trap microplastic particles at the edge position and susceptibility of macroplastics for transport to the saltmarsh interior may co-contribute to the preferential accumulation of macroplastics inside the saltmarshes and microplastics at the marsh edge of Linkun Island.

Although it is widely recognized that plastic degradation in the environment is slow, beaches are considered as a favorable environment for weathering degradation of plastics (Corcoran et al., 2009). Photochemical reactions caused by UV radiation induce oxidation, which makes plastics brittle and susceptible to shattering due to decreased elasticity. As the plastic surface weakens, friction/abrasion by wind, waves, and sand can generate microplastics from macroplastics (Andrady, 2011). Photooxidative and photothermal oxidative degradation may have a greater effect on plastics exposed on the beach surface than on the sea surface or floor due to higher UV radiation intensity and oxygen in combination with a higher temperature (Andrady, 2011). In addition, mechanical fragmentation of plastic debris deposited on beaches is likely greater due to sand abrasion associated with wind/wave action and tidal currents (Song et al., 2017). Therefore, the large amount of macroplastics trapped in the saltmarshes of Linkun Island may breakdown to form massive amounts of microplastics.

Large plastic debris trapped within the interiors of saltmarshes is isolated from direct wave/tide activity allowing the macroplastics to accumulate within the saltmarsh environment. While their ability for trapping microplastics was much weaker, microplastics entering saltmarshes during spring tides may tend to leave during ebb tides. The large plastic debris trapped in saltmarshes will experience a favorable environment for degradation to secondary microplastics, which are more easily transported to the ocean by tidal and wave processes (Andrady, 2017). These microplastics may be released to the coastal waters resulting in saltmarshes and beaches being a non-negligible "source" of microplastics to the ocean. When discussing the flux of plastics released into the ocean, we often report the total mass of plastics produced by humans, which mostly exists in the form of macroplastics in their early stage after release to the environment. Therefore, if macroplastics are readily retained on beaches and coastal wetlands, these environments may ultimately accumulate plastic materials returning to the land-sea interface from the ocean via wave, tide and current transport processes. This trapping of macroplastics may provide a possible explanation for the large imbalance between the flux of plastic entering the sea and the stock of plastic in the surface layer of the sea (Cózar et al., 2014).

In addition to the tendency of microplastics to be wash away from the coast by tidal processes, microplastics produced by degradation of large plastics in saltmarshes may be consumed by animals in the water column or benthos. Hundreds of species of marine animals have been documented to contain or ingest microplastics. Microplastics have been detected at the base of the food web in a large variety of zooplanktonic organisms, such as Chaetognatha, Copepoda, and Salpida (Setala et al., 2014; Cole et al., 2015). At higher trophic levels, both invertebrates such as Polychaeta, Crustacea, Echinodermata, and vertebrates such as (larval) fish, seabirds and marine mammals are known to ingest microplastics, either directly or via consumption of lower food web prey (Rochman et al., 2013; Barboza et al., 2018). Microplastics may even impact human health through consumption of seafood, such as mussel and oyster, which are confirmed to contain abundant microplastics (Bouwmeester et al., 2015; Wright and Kelly, 2017; Barboza et al., 2018; Rist et al., 2018). Saltmarsh wetlands in estuarine and coastal areas are key habitats for many marine and coastal organisms to feed and reproduce; they also act as a temporary transit habitat for a large number of migratory birds (Weinstein and Kreeger, 2007). Thus, materials produced in saltmarshes can play an important role in material cycling at the land-sea interface through food web dynamics (Odum, 2002).

5. Conclusion

Saltmarshes showed a strong ability to trap plastic debris in this study, which can significantly affect the distribution (e.g., size fractions) and transport of plastic pollution across the land-sea interface. The major question is whether saltmarshes act as a sink or source of various plastic size fractions to the ocean. Our data suggest different roles for different types of plastic debris. For large plastic debris, saltmarshes may serve as a strong "sink" or transformer of macro-to microplastics. Large plastic debris trapped within the interiors of saltmarshes is isolated from direct wave/tide activity allowing the macroplastics to accumulate within the saltmarsh environment. While their ability for trapping microplastics was much weaker, microplastics entering saltmarshes during spring tides may tend to leave during ebb tides. Given the favorable trapping and weathering environment for plastics in the saltmarshes, it is reasonable to speculate that saltmarshes are an important sink and/or transformer of macroplastics and a net source of microplastics to the ocean. Thus collection and proper handling of large plastic debris on beaches not only provide aesthetic benefits, but likely serves a critical role in attenuating marine microplastic pollution across the land-ocean interface.

Acknowledgement

This work was supported by the National Natural Science Foundation of China (No. 41601554 & No. 41807495), Natural Science Foundation of Zhejiang Province (No. LQ16C030004), and open fund of Key Laboratory of Watershed Sciences and Health of Zhejiang Province (No. KFKT2018003).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2019.110636.

References

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62 (8), 1596–1605. https://doi.org/10.1016/j.marpolbul.2011.05.030.
Andrady, A.L., 2017. The plastic in microplastics: a review. Mar. Pollut. Bull. 119 (1), 12–22. https://doi.org/10.1016/j.marpolbul.2017.01.082.
Auta, H.S., Emenike, C.U., Fauziah, S.H., 2017. Distribution and importance of microplastics in the marine environment: a review of the sources, fate, effects, and potential solutions. Environ. Int. 102, 165–176. https://doi.org/10.1016/j.envint.2017.02.013.
Barboza, L.G.A., Vethaak, A.D., Lavorante, B.R., Lundebye, A.K., Guilhermino, L., 2018. Marine microplastic debris: an emerging issue for food security, food safety and human health. Mar. Pollut. Bull. 133, 336–348. https://doi.org/10.1016/j.marpolbul.2018.05.047.
Boesch, D.F., Turner, R.E., 1984. Dependence of fishery species on salt marshes: the role of food and refuge. Estuaries 7 (4), 460–468. https://doi.org/10.2307/1351627.
Bouwmeester, H., Hollman, P.C., Peters, R.J., 2015. Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: experiences from nanotoxicology. Environ. Sci. Technol. 49 (15), 8932–8947. https://doi.org/10.1021/acs.est.5b01090.
Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: sources and sinks. Environ. Sci. Technol. 45 (21), 9175–9179. https://doi.org/10.1021/es201811s.
Browne, M.A., Dissanayake, A.G., Tamara, S., Lowe, D.M., Thompson, R.C., 2008. Ingested microscopic plastic translocates to the circulatory system of the mussel, *Mytilus edulis* (L.). Environ. Sci. Technol. 42 (13), 5026–5031. https://doi.org/10.1021/es800249a.
Browne, M.A., Niven, S.J., Galloway, T.S., Rowland, S.J., Thompson, R.C., 2013. Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity. Curr. Biol. 23 (23), 2388–2392. https://doi.org/10.1016/j.cub.2013.10.012.
Carson, H.S., Colbert, S.L., Kaylor, M.J., McDermid, K.J., 2011. Small plastic debris changes water movement and heat transfer through beach sediments. Mar. Pollut. Bull. 62 (8), 1708–1713. https://doi.org/10.1016/j.marpolbul.2011.05.032.
Cesa, F.S., Turra, A., Baruque-Ramos, J., 2017. Synthetic fibers as microplastics in the marine environment: a review from textile perspective with a focus on domestic washings. Sci. Total Environ. 598, 1116–1129. https://doi.org/10.1016/j.scitotenv.2017.04.172.
Chua, E.M., Shimeta, J., Nugegoda, D., Morrison, P.D., Clarke, B.O., 2014. Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, *Allorchestes compressa*. Environ. Sci. Technol. 48 (14), 8127–8134. https://doi.org/10.1021/es405717z.
Claessens, M., De Meester, S., Van Landuyt, L., De Clerck, K., Janssen, C.R., 2011. Occurrence and distribution of microplastics in marine sediments along the Belgian coast. Mar. Pollut. Bull. 62 (10), 2199–2204. https://doi.org/10.1016/j.marpolbul.2011.06.030.
Cole, M., Lindeque, P.K., Fileman, S.E., Halsband, C., Galloway, T.S., 2015. The impact of polystyrene microplastics on feeding, function and fecundity in the marine copepod *Calanus helgolandicus*. Environ. Sci. Technol. 49 (2), 1130–1137. https://doi.org/10.1021/es504525u.
Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62 (12), 2588–2597. https://doi.org/10.1016/j.marpolbul.2011.09.025.
Corcoran, P.L., Biesinger, M.C., Grifi, M., 2009. Plastics and beaches: a degrading relationship. Mar. Pollut. Bull. 58 (1), 80–84. https://doi.org/10.1016/j.marpolbul.2008.08.022.
Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, A.T., Navarro, S., Garcia-de-Lomas, J., Ruiz, A., Fernandez-de-Pelles, M.L., Duarte, C.M., 2014. Plastic debris in the open ocean. Proc. Natl. Acad. Sci. 111 (28), 10239–10244. https://doi.org/10.1073/pnas.1314705111.
Crawford, C.B., Quinn, B., 2017. Microplastic Identification Techniques – Chapter 10. Microplastic Pollutants. Elsevier, pp. 219–267. https://doi.org/10.1016/B978-0-12-809406-8.00009-8.
da Costa, J.P., Santos, P.S., Duarte, A.C., Rocha-Santos, T., 2016. Nano) plastics in the environment–sources, fates and effects. Sci. Total Environ. 566, 15–26. https://doi.org/10.1016/j.scitotenv.2016.05.041.
Eerkes-Medrano, D., Thompson, R.C., Aldridge, D.C., 2015. Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. Water Res. 75 (3), 63–82. https://doi.org/10.1016/j.watres.2015.02.012.
Eo, S., Hong, S.H., Song, Y.K., Lee, J., Lee, J., Shim, W.J., 2018. Abundance, composition, and distribution of microplastics larger than 20 μm in sand beaches of South Korea. Environ. Pollut. 238, 894–902. https://doi.org/10.1016/j.envpol.2018.03.096.
Foley, C.J., Feiner, Z.S., Malinich, T.D., Höök, T.O., 2018. A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. Sci. Total Environ. 631, 550–559. https://doi.org/10.1016/j.scitotenv.2018.03.046.
Gedan, K.B., Silliman, B.R., Bertness, M.D., 2017. Centuries of human-driven change in salt marsh ecosystems. Annu. Rev. Mar. Sci. 1 (1), 117–141. https://doi.org/10.1146/annurev.marine.010908.163930.
Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, e1700782. https://doi.org/10.1126/sciadv.1700782.
Gregory, M., 2009. Environmental implications of plastic debris in marine settings-entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philos. Trans. R. Soc. B. 364, 2013–2025. https://doi.org/10.1098/rstb.2008.0265.
Hengstmann, E., Tamminga, M., vom Bruch, C., Fischer, E.K., 2018. Microplastic in beach sediments of the Isle of Rügen (Baltic Sea) - implementing a novel glass elutriation column. Mar. Pollut. Bull. 126, 263–274. https://doi.org/10.1016/j.marpolbul.2017.11.010.
Hinata, H., Mori, K., Ohno, K., Miyao, Y., Kataoka, T., 2017. An estimation of the average residence times and onshore-offshore diffusivities of beached microplastics based on the population decay of tagged meso-and macrolitter. Mar. Pollut. Bull. 122 (1–2), 17–26. https://doi.org/10.1016/j.marpolbul.2017.05.012.
Isobe, A., Kubo, K., Tamura, Y., Nakashima, E., Fujii, N., 2014. Selective transport of microplastics and mesoplastics by drifting in coastal waters. Mar. Pollut. Bull. 89 (1–2), 324–330. https://doi.org/10.1016/j.marpolbul.2014.09.041.
Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347, 768–771. https://doi.org/10.1126/science.1260352.
Kataoka, T., Hinata, H., Kato, S., 2015. Backwash process of marine macroplastics from a beach by nearshore currents around a submerged breakwater. Mar. Pollut. Bull. 101 (2), 539–548. https://doi.org/10.1016/j.marpolbul.2015.10.060.
Klein, S., Worch, E., Knepper, T.P., 2015. Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-main area in Germany. Environ. Sci. Technol. 49 (10), 6070–6076. https://doi.org/10.1021/acs.est.5b00492.
Kneib, R.T., 1997. The role of tidal marshes in the ecology of estuarine nekton. Oceanogr. Mar. Biol. 35 (35), 163–220.
Lee, J., Hong, S., Song, Y.K., Hong, S.H., Jang, Y.C., Jang, M., Heo, N.W., Han, G.M., Lee, M.J., Kang, D., Shim, W.J., 2013. Relationships among the abundances of plastic debris in different size classes on beaches in South Korea. Mar. Pollut. Bull. 77 (1–2), 349–354. https://doi.org/10.1016/j.marpolbul.2013.08.013.
Leonard, L.A., Luther, M.E., 1995. Flow hydrodynamics in tidal marsh canopies. Limnol. Oceanogr. 40 (8), 1474–1484. https://doi.org/10.4319/lo.1995.40.8.1474.
Li, J., Zhang, H., Zhang, K., Yang, R., Li, R., Li, Y., 2018. Characterization, source, and retention of microplastic in sandy beaches and mangrove wetlands of the Qinzhou Bay, China. Mar. Pollut. Bull. 136, 401–406. https://doi.org/10.1016/j.marpolbul.2018.09.025.
Liebezeit, G., Dubaish, F., 2012. Microplastics in beaches of the east Frisian islands spiekeroog and kachelotplate. Bull. Environ. Contam. Toxicol. 89 (1), 213–217. https://doi.org/10.1007/s00128-012-0642-7.
Lo, H.S., Xu, X., Wong, C.Y., Cheung, S.G., 2018. Comparisons of microplastic pollution between mudflats and sandy beaches in Hong Kong. Environ. Pollut. 236, 208–217. https://doi.org/10.1016/j.envpol.2018.01.031.
Martins, J., Sobral, P., 2011. Plastic marine debris on the Portuguese coastline: a matter of size? Mar. Pollut. Bull. 62 (12), 2649–2653. https://doi.org/10.1016/j.marpolbul.2011.09.028.
Masura, J., Baker, J., Foster, G., Arthur, C., 2015. Laboratory Methods for the Analysis of Microplastics in the Marine Environment: Recommendations for Quantifying Synthetic Particles in Waters and Sediments. NOAA Technical Memorandum NOS-OR &R-48.
Mudd, S.M., D'Alpaos, A., Morris, J.T., 2010. How does vegetation affect sedimentation on tidal marshes? Investigating particle capture and hydrodynamic controls on biologically mediated sedimentation. J. Geophys. Res. 115, F03029. https://doi.org/10.1029/2009JF001566.
Nor, N.H.M., Obbard, J.P., 2014. Microplastics in Singapore's coastal mangrove ecosystems. Mar. Pollut. Bull. 79 (1–2), 278–283. https://doi.org/10.1016/j.marpolbul.2013.11.025.
Odum, E.P., 2002. Tidal marshes as outwelling/pulsing systems. In: Concepts and Controversies in Tidal Marsh Ecology. Springer, Dordrecht, pp. 3–7.
Oehlmann, J., Schulte, O.U., Kloas, W., Jagnytsch, O., Lutz, I., Kusk, K.O., Wollenberger, L., Santos, E.M., Paull, G.C., Van Look, K.J.W., Tyler, C.R., 2009. A critical analysis of the biological impacts of plasticizers on wildlife. Philos. Trans. R. Soc. B. 364, 2047–2062. https://doi.org/10.1098/rstb.2008.0242.
Oliveira, M., Ribeiro, A., Hylland, K., Guilhermino, L., 2013. Single and combined effects of microplastics and pyrene on juveniles (0+ group) of the common goby *Pomatoschistus microps* (Teleostei, Gobiidae). Ecol. Indicat. 34, 641–647. https://doi.org/10.1016/j.ecolind.2013.06.019.

Peng, G., Zhu, B., Yang, D., Su, L., Shi, H., Li, D., 2017. Microplastics in sediments of the changjiang estuary, China. Environ. Pollut. 225, 283–290. https://doi.org/10.1016/j.envpol.2016.12.064.
Qiu, Q., Peng, J., Yu, X., Chen, F., Wang, J., Dong, F., 2015. Occurrence of microplastics in the coastal marine environment: first observation on sediment of China. Mar. Pollut. Bull. 98 (1–2), 274–280. https://doi.org/10.1016/j.marpolbul.2015.07.028.
Rist, S., Almroth, B.C., Hartmann, N.B., Karlsson, T.M., 2018. A critical perspective on early communications concerning human health aspects of microplastics. Sci. Total Environ. 626, 720–726. https://doi.org/10.1016/j.scitotenv.2018.01.092.
Rochman, C., Browne, M., Halpern, B., Hentschel, B., Hoh, E., 2013. Classify plastic waste as hazardous. Nature 494, 169–171. https://doi.org/10.1038/494169a.
Rooth, J.E., Stevenson, J.C., Cornwell, J.C., 2003. Increased sediment accretion rates following invasion by *Phragmites australis*: the role of litter. Estuaries 26, 475–483. https://doi.org/10.1007/BF02823724.
Setala, O., Fleming-Lehtinen, V., Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. Environ. Pollut. 185, 77–83. https://doi.org/10.1016/j.envpol.2013.10.013.
Shi, Z., Hamilton, L.J., Wolanski, E., 2000. Near-bed currents and suspended sediment transport in saltmarsh. J. Coast. Res. 16 (3), 909–914.
Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Jung, S.W., Shim, W.J., 2017. Combined effects of UV exposure duration and mechanical abrasion on microplastic fragmentation by polymer type. Environ. Sci. Technol. 51 (8), 4368–4376. https://doi.org/10.1021/acs.est.6b06155.
Stolte, A., Forster, S., Gerdts, G., Schubert, H., 2015. Microplastic concentrations in beach sediments along the German Baltic coast. Mar. Pollut. Bull. 99 (1–2), 216–229. https://doi.org/10.1016/j.marpolbul.2015.07.022.
Stumpf, R.P., 1983. The process of sedimentation on the surface of a salt marsh. Estuar. Coast Shelf Sci. 17 (5), 495–508. https://doi.org/10.1016/0272-7714(83)90002-1.
Thévenot, D.R., Moilleron, R., Lestel, L., Gromaire, M.C., Rocher, V., Cambier, P., Bonte, P., Colin, J.L., de Ponteves, C., Meybeck, M., 2007. Critical budget of metal sources and pathways in the Seine River basin (1994–2003) for Cd, Cr, Cu, Hg, Ni, Pb and Zn. Sci. Total Environ. 375, 180–203. https://doi.org/10.1016/j.scitotenv.2006.12.008.
Thompson, R.C., Moore, C., vom Saal, F.S., Swan, S.H., 2009. Plastics, the environment and human health: current consensus and future trends. Philos. Trans. R. Soc. B. 364, 2153–2166. https://doi.org/10.1098/rstb.2009.0053.
Van Beusekom, J.E.E., de Jonge, V.N., 1998. Retention of phosphorus and nitrogen in the Ems estuary. Estuaries 21, 527–539. https://doi.org/10.2307/1353292.
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.
Vianello, A., Boldrin, A., Guerriero, P., Moschino, V., Rella, R., Sturaro, A., Da Ros, L., 2013. Microplastic particles in sediments of Lagoon of Venice, Italy: first observations on occurrence, spatial patterns and identification. Estuar. Coast Shelf Sci. 130, 54–61. https://doi.org/10.1016/j.ecss.2013.03.022.
Von Moos, N., Burkhard-Holm, P., Kohler, A., 2012. Uptake and effects of microplastics on cells and tissue of the blue mussel *Mytilus edulis* L. after an experimental exposure. Environ. Sci. Technol. 46 (20), 11327–11335. https://doi.org/10.1021/es302332w.
Wang, A., Chen, J., Jing, C., Ye, G., Wu, J., Huang, Z., Zhou, C., 2015. Monitoring the invasion of *Spartina alterniflora* from 1993 to 2014 with landsat TM and SPOT 6 satellite data in yueqing bay, China. PLoS One 10 (8), e0135538. https://doi.org/10.1371/journal.pone.0135538.
Wang, Z., Su, B., Xu, X., Di, D., Huang, H., Mei, K., Dahlgren, R.A., Zhang, M., Shang, X., 2018. Preferential accumulation of small (< 300 μm) microplastics in the sediments of a coastal plain river network in eastern China. Water Res. 144, 393–401. https://doi.org/10.1016/j.watres.2018.07.050.
Weinstein, J.E., Crocker, B.K., Gray, A.D., 2016. From macroplastic to microplastic: degradation of high density polyethylene, polypropylene, and polystyrene in a salt marsh habitat. Environ. Toxicol. Chem. 35 (7), 1632–1640. https://doi.org/10.1002/etc.3432.
Weinstein, M.P., Kreeger, D.A. (Eds.), 2007. Concepts and Controversies in Tidal Marsh Ecology. Springer Science & Business Media.
Wright, S.L., Kelly, F.J., 2017. Plastic and human health: a micro issue? Environ. Sci. Technol. 51 (12), 6634–6647. https://doi.org/10.1021/acs.est.7b00423.
Zhao, S., Zhu, L., Li, D., 2015. Microplastic in three urban estuaries, China. Environ. Pollut. 206, 597–604. https://doi.org/10.1016/j.envpol.2015.08.027.
Zhao, S., Zhu, L., Wang, T., Li, D., 2014. Suspended microplastics in the surface water of the Yangtze Estuary System, China: first observations on occurrence, distribution. Mar. Pollut. Bull. 86 (1–2), 562–568. https://doi.org/10.1016/j.marpolbul.2014.06.032.
Zhu, L., Bai, H., Chen, B., Sun, X., Qu, K., Xia, B., 2018. Microplastic pollution in north yellow sea, China: observations on occurrence, distribution and identification. Sci. Total Environ. 636, 20–29. https://doi.org/10.1016/j.scitotenv.2018.04.182.